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Evaluating the Swedish Producer Responsibility for Packaging Materials

Policy Design and Outcome

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Keywords: producer responsibility, collection rates, plastic packaging, cardboard packaging, waste management, recycling, cost effectiveness, inter-municipalities differences, cost benefit, supply chain management, recycling determinants, Sweden.

Abstract

During the last decades the promotion of recycling of used packaging materials has become an integral part of environmental policy. In 1994, a producer responsibility ordinance was introduced for different packaging materials in Sweden. The overall purposes of this thesis are to: (a) evaluate the design – and primarily the cost-effectiveness – of this waste management regime, and (b) attempt to explain differences in collection rate outcomes across Swedish municipalities. The thesis consists of an introductory part and two self-contained papers.

Paper [1] outlines a theoretical model that is designed to link the necessary conditions for a cost effective waste management policy to specific *policy designs*. The main purpose of the paper is to make use of this model and analyze the incentive structure and the effectiveness of the Swedish producer responsibility ordinance in the empirical context of paper packaging. A secondary purpose is to discuss if the empirical evidence suggests that an alternative waste management regime, i.e., the so-called UCTS-system, could be more effective. According to the results, both the Swedish producer responsibility scheme and the UCTS system fulfill two important cost effectiveness conditions. The packaging fee in the present Swedish system and the packaging tax in the UCTS system provide similar incentives to an *output effect*. Furthermore, both systems also give rise to *input substitution effects*, e.g., they encourage the use of secondary materials at the expense of virgin materials by subsidizing collection and recycling activities. However, in the Swedish producer responsibility system, waste collection entrepreneurs in areas with high marginal costs of collection often obtain high refunds, a situation that is in violation with the cost effectiveness criterion. Neither of the systems tends to encourage enough of *design for recyclability*, although the Swedish producer responsibility seems to perform somewhat better here. Our analysis of the transformation and transaction costs involved in the two waste management systems suggests that it is hard to a priori determine which system will minimize these costs. It will depend on, for instance, households' valuation of sorting efforts, and the presence of economies of scale in the waste collection system. This implies that different systems in different parts of the country can be preferred.

Paper [2] focuses on the actual *outcome* of the producer responsibility in the case of household plastic packaging collection. The paper investigates the main determinants of collection rates of household plastic packaging waste in Swedish municipalities. This is done by employing regression analysis based on cross-section data for 252 Swedish municipalities. The results suggest that pure cost, economic-demographic, and socio-demographic factors as well as environmental preferences all help explain inter-municipality collection rates. For instance, the collection rate appears to be positively affected by increases in the unemployment rate, the share of private houses, and the presence of immigrants (unless newly arrived) in the municipality. The impacts of distance to recycling industry, urbanization rate and population density on collection outcomes turn out, though, both statistically and economically insignificant. A reasonable explanation for this is that the compensation from the material companies varies depending on region and is likely to be higher in high-cost regions. However, if true, this also suggests that the plastic packaging collection is cost ineffective. Finally, the analysis also shows that municipalities that employ weight-based waste management fees have on average a higher collection rate than municipalities in which flat and/or volume-based fees are used.

A major conclusion of the thesis is that public policy in the waste management field ought to focus more on regional cost differences in the collection and recycling of packaging materials.

In memory of dad

List of Papers

The licentiate thesis contains this introduction and the following papers:

Paper 1: Producer Responsibility for Paper Packaging: An Effective Supply Chain Management Policy?

Paper 2: An Econometric Analysis of Regional Differences in Waste Collection: The Case of Plastic Packaging in Sweden.

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Of course, any remaining shortcomings and errors are solely mine.

Luleå, a dark Tuesday evening in November 2004

Olle Hage

Preface

1. Introduction

Recycling is no new phenomenon; mankind has always recycled used products when it has been economical.[1] However, during the last decades recycling has gained additional attention as an integral part of environmental policy and the policy strive for sustainable development. The Swedish Environmental Protection Agency (SEPA) (1998) suggests that one reason for this is that society has been relatively successful in reducing harmful emissions from factories and other point sources of emissions, but so far the diffuse environmental problems arising from the consumption of goods have been harder to monitor. One part of the finished product, the packaging, has also become a symbol for the industrialized countries' so-called "over-consumption" (Ibid.). The importance of recycling has also become visible in government work and policies, and in 1994 a producer responsibility ordinance was introduced in Sweden for packaging, waste paper and tires. Since then electronic products and cars have also been included in the producer responsibility legislation.

The main motive for introducing this type of legislation is that the producer responsibility would facilitate the fulfillment of several important environmental objectives such as: (a) reductions in the amount of waste; (b) reductions in the amount of litter; (c) increases in recycling rates; and (d) the creation and diffusion of energy and materially efficient products (e.g., SOU, 1991; and SOU, 2001).

The legislation implies that the producer has the physical and the economical responsibility for the packaging waste. The producers are obliged to provide suitable systems for collecting and recycling packaging waste, and for informing the consumers about these systems. Furthermore, the producer must consult with the municipalities about the systems for packaging collecting. The producers should also collect data on the results from the collection and recycling activities and report these to SEPA. The consumers, on their part, must clean and sort packaging waste from other waste and transport this waste to the recycling stations. (SFS 1997:185, 1997)

The producer responsibility ordinance is a law with few detailed instructions; the Swedish government preferred voluntary solutions for the industry (Prop. 1992/93:180). The producers

[1] For a comprehensive discussion, see Skottheim (2000).

also suggested a voluntary agreement. One reason for this was that it was assumed that the producers would be able to create cost effective solutions if given the freedom to design the responsibility. However, the laws that finally were introduced were motivated largely by the desire to ensure fair competition, i.e., reduce problems with free-riders. SEPA is authorized to give producers detailed instruction about what are considered to be a suitable collection system, but they also prefer flexibility on the part of the producers. For this reason the Agency has only decided to give one detailed instruction about the collection system; SEPA requires that the collection should be nationwide. The motivation for this cautious acting with the lack of detailed instructions is, again, that SEPA believes that detailed instructions could hamper the producers' possibility to create a cost effective collection system (SEPA, 1996).

As was noted above, the Swedish producer responsibility for packaging burdens producers and consumer with costs. In spite of this, no economic analysis was done prior to the introduction of the producer responsibility in 1994. This absence of economic analysis was also notified in official reports (e.g., SEPA, 1998; and RR, 1999). Since then, however, a few cost benefit analyses on the Swedish producer responsibility for packaging and newspaper have been conducted (see, for instance, Radetzki, 2000; Ekvall & Bäckman, 2001; and Bäckman et. al. 2001). In the case when the households' efforts are valued (in terms of the opportunity cost of time) all these studies, especially Radetzki's, indicate that the recycling of packaging waste is relatively costly for society. Bruvoll (1998) has drawn similar conclusions about households' paper and plastic waste recycling in Norway. This has given rise to a debate about the value of households' recycling efforts (Bruvoll et. al., 2002). The sceptics argue that many households consider recycling to be a meaningful and voluntary activity. Consequently, it is deemed unreasonable to place a cost on it. However, as a result of this debate Bruvoll et. al. (2002) and Berglund (2003) have presented results that indicate that the household on average consider their work as an effort (but the value of this utility loss can be considerably lower than, for instance, the net income).

However, even if one accepts the notion that it is socially worthwhile to promote recycling of packaging materials, there remains the question of what strategy is the most effective to accomplish this goal. Some question the ability of the producer responsibility legislation to stimulate waste minimization throughout the entire supply chain, especially since the responsibility in many cases only relates to the disposal chain (rather than to the supply chain as a whole) (e.g., Lamming and Hampson, 1996). In addition, some authors argue that there exist alternative policies, e.g., the so-called UCTS system (a combination of an output tax on goods and a recycling subsidy), which provide virtually the same incentives for resource conservation and recycling as a producer responsibility mandate but at lower administrative (transaction) costs (see, in particular, Walls and Palmer, 2001). Nevertheless, the above arguments have primarily been drawn from theoretical models and analysis, and there is still a lack of empirical evidence on exactly what types of incentives that are created by the producer responsibility legislation and if these incentives give rise to a cost effective recycling.

Following the above, the overall purposes of this thesis are to: (a) evaluate the design – and primarily the cost effectiveness – of the Swedish producer responsibility ordinance for packaging materials, and (b) attempt to explain differences in collection rate outcomes across Swedish municipalities. The first of these purposes is perhaps of special interest here, especially since the deliberate strategy to impose few detailed instructions in the ordinance has been motivated by both authorities and producers on cost effectiveness grounds. Before explaining the scientific methods to be used to fulfill the above purposes, a short explanation of the economic motives for state intervention in the waste sector is needed.

2. Efficiency/Effectiveness Criteria in Economics

Economists distinguish between different efficiency/effectiveness criteria. One central criterion is *social optimality* (or Pareto efficiency), and for our purposes it is present when all (private plus external) marginal costs of production equal the marginal benefits (avoided external costs) of the activity. In the recycling packaging markets, the marginal cost includes sorting, cleaning, transporting the packaging to the recycling stations, transaction costs for measuring what is being exchanged and of enforcing agreements, and finally the labor and capital cost needed to transform used packaging to new inputs in production. The marginal benefits consist of the avoided external costs as reduced waste disposal and reduced need for virgin resources. However, it is important to recognize that recycling activities may also consume virgin resources. If we have social optimality then the economic welfare of society is maximized. This criterion provides the cornerstone of the different cost benefit analysies reviewed above.

However, in practice it is often difficult to estimate the marginal external costs for society for a particular activity such as waste disposal. This means that politicians seldom have enough information to explicitly promote social optimality, i.e., an optimal balance between social benefits and costs. Instead they often decide upon “exogenously given” environmental goals such as a quantitative target for recycling. Thus, here the policy is an outcome of the political decision process rather than a “technical” issue to be decided using economic cost benefit analysis. Still, in such cases another important policy criterion, *cost effectiveness*, becomes central. Policies that attain a given goal at lowest possible costs are cost effective. It should be clear that cost effectiveness is a necessary – but not a sufficient – condition for social optimality. This study focuses mainly on the cost effectiveness of the Swedish producer responsibility ordinance.

3. The Methodological Approach

This thesis applies two different methods in its evaluation of the Swedish producer responsibility. These two methods are shortly discussed in this section.

In paper 1 a theoretical model is designed to link the necessary conditions for a cost effective waste management policy to specific *policy designs*, including the producer responsibility regulation and an alternative waste management system, the so-called UCTS system. This model is a modified version of a model developed by Fullerton and Wu (1998), and it provides the criteria for a cost effective waste management policy and thus forms the base of the empirical investigation. According to this model both a producer responsibility scheme and the UCTS system can, *under specific circumstances*, represent cost effective waste management schemes. Still, the maintained hypothesis in the study is that the way waste management schemes work in theory may differ a lot from how they work in practice. The core part of the empirical investigation relies on surveys and interviews of the companies involved in a specific supply chain, on surveys of the producer responsibility legislation, and reports from the responsible authorities. The aim of the empirical investigation is to explore in what way the producer responsibility legislation has affected the incentive structure and the costs in the respective companies. The empirical findings based on the present Swedish producer responsibility regulation are then confronted with the hypothetical UCTS system, and the study analyzes if this alternative system could provide a more cost effective approach to the waste management problem than the present one.

Paper 2 analyzes the actual *outcome* of the producer responsibility. Actual collection rates of households’ plastic packaging collection at the municipality level in Sweden are here analyzed, and the main determinants of household collection rates are identified. This is done using a regression analysis based on cross-sectional data from the year 2002, which include economic, demographic, institutional and policy-related variables for 252 municipalities. This approach permits us to isolate the factors that determine the variances in collection rates across Sweden and the paper particularly investigate whether different cost-related factors play an important role in shaping these regional differences.

4 Summaries of the Papers

Paper 1: Producer Responsibility for Paper Packaging: An Effective Supply Chain Management Policy?

The main purpose of this paper is to analyze the incentive structure and the effectiveness of the Swedish producer responsibility ordinance, i.e., the ability of the system to promote producers to economize with cardboard packaging and to fulfill the related environmental goals cost effectively. A secondary purpose is to discuss if the empirical evidence in any way suggests that an alternative supply chain management regime, i.e., the UCTS-system, could be more effective.

This purpose is fulfilled by designing a model for a socially optimal waste management policy and comparing the model assumption and result with empirical facts from a specific supply chain (i.e., the so-called Karin's lasagna packaging). Furthermore, the empirical findings based on the present Swedish producer responsibility regulation are then confronted with a hypothetical UCTS system, and the study analyzes if this alternative system could provide a more cost effective approach to the waste management problem than the present one.

According to the results, both the Swedish producer responsibility scheme and the UCTS system fulfill two important cost effectiveness criteria. The packaging fee in the present Swedish system and the packaging tax in the UCTS system provide similar incentives to an *output effect*. Furthermore, both systems also give rise to *input substitution effects*. For instance, the two systems encourage the use of secondary materials at the expense of virgin materials by subsidizing collection and recycling activities. However, in the Swedish producer responsibility system, waste collection entrepreneurs in areas with high marginal costs of collection also often obtain high refunds, a situation that is in violation with the cost effectiveness criterion. Neither of the systems tends to encourage enough of *design for recyclability,* but here the Swedish producer responsibility seems to be somewhat more effective than the UCTS system. The analysis of the *transformation and transaction costs* involved in the two waste management systems suggests that it is hard to a priori determine which system will minimize waste management costs. It will depend on, for instance, households' valuation of sorting efforts, and the presence of economics of scale in the waste collection system. This implies in fact that different systems can be preferred in different parts of the country.

Paper 2: An Econometric Analysis of Regional Differences for Household Plastic Packaging Waste Collection in Sweden.

The Swedish producer responsibility ordinance mandate producers to collect and recycle packaging materials. Consumers are also obliged to clean, sort and transport used packaging to the producers' collection system. According to the ordinance, producers have relatively great opportunities to design how they should fulfil their responsibility. However, SEPA requires that this collection should be nationwide.

This paper investigates the main determinants of collection rates of household plastic packaging waste in Swedish municipalities. This is done by employing regression analysis based on cross-section data for 252 Swedish municipalities.

The results suggest that pure cost, economic-demographic, and socio-demographic factors as well as environmental preferences all help explain inter-municipality collection rates. For instance, the collection rate appears to be positively affected by increases in the unemployment rate, the share of private houses, the presence of "green" households and the presence of immigrants (unless newly arrived) in the municipality. The impacts of distance to recycling industry, urbanization rate and population density on collection outcomes turn out, though, both statistically and economically insignificant. A reasonable explanation for this is that the compensation from the material companies varies depending on region and is likely to be higher in high-cost regions. However, if true, this also suggests that the Swedish plastic packaging collection is cost ineffective. The analysis also shows that the actors within the existing waste management regime can affect the collection outcome in various ways. For example, municipalities that employ weight-based waste management fees have on average a higher collection rate than municipalities in which flat and/or volume-based fees are used. Finally, the results from the study also suggest that it is reasonable to put a positive value on households' sorting activities when, for instance, analyzing the social benefits and costs of the producer responsibility ordinance.

5. Overall Conclusions and Implications

This thesis has highlighted a number of effectiveness issues in the design of different waste management systems, and it focuses in particular on the design and the actual outcomes of the Swedish producer responsibility for packaging materials. The most important implication that stems from this thesis is that society ought to be focusing more on the regional collection cost differences. It also suggests the importance for policy makers to search for a policy that stimulates so-called design for recyclability in packaging materials.

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Producer Responsibility for Paper Packaging: An Effective Supply Chain Management Policy?[*]

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ABSTRACT

The main purpose of this study is to analyze the incentive structure and the effectiveness of the Swedish producer responsibility ordinance, i.e., the ability of the system to promote producers to economize with cardboard packaging and to fulfill the related environmental goals cost effectively. A secondary purpose is to discuss if the empirical evidence in any way suggests that an alternative supply chain management regime, i.e., the UCTS-system, could be more effective. According to the results, both the Swedish producer responsibility scheme and the UCTS system fulfill two important cost effectiveness criteria. The packaging fee in the present Swedish system and the packaging tax in the UCTS system provide similar incentives to an *output effect*. Furthermore, both systems also give rise to *input substitution effects*. For instance, both systems encourage the use of secondary materials at the expense of virgin materials by subsidizing collection and recycling activities. However, in the Swedish producer responsibility system, waste collection entrepreneurs in areas with high marginal costs of collection also often obtain high refunds, a situation which is in violation with cost effectiveness. Neither of the systems tends to encourage enough of *design for recyclability*, but here the Swedish producer responsibility seems to be somewhat more effective than the UCTS system. Our analysis of the *transformation and transaction costs* involved in the two waste management systems suggests that it is hard to a priori determine which system will minimize waste management costs. It will depend on, for instance, households' valuation of sorting efforts, and the presence of economies of scale in the waste collection system. This implies in fact that different systems can be preferred in different parts of the country.

Keywords: producer responsibility, supply chain management, waste management, cardboard packaging, recycling, cost effectiveness.

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1. Introduction

1.1 Background and purpose

During the last decades the diffusion of environmental management techniques along the entire supply chain of a product has become a common way of encouraging improved environmental performance of an industry. This strategy is known as environmental supply chain management and involves the inclusion of environmental aspects in integrated management of industrial chains for manufactured goods (e.g., Lamming and Hampson, 1996). For example, the academic literature has focused a lot of attention on the way in which environmental concerns are integrated into the purchasing functions of companies (e.g., Min and Galle, 1997), including the use of environmental criteria in supplier assessment as well as collaborations on environmental standards between suppliers and customers.

Most analysts agree that there are basically two reasons why a private company decides to include environmental issues into the entire management of the company's supply chain (e.g., Berger et al., 2001). Either the company finds such practices profitable due to potential cost savings or risk avoidance or they are forced by government policy and legislation to implement them. This study focuses primarily on the latter case, namely that of the producer responsibility legislation for packaging materials. These policies have become increasingly common in many countries (e.g., Germany and Sweden), and discussions have also been underway at the European Commission for similar schemes on an EU-wide basis.

In general producer responsibility legislation states that producers who disseminate packaging are entrusted with the responsibility for collection and handling of their products at the end of their useful lives (Palmer and Walls, 1999). For this purpose, the producers are required to establish collection systems for the respective products and in some cases the legislation also mandates that a specific share of the materials consumed are recycled. The aim of the policy is clearly to influence supply chain practices and stimulate the diffusion of environmentally sound practices throughout the complex network of industrial buying and selling.

However, some analysts have questioned the ability of the producer responsibility legislation to stimulate waste minimization throughout the entire supply chain, especially since the responsibility in many cases only relates to the disposal chain (rather than to the supply chain as a whole) (e.g., Lamming and Hampson, 1996). In addition, some authors argue that there exist alternative policies, e.g., the so-called UCTS system (a combination of an output tax on goods and a recycling subsidy), which provide virtually the same incentives for resource conservation and recycling as a producer responsibility mandate but at lower administrative (transaction) costs (see, in particular, Walls and Palmer, 2001). Nevertheless, the above arguments have primarily been drawn from theoretical models and analysis, and there still exists a lack of empirical evidence on exactly what types of incentives that are created by the producer responsibility legislation and on whether alternative systems would constitute even more effective environmental supply chain management strategies.

