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The Economics of Household Packaging Waste

Norms, Effectiveness and Policy Design

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The Economics of Household Packaging Waste: *Norms, Effectiveness and Policy Design*

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Abstract

This thesis consists of an introduction and four self-contained papers, which all deal with the economic effectiveness of the Swedish producer responsibility for packaging materials. **Paper I** analyzes the determinants of household packaging recycling efforts in Sweden by employing data on households' self-reported behavior. This is analyzed in an ordered probit regression framework. Theoretically the paper draws heavily on recent developments in the literature on integrating norm-motivated behavior into neoclassical consumer theory. The results show that both economic and moral motivations are important in explaining household recycling outcomes. This indicates that recycling campaigns could be effective in increasing recycling efforts, not the least by influencing individuals' perceptions about others' (positive) contributions in the recycling field. The results also indicate that the importance of moral motivation partly diminishes if improved collection infrastructure makes it easier for households to recycle. **Paper II** investigates the main determinants of collection rates of household plastic packaging waste across Swedish municipalities. This is done by the use of a regression analysis based on cross-sectional data for 252 Swedish municipalities. The results suggest that local policy measures, geographic/demographic variables, socio-economic 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 on collection outcomes of differences in distance to recycling industry, urbanization rate and population density turn out, though, both statistically and economically insignificant. This implies that the plastic packaging collection in Sweden may be performed cost-ineffectively. Finally, the analysis also shows that municipalities that employ weight-based waste collection fees overall experience higher collection rates than those municipalities in which flat and/or volume-based fees are used. **Paper III** extends the analysis in Paper II primarily by: (a) adding 30 more municipalities; (b) including additional data on local policy variables; and (c) adopting spatial econometric methods. The empirical results suggest that the collection of plastic packaging is positively related to collection in neighboring municipalities. The analysis also shows that municipalities that employ weight-based waste fees generally experience higher collection rates than those municipalities in which volume-based fees are used. The presence of curbside recycling and a high intensity of recycling drop-off stations also provide important explanations for why some municipalities perform better than others. Paper III also confirm the finding from paper II that the plastic packing collection in Sweden seems to be conducted in a cost-ineffective manner. Finally, **Paper IV** analyzes the incentive structure and the cost-effectiveness of the Swedish producer responsibility ordinance. 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 give rise to *output* and *input substitution effects*. However, none of the systems tends to encourage enough of *design for recyclability*. 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. This will depend on, for instance, households' valuation of sorting efforts and the presence of economies of scale in the waste collection system. The above also suggests that different systems can be preferred in different parts of the country, and that the cost-effectiveness of the Swedish packaging collection scheme could be improved.

To mum

List of Papers

This doctoral thesis contains an introduction and the following papers:

Paper I: Norms and Economic Motivation in Household Recycling: Evidence from Sweden.

Paper II: An Econometric Analysis of Regional Differences in Household Waste Collection: The Case of Plastic Packaging Waste in Sweden. Forthcoming in *Waste Management*. (with Patrik Söderholm).

Paper III: Household Plastic Waste Collection in Swedish Municipalities: A Spatial-Econometric Approach. (with Krister Sandberg, Patrik Söderholm and Christer Berglund).

Paper IV: The Swedish Producer Responsibility for Paper Packaging: An Effective Waste Management Policy? Reprinted from *Resources, Conservation and Recycling*, Vol. 51, (2007), 314-344, with permission from Elsevier.

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Luleå, December 2007

Olle Hage

Preface

1. Policy Background and Research Focus

The *research focus* of this thesis is to investigate why Swedes recycle packaging materials, and to evaluate the design – and primarily the cost effectiveness – of the Swedish producer responsibility for packaging materials. Of course, in all times people have recycled old items when it has been profitable, and this driving force is still present. However, items that are not profitable to recycle become waste, and end up at landfills. In the 1970s politicians started to pay increased attention to the so-called waste problem. For example, in countries such as Sweden and the US, people worried that the accumulation of waste would create a shortage of landfills (e.g., Skottheim, 2000; Ackerman, 1997). In time, the importance of recycling also became visible in Swedish government policy and legislation. For instance, in 1994 producer responsibility ordinances were introduced for packaging, paper and tires.[1] It is useful to analyze the effects that this policy has had on household packaging recycling because household packaging waste is a low-value product that normally was not recycled before the producer responsibility legislation was introduced. This study will only consider recycling of packaging (without deposit-refund system) in Sweden, and for this reason only the Swedish producer responsibility for packaging will be considered below.