An accurate understanding of the producer responsibility legislation and its incentive structure is important for the evaluation of environmental policy. Comparisons with alternative waste regimes are of course also necessary when considering efficient policies for society. In Sweden today only a few goods are affected by the producer responsibility (see section 2). However, there are reasons to believe that additional products will be affected by the producer responsibility legislation in the future. For instance, the Swedish Ecocycle Commission, SEC (1997) suggests that the producer responsibility should be extended to all goods in Sweden. Also the Swedish Environmental Protection Agency (e.g., SEPA, 1999) is positive towards the implementation of producer responsibility legislation for additional goods even if they also anticipate some conflicts with EC law and international trade agreements. Consequently, all producers, not only currently affected ones, should be interested in improved information about the nature and the impacts of the producer responsibility legislation.

Following the above, the *main purpose* of this study is to analyze the incentive structure and the effectiveness of the Swedish producer responsibility, i.e., the ability of the system to promote producers to economize with paper packaging and to fulfill the related environmental goals cost effectively. A secondary purpose is to discuss if the empirical evidence in any way suggests that an alternative supply chain management regime, i.e., the UCTS-system, could be more effective.

1.2 Methodology, Scope and Outline

The study follows five consecutive steeps. *First*, the structure of the Swedish producer responsibility system for packaging materials is presented (section 2). *Second*, the packaging for the product Karin's Lasagna, produced by the Swedish company Dafgård, is chosen as the case material in this study. This lasagna packaging is mainly made out of cardboard packaging. There are several reasons for this choice: (a) it is important to focus on a product produced by a company, which must comply with the Swedish producer responsibility ordinance, and Dafgård is such a company; (b) paper and cardboard packaging is a common packaging type in Sweden, making up approximately 20 percent of the packaging market

(SEPA, 2003); and (c) most of the packaging waste has a very low economic value. Thus, to be able to draw representative conclusions it was important to focus on a packaging with such a low value, and Dafgård's lasagna packaging is such a packaging. In this study the entire life cycle of this specific packaging is identified and all significant companies that are involved along the supply chain are identified (section 3). The rather detailed descriptions of the producer responsibility system in Sweden and its impacts on a specific product is deemed to be important since the way producer responsibility systems are designed in theory and the way they work in practice often differ (e.g., Palmer and Walls, 1999).

Third, we discuss some important criteria for a cost effective waste management policy, and a theoretical model is designed to link these necessary conditions for a cost effective waste management policy to specific policy designs, including the producer responsibility regulation and the UCTS system (section 4). This model is a modified version of a model used by Fullerton and Wu (1998), and provides the criteria for a cost effective waste management policy and thus forms the base for the empirical investigation.

Fourth, the core part of the empirical investigation relies on surveys and interviews of the companies involved in the chosen supply chain and on surveys of the producer responsibility legislation and reports from the responsible authorities. The aim of the empirical investigation is to explore in what way the producer responsibility legislation has affected the incentive structure and the costs in the respective companies (section 5).

The *fifth* and final part of the investigation focuses on the cost effectiveness of different waste management systems (section 5). The empirical findings based on the present Swedish producer responsibility regulation are confronted with a hypothetical UCTS system, and we analyze if this alternative system could provide a more cost effective approach to the waste management problem than the present one. Section 6 concludes the paper, summarizes the main findings and outlines some important implications.

2. The Swedish Producer Responsibility Ordinance: Design and Outcomes

This section review central components of the Swedish producer responsibility ordinance, with special emphasis on paper and cardboard packaging waste. Specifically, the section explains how the producers have organized the producer responsibility in practice, and discusses the extent to which the producers have fulfilled the goals in the ordinance.

2.1 The Ordinance in Brief

The focus on reuse and recycling as important strategies to reduce waste and resource scarcity gained strong political support in Sweden during the 1980s. In May 1990 an investigation on how to support returnable packaging systems was commissioned (SOU, 1991). One of the proposals was that legislation on producer responsibility should be introduced in Sweden. The main motive was that the producer responsibility would facilitate the fulfillment of several important environmental objectives such as: (a) reductions in the amount of waste; (b) reductions in the amount of litter; (c) increases in recycling rates; and (d) the creation and diffusion of energy and materially efficient products (see SOU, 1991; and SOU, 2001).

In October 1994 the Swedish producer responsibility legislation was introduced for packaging materials (SFS 1994:1235), waste paper (SFS 1994:1205) and tires (SFS 1994:1236). In 1998, the producer responsibility for packaging waste was revised (SFS 1997:185). One reason for the revision was that in 1998 Sweden carried out the EC Directive in the Swedish legislation. The same year producer responsibility for cars was also introduced (SFS 1997:788), and in 2001 a producer responsibility for electric and electronic products was implemented (SFS 2000:28).

The Swedish producer responsibility implies that the producers should collect their products after they have been used and they are also responsible for that these used products should be reused, recycled and/or energy recovered. The producers have both a physical and an economic responsibility for the used products. The physical responsibility implies that the producers should collect, remove and take care of the waste, while the economic responsibility means that the producers should cover the cost for this end-of-life management. The producer responsibility is an ordinance with no detailed instructions about the means of compliance, but instead leaves a significant amount of freedom to producers to organize the system themselves.

2.2 Producer Responsibility for Packaging Waste

The main aim of the producer responsibility for packaging is to decrease the packaging weights and volumes to levels that do not jeopardize safety and hygiene standards. Packaging should therefore be designed, produced and marketed so that they can be reused or recycled. Packaging should also be produced in ways that minimizes the pollution impacts that arise when the packaging is burned or dumped on a landfill (SFS 1997:185, 1§).

As was noted above, the producer has the physical and economic responsibility for the packaging waste. *Producers* are all those who professionally produce, import or sell a good in a packaging. All products that are produced to content, protect, treat, deliver and present goods from raw material to final goods and from producers to consumers shall be regarded as *packaging*. The producers are obliged to provide suitable systems for the collection of packaging waste and to inform consumers about these systems. Furthermore, the producers must consult and cooperate with the respective municipalities about the systems for collecting and the producers should also collect data providing information about the outcome of the collecting, recycling and energy recovering activities and report these result to SEPA. The *consumers* are obliged on their part to sort out packaging waste from other waste, clean the packaging and finally transport the packaging waste to the systems for collecting. The producers are not forced to pay the consumer for these work effort in spite of their economic responsibility for fulfilling the ordinance (SFS 1997:185). The decree about the producer responsibility also regulates for what purpose the collected packaging waste should be recycled and/or energy recovered and to what degree, see Table 1.

Table 1: Swedish Targets in the Producer Responsibility for Packaging (% by weight)

Type of *packaging* until 29 of June 2001 and type of *packaging waste* after 30 of June 2001.	**Recycling and reuse after 1 of January 1997 (%). Reuse in brackets.**	**Recycling or energy recovering after 30 of June 2001 (%). Energy recovery in brackets.**
Aluminum, not drink packaging	50	70
Aluminum, drink packaging	90	90
Board, paper and cardboard	30	40 (30)
Corrugated cardboard	65	65
Plastic, excluding PET-bottles	30	30 (40)
Plastic, PET-bottles	90	90
Steel plate	50	70
Glass, excluding reusable glass	70	70
Reusable glass, beer and soft drinks	(95)	-
Reusable glass, wine and spirits	(90)	-
Wood	-	15 (55)
Other materials	-	15 (15)

Sources: SFS 1997:185 and RR (1999).

The motive of these targets is primarily to attain the main aim of the legislation. Lindhqvist (2000) notes: “Collection and recycling targets are in these context secondary goals that are justifiable if they give the proper incentives for change of the products and product systems,” (p. 129).

In this paper, the definitions of the EC directive for packaging and packaging waste have been used. Reuse means to refill a packaging and use it according to the original purpose. Recycling imply that the waste should be processed and be used as input in new production. Energy recovering means that burning of the waste is permitted, provided that the energy content is recovered. For example, the packaging producers are required to collect at least 70 percent of all cardboard packaging. At least 40 percent of all this packaging should be recycled, hence used as input in new products. The rest of the collected packaging, 30 percent of all cardboard packaging, is not allowed to end up on landfills but energy recovering is seen as a suitable treatment.

2.3 The Organization and Financing of the Packaging Producer Responsibility

To fulfill the producer responsibility for packaging, in 1994 the retailers and producers founded four joint material companies that administrate the collection and recycling activities. These are Svensk Kartongåtervinning AB (SKAB) (cardboard), Plastkretsen AB (PAB) (plastic), Svenska Metalkretsen AB (SMAB) (metal) and RWA Returwell AB (RWAB) (corrugated board). Together they all form the service organizations Svenska Förpacknings-insamlingen AB (SFAB) and Reparegistret AB (REPA). Already in 1986, when the collection of glass took off, the industry founded the joint material company Svensk Glasåtervinning AB (SGAB). SGAB is separated from REPA and SFAB but cooperate with them. All these companies run with no profit interests and they do not distribute any returns to their owners.

SFAB’s task is to coordinate the operations of the material companies. For example, they establish and operate recycling stations and they also inform packaging consumers about the collection and recycling system. REPA is the material companies’ representative towards the members. Through REPA the material companies can offer a nationwide coverage of recycling systems for their packaging waste. Individual producers can fulfill their producer responsibility if they join REPA. Because the collection and recycling of packaging cannot carry its own costs the companies must pay a non-recurring membership fee, an annual fee for register maintenance as well as a packaging fee. The non-recurring membership amount is SEK 400 (in 2002) for companies with an annual turnover less than SEK 5 millions and SEK 2,000 for companies with an annual turnover higher than SEK 5 millions. All companies irrespective of annual turnover must also pay SEK 500 (in 2002) every year to REPA for register maintenance. The packaging fee is based on weight and it varies across different materials, see Table 2. (Repa, 2002)

Table 2: Packaging Fee for Different Packaging Materials (as of January 2002)

Packaging material	Packaging fees (SEK per kg)
Plastic	1.50
Metal (steel plate and aluminum)	1.50
Paper/Cardboard	0.35
Corrugated cardboard	0.23
Steel barrel (30-250 liter)	0.06

Source: REPA (2002).

The packaging fees are charged to only one part of the supply chain, this to avoid that more than one company pays for the same packaging material. As a rule, the fees are paid by the packaging filler, packer or re-packer for products made in Sweden, and by the importer for foreign products. However, in order to avoid that every retailer, pizza restaurant, hot-dog stand, consumer, office and so on must join just because they fill a packaging, the packaging for certain service packaging (e.g., carrier bags, pizza cartons, wrapping paper etc.) is charged to the manufacturer or importer of the service packaging. There is also an annual standard fee that producers with a low turnover can pay if they do not want to pay the standard packaging fee. Packaging fillers with an annual turnover of less than SEK 0.5 million do not need to join REPA and consequently do not pay anything for their packaging material. Finally, only packaging materials sold in Sweden are charged by REPA.

In 1999 REPA had about 11,000 producer members and they accounted for about 90 percent of all packaging in Sweden. It is worth noting that packaging fillers need not join REPA; they can choose to organize the producer responsibility on their own. In 1999, there were about 80 companies that had reported to SEPA and about 160 companies that had reported to REPA that they intended to organize the producer responsibility on their own. SEPA did a survey of these companies and found that many of them, especially the small ones, did not fulfill their producer responsibility. However, according to SEPA this “free rider” behavior had an insignificant effect on the environment (SEPA, 1999).

2.4 To What Extent Have the Producers Been Successful?

As has been noted above, the ultimate purpose of the producer responsibility is to achieve environmental improvements of the packaging and packaging systems through design and product development. Since this goal is difficult to evaluate, the authorities have set recycling and energy recovery targets for packaging waste, see Table 1. Table 3 displays to what extent the producers (and material companies) have fulfilled the recycling targets.

In 2002 the producers of corrugated cardboard, steel plate and glass all fulfilled their responsibility. It is in fact the case that they have a history of almost always attaining the required targets. Producers of aluminum cans and paper and cardboard have almost fulfilled their responsibility in 2002, at least for recycling. However, the producers of plastic, PET-bottles and other aluminum have not fulfilled their responsibility. Table 3 also reveals that the recycling rates have increased for most of the materials after the packaging ordinance was introduced. The above shows that overall many of the producers have managed to fulfill the secondary purpose (i.e., the required targets) of the producer responsibility ordinance.

Table 3: Reuse and/or Recycling of Packaging (Percent of Production for Selected Years)

Packaging material	1992	1994	1995	1996	1997	1998	1999	2000	2001	2002
Aluminum, not cans	-	-	-	19	12	27	33	25	22	24
Aluminum, cans	85	91	92	92	91	87	84	86	85	86
Paper and cardboard	-	10	19	45	34	37	40	34	41	37
Corrugated cardboard	67	74	77	81	84	85	84	84	85	86
Plastic, not PET-bottles**	-	-	-	-	13	19	34	43	13	16
PET-bottles***	-	49	73	76	78	80	91	91	78	77
PET- bottles, reusable	-------------- See PET-bottles up to 2001 ----------------								98	n.a
Steel plate	-	-	-	64	64	71	62	61	71	70
Glass	55	56	61	72	76	84	84	86	84	88
Reusable glass	n.a	97	98	98	97	98	98	99	99	n.a

* The figures include reuse and recycling before 30 June 2001 and thereafter only recycling.

** The decrease in the "recycling" of plastic packaging in 2001 is explained in large by the fact that for this year reuse is not defined as recycling.

*** The decrease in the "recycling" of PET-bottles in 2001 is explained in large by the fact that for this year reuse is not defined as recycling.

Sources: SEPA (2003), SEPA (2002a), SEPA (2002b) and SEPA (2001b).

However, Table 3 does not reveal how well the producers have managed to fulfill the main purpose of the legislation, namely to achieve environmental improvements in the packaging and packaging systems through design and product development.

There exist quite a few studies, which conclude that packaging has become lighter after the introduction of the producer responsibility ordinance. For instance, SEC (1998) reports that most of the packaging has become lighter during the 1990s. Packforsk (1999) has studied the packing weight development for 20 common everyday commodities in Sweden from 1994 to 1999. This study shows that the packaging weight for these commodities halved during these five years. This fact has, in spite of an economic growth with higher consumption rates, led to a decrease in the total packaging weight by approximately 17 percent. In 1991 the total amount of packaging was estimated at 1 141 000 ton and in 2000 it had decreased to 950 000 ton (SEPA, 1998). Also Hjern and Plogner (1999); and Sandström and Soutokorva (2001) conclude that packaging weights have decreased since the producer responsibility was introduced. According to Lindhqvist (2000), the experiences from other countries with similar legislation correspond well to the Swedish situation. Consequently, some evidence of an *output effect* seems to exist, implying that the use of packaging has decreased after the producer responsibility was introduced.

According to SOU (2001), about 600,000 ton of packaging waste went to recycling in 2000, which represents approximately 70 percent of the total amount of packaging in the Swedish market. This is an increase from the situation in 1994 when the producer responsibility ordinance was introduced, and when the recycling amounted to 250,000 ton of packaging waste. Consequently, this indicates that there exists evidence of an *input substitution effect* so that recycled inputs have replaced virgin inputs after the producer responsibility was introduced. On the other hand, SEPA (1998) reports that the producer responsibility ordinance has not stimulated growth in the use of returnable products. SEPA (2002a) reports that the use of plastic laminate packaging has increased substantially, and this leads to problems for the recycling firms. SEPA therefore requests producers to seriously consider "recyclability" in the production of packaging. This shows that it is not likely that changes in packaging design have made packaging easier to recycle after the producer responsibility ordinance was introduced.

It is also difficult to know to what degree the above changes in packaging depend on the producer responsibility ordinance as such. During the years since the ordinance was introduced other factors have changed and this influences the packaging weight, and the fact that packaging material is becoming lighter is not a new phenomenon. For instance, the weights of Swedish soft drinks and beer glass bottles have been reduced by nearly 60 percent during the last 70 years and the Swedish soft drink and beer can has become 25 percent lighter between 1983 to 1995 (Sandström and Soutokorva, 2001). According to SEC (1998); and Sandström and Soutokorva (2001) the main reasons for the recent weight decreases were not any fundamental technical development in packaging construction. Instead it has depended on small constant improvements due to cost optimizing with existing technologies and packaging material. There are also other reasons than cost factors that help explain the recent changes in packaging. SEC (1998); and Sandström and Soutokorva (2001) conclude that consumer preferences are an important reason. A survey also shows that there was a trend towards “green” products in Sweden from 1993 to 1995 (SEPA, 1998). In addition there has been other new legislation in the waste area during the last decade. For instance, a new Environmental Code (miljöbalken) was introduced in 1999. In 2000 and 2002, respectively, a tax on waste disposal and a prohibition to deposit burnable waste were introduced (e.g., SEPA, 2001a; and SOU, 2001). A new EEC directive on waste and waste incineration has also been introduced (SEPA, 2001a). The organization for waste collection can also be a reason behind the changed behavior. Many municipalities have implemented different volume based pricing programs, at least for private house owners (SOU, 2001). In addition, 20 municipalities (7 percent) have introduced weight based fees for the garbage, at least for the private house owner (SEPA, 2001a).

In sum, it is still unclear to what extent the producer responsibility generally provides incentives for an *output effect* and an *input substitution effect* and if this goal is attained *cost effectively*. The question of whether the producer responsibility induces changes in the recyclability of packaging materials also remains to be resolved. The remainder of this paper addresses these issues in the empirical context of Karin’s lasagna packaging.

3. Producer Responsibility in Supply Chain Management: The Case of Karin’s Lasagna[1]

In this section the supply chain for the packaging for Karin’s lasagna is briefly presented. Figure 1 provides an overview of the most important components of the supply chain. Since cardboard is the dominating packaging material throughout this supply chain, the analysis will be concentrated around this material.

In the case of Karin’s Lasagna, Dafgård is the *packaging filler* and consequently this company must pay the packaging fee to REPA. Dafgård buys their packaging from different *packaging producers*. There are three suppliers of packaging involved. CC Pack and Ekmans produce the consumer packaging. The consumer packaging consists of a cardboard tray (polyester coated cardboard) and a cardboard box. The cardboard tray is produced by CC Pack and after it has been filled with lasagna at Dafgård it is sealed with plastic folia. The paperboard tray is then placed in a cardboard box produced by Ekmans. Both CC pack and Ekmans buy their cardboard from the *material producer* Stora Enso (Olbrich, 2002; and Lindgren, 2002). Smurfit Dalwell produces the transport packaging of corrugated cardboard.

[1] This section is largely based on personal communications with industry representatives. See, in particular, Asp (2002), Pihl (2002) and Vretman (2002).

Dafgård fills every corrugated cardboard box with nine cardboard boxes. Then they pile up 80 corrugated cardboard boxes on a euro pall and secure this packaging with plastic film. The supply chain starts when Stora Enso harvests or buys forest *raw material* (Engström, 2002).