Given the political intention to promote household recycling it is essential to understand the households' motivation to recycle: Why do people recycle? Does economic motivation matter for the recycler or is recycling an altruistic behavior where economic motives and incentives do not matter? This implies that it is important to have a proper understanding of how local waste management policies and recycling infrastructure implemented at the municipal level influence recycling levels. What happens if the municipality switches from volume-based waste fees to weight-based waste fees? What happens if the waste operators start to collect the waste from the household instead of relying on drop-off recycling stations? These questions are of current interest for actors within the existing waste management system in Sweden. The municipalities could by lowering household waste save money, especially today when they need to pay a tax on waste that end on landfills or are energy recovered.

[1] Since 1994 electronic products and cars have also been included in the producer responsibility legislation.

The national recycling authorities have also recently started to question the collection system chosen by the producers and have suggested that households need a more convenient recycling infrastructure. However, they will not give detailed instructions on this matter yet, because they are of the opinion that there is a lack of information about the effects of such measures. A proper understanding of the producer responsibility ordinance is also important for the evaluation of environmental policy, especially: Does the ordinance give the producers cost-effective incentives to economize on virgin resources? Have the producers implemented a cost effective collection? If the answer is no to these question, this suggests that authorities and producers could save money without sacrifice the environmental goals. These are some of the issues that are addressed in this thesis. Given that Sweden has been one of the international forerunners in the promotion of recycling, the results presented in the thesis could also provide important lessons for other countries.

In Sweden today, only a selected number of materials are affected by the producer responsibility. 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) has suggested that the producer responsibility should be extended to all goods in Sweden. Also the Swedish Environmental Protection Agency (e.g., SEPA, 1999; 2006a) has been positive towards the implementation of producer responsibility legislation for additional goods. Hence, all producers, not only the currently affected ones, should be interested in gaining knowledge about the nature and the impacts of the producer responsibility legislation.

2. Polices Aiming to Influence Household Recycling Behavior in Sweden

This section will describe the design and the implementation of policies and infrastructure facilities that influence the recycling decisions of household packaging waste in Sweden. The producer responsibility ordinance is the cornerstone of these polices, and an understanding of this policy is necessary for understanding household packaging waste recycling efforts. However, Swedish household packaging recycling decisions are also strongly influenced by local policies aimed to lower household waste flows and hence, intentionally or accidentally, stimulating recycling. These policies include waste fees and infrastructural measures that lower the time cost for the recycler. This section also briefly discusses the extent to which the producers have fulfilled the recycling rate targets in the producer responsibility ordinance. Finally, the recycling authorities' views about how the producers should develop the future household packaging collection will be discussed.

Before introducing the producer responsibility ordinance it is appropriate to introduce some important definitions. In this thesis, the definitions of the EC directive for packaging and packaging waste have been used. *Reuse* means in this case to refill a packaging and use it according to the original purpose. Packaging materials that are reused are not considered waste and consequently will not be included in the recycling results. *Recycling* implies that the waste should be processed and used as input in new production. *Energy recovery* means that burning of the waste is permitted, provided that the energy content is recovered.

2.1 The Design of the Producer Responsibility Ordinance

The Producer responsibility ordinance implies that the producer has the physical and the economic responsibility for the packaging waste. The producers are obliged to provide and operate suitable systems for collecting and recycling packaging waste. 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 the Swedish Environmental Protection Agency (SEPA). The responsibility for informing households is today shared between the producers and the municipalities. The producers are responsible for national recycling campaigns, for instance television advertising and for providing information on the internet, while the municipalities are responsible for municipality-specific information.[2] The packaging consumers, on their part, must clean and sort packaging waste from other waste and transport this waste to the recycling stations (without any financial compensation from the producers). The ordinance also prescribes specific recycling rates targets for the producers, see Table 1 (SFS 2006:1273; SEPA, 2006a).

Table 1: The Swedish Producer Responsibility Targets for Packaging in 2007

Type of packaging waste	Recycling (%) (only material recycling)	Recycling including energy recovery (%)
Metal, not drink packaging (aluminum and steel)	70	70
Paper (paper, cardboard and corrugated cardboard)	65	65
Plastic, not drink packaging	30	70
Glass, excluding reusable glass	70	70
Metal, drink packaging (aluminum cans)	90	90
Plastic, drink packaging (PET-bottles)	90	90
Wood	15	70
Other materials	15	30

Note: These targets are intended for all packaging waste, not only packaging waste from households.

Source: SFS 2006:1273.

[2] Before 1 of January 2005, the producers were responsible for all information activities. Typically they then paid the municipalities for informing the households about local conditions.

The producer responsibility ordinance is a law with few detailed instructions; the Swedish government preferred voluntary solutions for the industry (Government Bill, 1992/93:180). The producers 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 (flexible) 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 is considered to be a suitable collection system, but they also prefer flexibility on the part of the producers. For this reason SEPA has only decided to give one detailed instruction about the collection system; the Agency requires that the collection should be nationwide. The motivation for this cautious usage 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).

2.2 The Implementation of the Producer Responsibility Ordinance

Figure 1 shows the organization of the Swedish household packaging waste. In order to comply with the producer responsibility, in 1994 the retailers and the producers founded four joint material companies that administrated the collection and recycling of packaging waste.[3] These four material companies form the service organizations Förpacknings- och tidnings-insamlingen AB (FTI)[4] and Reparegistret AB (REPA). FTI's task is to coordinate the different responsibilities of the material companies. For instance, they establish and operate recycling stations and inform packaging consumers about the collection and recycling system (FTI, 2006). Via FTI, the material companies can offer a nation-wide coverage of packaging waste collection. Individual producers can fulfill their producer responsibility if they join FTI; they then pay a packaging fee[5] to FTI based on the weight of their packaging.[6] Already in 1986, when the collection of glass packaging (without deposit-refunds) took off, the industry

[3] They were; Plastkretsen AB (PAB) (plastic packaging), Svensk Kartongåtervinning AB (SKAB) (paper and cardboard packaging), Svenska Metalkretsen AB (SMAB) (metal packaging), and RWA Returwell AB (RWAB) (corrugated cardboard packaging). In 2006 SKAB and RWAB merged to form the new entity Returkartong AB (RAB) and starting in August 2007 the plan was for FTI to take over the responsibility for the household collection from PAB, SMAB, and RAB. However, this reorganisation is not yet completed.

[4] Before November 2004 FTI was named Förpackningsinsamlingen (SFAB).

[5] This fee differs for different packaging materials and it is the packaging filler (for products produced in Sweden) and/or the importer (for products produced abroad) that should pay the packaging fee.

[6] The packaging fees were administrated by REPA before August 2007 but today REPA have moved all of their operations to FTI.

founded the joint material company Svensk Glasåtervinning AB (SGAB) (Glass packaging).[7] All these companies – the producer responsibility organization (PRO) - run without profit interest and they do not distribute any returns to their owners.

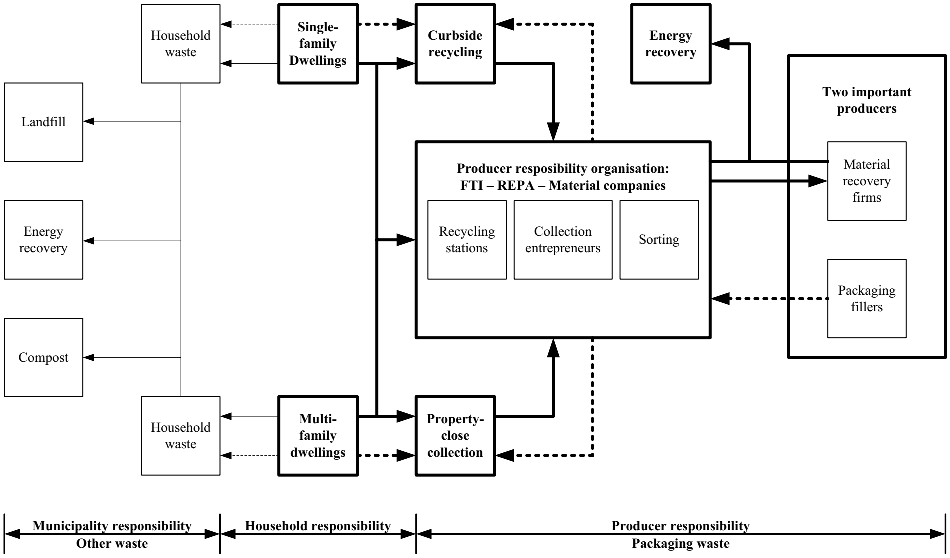

Figure 1 Household Waste Organization in Sweden: The Case of Packaging
Note: The flowchart shows the direction of all household waste ("complete" lines) and the payment for these services (broken lines). The households should deliver the packaging waste to the producer responsibility organization (bold lines) by using the recycling stations or by using curbside recycling/property-close collection provided by other actors. However, the household could also "defect" and put the packaging in the household (non-packaging) waste bin (thin lines).