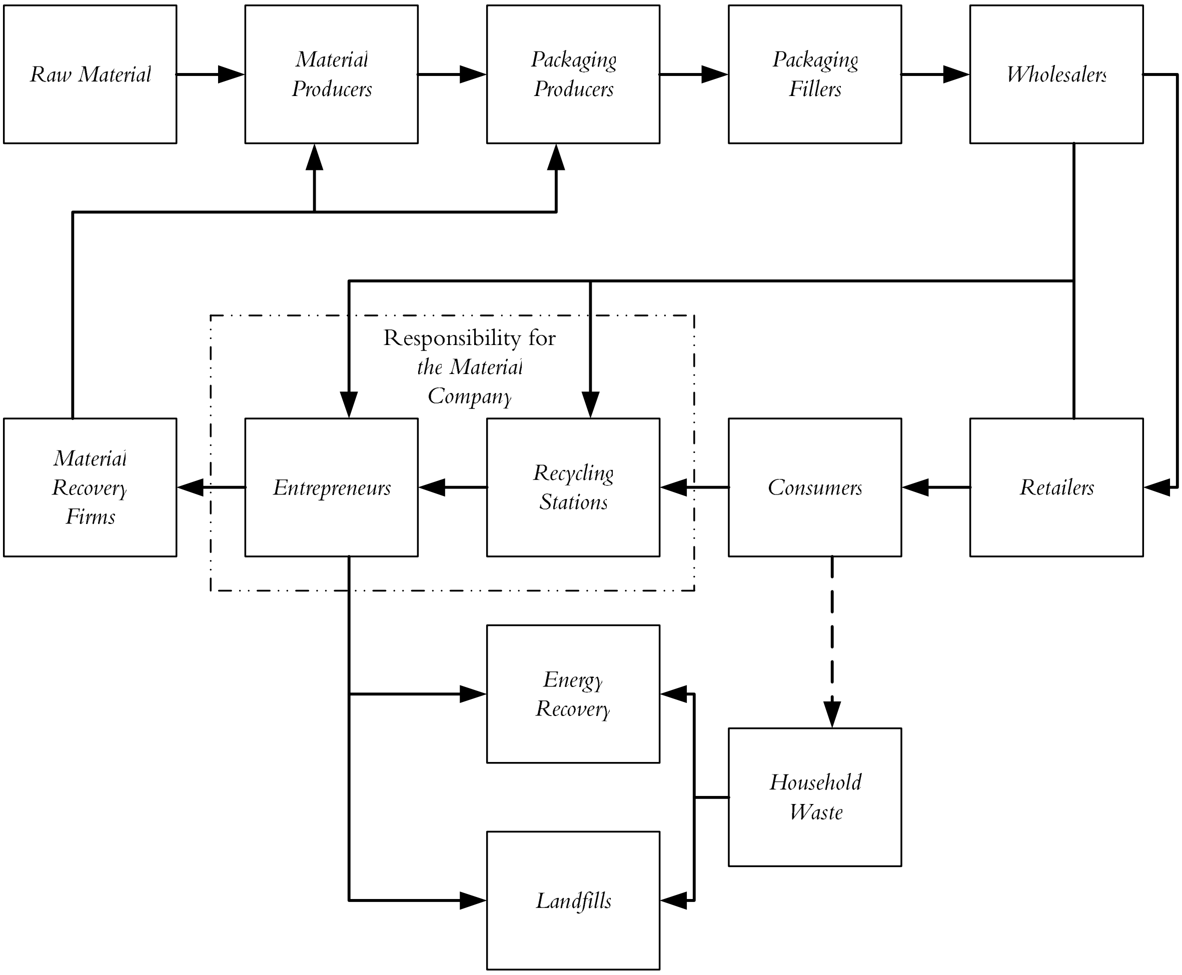

Figure 1: The Supply Chain for Paper Packaging

After the packaging has been sealed, the lasagna is transported to different *wholesalers* and *retailers*. In Sweden the food wholesale and retail sectors are dominated by ICA, Coop Sweden and Axfood. Every euro pall comprises 720 portion boxes of lasagna, so naturally some repacking at the wholesalers will normally take place because very few retailers want to buy so many lasagna boxes at once. However, this process is simple and does not involve paying more packaging fees to REPA. The wholesalers remove the plastic film and sell the corrugated cardboard boxes piece by piece to the retailers. The retailers unpack the lasagna boxes from the corrugated cardboard boxes and place the lasagna boxes in freezers.

This means de facto that the wholesalers become the *consumers* of the plastic film and that the retailers are the consumers of the transport packaging. Hence, they are, in line with the ordinance, responsible for the proper treatment of this packaging waste. Firms that are consumers of packaging materials have several alternatives when they hand over the packaging waste to the producers. They can choose to dispose of it at *recycling stations*,[2] they can hire a contractor that takes care of their packaging waste or they can sell it to the material companies' entrepreneurs.

The next step in the supply chain is that the *consumer* buys the product including the lasagna boxes. After the consumer has eaten the lasagna and has no alternative use of the paperboard tray and the cardboard box, his/her responsibility is to sort out this packaging waste from the household waste (garbage) and to clean it. Both the paperboard tray and the cardboard box must then be transported to the *recycling stations* (and there are approximately 7000 of these in Sweden), and put in the containers for cardboard and for corrugated cardboard (SFAB, 2000). However, every household does not necessarily fulfill their obligations; instead some leave the cardboard waste with the *household waste*. The municipalities are responsible for this garbage and they collect this misplaced packaging waste and leave it for *energy recovery* purposes or to *landfills*. The plastic folia should generally be disposed of with the household waste.

As mentioned above, SKAB is responsible for the cardboard packaging waste and the corrugated cardboard waste from households. The company has engaged different *entrepreneurs* to deal with this waste. The entrepreneurs put out and empty the containers, sort the cardboard in different fractions and finally compress the cardboard waste. IL Recycling is the main entrepreneur in about 160 municipalities and Stena Scanpaper is the entrepreneur in about 110 municipalities. In 5 municipalities the municipalities have the contract themselves. The entrepreneurs own the cardboard waste and sell it to *recyclers*, in this case paper and board mills. In the case of paper packaging these are the Fiskeby board mill in Norrköping that produces new cardboard and the Örebro board mill that produces surface layer on plasterboard. The recyclers pay the entrepreneurs according to quality. They check the quality with random drill testing. However, as already mentioned, the payment from the recyclers is not enough to finance the collection of waste packaging so SKAB uses large parts of the packaging fees to pay the entrepreneurs to collect and recycle the cardboard

[2] In Sweden companies use different recycling stations than do households. These recycling stations are named "mottagningspunkter" ("points of reception") and there is at least one such station in each municipality.

waste. The recycled cardboard from Fiskeby board mill is sold to the packaging producers and will eventually become new packaging.[3]

4. The Economics of Packaging Waste

In this section the economic motives for state intervention – and policy tools for improving economic efficiency – in the packaging waste sector is first discussed. Furthermore, selected earlier studies on economically efficient waste management system are reviewed. The section ends by outlining a simple general equilibrium model, which can be used to evaluate the economic efficiency of different waste management regimes. This model forms the basis of the empirical analysis that follows in section 5.

4.1 Addressing Market Failures in the Packaging Waste Sector

Economists often distinguish between different efficiency criteria. One central criterion is *social optimality*, and it is present when the private marginal cost of production equals the societal marginal costs of production (and these in turn equal the marginal benefit of the activity). Unfortunately there are several facts indicating that most waste disposal "markets" do not function in this ideal way, i.e., one or several market failures are present (e.g., Radetzki, 2000). This motivates state intervention of some kind in this market.

One of the reasons why the waste disposal market does not function well is the presence of external costs in deposition in landfills. External costs are present when the consumption or production decisions of one agent have a negative impact on the welfare of another agent, and if no compensation is paid by the generators of the impacts to the affected part.

These external costs imply that the private marginal cost for waste disposal will be lower than the marginal cost for society. Consequently the deposition in landfills will be too large. In the case of deposition in landfills, there are several external costs present. Some waste will give rise to contaminated leachate and methane emissions (SEPA, 2001a). Smell and noise from landfills will disturb the people who live close to the landfill. Landfills can also destroy the view of the landscape. The primary problem in these cases are too much waste, not that recycling is too low, but policies that stimulate recycling should of course also decrease the problem with too much waste. It may also be the case that the actors in the market are too myopic, and do not take into account the full costs of the depletion of natural resources.

[3] According to Vretman (2002), the packaging producer Dafgård does not use this recycled packaging because the company's management is of the opinion that recycled cardboard does not meet their quality standards.

Production based on virgin materials, especially if it is an exhaustible (non-renewable) material, denies future generations these materials and consequently implies an external cost.[4] This external cost can be avoided through more intense recycling activities.

Another potential source of market failure is that public policy often implies a bias against the recycling of materials. Historically, waste deposition in many countries, among them Sweden, has been financed by taxes or fixed fees (flat prices). Under such funding mechanisms households and firms face zero prices for additional waste deposition. Consequently, the marginal private cost for waste deposition is zero although the marginal cost for society is positive. This leads to an inefficiently large amount of waste deposition since households and firms do not have any economic motives to consider deposition costs in their purchasing and recycling decisions. This in turn does not create enough incentives for producers to construct products that are less costly to dispose of and/or that can easily be recycled into new products at the end of their lifetime.

However, it is often difficult to estimate the total marginal cost for society for a particular activity such as waste disposal. This means that politicians seldom know how to attain social optimality, i.e., an optimal balance between social benefits and costs. Instead they often decide on "exogenously given" environmental goals such as a quantitative target for recycling. In such cases another important policy criterion, *cost effectiveness*, becomes central. Policies that attain a given goal at the lowest possible cost are cost effective. It should be clear that cost effectiveness is a necessary – but not a sufficient – condition for social optimality. This study concerns mainly with cost effectiveness. Thus, the analysis in chapter 5 does not, for instance, deal with questions related to the benefits of recycling (versus landfill or incineration). In spite of this it is important to know if a regulation has the theoretical opportunity to be socially optimal, because if so we also know that the regulation provides the correct incentives for waste management to be cost effective.

Economists usually recommend the use of different economic instruments to correct for the market failures that are present. Commonly used economic policy instruments are Pigovian taxes, subsidizes and tradable pollution permits. These incentive-based instruments have several advantages over "command-and-control" policies that involve the setting of technology or emission standards enforced with legislation without the aid of market-based incentives. An economic instrument promotes the *least cost way* of achieving any pre-determined

[4] Economically the presence of non-optimal natural resource use could be the result of the use of private discount rates that are higher than the preferred social discount rate.

environmental goal. For instance, a tax on waste deposition will raise the cost of this activity and consequently the producers with the lowest marginal cost for recycling will recycle more than the producer with high cost for recycling. Furthermore, economic instruments induce what are often called *dynamic efficiency* effects. Because the waste producer must pay for every unit of waste deposition, it will have an incentive to develop more efficient ways of reducing waste deposition. Revenues from environmental taxes can also be used to reduce other distortionary taxes in the economy such as income taxes. (Palmer and Walls, 1999)

4.2 Earlier Economic Studies on Alternative Waste Management Strategies

The easiest method, at least in theory, to correct for market failures in the waste market would be to charge households according to the amount of waste (e.g., according to weight) that they generate (so-called unit-based pricing). This method solves the problem with zero pricing for additional waste deposition and makes it possible to charge for the external cost if it is a public landfill. If the landfills are private, the society can easily reach the same result with a Pigovian tax on the (private) cost. However as Dinan (1993) notes, the unit-based pricing system will lead to problems with illegal waste disposal and high transaction costs. Fullerton and Kinnaman (1996) provide some empirical support for this notion in their case study of the U.S. city Charlottesville.

High transaction costs depend on the need for a complicated system for measuring and controlling waste disposal. To function well a system with unit-based pricing must have different prices on different waste materials because the external effects vary for different waste materials. Every waste fraction must also be measured with a “fair” method. For instance, Fullerton and Kinnaman (1996) show that the system in Charlottesville, where the households were charged per volume, leads to stomping. This was unnecessary work for society since collectors compact their waste anyway. Calcott and Walls (2000) also show with a theoretical model that unit-based pricing only creates incentives for “design for environment” when there is a fully functioning recycling market.

A regulation with the same incentives as unit-based pricing is a deposit-refund system. In Sweden the glass bottles for soft drinks and beer are the best-known examples of a deposit-refund system. In traditional deposit-refund systems the consumer pays a deposit (tax) on a product at the time of purchase and receives a refund (subsidy) when he/she returns the product for recycling. Since the consumer ends up paying the tax only for the products that they do not recycle, the system is equivalent to unit-based pricing but without the problem of illegal waste disposal. Unfortunately, high transaction costs can often be a problem with traditional deposit-refund system (Palmer and Walls, 1999).

A special case of the deposit-refund system is the “upstream combination tax/subsidy” (UCTS) system. This method utilizes an output tax on the producer of goods and combines it with a subsidy to the producer that recycles the waste. This method can, it is argued, impose lower transactions cost than the deposit-refund system, primarily because there will be fewer participants and products involved. Dinan (1993), Palmer and Walls (1997), and Calcott and Walls (2000) show with different (theoretical) partial equilibrium models that this UCTS should create a socially optimal resource allocation if the tax and subsidy are set to reflect the marginal social cost of disposal. Fullerton and Wu (1998) come to the same conclusion with a (theoretical) general equilibrium model.

Virgin material taxes, output taxes or recycling subsidies are often put forward as effective policy tools to address market failures in the waste market (e.g., Dinan, 1993; Palmer and Walls, 1997). All these methods give rise to the same problem if they (unlike the UCTS system) are enforced in isolation; they will not price the primary problem, that is the marginal external cost from waste deposition, and they will therefore not be efficient. For instance, Dinan (1993) shows that a virgin material tax would not stimulate the use of recycled materials in products that do not use virgin materials as input. Hence, a virgin material tax would not promote enough of recycled materials in such products.

Palmer et al. (1997) use a theoretical model calibrated with elasticity estimates for demand and supply to compare the UCTS-policy with output taxes (ADF) and with recycling subsidies in the American waste markets for paper, glass, plastic and steel. They conclude that the UCTS system is much more cost efficient than the other two regulations. Furthermore, they conclude that the least-cost approach, an overall decrease of waste with 10 percent, has a marginal cost of USD 45 and that a 10 percent decrease in each waste material had a marginal cost of USD 70. Clearly, a producer responsibility system – such as the Swedish one – represents yet another way of addressing market failures in the waste sector. The remainder of this section builds on the above studies, and outlines a modified version of Fullerton and Wu’s (1998) general equilibrium model that permits a consistent theoretical comparison of the UCTS system and a stylized producer responsibility system. In section 5 we then make use of these theoretical insights and analyze to what extent any similarities and differences between the two systems tend to be important in practice.

4.3 A General Equilibrium Model of Packaging Waste

This section outline a model that can be used as the basis for an evaluation of the effectiveness of the Swedish producer responsibility ordinance and compare it to the UCTS system. The model is a revised version of a model presented by Fullerton and Wu (1998) and it is designed to convey basic intuition about the supply chain for paper packaging and it displays necessary conditions for the implementation of efficient waste management regimes.

The model has several important attributes. *First*, it covers the entire lifecycle of each product. *Second*, the distinction between economic and regulatory instrument can be analyzed with the model. *Third*, the model acknowledge the fact that neither recycling nor waste reduction are the primary goals of an efficient waste management strategy. Instead it stipulates that all products and all forms of waste should be priced so as to reflect the full social cost. (Fullerton and Wu, 1998)

In the model, firms use packaging to protect their main products. The packaging is produced by a mix of primary resources and recycled materials. Different firms use different amounts of packaging and they also choose different levels of recyclability. The households supply primary resources and hold back some resources for leisure in the model. They also generate the amounts of waste that end up in landfills or in recycling, which in turn depends on the firm's choice of packaging and the firm's choice of product recyclability.

4.3.1 Model Assumptions and Specification

In the model the economy has *n* identical households that buy a single composite commodity with a packaging rate *q*. The packaging has one single attribute; a degree of recyclability *ρ*. Households choose to dispose the packaging waste in the form of garbage *g* that end up in a landfill or in the form of recycling *r* which end up in new products. The household technology will determine the generation of *g* so that:

$$g = g(q, \rho) \tag{1}$$

where *g(….)* is continuous and quasi-concave, with the first derivatives: $g_q > 0$ and $g_\rho < 0$. In a similar way the generation of *r* is given by:

$$r = r(q, \rho) \tag{2}$$

where *r(....)* is continuous and quasi-concave, with first derivatives: $r_q > 0$, $r_\rho > 0$. The households' utility functions u are given by:

$$u = u(q, h, G) \tag{3}$$

where q denotes packaging, h denotes another good produced and consumed at home and G denotes the negative externality arising from the production of garbage. G is a function of all garbage, $G = ng$. The first derivatives for the two last terms are straightforward: $u_h > 0$, $u_G \leq 0$. The impact of changes in q on household utility is more complicated. If products have low degrees of packaging, they will be harmed and consumers must replace broken units. Hence, more packaging has a positive marginal utility ($u_q > 0$). On the other hand, if products have higher levels of packaging, they will eventually be difficult to handle (e.g., take up space and be heavy to transport). Hence, too much packaging has negative marginal utility ($u_q < 0$). Furthermore, the production function for packaging can be expressed as:

$$q = f(k_q, r, \rho) \tag{4}$$

Competitive firms produce packaging q under the condition of constant return to scale, using production factors k_q and recycled materials r. In equilibrium, the firm's use of r must equal the households' generation of r. The firm must also choose the product's recyclability, ρ. The first derivatives for the three terms are straightforward: $f_k > 0$, $f_r > 0$, $f_\rho > 0$. Fullerton and Wu (1998) assume that the garbage collection industry uses the resource k_g as the only input under constant returns to scale, implying that the production function for g will be linear so that:

$$g = k_g \gamma \tag{5}$$

where γ is a constant. The households produce the good h, which can be interpreted as leisure, from home use of time and resources, k_h.

$$h = k_h \tag{6}$$

Finally, the following resource constraint closes the model:

$$k = k_q + k_g + k_h \tag{7}$$

where k denotes an aggregate fixed total resource.

4.3.2 Model solutions

We here distinguish between the solution in a social planning context ("first-best" solution) and the outcome in a decentralized economy. In a social planning model the planner's goal is to maximize the utility of a representative household, Eq. (3), subject to the resource constraint in Eq. (7), the production functions in Eq. (4-6) and the waste generation technologies Eq. (1-2). Using the appropriate Lagrangian the first-order conditions become:

$$\frac{u_q}{u_h} = \frac{1}{f_k} + \left(\frac{1}{\gamma} - \frac{nu_G}{u_h}\right)g_q + \left(-\frac{f_r}{f_k}\right)r_q \tag{8a}$$

$$-\frac{f_\rho}{f_k} + \left(\frac{1}{\gamma} - \frac{nu_G}{u_h}\right)g_\rho + \left(-\frac{f_r}{f_k}\right)r_\rho = 0 \tag{8b}$$

where the first bracket in every equation is defined as the marginal social cost per unit garbage (*MSCg*). This cost includes both the direct resource cost/internal cost ($1/\gamma$) and the external cost ($-nu_G/u_h$). Eq. (8a) says that the marginal utility from another unit of q (packaging rate) would equal the marginal social cost of producing and disposing of it. Eq. (8b) shows that in optimum recyclability ρ should increase until its marginal resource cost offsets the savings in disposal costs.

The above is a "first-best" solution because it does not incorporate any distorting taxes on the production factors. Fullerton and Wu (1998) here assume that the government can use lump sum taxes to finance subsidies needed for garbage collection and recycling, and they also assume that revenues from taxes on packaging or garbage disposal are returned to consumers as lump sum transfers. This assumption simplifies the analysis without changing the main result that the marginal benefit of having more of the good equals the marginal cost. On the other hand the model captures other market failures due to illegal dumping, transaction costs and enforcement problem. The household budget is altered as follow when the government is allowed to provide different tax incentives:

$$\left(k-k_h\right)+\left(p_r-t_r\right)r=\left(p_q+t_q\right)q+\left(p_g+t_g\right)g \tag{9}$$

Each household owns k resources and sells (k-k_h) to the market at a price of one (since k is a numeraire). The payment for each unit of recycling is p_r which can be taxed at rate t_r per unit. All taxes can be positive or negative. This income can be used to buy the packaging q at a price p_q with a per-unit tax rate t_q. Eventually, the household might have to pay the price p_g and the tax t_g for each unit of garbage. Firms' profit functions will also be affected by the introduction of taxes:

$$\pi=p_q q-p_r r-k_q-qt_\rho \rho \tag{10}$$

where t_ρ is the tax per unit of recyclability. Individual firms will in aggregate face "demand" schedules for ρ that are reflected in the price p_q. The consumer will be willing to pay more for a product with greater recyclability ($\partial p_q/\partial\rho \geq 0$) if they must pay for garbage disposal. Using the appropriate Lagrangian the first-order conditions give:

$$p_q=\frac{1}{f_k}+\rho t_\rho \tag{11}$$

$$p_r=\frac{f_r}{f_k} \tag{12}$$

$$\frac{\partial p_q}{\partial\rho}\cdot q=qt_\rho-\frac{f_\rho}{f_k} \tag{13}$$

Consequently, the price p_q just covers resource cost plus taxes per unit of output if the market is characterized by competition. Firms will also use more r until its marginal cost is offset by its cost to the firm. Competitive firms in the garbage collection industry maximize profit ($p_g g$ – $p_k k_g$), where $g = \gamma k_g$ and $p_k = 1$, so $p_g = 1/\gamma$. Consequently, the price on garbage just covers costs. In this decentralized economy, household maximizes utility in Eq. (3) subject to the budget constraint Eq. (9) by choosing h, q and attributes ρ (which together determine g and r).

Using the appropriate Lagrangian gives the first-order conditions in terms of the prices and tax rates faced by households, but using Eq. (11)-(13) above to replace each price with the corresponding cost of production gives:

$$\frac{u_q}{u_h} = \frac{1}{f_k} + \rho t_\rho + t_q + \left(\frac{1}{\gamma} + t_g\right) g_q + \left(\frac{f_r}{f_k} + t_r\right) r_q \quad (14a)$$

$$q t_\rho - \frac{f_\rho}{f_k} + \left(\frac{1}{\gamma} + t_g\right) g_\rho + \left(\frac{f_r}{f_k} + t_r\right) r_\rho = 0 \quad (14b)$$

$$q t_\rho - \frac{f_\rho}{f_k} = 0 \quad (14c)$$

Eq. (14a-c) shows a general equilibrium where all firms are on their supply curves and all households are on their demand curves for each commodity and attribute. Eq. (14a) states that marginal utility should be equal to the "full effective price" of consumption. The consumer must pay the firm's cost for resources and taxes plus their own cost for disposal.

4.4.3 Market Failures and Corrections

In this section, the necessary conditions for socially efficient producer responsibility and UCTS systems are evaluated by solving for the tax rates that induce private behavior in Eq. (14) to equal the social optimum conditions in Eq. (8a-b).

Case A: A Producer Responsibility Scheme

It can be shown that a producer responsibility scheme could induce social optimality. This is the case if we assume a system where the producer must take back the own packaging waste from packaging consumers and where the producers have the option whether to recycle or to dispose of the packaging waste. The producer responsibility causes some modifications in the model, because the responsibility for garbage disposal and recycling shift from the household to the firm. Hence, the household budget constraint will change as follows:

$$k - k_h = \left(p_g + t_g\right) q \quad (15)$$

In addition, the firm profit function becomes:

$$\pi = p_q q - k_q - \left(p_g + t_g\right) g - t_r r - q \rho t_\rho \quad (16)$$

It can be shown that this system will lead to a social optimal solution if the producer has to pay the social marginal cost of garbage. This means that the firm should pay $p_g = 1/\gamma$ and $t_g = -nu_G/u_h$ and all other taxes should be set to zero. The tax on garbage will increase the cost for waste disposal and hence lower the amount of disposal and increase the amount of recycling. However, this is not enough to induce social optimality. In order to provide optimal incentives to reduce packaging and to design for recyclability it is important that producers actually collect their own packaging. This means that all packaging costs for the society are internalized in the firms' production decision.

Case B: A UCTS Scheme

As mentioned above, one approach to solve for the presence of market failures in the garbage market could be a special form of a deposit-refund system, the so-called "upstream combination tax/subsidy" (UCTS) system. It can be shown that this method can also solve the problem when the disposal fee is zero ($p_g + t_g = 0$). This time the government needs to enforce taxes on packaging t_q and the commodity together with a subsidy for recycling t_r as follow:

$$t_q = \left(\frac{1}{\gamma} - \frac{nu_G}{u_h}\right)\left(g_q - \frac{g_\rho r_q}{r_\rho}\right) \tag{17}$$

$$t_r = \left(\frac{1}{\gamma} - \frac{nu_G}{u_h}\right)\frac{g_\rho}{r_\rho} \tag{18}$$

The tax on packaging will increase the cost for packaging and hence lower waste disposal. Furthermore, the subsidy on recycling will decrease the cost of recycled material and stimulate the use of recycled inputs. To generate a socially optimal solution this model also need to assume a perfect market for recycling collection. This means that there should be a positive price on recyclable items and this price should reflect recyclability. If so the consumer will demand 'design for environment' in their buying decision. If there is some failure in the market for recycled material ($p_r = 0$), the consumer will not demand enough of recyclability. The government could, at least in theory, compensate this market failure with a subsidy on recyclability. In reality it seems difficult to quantify recyclability; at the least it will induce very high transaction costs to measure and monitor this concept.

4.4 The Importance of Transaction Costs

According to North (1990), the total production cost is the sum of transformation and transaction costs. The importance of transactions costs in production has been known since Coase (1937) explained the existence of firms. He defines transaction costs as the costs of measuring what is being exchanged and of enforcing agreements. Transformation costs are the cost for using production factors. In the case of paper packaging recycling, the transformation costs is the labor and capital cost that are needed to transform used paper packaging to new packaging. The transaction cost is all cost that is needed to control and enforce that this really takes place. The model outlined above only addresses transformation costs, but in the choice between different waste management policies, transaction costs need also to be evaluated. In the case of the Swedish producer responsibility system there are several types of transaction costs. For instance, producers have organized themselves and created a nation-wide collection system. The material companies negotiate with and contract packaging fillers and waste entrepreneurs and they also enforce the agreements. Moreover, the material companies must inform packaging consumers about the collection of used packaging and the government must monitor and evaluate the system. Given an environmental goal, cost effectiveness implies the minimization of total production cost. This is not the same as minimizing the transaction costs. It is possible that a system with low transaction cost can have higher total production cost than other systems because of higher transformation costs (and vice versa). Consequently, it is important to evaluate the transformation and transactions costs associated with different waste management schemes. This will be done in section 5.2.4.

4.5 Brief Summarizing Remarks

The model outlined in this section shows that both a producer responsibility system and a UCTS system can induce a socially efficient waste management regime and consequently also promote cost effectiveness. As Palmer and Walls (1999) point out, the UCTS-system is also consistent with the objectives of producer responsibility. *First*, the tax can finance some of the costs for municipalities to manage the waste. *Second*, the tax reduces output/packaging and hence reduces waste. *Third*, the subsidy for recycling stimulates the use of recycled material. However, the way producer responsibility schemes are designed in theory and the way they work in practice can be quite different. There are also reasons to believe that the transaction costs will differ for these two types of systems (Ibid.). The above motivates an empirical investigation of the cost effectiveness of the Swedish producer responsibility system. In the empirical part, the following evaluation criteria, which all can be drawn from the above model, guide the analysis:

- To be cost effective a waste management policy should give incentives to an *output effect*, i.e., it should provide firms with an incentive to produce less output (and hence less waste) and/or encourage greater material efficiency.
- The policy should stimulate to an *input substitution effect*, i.e., encourage substitution of recycled inputs for virgin ones.
- Producers should also have an incentive to undertake cost-effective changes in product design, including increasing *product recyclability,* i.e., "design for environment".
- The transaction costs (e.g., contract costs, monitoring etc) of the system must be kept as low as possible.

It is important to point out, that at the same time as a cost effective policy must provide incentives for the firms to fulfill the three first criteria it is not necessary that a single firm actually fulfills all criteria. For instance, the policy could be cost effective even if the output effect is zero for a single company (even in the presence of an output tax); this would be the case if this particular company finds it more economical to fulfill the authority's requirements by, say, input substitution.

5. Does the Swedish Producer Responsibility Constitute an Effective Waste Management Policy?

The general equilibrium model outlined in the previous section identified the necessary conditions for a socially efficient waste management policy. This model will be used as a benchmark when analyzing if the existing Swedish producer responsibility regime and/or the UCTS approach could be labeled as cost effective waste management policies. This section begins by comparing the above theoretical producer responsibility scheme with the Swedish producer responsibility (section 5.1); this analysis is important since the way the producer responsibility is designed in the theory may differ substantially from the existing scheme in Sweden. The study then focuses on the cost effectiveness aspects of the Swedish producer responsibility system and compares these to the alternative UCTS system. Throughout this analysis the practical experiences from the recycling of our case packaging material, i.e., the supply chain for Karin's Lasagna, will be emphasized.

5.1 Remarks on Producer Responsibility in Theory and in Practice

As was noted in section 4, the producer responsibility can be a socially optimal solution for addressing the presence of market failures in the used packaging field. A producer responsibility can simply force the producers to collect the used packaging they have sold and then give them the option to recycle it or to dispose of it on a landfill. In such a case it is enough to let producers pay the social marginal cost of garbage when they dispose of the used packaging on landfills. This will give them the right incentives to use a socially optimal amount of packaging and to design the packaging with a socially optimal recyclability. Eventually they will also recycle the optimal amount of used packaging. For this type of system to work well it is not necessary for producers to actually collect all packaging they have sold. A more cost effective solution could be to measure the used amount of packaging in the respective firm's production and also measure the amount of packaging each firm is recycling. Then this producer responsibility should create social optimality if each firm paid a optimal tax whose level was based on the difference between the used and recovered amount of packaging. The differences between the Swedish producer responsibility and the theoretical scheme outlined are however rather large.

As was noted in section 2, the Swedish producer responsibility mandates the producers to collect and recycle the packaging, exactly as in the model. However, this does not mean that the Swedish producers are forced to collect and recycle all packaging. Instead they must fulfill some required levels of recycling for different packaging materials (e.g., 40 percent of the paper packaging must be recycled). These different solutions could in theory give the same result if the required level for recycling in the Swedish system is socially optimal and if the garbage tax in the theoretical model is socially optimal, at least when we analyze aggregate levels. However, when we analyze individual firms it is not reasonable to assume that the Swedish system could reach this optimal solution. Of course different firms will have different costs for recycling their packaging, since they use different paper materials and designs, which in turn induce different costs for the recycling. There will also be different distances for different firms to collect, recycle and dispose of the packaging. Hence, to be cost effective the system should induce firms with low cost for collection and recycling to collect and recycle relatively much (and vice versa). In the theoretical model the garbage tax actually stimulates this type of cost effective behavior. If the firm has low costs of recycling they will recycle a lot and be spared from garbage taxes, and if they have high costs for recycling they will recycle less and instead pay the garbage tax. Nothing in the Swedish scheme induces this type of behavior on the individual firm level, and this is of course a shortcoming of the Swedish producer responsibility system.

In January 2000 a waste tax was introduced for waste that ends up at a landfill. This tax is paid to the government by the landfill firms. These firms are typically owned by the municipalities, and they have responded to the new tax in primarily two ways. They burn and compost more waste because they do not need to pay tax for this usage, and they have increased the waste fees (SEPA, 2001a). However, this tax does not function in the same way the tax in Fullerton and Wu's model does, primarily because producers are still forced to recycle the packaging according to the producer responsibility.

On the other hand, there is one important advantage with the Swedish model. For Fullerton and Wu's theoretical model to work properly in practice, each producer must collect the packaging they have sold. This implies that every producer must have their own system to collect the packaging they have sold, implying very high average collection costs for small producers. The Swedish producers are permitted to set up their own system for packaging collection but as we have noted above, very few companies have chosen to do this by their own. Instead they choose to join REPA and pay them a packaging fee, something which de facto means that they hand over the responsibility to REPA and to SKAB. The cost for this collection system is probably much lower than would be the case if each producer would be forced to collect their sold packaging. It is possible to identify at least three observations that support this conclusion. *First* of all, the fact that the majority of the Swedish packaging fillers have joined a voluntary system is an indication of lower costs within the present system. *Second*, the collection of used packaging involves rather high capital costs (Pihl, 2002). *Third*, Sweden is a rather large country with a low population density. Both the second and the third reason imply that the collection of packaging is characterized by economies of scale. Consequently, the Swedish system with one large collector should have much higher levels of collected material per collector and thus much lower average collection costs.

As was noted in section 4, the UCTS system can also be a socially optimal solution. In the theoretical UCTS system outlined above it is enough for the government to enforce a tax on packaging and a subsidy on recycling (at optimal levels), provided that we have a well functioning recycling market. In fact this means that the Swedish producer responsibility has rather large similarities with this UCTS system. The Swedish packaging fee directly works as an upstream packaging tax and the payment from REPA to the collectors essentially works as a subsidy to the collectors. However, there are also important differences and in the next section we address these in more detail.

5.2 Cost Effectiveness in the Swedish Producer Responsibility Scheme and in the UCTS System

This section analyzes each of the criteria for a cost effective waste management system in the context of: (a) the present Swedish producer responsibility system; and (b) a hypothetical UCTS system as it could be designed to work in practice.

5.2.1 Output Effect

According to the interviews with industry representatives, individual firms in Sweden do not generally consider themselves responsible for fulfilling the authorities' recycling requirements. They fulfill, they argue, their responsibility when they pay the packaging fee. This packaging fee will essentially work as a packaging tax and there is no such tax in Fullerton and Wu's producer responsibility scheme. The packaging fee increases the cost for packaging and hence induces reductions in the amount of packaging. Consequently it has an output effect, but it can of course be low. It is also obvious when talking with industry representatives that this fee represents one – out of many – economic reasons to decrease the amount of packaging (see, in particular, Vretman, 2002; Lindgren, 2002; and Olbrich, 2002).

The packaging tax in the UCTS system will function more or less as the packaging fee in the Swedish producer responsibility system. However there is one difference. The upstream tax in the UCTS system is decided directly by the government and the Swedish packaging fee is decided by REPA. Still, the Swedish packaging fee is indirectly set by the government through the required targets for collection and recycling. Hence, there should be no problem for the government to reach the same goals regardless of the system chosen. For instance, if the fee is to low, the government only needs to increase the targets, REPA will be forced to increase the fee, and the incentives to reduce the amount of packaging will be maintained. Consequently, the UCTS system will provide virtually the same incentives for output reduction as will the Swedish producer responsibility scheme.

5.2.2 Input Substitution Effect

As was noted in section 2, the packaging fees differ for different packaging materials. This means that the fees also induce material substitution between different packaging materials. In theory these fees could promote a socially optimal mix of packaging materials but in practice this is unlikely to be the case. One reason is simply that it is very difficult to assess all environmental impacts involved in monetary terms, and thus translate them into optimal fees. Still, it is worth noting that in the Swedish system plastic packaging, which is produced by fossil fuels, has a much higher fee than paper packaging, often produced by renewable resources.

One problem with the Swedish packaging fee is that its level does not depend on to what extent the producers use recycled inputs. In order to be socially optimal the packaging fee should probably be low if the production is intensive in the use of recycled inputs and high if new packaging is made of virgin materials. This relation can be seen in the Fullerton and Wu producer responsibility scheme, in which the producers do not need to pay tax for the use of recycled inputs. Consequently, the packaging fee in Sweden does not necessarily give the individual firm a direct incentive to use recycled inputs. Nevertheless, it is of course incorrect to assert that the Swedish producer responsibility as a whole does not encourage the use of recycled inputs. As we have seen, REPA/SKAB fulfills the Swedish requirements, primarily by using the packaging fee to subsidize the collection of used packaging and hence lower the cost for products that use recycled inputs.

In the Swedish producer responsibility there is no uniform payment for collectors. As was noted in section 3, SKAB/REPA negotiates with different collection entrepreneurs to organize the collection system in different geographical regions, implying that the "subsidy" varies across regions. In most cases they negotiate with the municipalities. According to Pihl (2002), the payment is generally higher in regions with low population densities and with long transport distances to the end use market. This could be rational for society if the social costs for deposits and burning were higher in these areas. However, in reality the opposite seems to be more likely. Reasonably, the social cost for landfills and burning should be lower in sparely populated areas. Consequently, a socially optimal and a cost-effective collection of a given amount of packaging do not imply lower payments to collectors in densely populated areas.

The fact that collectors in sparely populated areas get higher payments is somewhat unexpected considering that the industry has organized the collection. One reason to have an ordinance without any detailed instruction was also to let the industry create a rational and cost effective system. Still, it is not entirely true that the industry is allowed to organize the collection on its own. The ordinance prescribes that the producers should provide suitable collection systems (SFS 1997:185, 4§), and SEPA has the authority to announce instructions about what a suitable collection system is (SFS 1997:185, 13§). Although SEPA has not announced detailed instructions, the Agency does demand that the collection system should be nationwide (SEPA, 1996).

The packaging tax in the UCTS system could be differentiated in the same manner as the packaging fee in the Swedish producer responsibility scheme. Consequently, the Fullerton/Wu UCTS system can provide exactly the same incentives for input substitution between different packaging materials as the present Swedish system does. The subsidy for recycling in the UCTS system will also encourage substitution of recycled inputs for virgin materials. However, there is one rather large difference between the two systems. In the UCTS system the subsidy for recycling will be uniform, and thus all collectors – irrespective of location – would receive the same payment. This means that society will reach the recycling targets in a cost effective manner; the uniform subsidy will stimulate collectors with low marginal cost of collection to collect more than high cost collectors. This is consistent with a socially optimal policy, but only as long as there are no differences across Swedish regions in the social costs for landfills or burning.

5.2.3 Design for Recyclability

The packaging fee in the present Swedish system is not a function of recyclability, at least when viewed from the perspective of the individual firm. In order to be socially optimal the packaging fee should be low if the packaging has a high degree of recyclability and high when it has lower degree of recyclability. Recyclability has, however, several aspects. A product with a high recyclability should be easy to clean, sort and transport. Furthermore, the used packaging should be easy to use as an input in the production of new products (i.e., easy to recycle). Clearly, on an aggregate level, the packaging fee will in some sense be a function of recyclability. REPA/SFAB could fulfill the requirements from the authorities rather easy if producers design packaging that is easy to recycle and hence lower the packaging fees. However, since there are about 20,000 packaging fillers in Sweden it will not be rational for the individual firm to consider recyclability in their production decision. The industry representatives also confirm this behavior (Vretman, 2002; and Olbrich, 2002).

In Sweden the consumer has a large responsibility. As was mentioned in section 2, the consumer should clean and sort the packaging and finally transport it to assigned recycling centers. The fact that the ordinance gives the consumers this responsibility should reduce the above-mentioned problem that the packaging fee is not a function of recyclability (and does thus not promote enough of "design for environment"). There are few reasons to doubt that those consumers that fulfill their responsibility should demand packaging that is easy to clean, sort and transport and hence stimulate the production of this type of packaging. However, in reality only 37 percent of the paper and cardboard boxes were recycled in 2002 (SEPA, 2003). This means that rather many consumers throw their paper packaging with the garbage (i.e., burnable waste), and hardly consider "design for environment" issues in their consumption decisions. In addition, there are also few reasons to believe that the consumers care – or even have enough knowledge to decide – which type of packaging is easy to recover for firms. Consequently, the ordinance provides too few incentives for the packaging producer to increase the packaging recyclability.

When we compare this to the proposed UCTS system a number of issues are important. One issue that is troublesome for an efficient implementation of the UCTS system is that the value of each paper packaging waste item is likely to be low while the cost of counting the value for the collected items is relatively high. This means that the paper packaging consumer will not get any compensation for their paper packaging waste from the paper packaging collectors. Consequently this means that the consumer will not have an incentive to demand paper packaging with increased recyclability. It is also not obvious how Fullerton and Wu perceive that the waste collection and recycling should be organized in the UCTS system. However, Palmer and Walls (1999) mean that if the UCTS policy would be implemented in the USA, the existing waste collection arrangement could be used. This means that the municipality would manage to collect all waste from households without sorting from the packaging consumer, implying that there will not be any incentives for the consumer to demand recyclability when they buy products.

5.2.4 Transaction costs

In the present Swedish system the material company (SKAB) must negotiate and contract different entrepreneurs to collect the paper packaging waste in the different municipalities. These negotiations imply transaction costs. However, the contracts are generally long term (3-5 years) and the collection area for each contract is rather big (Pihl, 2002). Hence, the total costs for the negotiations are unlikely to be particularly high in relative terms.

REPA also contracts individual packaging fillers. This is however a simple process since no negotiations are needed. The packaging fillers join REPA and declare (on a quarterly basis) their use of packaging and pay the weight based standard packaging fee. REPA and SKAB also inform the packaging fillers using different methods; they print brochures, organize seminars and have salesmen that visit packaging fillers (Pihl, 2002). SKAB/REPA must also collect the packaging fee from the packaging fillers and distribute the compensation to the entrepreneurs. This is also a rather simple process where the fillers' payments are based on their own reporting, and the compensation to the entrepreneurs is based on the delivery to the material recovery companies. There is of course the risk that selected packaging fillers present false reports about their use of packaging. Still, REPA's sales people do inspect the fillers and their experience is that the system is working well and that the monitoring costs are fairly low (Ibid.).

There have been – and there still are – problems with litter on the recycling stations. Two issues tend to be particularly troublesome. *First*, the entrepreneurs sometimes empty the containers too late, and, as a result, the packaging consumers place waste outside the containers. This gives rise to rather high costs. SFAB has constructed a new container with a level indicator based on GSM technique (mobile phone) that signals to the entrepreneurs when they need to empty the containers. If the entrepreneur fails to do this within a twenty-four hour time period a "fine" is imposed. *Second*, households and companies sometime dump non-eligible waste at the recycling stations. This illegal dumping creates two different problems. It increases the sorting costs for the material companies as well as the cost of monitoring and controlling the recycling stations. SFAB has employed former police officers that observe the recycling stations, identify any illegal dumping and report the culprits to the police. SFAB has also initiated cleaning contracts with cleaning firms and sports clubs. The total cost for amounts to about 50 million SEK per year for the material companies. The five material companies (including SGAB) share this cost. (Pihl, 2002; and SFAB, 2003b)

As mentioned section 2, SFAB and SKAB must inform packaging consumers about the recycling stations. This is done partly through cooperation with the municipalities, and SFAB typically compensates the municipalities for information activities undertaken. In most cases the municipalities inform households about the packaging collection together with other waste information. SFAB also informs through the telephone catalogue (SEPA, 2002a; and SFAB, 2003a). Furthermore, SFAB and each material company use advertising campaigns in newspapers and on television to inform the packaging consumers. Pihl (2002) claim that these campaigns give results but are expensive. Finally, each material company must report the collection and recycling outcome to SEPA. This is however a rather simple process in which they basically summarize the reports from packaging fillers and the entrepreneurs (Pihl, 2002).

To run all the above-mentioned operations SKAB has only two employees (man-year). The total salary and pension costs to employees, board and accountant were about 1.7 million SEK in 2001 (SKAB, 2002). Some of the tasks are also undertaken by REPA and SFAB and they have a total of 36 employees (SFAB, 2003a). It is of course hard to know how large part of these costs should burden SKAB, but one approximation is to relate SKAB's total costs to the total payment to REPA. SKAB's total cost in 2001 was about SEK 83 million and the total payment to REPA in 2001 amounted to SEK 445 million. This implies that 18.7 percent of the administration cost for REPA and SFAB should burden SKAB (corresponding to about 6-7 employees).

SEPA is the central authority for the Swedish producer responsibility. It aims at supporting, controlling and evaluating the ordinance. Furthermore, SEPA is the government's expert authority guiding the government in their evaluations of the system. In 1998, SEPA needed two man-years to run this task (RR, 1999). Finally, transaction costs also arise at the material recovery companies. For instance, they need to control the quality on the paper packaging with random drill tests. In sum, the above indicates that the Swedish producer responsibility system incurs transaction costs at several stages of the supply chain. We do not attempt here to estimate the total transaction costs involved (and neither do we attempt to identify all types of transaction costs). Still, it is useful to briefly discuss if the proposed UCTS system would imply lower aggregate transaction costs.

The material companies are unnecessary in the UCTS system discussed by Palmer and Walls (1999), but of course some one must do the work. In addition, there will exist no need for recycling stations and accordingly littering problems on the recycling stations will disappear. Furthermore, there will not be any need to inform the packaging consumer about the system. The municipalities will also be spared from there monitoring costs in this system.

In the UCTS system the government will collect the upstream packaging tax and also administrate the upstream recycling subsidy. This could be a system with lower transaction costs than the present one because fewer negotiations may be necessary on the collection side. According to the Swedish experience, the authorities' administration costs for environmental taxes are lower than 1 percent of the "tax turn-over" (Ds, 1994). The Swedish nitrogen oxide fee is from an administrative perspective rather similar to a UCTS system; firms pay a fee according to their declared use of nitrogen oxide and they then obtain a refund based on their declared energy effectiveness. In 1992 the administration cost for this system was lower than 0.5 percent of the “tax turn-over”. This would, if the parallel holds, imply a cost to administrate the paper packaging tax/refund in the UCTS-system of less than SEK 1 million. Clearly, there will also be some administrative costs for the packaging and collection firms, but these costs would essentially be the same as in the present system. The costs for informing the industry should also be the same for both systems. The only difference is that the operational responsibility for providing this information shifts from SKAB and REPA to the central authorities. Similarly, the monitoring costs for the material recovery industry would be the same in the UCTS system compared to those in the present system, and SEPA, or some other authority, would still be needed to supervise and evaluate the new system.

In sum, the above suggests, in line with arguments put forward by Palmer and Walls (1999), that the overall transaction costs of the UCTS system may well be lower than those of the present Swedish producer responsibility scheme. However, while it is relatively easy to identify the transaction costs, in the current system, which would disappear if the UCTS system was introduced instead, it is harder to predict what new transaction costs would be incurred by the implementation of a new UCTS system.

5.3 Brief Comments on the Transformation Costs

The model outlined in section 5 essentially assumes that the transformation costs are the same regardless of the waste management strategy chosen. For instance, the same waste collection activities would have to be undertaken. In this section we comment briefly on the size of some of these costs and discuss to what extent the overall transformation costs really are the same for the two waste management systems.

The Swedish producer responsibility induces several types of transformation costs. The consumer must sort, clean, store and finally transport the packaging waste to the recycling stations. Two cost benefit analyses of the Swedish producer responsibility have been conducted during the last few years. The results differ but both studies point at potentially substantial costs for the households. Radetzki (2000) estimate the marginal cost for packaging consumers at SEK 32 500 per additional ton recycled. Ekvall and Bäckman (2001) estimate the average cost at a minimum of SEK 410 per ton and a maximum of SEK 6,950 per ton. The 410 figure is based on the assumption that leisure has no value. According to SEPA (2003), in 2002 74,882 tons of paper packaging were recycled in Sweden. This implies that the total costs for the paper packaging consumer range between SEK 31 million and SEK 2,434 million.

SKAB/SFAB is responsible for the nationwide system of recycling stations. They rent sites for the stations and apply for building permits. Furthermore, they cooperate with the entrepreneurs to construct and place suitable collection containers. They are also responsible for the cleaning of the stations. The entrepreneurs must collect the packaging from the recycling stations and transport the packaging to regional sorting plants. After they have secured the quality in the sorting plant, they deliver the packaging to the material recovery companies. This collection work involves rather high capital costs with containers, special lorries, and sorting plants. The majority of SKAB's total costs are allocated to these entrepreneurs (Pihl, 2002).

The producer responsibility system is likely to have decreased the garbage costs for the Swedish municipalities. They are still responsible for the garbage but the collection costs should have decreased when packaging waste (at least partly) disappeared from the garbage. However, the producer responsibility has also increased some cost for the municipalities. The municipalities are obliged to supervise how the producers manage the producer responsibility at the local level (RR, 1999). Some of the municipalities have also complained that they bear parts of the cost that the producers are responsible for (Ibid.). Many municipalities are not getting full compensation for costs related to information activities aimed at households, cleaning and snow clearing at the recycling stations as well as the establishment of recycling stations.

According to Palmer and Walls (1999), the UCTS system will also induce transformation costs and some of them will be the same as in the Swedish producer responsibility. However, there will be differences. In the UCTS system, Palmer and Walls argue, no sorting, cleaning and transporting activities on the part of the consumer will be necessary. They can simply throw the packaging waste in the garbage. Naturally, however, some transformation cost will increase as a consequence. In the UCTS system the municipalities will have the responsibility for all waste. This implies that their cost will increase but it is harder to estimate what will happen to total collection costs. The UCTS system builds on the idea that the municipalities collect the waste at each household, while in the present system the entrepreneurs collect the packaging waste at the recycling stations. This fact may imply that the UCTS system will impose higher collection costs. At same time there should be some economies of scale when we have only one system for collection of all household waste. Moreover, the Palmer and

Walls system will imply higher sorting costs for the waste industry. In the present system the consumer has cleaned and sorted the packaging waste so the material companies need only conduct minor sorting work. Material recovery processes typically manage to eliminate contamination but more chemicals are needed when the packaging is more contaminated. The work environment in the sorting plants will also be worse if the packaging is more contaminated.

6. Concluding Remarks

The main purpose of this study has been to analyze the incentive structure and the effectiveness of the Swedish producer responsibility, i.e., the ability of the system to promote producers to economize with paper packaging and to fulfill the related environmental goals cost effectively. A secondary purpose has been to discuss if the empirical evidence in any way suggests that an alternative supply chain management regime, i.e., the UCTS-system, could be more cost effective.

Both the Swedish producer responsibility scheme and the UCTS system fulfill two of the cost effectiveness criteria outlined above. The packaging fee in the present Swedish system and the packaging tax in the UCTS system provide similar incentives to an *output effect*. Furthermore, both systems can also give rise to two types of *input substitution effects*. The actual packaging fees in the existing Swedish system vary for different materials and induce a substitution effect between different packaging materials. Exactly the same effect can be attained in the UCTS system if the government employs different packaging taxes for different materials.

Both systems also encourage the use of secondary materials at the expense of virgin materials. This is done by subsidizing collection and recycling that in turn decrease the cost of secondary materials. There is however one important difference between the two systems. The subsidy to waste collection in the Swedish producer responsibility system differs for different collectors. In general entrepreneurs in areas that are assumed to have the highest marginal cost of collection obtain the highest refund, while in the UCTS system the subsidy is equal for all collectors irrespective of collection costs. This implies that the UCTS system would promote a cost effective collection of packaging waste. Still, it ought to be “easy” to implement such an approach also in the Swedish producer responsibility scheme. If the authorities give up the requirement on a nationwide collection it is reasonable to assume that the industry would collect the packaging in regions where they found it most profitable.

Neither of the systems tends to encourage enough of *design for recyclability*. However, here the Swedish producer responsibility seems to be somewhat more effective than the UCTS system. The present Swedish system will at least partly stimulate the demand for packaging that is easy to clean, sort and transport for the packaging consumer. The UCTS system will not give the paper packaging producer any incentive to design the packaging for recyclability. This depends on a failure in the market for collection. The value of paper packaging waste is low but the consumers' cost for paper packaging after recyclability is high in comparison so there is unlikely to be incentives for the consumer to demand paper packaging that is easy to recycle (at least as a result of the producer responsibility system as such).

The analysis of the transformation and transaction costs involved in the two waste management systems suggests that it is hard to a priori determine which system will minimize waste management costs. It will depend on, for instance, households' valuation of sorting efforts and the presence of economies of scale in the waste collection system. This implies in fact that different systems can be preferred in different parts of the country.

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II

An Econometric Analysis of Regional Differences in Household Plastic Packaging Waste Collection in Sweden*

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ABSTRACT

The Swedish producer responsibility ordinance mandates producers to collect and recycle packaging materials. This paper investigates the main determinants of collection rates of household plastic packaging waste in Swedish municipalities. This is done by employing regression analysis based on cross-section data for 252 Swedish municipalities. The results suggest that pure cost, economic-demographic, and socio-demographic factors as well as environmental preferences all help explain inter-municipality collection rates. For instance, the collection rate appears to be positively affected by increases in the unemployment rate, the share of private houses, and the presence of immigrants (unless newly arrived) in the municipality. The impacts of distance to recycling industry, urbanization rate and population density on collection outcomes turn out, though, both statistically and economically insignificant. A reasonable explanation for this is that the compensation from the material companies varies depending on region and is likely to be higher in high-cost regions. However, if true, this also suggests that the plastic packaging collection is cost ineffective. Finally, the analysis also shows that municipalities that employ weight-based waste management fees have on average a higher collection rate than municipalities in which flat and/or volume-based fees are used.

Keywords: collection rates, recycling, plastic packaging, inter-municipalities differences, producer responsibility, waste management, cost effectiveness, cost benefit analysis.

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1. Introduction

During the last two decades, environmental policies have shifted focus from the production processes to the finished products (e.g., packaging). The Swedish Environmental Protection Agency (SEPA) suggests that one reason for this is that society has been relatively successful in reducing pollution from factories and other stationary sources of emissions (SEPA, 1998). This shift in preferences is visible in policy documents. One part of finished products, the packaging, has even become the symbol of "unsustainable" consumption in the industrialized countries (Ibid.). In 1994, a producer responsibility ordinance for packaging (SFS 1994:1235)[1] was introduced in Sweden and also for some other products, (e.g., tires).

The Swedish producer responsibility burdens producers and consumers with different types of costs. The legislation implies that the producer has the physical and the economic responsibility for the packaging waste. The producers are obliged to provide suitable systems for the collection and the recycling of packaging waste, and inform the consumers about these systems. Furthermore, the producer must consult with the municipalities about these systems for collecting. The producers should also collect data on the outcomes of the collection and recycling activities and report these to SEPA. The consumers must clean and sort out packaging waste from other waste, and transport this packaging waste to the systems for collecting (SFS 1997:185, 1997).

After the introduction of the ordinance a few cost benefit analysis on the Swedish producer responsibility for packaging and newspaper have been conducted. One of them, Radetzki (2000), concluded that the recycling of packaging waste is relatively costly for the society. Radetzki suggests that if recycling reduces landfilling and burning in equal proportion, an environmental benefit valued at 405 million SEK is obtained by spending as much as 7,620 million SEK. Ekvall and Bäckman (2001) evaluate the social costs and benefits of different treatments of Swedish cardboard packaging waste. Bäckman et al. (2001) perform similar studies for glass, metal, plastic packaging, and for newspaper. These two studies present result that overall are more beneficial for recycling but recycling still comes out as socially inefficient waste management strategy for society in the case when households' time inputs are valued. Bruvoll (1998) has drawn similar conclusions about household paper and plastic waste recycling in Norway. The

[1] This legislation was revised in 1998 (see SFS 1997:185).

above studies have fueled the debate on the significance of households' recycling efforts (se also Bruvoll et al., 2002). The skeptics argue that many consider recycling to be a meaningful and voluntary activity in itself and as a consequence it is unreasonable to place a cost on it, while others – not the least many economists – stress that it is unlikely that the opportunity cost of household time spent upon recycling activities is zero.

While the above studies focus on the national outcome, it is reasonable to assume that both private and environmental costs for different waste management schemes differ across regions and counties. For example, the external costs arising from landfilling and burning are probably lower in sparsely populated areas than in urban areas, while the marginal collection costs are likely to be relatively high in the former regions. This implies that it could be socially worthwhile to have different targets for waste management treatment in different parts of Sweden (see, for instance, Berglund, 2004). The authorities do not seem to consider this in their instructions to the producers. The only instruction the producers receive in this case is that the packaging collection should be nationwide (SEPA, 1996). However, in reality we can observe rather large differences in collection rates for different municipalities. For example, the collection of household plastic packaging waste in Swedish municipalities differs from a minimum of 0 kg per resident up to a maximum of 7.8 kg (SFAB, 2004). What explains these differences in collection rates? For instance, can the observed differences be attributed to important cost elements in the respective municipalities, or do the results suggest that the cost effectiveness of the Swedish producer responsibility system could be improved? What is the role of differing municipal waste management policies in determining these outcomes in collection? These are some of the most central questions that will be discussed and analyzed in this paper.

The purpose of this paper is to identify the main determinants of household collection rates of plastic packaging waste in different municipalities. This is done using regression analysis based on cross-sectional data from 2002, which include economic, demographic, institutional and policy-related variables. The focus on plastic packaging is motivated out of several reasons. *First*, as noted above the collection of plastic packaging shows large differences between Swedish municipalities. *Second*, the plastic collection has not fulfilled the set targets of the producer responsibility ordinance. Consequently, it should be important to learn more about what determines plastic waste collection in the country. *Third*, producers compile collection data for all packaging fractions at municipality level (SFAB, 2004), but it is merely plastic packaging that only stem from households. The data for other packaging materials include collection from both households and firms. The focus on household packaging is in turn motivated for two reasons. *First*, households generally consume more plastic packaging than firms do. *Second*, there exist two completely different systems for the collection of packaging waste from households and firms. Thus, it would be difficult to model collection behavior and draw reliable conclusion if not permitted to separate between these two actors. Furthermore, it is suitable to use cross-sectional or panel data when inter-municipality differences are investigated. In our case, the choice of cross-sectional data was obvious because packaging collection data was only attainable for the year 2002.

Previous economics research on the determinants of household recycling has mainly focused on U.S. conditions (e.g., Fullerton and Kinnaman, 1996; Callan and Thomas, 1997; Jenkins et. al. 2003). Berglund and Söderholm (2003) investigate the determinants of inter-country waste paper recovery, and use a panel data set covering a total of 49 countries. This study differs from their research effort in that it focuses solely on Swedish conditions. This is an important difference because after 1994 the legal framework and the packaging collection system are more or less the same all over Sweden. Furthermore, Sweden has a rather low and homogenous population, and an even income distribution (Björklund and Freeman, 1995). There is also a study that investigates the determinants of household waste generation and recycling in Sweden, namely the one by Sterner and Bartelings (1999). However, their research differs from this study in several respects. *First*, they perform case studies in four municipalities while this paper has a nationwide scope. *Second*, the Sterner/Barteling study was done in the early stages of the producer responsibility (1994) while this study is based on 2002 data. Compared to all previous research efforts, this analysis also includes new independent variables in the data set (e.g., immigrant rate and environmental preference variables). Finally, even though the paper does not focus on the social benefits and costs of recycling efforts it ought to represent an appropriate starting point for understanding – and later on assessing – the regional differences in recycling activities.

Increased knowledge about the factors that drive household collection rates of packaging waste in different regions is important for both policy makers and producers. For *policy makers*, it is important when evaluating environmental policy and designing new legislation. For example, when evaluating if it is socially worthwhile to collect and recycle packaging waste it is important to address regional features for households and producers in the cost benefit analysis. Moreover, the municipalities can make use of the results when designing new waste fee systems for allocating more household packaging waste to the producers and thus lower municipality waste management costs. For producers this knowledge could be important when attempting to increase the effectiveness in collection in different respects; the information could, for instance, help producers design a cost effective collection system and also help them to actually fulfill the producer responsibility targets.

The paper proceeds as follow. Section 2 provides a background to packaging recycling in Sweden with special focus on the collection of household plastic packaging waste. Some important determinants of household plastic waste collection are discussed in section 3, and this discussion is used to outline an econometric model in section 4, which also discusses data and model estimation issues. The empirical results are presented and discussed in section 5. Finally, section 6 provides some concluding remarks and outlines a number of important implications.

2. Packaging Waste Collection in Sweden

This section begins with a brief overview of the Swedish producer responsibility ordinance, especially for plastic packaging waste. Furthermore, the section also explains how the producers have organized the producer responsibility in practice. Finally, the outcome of the plastic waste collection efforts is presented.

2.1 The Producer Responsibility Ordinance

As was mentioned above, packaging recycling in Sweden is regulated by the producer responsibility ordinance. This regulation implies that *producers* should collect, remove and recover the packaging waste from consumers. However, they are not required to take care of all packaging. Instead the ordinance also regulates to what degree used packaging should be collected and for what purpose the collected packaging waste should be used (recycled and/or energy recovery). Table 1 summarizes these policy targets.

Table 1: Targets in the Swedish Producer Responsibility for Packaging (Percent of Consumed Packaging Weight)

Type of packaging	Recycling and energy recovering after 30 June, 2001. Energy recovery rates in brackets.
Aluminum, excluding drink packaging	70%
Aluminum, drink packaging	90 %
Board, paper and cardboard	40 % (30 %)
Corrugated cardboard	65 %
Plastic, excluding PET-bottles	30 % (40 %)
Plastic, PET-bottles	90 %
Steel plate	70 %
Glass, excluding reusable glass	70 %

Sources: SFS 1997:185 (1997) and RR (1999).

In this paper, the definitions from the EC directive for packaging and packaging waste have been used. Recycling imply that waste should be processed and be used as input in new plastic production. Energy recovering means that it is allowed to burn the waste, if recovering the energy content. For example, the packaging producers are required to collect at least 70 percent of all plastic packaging. At least 30 percent of all plastic packaging should be recycled, hence used as input in new plastic products. The rest of the collected packaging, 40 percent of all plastic packaging is not allowed to end up on landfills but energy recovering is seen as a suitable treatment.

The Swedish producer responsibility is an ordinance with few detailed instructions. However, the ordinance obliges producers to provide suitable systems for collecting packaging waste and also to inform the consumers about these systems (SFS 1997:185). *SEPA* – that has the authority to outline instructions for the producers – has required that the collection should be nationwide (SEPA, 1996). The *municipalities* are responsible to supervise the collection within their own borders (RR, 1999).

The *consumer* has the responsibility to clean and sort the packaging waste and transport it to the recycling stations (SFS 1997:185). Although producers have the economic responsibility for the packaging waste, the consumers do not get any economic compensation from the producer for this effort. However, every household does not fulfill their obligation; instead they leave the packaging waste with the household waste. Consequently, the municipalities collect this misplaced packaging waste and leave it to energy recovery or landfills.

2.2 The Practical Implementation of the Producer Responsibility

In order to fulfill the producer responsibility for packaging, the retailers and the producers have founded four joint material companies[2] that administrate the collection and recycling of packaging. Together they all form the service organizations Svenska Förpackningsinsamlingen AB (SFAB) and Reparegistret AB (REPA). SFAB's task is to coordinate the material companies. For example, they establish and operate recycling stations[3] and they also inform packaging consumers about the collection and recycling system. Through *REPA* the material companies can offer a nation wide coverage of packaging waste collection. Individual *producers* can fulfill their producer responsibility if they join REPA. If they do so, they must pay a packaging fee to REPA[4] based on the weight of their packaging (see Table 2), primarily because the collection and recycling of packaging cannot carry its own costs. These packaging fees are charged to only one part of the supply chain, this to avoid that more than one company pay for the same packaging material. As a rule, the fees are paid by the packaging filler, packer or re-packer for products made in Sweden, and by the importer for foreign products. According to SEPA (2002), 10,000 firms have joined REPA and together they represent about 90 percent of all packaging materials used in Sweden.

Table 2: Packaging Fee for Different Packaging Materials (2004)

Packaging material	Packaging fees (SEK/kg)
Plastic	2.70
Metal (steel plate and aluminum)	1.20
Paper/Cardboard	0.55
Corrugated cardboard	0.23
Steel barrel (30-250 liter)	0.06

Source: REPA (2004)

Plastkretsen AB (*PAB*) administrates and organizes the collection and recycling of all plastic packaging waste from households and producers, besides returnable PET-bottles.[5] The collection and recycling from households and producers differ in some aspects but since the recycling from producers is beyond the scope of this study, only the household collection and recycling will be

[2] These include Svensk Kartongåtervinning AB (SKAB) (paper and cardboard packaging), Plastkretsen AB (PAB) (plastic packaging), Svenska Metalkretsen AB (SMAB) (metal packaging), and RWA Returwell AB (RWAB) (corrugated cardboard packaging).

[3] There are approximately 7,000 recycling stations in Sweden (SFAB, 2000).

[4] These packaging fees are redistributed to the four material companies by REPA.

[5] The recycling of returnable PET-bottles is organized by Svenska Returpack.

described. This is done schematically in Figure 1. The arrows marked HPPW show the flow of rigid plastic packaging waste and the "payment arrows" show which actors are compensated for their work.

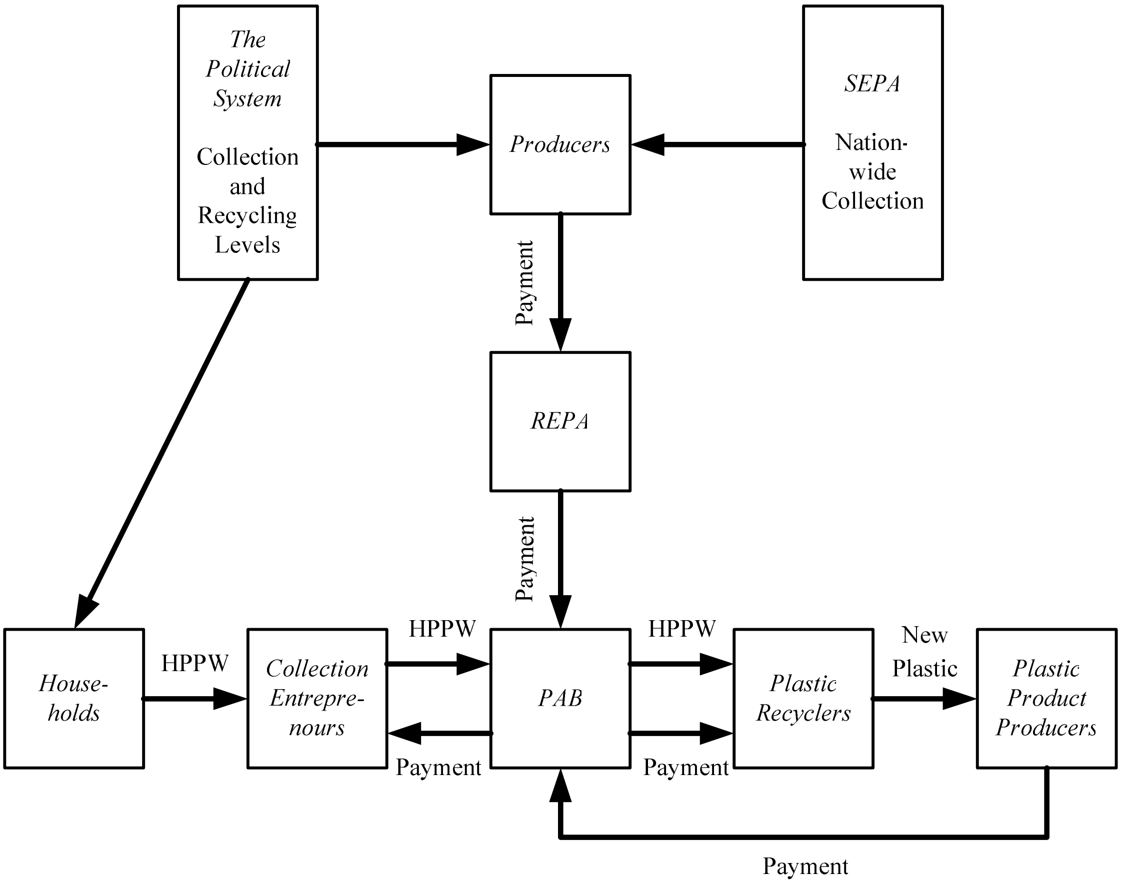

Figure 1: The Structure of Household Plastic Packaging Waste Collection in Sweden

Because of production constraints, PAB is only interested in household rigid plastic packaging (e.g., bottles and containers). Hence, only rigid plastic packaging should be sorted, cleaned and transported to recycling stations while flexible plastic packaging waste (e.g. bags and film) should be placed in the household burnable waste.[6] In order to facilitate the collection of plastic

[6] Flexible plastic packaging waste from households is typically energy recovered, and PAB pay the municipalities for this service (SEPA, 2004b).

packaging waste from recycling stations, PAB has engaged different *collection entrepreneurs*.[7] These entrepreneurs put out and empty the containers at the recycling stations and transport the plastic waste to *plastic recyclers*. These recyclers[8] are also engaged by PAB. They will sort, clean and process the plastic waste into new plastic material. The remaining plastic waste that is not processed to new plastic will be energy recovered. PAB owns the plastic up to this stage and compensates both collection entrepreneurs and plastic recyclers. After these stages, PAB sells the new plastic material to *plastic product producers*.[9] PAB's operations are to 90 percent financed by the packaging fees and to 10 percent by the selling of new plastic material (PAB, 2004; SEPA, 2004b; and SOU 2001)

2.3 The Outcome of the Plastic Packaging Collection and Recycling

According to SEPA (2004a), the annual consumption of plastic packaging has been approximately 150,000 ton in Sweden during 1999-2002 and in 2003 151,453 ton was consumed. As seen in Table 1, the producer responsibility ordinance requires that producers should recycle at least 30 percent and energy recover at least 40 percent of the plastic packaging consumption. Table 3 shows how well PAB have fulfilled these targets.

Table 3: Recycling and Energy Recovery of Plastic Packaging excluding PET-bottles (Percentage of Total Consumption for Selected Years)

Treatment	1996	1997	1998	1999	2000	2001	2002	2003
Recycling from PAB	11	13	20	16	15	13	16	18
Energy Recovery from PAB	2	8	16	16	17	15	17	18
Energy Recovery from household waste*	?	?	?	(27)	(34)	(32)	(31)	32

* Data collected by the municipalities. The data for 1999-2002 are not included in the official packaging collection result.

Sources: SEPA (2004a), SEPA (2003), SEPA (2002), and SEPA (2001b).

[7] These entrepreneurs can be divided into three categories (SFAB, 2004). First, there exist three nation wide entrepreneurs, IL Recycling, Stena and Sita, that each collects plastic packaging waste in at least 50 municipalities. Second there are a number of regional entrepreneurs that all serve at least three municipalities each. Third, there are also 22 entrepreneurs that only act in one municipality.

[8] PAB has engaged three companies to sort and recycle rigid plastic waste. These are Plaståtervinning i Arvika, Plaståtervinning i Strömsbruk and Swerec AB in Lanna/Bredaryd (PAB, 2004).

[9] Examples of new plastic products based on recycled plastic packaging include bags, furniture and building material.

Table 3 indicates that PAB collects 36 percent of total plastic packaging material consumed. However, merely 18 percent of total plastic packaging consumption is recycled. Hence, PAB experiences problems in fulfilling the producer responsibility targets. However, the recycling level shows a slowly increasing trend. PAB almost fulfills the goal for total recovery including recycling in 2003 when burning of household plastic waste collected by municipalities is included. At least two reasons for the observed low levels of recycling are possible. First, rather low levels are collected from households and producers. Second, the fact that only 18 percent are recycled in spite of the fact that 36 percent is collected makes it reasonable to suspect that it is economically difficult to recycle plastic waste. PAB also reports to SEPA that one reason for burning a large part of the collected plastic packaging waste relates quality problem with the plastic waste (SEPA, 2002). For example, non-marked plastic packaging and especially plastic laminate are not only uneconomical to recycle but also technically impossible. Other experienced problems relate to contaminated and badly sorted plastic waste.

SEPA (2002) reports that Swedish households consumed about 95,000 tons of plastic packaging in 2001 and that the producers consumed about 50,000 ton. In 2001, 42 percent of all collected packaging waste was collected from households and 57 percent was collected from producers in 2001. Consequently, only 19 percent of the household plastic packaging waste was collected by PAB in 2001.

Figure 2 indicates how much household plastic packaging waste per resident that was collect in different municipalities in 2002. The average municipality collects about 1.33 kg of plastic packaging waste per resident. However, it is obvious that the collection rates differ significantly across municipalities. Nine municipalities (e.g., Bengtfors) do not collect any household plastic packaging waste at all and Älvdalen is the municipality that collects the most, 7.8 kg per resident. Figure 2 also shows, looking left to right, that the collection rate shows a smooth but exponential increase. This is essentially the expected outcome for collection activities in different regions according to economic theory, assuming continuous increasing marginal costs for collection (Berglund, 2004).

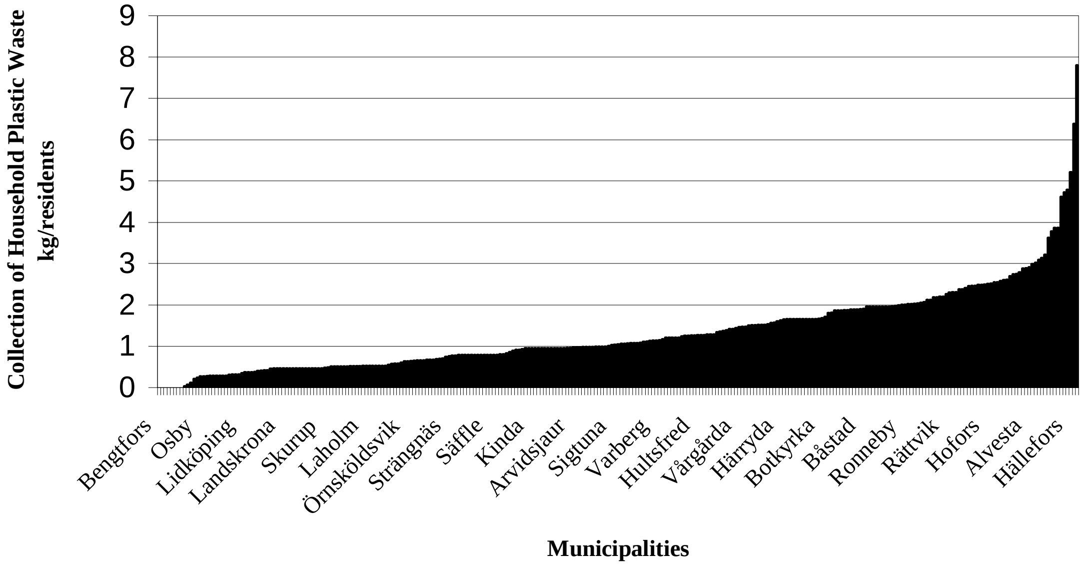

Figure 2: Collection of Household Plastic Waste in 289 Swedish Municipalities for 2002 (kg/resident)

Source: SFAB (2004)

The discussion in this section shows that PAB must increase their recycling activities to fulfill the targets in the producer responsibility. One solution could be to develop better sorting techniques and better recycling techniques and thereby be able to recycle more of already collected plastic packaging waste. Another – and perhaps more effective method at least in short term – could be to increase the collection of plastic packaging waste. The analysis also shows that at the same time as households are the dominant plastic packaging consumers, the collection levels from households are rather low. Hence, in the efforts to increase collection rates, it is worth noting that there is a large reserve of non-collected plastic packaging waste in the households. Furthermore, there also exist great differences in household plastic packaging waste collection when comparing across different municipalities. The next section discusses a number of possible explanations to these differences.

3. Determinants of Packaging Collection: Lessons from the Literature

This section begins with a presentation of some relevant findings from the empirical research literature. Furthermore, appropriate behavioral assumptions concerning the agents that influence packaging collection in Sweden are discussed. Finally, the section summarize the main types of determinants that are hypothesized to influence the rate of plastic packaging collection in Swedish municipalities.

3.1 Earlier Empirical Findings

As was noted above a number of earlier empirical studies investigate household recycling determinants. We focus here on the main findings from these studies. Sterner and Bartelings (1999) analyze observed household waste generation together with survey data on households attitudes in four Swedish municipalities. They find that a proper infrastructure for recycling as well as strong economic incentives are important conditions for attaining reductions in household waste generation and hence an increase in recycling rates. Their results also suggest that weight-based wastes fees are more effective in reducing household waste than are volume-based waste fees.

Several of the U.S. studies analyze how different waste management schemes and waste fees influence household recycling. Fullerton and Kinnaman (1996) perform a case study in Charlottesville, U.S., and find that volume-based waste pricing decreased garbage volumes but not the garbage weight. Callan and Thomas (1997) investigate how state and local polices influence household recycling in Massachusetts. Their results indicate that unit-based waste pricing and curbside recycling increase recycling activities, especially if these two policies are used in combination. They also find that demographic factors matter for recycling behavior. For instance, income, population and education were positively correlated with recycling efforts. Jenkins et al. (2003) analyze how solid waste programs affect household recycling behavior for five different materials in 20 metropolitan areas across the USA. They find that policies that improve the convenience of recycling – especially curbside recycling – increase household recycling. Their results also indicate that the length of recycling programs has a positive effect on recycling. Making recycling mandatory and the use of volume-based waste pricing did not appear to have a significant effect on the rate of recycling. The authors also suggest that the impacts of different waste management policies differ across materials.

Berglund and Söderholm (2003) investigate the main determinants of inter-country waste paper recovery and utilization. They find, for instance, that economic factors such as waste paper price and income (GDP) together with demographic factors (e.g., urbanization rate and population density) are important determinants of inter-country differences in waste paper recovery.

In a report from the Swedish Consumer Agency (2001) Swedish household recycling behavior is analyzed using eight focus group deliberations. This study concludes that economic factors and convenience factors are the most important for promoting household recycling efforts in the country. For example, recycling activities should in order to attract households' attention benefit from economic incentives, must not consume too much time and should also be simple and hygienic. It is also found that information and habits are relatively important, and that the wish for a better environment and altruistic motives tend to be subordinated to individual motives.

Hornik and Cherian (1995) employ a meta-analysis of 67 empirical studies on recycling behavior in the USA. They conclude that consumer knowledge and commitment to recycling best predict the propensity to recycle. Their results also suggest that economic incentives and social influence are the next best predictors. Schultz et al. (1995) review psychologists' empirical studies of US recycling behavior. Some of their findings are that environmental concern is important only when recycling requires a high degree of effort, and that high incomes and good knowledge about the recycling programs are important for participation.

3.2 Assumptions about the Behavior of the Agents Involved

The collection levels of household plastic packaging waste in Sweden are mainly explained by the behavior of four different agents: the material companies, collection entrepreneurs, households and the municipality authorities. In order to sort out the potentially most important determinants that affect plastic packaging collection in Sweden it is useful to discuss what the main objectives and motives of these agents are.

The *material companies* (e.g., PAB) are owned by larger firms in the packaging industry. All these material companies run without profit interests and do not distribute any returns to its owners so it is thus unreasonable to assume that they are profit maximizers. The only explicit goal that is revealed in their annual statements is that they should fulfill the requirements of the producer responsibility ordinance (e.g., SKAB, 2002). However, since the material companies are owned by (essentially) profit maximizing firms it is reasonable to assume that their goal is to minimize the cost of fulfilling the collection and recycling levels. One reason for why the government chose to have an ordinance with few instructions was also to provide room for the producers to create rational and cost effective collection systems (SEPA, 1996).

There are essentially two types of *collection entrepreneurs* in Sweden, privately owned and municipality owned (SEPA 2004b). In this study it is assumed that all entrepreneurs are profit-maximizing firms. Furthermore, though, it is also assumed that they have very limited opportunities to influence household plastic packaging collection. The rationale for the latter assumption is that the material companies administrate and organize the collection and they also establish recycling stations. Furthermore, different material companies often contract different entrepreneurs for the same recycling stations. These reasons imply that it is difficult for individual entrepreneurs to have an impact on the design and the total number of recycling stations.

In line with standard economic theory this study assumes that the *households* maximize utility. This is the case when the marginal utility for sorting packaging equal the marginal cost for the sorting. There exists no economic compensation for the households for their collection, and the possibility for the authorities to monitor household recycling behavior, and, if needed, impose sanctions is limited. However, there could still be several reasons for why households choose to help fulfill the obligations of the ordinance. *First*, their motives for recycling can be due to their self-image as a responsible person, i.e., they get satisfaction when acting according to a social norm.[10] *Second*, since recycling often is seen as beneficial for the environment it is reasonable to assume that households that value environmental goods get utility when they contribute to recycling. *Third*, some households also have an economic incentive to leave their packaging at the recycling stations. Some municipalities have implemented different household waste volume-based pricing programs,[11] at least for private house owners (SOU 2001:120). In addition, 20 municipalities, 7 percent, has introduced weight-based fees for the garbage, in particular for private house owners (SEPA, 2001a). This implies that many private house owners could get lower garbage fees if they leave their packaging at the recycling stations. On the other hand, the households' costs (in terms of time and effort) for sorting activities could sometimes be lower for people living in apartments. The reason for this is that several apartment houses have packaging

[10] This social norm can stem from legislation and thus be exogenously given (e.g., Bruvoll and Nyborg, 2002), or be determined by moral motives and thus be self-imposed and endogenous (Brekke et. al., 2002). Berglund (2003) find evidence for the presence of self-image in the case of household waste sorting in the Swedish municipality of Sweden.

[11] The volume-based fees include: (a) the opportunity to choose longer garbage collection intervals and hence pay less; (b) the opportunity to share garbage container and the garbage fee with neighbours; and (c) the opportunity to pay for the size of the garbage container (Villaägarnas riksförbund, 2004).

collection within walls of the property. According to SEPA (2003), in 2002 about 25 percent of all apartment houses had packaging waste collection within the property.

The fourth agent, the *municipalities*, is in this case assumed to partly minimize their cost for waste disposal and partly to fulfill environmental objectives that more or less prioritized on the environmental preferences. There is at least one economic reason for the municipalities to "support" the packaging collection. It reduces the need for household waste (garbage) collection and waste disposal. Consequently, the cost for these activities will decrease if they support packaging collection. In 2000, a tax was introduced on waste disposal and a prohibition to deposit burnable waste was introduced in 2002 (e.g., SEPA, 2001a; and SOU, 2001). The waste disposal tax and the prohibition to disposal burnable waste have of course strengthened the municipalities' motive to support packaging collection.

3.3 Possible Determinants of Plastic Packaging Collection in Sweden

As earlier research indicates it seem as if economic factors are important for recycling behavior. However, socio-demographic factors matter as well, so in the empirical investigation a number of different types of explanatory variables will be tested.

Households and entrepreneurs supply the household plastic packaging waste to PAB. Economic theory suggests that supply should increase if the *price* of plastic packaging waste increases. However, since the households do not get any payment for their work, their decision should not be directly influenced by a price increase for the plastic packaging waste. Nevertheless, when entrepreneurs get higher compensation for their collection, it ought to be more profitable to provide better service to households (e.g., more recycling infrastructure within the property), and consequently the households would probably supply more plastic packaging waste. Both the price of plastic packaging and the compensation to entrepreneurs are not decided by the value for the recyclers; instead it is mainly the result of political decisions. *First*, the targets for recycling imply a higher price to entrepreneurs than the market value for the plastic packaging waste. *Second*, the goal that the collection should be nationwide give reasons to suspect that entrepreneurs that work in high cost municipalities get higher compensation for their plastic packaging waste than entrepreneurs that work in low cost municipalities. Pihl (2002) confirms that entrepreneurs operating far away from recycling industries and in sparely populated areas get higher compensation for their collection of paper and cardboard waste (compared to those operating in densely populated areas). This is clearly a violation of the cost effectiveness principle. However, this plastic packaging waste compensation is decided through secret negations between PAB and the respective entrepreneurs. Thus, the compensation for plastic waste collection in different municipalities is probably important for explaining differences in collection rates, but since these plastic packaging waste compensation fees are not public we cannot test this hypothesis empirically. However, if it is the case that collection in high-cost municipalities is compensated through higher fees, it is also reasonable to presume that important cost factors will only have a minor impact on collection rates.

The *distance* between the municipalities and the recovery industries affects the transportation costs for the material companies. The longer this distance is the lower should the incentives be for the material companies to collect packaging. However, as noted above, this cost disadvantage can be neutralized by higher compensation levels for the collected household plastic packaging waste.

As was mentioned in section 2, many municipalities have implemented different *volume-based pricing* programs and some have implemented *weight-based fees* for household waste collection. This means that for some households there exist economic incentives to leave their packaging at the recycling stations. Consequently, the presence of both volume- and weight-based pricing for the household waste can be assumed to increase plastic packaging waste collection compared to the situation in municipalities that use fixed fees.

Demographic variables could also influence the behavior of both the households and the material companies. High *urbanization rates* and/or are *densely populated* municipalities imply shorter distances for households to recycling stations and for material companies when collecting used packaging. Consequently, high urbanization rates and/or more densely populated areas should lower the transport cost for both households and material companies. Still, high population urbanization rates and/or densely populated areas could also drive up land prices and hence the material companies' costs for establishing recycling stations. This implies that there ought to be one positive transport cost effect and one negative land cost effect associated with high urbanization rates population densities. Which of them dominates in practice remains thus an empirical question.

One hypothesis is that the relationship between population density/urbanization rate on the hand and collection costs on the other is non-linear; the transport cost effect dominates when the municipalities are relatively sparsely populated, while in very densely populated cities the land cost effect dominates. There could be at least four explanations for this relationship. *First*, in small- and medium-sized cities it is reasonable to assume that the municipalities' posses enough land that could be rented to the material companies at relatively favorable charges, but that this is harder in very dense cities. *Second*, small- and medium-sized cities in Sweden generally have relative small city centers. Hence, here it is possible for the material companies to establish their recycling centers just outside the city center but still avoiding long transport distances from households to recycling centers. In big cities, the establishment of recycling stations outside the city center implies much longer transport distances for households. *Third*, the possession of cars is typically less frequent in large cities and because many use their car for leaving household packaging waste at recycling stations, this could reduce collection rates. *Fourth*, congested cities often have problems with the traffic situation. The above suggests that overall there should be a positive urbanization rate and a positive population density effect but there should also be a negative *big city* effect. This assumption about a negative big city effect is also supported by the current debate in Stockholm, the capital of Sweden, about who should pay for collection within the borders of the property (e.g., Fastighetstidningen, 2004).[12] However, as suggested earlier, the impact of these regional cost differences in collection can be offset by PAB's pricing policy.

There exist also other demographic factors that probably affect households' packaging collection behavior. The *type of housing* should probably matter. As mentioned above, private house owners often have economic incentives for packaging collection. It is also reasonable to expect that private house owners have more space for storing used packaging. These two factors imply that collection should, ceteris paribus, be higher in areas with relatively many private house owners. However, the fact that some apartment houses have packaging collection within the borders of the property may also offset this impact. The rate of *unemployment* could also matter. The opportunity cost of the time spent on waste packaging collection could be lower for the unemployed, and one can therefore expect that they will collect more waste packaging than

[12] It can also be noted that higher land prices should increase landfill costs. However, this should not influence the collection costs for the material companies because all their packaging materials are burnt or recovered.

would employed persons. The Swedish Consumer Agency (2001) also concludes that time is one of the most important determinants for recycling behavior.

The final economic-demographic variable considered in this paper is *income*. Kriström and Riera (1996) and Hökby and Söderqvist (2003) find that the demand for environmental improvements is a “necessary” good; hence the income elasticity for environmental improvements should be positive but less than one. This implies that households with higher income will generally be willing to allocate more resources for environmental improvements and hence collect more packaging than low-income households. However, the opportunity cost for the households must also be considered. The collection of packaging is a time consuming activity for the household[13] so the opportunity cost for packaging collection should also increase with income. Consequently, this means that impact of income on collection behavior will be determined by a positive income effect and a negative opportunity cost effect. Which of these two effects that dominates is an empirical question. However, as noted above, many of the earlier empirical studies found that recycling in the USA was positively correlated with income.

Socio-demographic variables that could influence household packaging collection include *education*, *age*, *gender* and *rate of immigrants*. Economic theory does not provide any predictions about how these variables could affect the recycling levels. However, applied economic researchers have shown that education is often positively related to the willingness to pay for different environmental improvements (e.g., Carson, 1995). Empirical researchers in US have also found evidence for a positive relationship between education and household recycling (e.g., Schultz et al., 1995; Callan and Thomas, 1997). An explanation for this could be that higher education implies increased awareness of the importance of recycling for the environment. Consequently, higher education should imply higher packaging collection rates if the household believes in the notion that packaging collection is important for attaining environmental improvements. Schultz et al. (1995) also report that the relationship between age and U.S. household recycling appears to be ambiguous. Some studies find no relationship, some find a positive relationship and some found a negative relationship. Finally, Schultz et al. (1995) find no

[13] Several studies that investigate how much time a Swedish household need for sort, clean and transport their packaging waste each week. For example; Berglund (2003) found that households in Piteå need 49 minutes and Carlénius (2004) found that pensioner households in Luleå need 53.7 minutes. In addition, The Swedish Consumer Agency (1997) found that household in four municipalities need between 15 to 25 minutes to clean the packaging waste.

relationship between gender and household recycling. Immigrants, especially newly arrived immigrants from outside the Nordic countries, are not used with Swedish laws, regulations and may have difficulties in understanding the language; this makes it reasonable to believe that their participation in packaging collection program are generally lower than for people that have lived in Sweden for a long time. The empirical literature lacks tests of this hypothesis.

Collection levels could also be influenced by the municipalities' and the households' *environmental preferences*. The municipalities could affect the collection levels in different ways, and the more emphasis the local government puts on environmental issues the more likely it is that they will attempt to facilitate packaging collection. And there exist ways through which this can be achieve. The material companies often inform packaging consumers through the municipalities' waste information. Better waste information from the municipality should naturally increase the packaging collection levels. The municipalities also rent sites for the recycling stations and provide building permits. It is also reasonable to believe that households that value the environment above average should collect more packaging. As we have seen, Schultz et al. (1995) also found that environmental concern is an important determinant of US household recycling behavior if recycling requires a high degree of effort.

Figure 3 summarizes main factors that are hypothesized to influence plastic packaging collection in Swedish municipalities. These variables form the basis of the empirical investigation in the remainder of this paper. The expected impacts of the variables (+/-) on collection rates are put in brackets.

4. An Econometric Model of Swedish Plastic Packaging Collection

This section begins with a presentation of the empirical regression model and the variable definitions that are used in the estimations. After that data and model estimation issues are discussed.

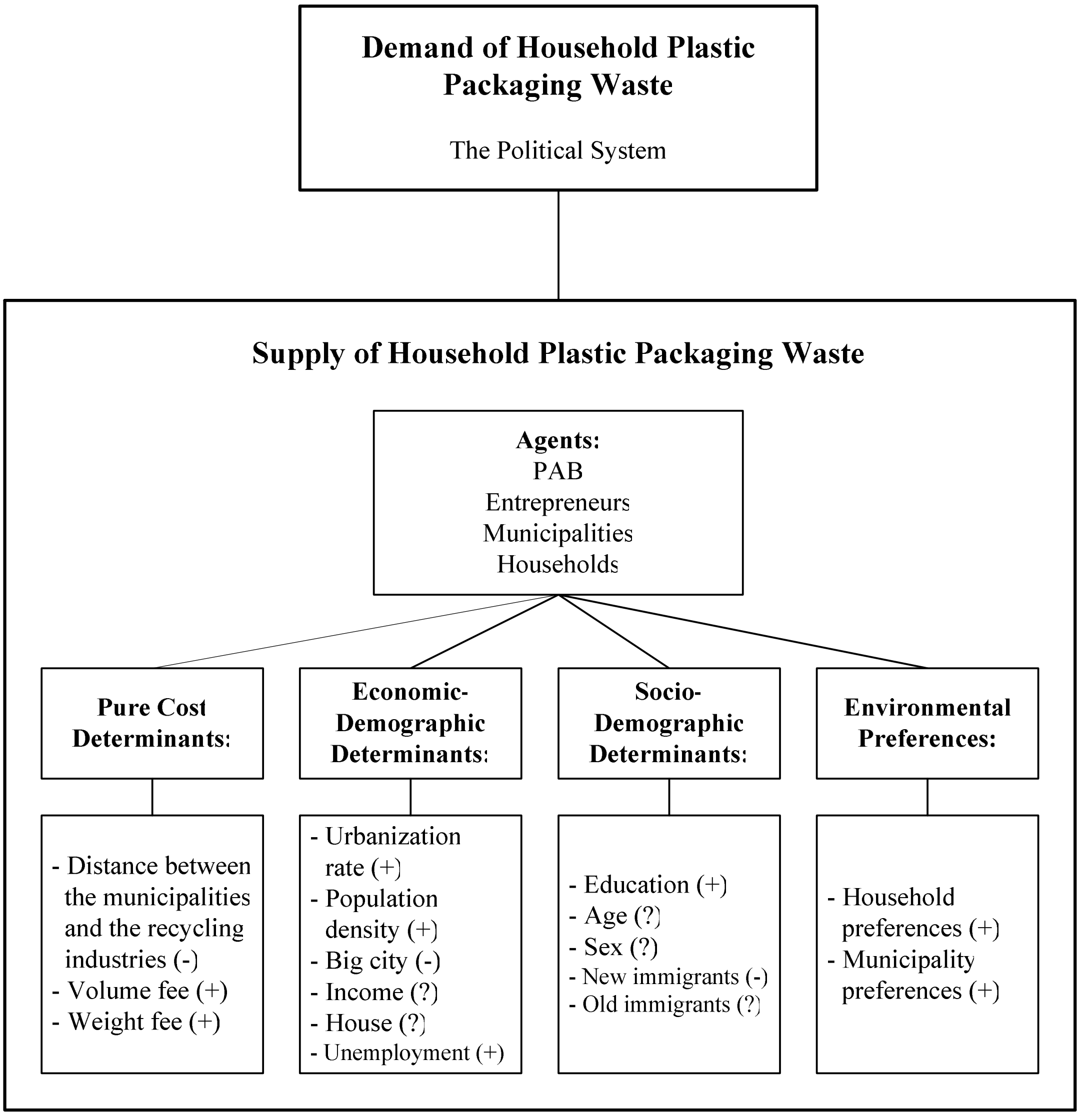

Figure 3: Determinants for Household Plastic Packaging Waste Collection

4.1 Empirical Specification

The empirical model outlined in this section builds on the discussion in section 3 and it attempts to characterize inter-municipality differences in household plastic packaging waste in Sweden. There are no theoretical reasons to suspect some specific functional form for the regression model, and in this paper we therefore assume a simple linear relationship (other specifications of the model were, however, also tested. See section 4.2). The model to be estimated can be written as:

$$
\begin{aligned}
CL_i = {} & \alpha_0 + \alpha_1 INC_i + \alpha_2 DIST_i + \alpha_3 URB_i + \alpha_4 POP_i + \alpha_5 BIGCITY_i \\
& \alpha_6 HOUSE_i + \alpha_7 UNEMPL_i + \alpha_8 EDU_i + \alpha_9 WOMEN_i + \alpha_{10} AGE_i \\
& + \alpha_{11} ENVM_i + \alpha_{12} ENVH_i + \alpha_{13} NEWIM_i + \alpha_{14} IM_i + \alpha_{15} FEEVOL_i \\
& + \alpha_{16} FEEWE_i + \sum_{j=1}^{17} \beta_j E_j
\end{aligned}
\tag{1}
$$

$i=1,\ldots\ldots,252 \qquad j=1,\ldots\ldots,17$

where CL_i is the plastic packaging collection level in kg per inhabitant in municipality *i*. The independent variables are presented and defined in Table 4, which also presents the data sources used for each variable.

4.2 Data and Model Estimation Issues

In this paper cross section data for 252 Swedish municipalities for the year 2002 are used. In 2002 there were 290 municipalities in Sweden, but due to data limitations 38 municipalities had to be excluded from the sample. However, this still means that almost 87 percent of all Swedish municipalities are included. The plastic packaging waste is sometimes collected from more than one municipality. In these cases, the total collection from these areas is divided with the total population in these areas (SFAB, 2004).

According to Table 4, it is obvious that some independent variables are not available for the year 2002. In one case this was a deliberate strategy (see below), but in some cases the choice depends on the fact that the dependent variable for 2002 was not attainable. The selection of the year 1999 as the variable for environmental preferences in the municipalities (*ENVM*) was intentional. The hypothesis is that green politicians could influence the behaviour of the municipality but for doing so they need time. Average income (*INC*), urbanization rate (*URB*), education level (*EDU*), immigrants (*IM*) are all variables that were not found for 2002 but instead for 2000, 2001 or 2003. These variables are, however, all relatively stable from one year to another. Consequently, the validity of this study should only have been marginally reduced as a result of these data limitations. The variable on the share of private houses (*HOUSE*) is however, based on a seven-year-old information. Still, since housing construction activities in Sweden have been unusually low after 1992 this also ought not to imply major problems for the investigation.

Table 4: Definitions and Sources of the Variables

Variable	Description	Sources
CL	The amount of plastic packaging collected per resident in 2002 (kg).	SFAB (2004)
INC	Average income for people between 20 and 64 years in 2000 (SEK).	SCB (2003)
DIST	Distance between each municipality and the nearest recycling industry (km).	Vägverket (1999)
URB	Urbanization rate, i.e., the share of the population living in densely populate areas in 2001. A densely populated area is defined as a group of buildings not more than 200 meters apart from each other and having at least 200 inhabitants.	SCB (2002a)
POP	The population density, i.e., the total population divided by the municipality's total land area in 2002.	SCB (2003)
BIG CITY	Dummy (1) for municipalities with 800 residents per km^2 or more in 2002, and 0 if less than 800.	SCB (2003)
HOUSE	The share of private houses in each municipality 1995 (%).	KFAKTA (2003)
UNEMPL	Open unemployment for people between 16 and 64 years in 2002, annual average (%).	AMS (2004)
EDU	People with at least 3 years at university divided by total population (%) in 2003.	KFAKTA (2003)
WOMEN	Women as share of total population (%) in 2002.	SCB (2003)
AGE	Average age (year) in 2002.	SCB (2004)
ENVM	Dummy for environmental preferences in the municipalities' governmenta, 1 if green party was represented in the municipality government (in 1999) and 0 if not.	KFAKTA (2003)
ENVH	Environmental preferences in households, measured by the share of votes on the Green party in the 2002 parliamentary election (%).	SCB (2002b)
NEWIM	New immigrants, foreign citizens with 0-4 years in Sweden as a share of total population in 2002 (%).	SCB (2003)
IM	Immigrants, foreign born outside the Nordic countries as a share of total population (%) in 2001.	KFAKTA (2003)
FEEVOL	Dummy for volume based waste pricing, 1 if yes and 0 if no.	Villaägarnas riksförbund (2004)
FEEWE	Dummy for weight based waste pricing, 1 if yes and 0 if no.	Villaägarnas riksförbund (2004)
E	Dummy for different packaging collection entrepreneurs.	SFAB (2004)

Another problem is that the information on waste fees is based on the situations in 2003 and 2004. Data for 2002 were not attainable in secondary sources and the alternative would have been to use information from 1999. According to Villaägarnas riksförbund (2004) approximately the same amount of municipalities applied volume-based household waste fees in 1999 compared to 2004. In 1999 there were 20 municipalities that used weight-based household waste fees and in 2004 this amount had increased to 22 (Villaägarnas riksförbund, 2004; and SEPA, 2001a). This implies that also these two variables seem to have been stable over time and our choice of year should not imply any major problems for the econometric estimation. It should also be noted that the urbanisation rate and the population variables are not highly correlated (the correlation rate equals 0.4). This implies that it makes sense to include both these economic-demographic variables in the regression analysis.

As described in Table 4, dummy variables are used in four cases; density (*BIGCITY*), environmental preferences in the municipalities (*ENVM*), waste fees (*FEEVOL* and *FEEWE*) and entrepreneurs (*E*). To avoid the dummy variable trap (see, for instance, Gujarati, 2003) all these variables has a reference category. For the *BIGCITY* variable all municipalities with less than 800 residents per km^2 are used as the reference category. In the case of *ENVM*, all municipalities with no representative from the green party in the municipality government are used as reference category. As reference category for waste fees we use municipalities that use fixed prices (flat rates). Finally, for the *E* variable entrepreneurs that only collect household packaging waste in one municipality is used as reference category.

All factors that are hypothesized to influence household plastic collection in this model are likely to be exogenously given. This implies that it is possible to estimate the model with ordinary least square (OLS) techniques. Different specifications of the model have been tested. Following the hypothesis of a non-linear relationship between population density and collection rates, it was tested if a quadratic form of the population density variable could explain inter-municipality differences better. Different variables were also used to measure environmental preferences in the municipality. For example, an environmental municipality ranking was used (see Miljöeko, 2001). Another test used the votes on the Green party in the municipality election (2002). However, none of these specifications added any new information.

5. Empirical Results and Discussion

This section presents the results from the model outlined in section 4. The parameter estimates of household packaging waste determinants are presented in Table 5.

Table 5: Parameter Estimates Adjusted for Heteroscedasticity

Variable	Coefficient	Expected Sign	*t*-statistics	Significance
Constant	-1.493			
Pure Cost Determinants				
Weight fee	**0.372	+	2.007	0.046
Volume fee	-0.412	+	-1.596	0.112
Distance	-0.000	-	-0.177	0.860
Economic-Demographic Determinants				
Urbanization rate	0.001	+	0.121	0.904
Population density	0.000	+	1.284	0.200
Big city	*-0.530	-	-1.835	0.068
Private house	*0.016	?	1.959	0.051
Unemployed	**0.233	+	2.382	0.018
Income	0.003	?	1.074	0.284
Socio-Demographic Determinants				
Education	**-0.027	+	-2.156	0.032
Women	-0.023	?	-0.212	0.832
Age	0.033	?	0.999	0.319
New Immigrants	***-0.183	-	-3.083	0.002
Immigrants	***0.075	?	3.292	0.001
Environmental Preferences				
Municipalities	0.065	+	0.492	0.623
Households	***0.201	+	3.510	0.001
Collection Entrepreneurs				
IL	-0.220	?	-0.688	0.492
Stena	0.099	?	0.409	0.683
Sita	*-0.442	?	-1,762	0.079
Återvinning & miljö	0.433	?	1.163	0.246
Gästrike återvinnare	0.151	?	0.313	0.754
LBC	-0.119	?	-0.418	0.676
Kangos	**-1.534	?	-2.154	0.032
Merab	*-0.663	?	-1.898	0.059
NSR	**1.123	?	1.967	0.050
Östgötafrakt	*-0.456	?	-1.897	0.062
Rambo	-0.529	?	-1.366	0.173
Renova	0.098	?	0.253	0.800
SRV	0.055	?	0.206	0.837
Vafab	***0.976	?	2.720	0.007
VMR	-0.445	?	-1.554	0.122
Västblekinge miljö	-0.230	?	-0.593	0.554
Karskoga energi & miljö	***-1.622	?	-4.801	0.000
R^2-adjusted	0.222			
F-statistics	3.17 (0.000)			
Breusch-Pagan-Godfrey Test	221.13 (0.000)			

Note: *, ** and *** indicate statistical significance at the ten, five and one percent levels, respectively.

Since the presence heteroscedasticity is a common problem in cross-sectional data (e.g., Green, 1997) this was tested for. There exist many different types of heteroscedasticity tests. Three common tests include White's General test, the Goldfeld-Quandt test and the Breusch-Pagan-Godfrey test (Ibid.). According to Gujarati (2003), White's test is not suitable when the regression has many regressors because the method will quickly consume degrees of freedom. This implies that White's test is not appropriate in our case. The Goldfeld-Quandt test has two weaknesses according to Gujarati (2003); first it is only applicable if it can be assumed that the heteroscedastic variance is positively or negatively related with one of the regressors. Second, one must identify the correct explanatory variable, the one that explain the heteroscedasticity. These limitations are however avoided if using the Breusch-Pagan-Godfrey test. Consequently, the Breusch-Pagan-Godfrey test is applied in this study. Based on this test the null hypothesis of homoscedasticity was rejected on the 1 percent level (see Table 5). The presented *t*-statistics and significance levels in Table 5 have therefore been calculated using the White estimator for the heteroscedasticity-consistent covariance matrix (Green, 1997).

The goodness-of-fit measure, R^2-adjusted, for the household plastic packaging collection regression model is 0.22. This means that 78 percent of the inter-municipality differences was left unexplained. In other words, a great deal of the variance is due to the error terms or to variation in non-observed variables. Still, according to Green (1997), low R^2-adjusted are very common when using cross-sectional data. It is also obvious that all five categories of determinants explain parts of the inter-municipality differences in collection rates.

The *pure costs determinants* show some interesting results. The coefficient for weight fee is expected and it is statistically significant at the 5 percent level. A municipality that has introduced a weight-based fee has on average 372 gram more plastic packaging waste collected per resident than municipalities in which flat rates for household waste are used. This is a substantial difference if we consider the fact that an average municipality collects 1.33 kg per resident. The coefficient for the volume-based fee is, surprisingly, negative, but it is not statistically significant. The result that weight fees are more effective in increasing collection rates of plastic packaging is perhaps not so unexpected if consider that it is relatively easy for households to compress plastic bottles and containers. These findings are also supported by some of the earlier empirical literature (e.g., Sterner and Bartelings, 1997; Jenkins et al., 2003).

The third pure cost determinant, *distance*, has the expected sign but is neither statistically nor economically significant. Hence, the distance between the municipality and the recycling industry does not seem to matter for collection levels. An explanation for this result could be that the coefficient for the dummy variables for some entrepreneurs may explain some of the differences in plastic packaging collection that could be due to differences in distance. For example, the entrepreneur that has the longest distance to the recycling industry is Kangos and the municipalities in which this company operates have lower recycling levels than the municipalities covered by the reference entrepreneurs. The coefficient for distance actually becomes statistically significant at the five percent level if excluding all entrepreneurs from the econometric estimation.

The second category, *economic-demographic determinants*, seems overall to be important in explaining collection differences. All coefficients have the expected signs and three of these are statistically significant. Still, out of three coefficients for urbanisation rate, population density and big city only the coefficient for big city is statistically significant (at the ten percent level).[14] The coefficient for the variable "big city" implies that there is a negative congested effect for the collection of household plastic packaging as was hypothesized in section 3. The results indicate that households in a congested city collect on average 530 gram less plastic packing per resident than do households in smaller cities. The results also indicate that the urbanization rate and the population density do not seem to matter for collection of household plastic packaging waste. This result contradicts the findings from earlier research focusing on inter-country differences (Berglund and Söderholm, 2003). Berglund (2004) also concludes that cost differences within Sweden ought to exist and – if cost effectiveness is a major policy goal – different collection rates for different municipalities are motivated.

The above suggests that the overall costs of packaging collection do not seem to matter much for the collection outcome although there exist reasonable arguments for why it should matter. This may imply that the collection of household plastic packaging waste in Sweden is not performed in a cost effective manner. One explanation for this could be the pricing negotiations between PAB and the entrepreneurs (see section 3). It is reasonable to suspect that entrepreneurs that collect plastic packaging in "high cost" municipalities obtain higher compensation for their

[14] Also the coefficient for the big city variable becomes more statistically significant when excluding the entrepreneur dummies from the regression model.

collection activities. Thus, it becomes more profitable for entrepreneurs to reach relatively high collection rates in areas located far away from the recycling industries and with low population densities and urbanization rates.

Two of the three other economic-demographic variables are both economically and statistically significant. The coefficient for private house is positive and statistically significant at the ten percent level. The model suggest that a one percent increase in the percentage share of private houses in a municipality induces an increase in household plastic packaging waste collection by 16 gram per resident. The coefficient for unemployment is statistically significant at the five percent level. A one percent increase in the open unemployment rate yield a 233 gram increase in household plastic waste collection per resident. Consequently, the opportunity costs for households seem to be influence their collection activities. Finally, the coefficient for income is statistically insignificant, implying that income does not explain household plastic packaging waste collection in Sweden. A possible explanation for this could be that the positive environmental preference effect is neutralized by the negative opportunity cost effect.

The results for the third category of independent variables, *socio-demographic determinants*, are partly somewhat surprising. The coefficient for the education level is not only unexpected (negative sign), but it is also statistically significant at the five percent level.[15] The coefficient for women and age are both statistically insignificant. In other words, we could not reject the null hypothesis that these coefficients equal zero. The coefficients for immigrants show both expected and unexpected results. The coefficient for new immigrants has the expected sign and is statistically significant at the one percent level. If the share of new immigrants in a municipality increases by one percent the amount of collected household plastic packaging waste will decrease by about 183 gram per resident. However, the coefficient for (all) immigrants is positive and also statistically significant at the one percent level. A possible explanation for these results could be that when immigrants arrive to Sweden they are not well acquainted with the Swedish laws and regulations and neither are they able to understand Swedish very well. However, over time they learn and the results suggest thus that immigrants as a group collect even more plastic packaging waste than the average Swedish citizen.

[15] Education and income is relatively highly correlated with a correlation coefficient of 0.69. Still, the education coefficient remains statistically significant if income is excluded from the model estimation.

The fourth category of determinants, *environmental preferences*, seems to be important in explaining collection outcomes. Both the coefficient for environmental preferences in the municipality government (*ENVM*), and the coefficient for the environmental preferences in households (*ENVH*) have the expected signs. The coefficient for *ENVM* is not statistically significant, but the coefficient for *ENVH* is statistically significant at the one percent level. A one percentage point increase for the Green party in the election for the parliament implies that the collection of household plastic packaging tends to increase by 193 gram per residents.

The fifth category of variables, *the entrepreneur dummies*, also seems to explain some of the variance in household plastic packaging waste collection. As mentioned in section 3.1, it is difficult to form any *a priori* expectations about the signs of these coefficients. The results suggest that the coefficients for Sita, Kangos, Merab, Östgötafrakt and Karskoga are all negative and statistically significant at either the one, five or ten percent levels. The coefficients for Vafab and NSR are positive and statistically significant at the one and five percent levels, respectively. For example, municipalities in which Sita operates, collect on average 436 gram less household plastic packaging per resident than municipalities that have entrepreneurs that only act in one municipality (the reference category).

It is possible to identify several explanations for these differences in collection results for different entrepreneurs. *First*, it is possible that these results simply are due to differences in municipality characteristics, which however are not entirely accounted for in the model used. For example, Sita, one of the three nation-wide entrepreneurs, tend to operate in very densely populated areas. As has been noted above this is likely to reduce household plastic waste collection, and our "big-city" variable may not fully reflect these impacts. The coefficient for Sita also becomes more economically and statistically significant if the big city dummy variable is excluded from the econometric estimation. The other entrepreneurs that have statistically significant coefficients are all regional companies that operate within a constrained area (e.g., a specific county). For example, Kangos act in sparsely populated areas in the most northern parts of Sweden (located far away from the recycling industries). This should decrease collection rates. In the model we attempt to capture these impacts using the distance and the economic-demographic variables. Again, however, it is possible that the variables used in the econometric estimation are not fully reflecting these impacts. One example could be the urbanization rate variable. According to the definition of an urban area in Sweden it is enough if a village have 200 residents if there are not more than 200 meters between the buildings. In common, the municipalities are also very large in Sweden, especially in the northern parts of the country. This implies that if a municipality has many small villages evenly scattered over the land area, it could very well be defined as a rather urban area but the collection cost for household plastic packaging collection could still be high.

Second, it is possible that there exist regional differences in Sweden that are not captured by the independent variables included in the econometric estimation. For example, it is conceivable that in general politicians and/or households in the "NSR-area" have a more positive attitude towards recycling than those in the "Kangos-area".

Third, the inter-entrepreneur differences could also be explained by the presence of firm-specific circumstances. It is possible that there exist differences in collection productivity and/or negation skills between different entrepreneurs. Nevertheless, as mentioned above, the entrepreneurs do not seem to have very large opportunities to affect collection rates. Thus, in practice firm-specific attributes are unlikely to explain a great deal of the observed variance.

6. Concluding Remarks and Implications

The purpose of this paper has been to study the main determinants of inter-municipality differences in the collection of household plastic packaging waste in Sweden. Overall the results suggest that pure cost, economic-demographic, and socio-demographic factors as well as environmental preferences all influence collection rates to various degrees. Some particularly interesting findings are worth noting.

First, the impacts of distance to recycling industry, urbanization rate and population density on collection outcomes turn out to be both statistically and economically insignificant. This result contradicts, though, both economic theory and earlier research (e.g., Berglund and Söderholm, 2003). Still, a reasonable explanation for this is that the compensation from the material companies varies depending on region and is thus quite effective in reducing cost disadvantages in collection. However, if true, this also suggests that the collection is cost ineffective. Consequently, it should be possible to collect the same amount of household plastic packaging waste at lower costs and/or increase collection rates without imposing higher costs. *Second*, the "big city" variable has a statistically and an economically significant impact, suggesting that there exists some critical level for population density, at least within the existing system, at which collection costs starts to increase with increases in the density.

The above evidence indicates that society could save resources by focusing more on attention on regional collection cost differences. This could be organized in different ways, both within the realms of the existing system or with an alternative waste packaging regime. Within the existing system it could be possible for the authorities to give up the *de facto* requirement that collection should be nationwide, or at least set different collection targets for dense and sparsely populated regions. This would give PAB an increased possibility to organize the collection in a cost effective manner, and equalize the marginal costs of collection in different municipalities. However, there are reasons to doubt that this policy shift should be sufficient for inducing cost effective collection activities. The requirement of nationwide collection is in practice actually a relative flexible; PAB already has relatively large opportunities to collect much in low cost municipalities and less in high cost ones. Another solution could therefore be to consider an alternative waste packaging management regime. An alternative suggested in the literature is the so-called UCTS system, a combination of tax on packaging and a recycling subsidy (e.g., Walls and Palmer, 2001). For a discussion of how this system could affect the collection of Swedish packaging collection, see Hage (2004) who, however questions the notion that this alternative overall would be a more effective waste management system than one based on producer responsibility.

The paper also indicates that the actors within the existing waste management regime can affect the collection outcome in various ways. *First*, the municipalities can increase the collection by introducing weight-based waste fees. However, at the same time as this seems to be an effective method to increase collection of packaging, the municipalities should consider undesirable side-effects of such fees. A weight-based waste fee will give households an incentive for illegal waste disposal. Empirical research in U.S. has also provided some support for this notion (Fullerton and Kinnaman, 1996). This analysis simply shows that municipalities that employ weight-based fees have on average a higher collection rate, but we have no information about illegal dumping in the respective municipalities. *Second*, it also seems as if the material companies could increase the collection of packaging collection by clearly informing new immigrants about the collection system. However, the costs for this information should of course also be considered.

The results from the study also suggest that it is reasonable to put a value on household contributions when, for instance, analysing if the producer responsibility ordinance is socially worthwhile. *First*, the paper shows that unemployed people tend to collect more plastic packaging waste than employed persons. Hence, people with a higher opportunity cost of time tend to collect less. *Second*, the study also shows results that are consistent with the notion that "green" households recycle more of their plastic packaging waste than the average household.

Finally, the paper also indicates the presence of a potential problem for the producer responsibility ordinance, at least for plastic packaging. Higher education levels are often correlated with a higher willingness to pay for environmental improvements and empirical research from U.S. also shows, at least partly, that high education and recycling tend to be positively correlated. This study shows, however, the opposite.

This study also indicates that several issues should be in focus for future research efforts. *First*, it is important to investigate the determinants of the collection of other packaging materials than plastic. It is possible that, for instance, the effects of different policy schemes differ across materials. Jenkins et al. (2003) also confirm this for U.S. recycling behaviour. *Second*, U.S. studies typically show that curbside recycling is the most effective policy to implement in order to increase recycling in the USA. Increased knowledge about how this policy would affect Swedish recycling activities is important. *Third*, this study, and others as well, show that weight-based waste fees are effective in promoting waste collection. However, it is also important to obtain more knowledge about the costs of this system, including the risks for illegal disposal. *Fourth*, we also need more information about whether the socially optimal mix of different waste treatments (e.g., incineration versus recycling) will differ across regions. Cost-benefit analyses that consider regional aspects are therefore desirable.

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