For managing the household collection of household packaging waste the PRO has chosen to establish recycling stations (drop-off stations) scattered around Sweden. Commonly, a recycling station includes different recycling bins for the respective packaging materials, and these stations should only be used by households. In 2006 there existed about 6000 drop-off stations for a total Swedish population of 9 million people (Funck, 2006).[8] This means that on average about 1500 individuals "share" a station, and since Sweden is a quite sparsely populated country some households may be located far away from their nearest drop-off station. In order to facilitate the collection of packaging waste from recycling stations, the material companies have engaged different collection entrepreneurs. These entrepreneurs

[7] SGAB is separated from REPA and FTI but cooperate with them.

[8] Usually a recycling station has at least one container for each packaging waste material. However this is not always the case, so the number of materials that are collected at each recycling station may differ slightly.

collect the packaging waste at the recycling stations and transport the packaging waste to packing recyclers. The entrepreneurs are compensated by the material companies through a fixed compensation and one variable compensation. The fixed compensation is decided in (secret) negotiations between the entrepreneur and the material company, and hence differs for different material and entrepreneurs. On the other hand, the variable compensation is the same for all entrepreneurs that collect the same packaging material, but it varies across different packaging materials (Pihl, 2002; Schyllander, 2007).

2.3 Local Recycling Facilities and Waste Polices

According to Figure 1, also other actors have, by introducing packaging collection services, become involved in the household packaging collection after the introduction of the producer responsibility. Mainly there are two forms of this type of collection in Sweden. First, many multi-family dwelling owners have installed central sorting houses or rooms within the property, hereafter called *property-close collection*.[9] In these facilities people living in the apartments can leave their packaging waste in specially assigned bins for different packaging materials. Second, in about 15 municipalities the local authorities have organized *curbside collection* for single-family dwellings.[10] In both these cases the packaging materials are collected by waste entrepreneurs that sell the packaging waste to the material companies. These entrepreneurs get the same variable compensation as the entrepreneurs that are engaged by the material companies (Forselius, 2007). However, they do not get the fixed compensation from the material companies for their work. In most cases, these forms of collection also imply higher collection costs than the PRO collection. Consequently, this means that these multi-family owners and municipalities take on parts of the producers' responsibility without full compensation. Their behavior is often motivated by environmental concern and/or aims to increase the convenience for the households. In recent years these circumstances have been criticized by interest groups,[11] which argue that the material companies should organize and pay for these types of packaging collection.

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[9] Some rough estimations suggest that 46 percent of all multi-family dwellings in Sweden have access to this service (SEPA, 2006a).

[10] The number of municipalities that offer curbside recycling differs slightly for different packaging material.

[11] Some of these are: Villaägarna (The single-family dwellings organization in Sweden), 2007; Waste Sweden (The Waste Companies Organization) (RVF, 2006); large Swedish multi-family dwellings organizations (e.g. Fastighetsägarna, HSB, SABO and the Swedish tenant organization (Hyresgästföreningen), see e.g., Eriksson et al. (2006).

As we could see in Figure 1, the municipalities are responsible for handling the non-producer responsibility household waste in Sweden.[12] Normally, this (non-packaging) waste is collected from the property. This means that households – especially the household that do not have the opportunity to use property-close collection or curbside collection – could save time and inconvenience if they put their packaging waste in this municipality waste stream instead of using the recycling stations. However, local authorities have also introduced new waste policies establishing financial incentives for households to lower waste levels and hence stimulate recycling (because household do not pay for leaving their packaging waste at recycling stations). According to Hage et al. (2007), in 2005 all municipalities had abandoned the flat fee pricing for waste and a majority hade introduced some type of volume-based fees.[13] Furthermore, 25 municipalities had introduced weight based waste fees.[14] However, these policies will mainly affect the behavior of households in single-family dwellings. The residents in multi-family dwellings will continue to pay the waste management as a fixed part of the rent; hence they face no monetary incentive to increase recycling efforts.

2.4 The Producer Responsibility Ordinance: The Outcome

Swedish recycling of packaging waste shows quite impressing results. The total rate of recycling – if excluding wood and other packaging – was 68 percent in 2005 and if we include energy recovery, 75 percent was recycled. (SEPA, 2006b) The recycling outcomes for each packaging material are presented in Table 2.

If we compare these results with the recycling targets in Table 1 we can observe that the material companies that are responsible for paper and glass packaging have fulfilled their responsibility in 2005. However, it is obvious that the goal fulfillment in (total) paper packaging recycling can to a great extent be explained by the success of corrugated cardboard recycling. SEPA (2005b) also notes that the material company (SKAB) should increase their effort in increasing the recycling of paper and cardboard packaging. On the other hand, the material companies that are responsible for metal and plastic packaging do not fulfill the recycling targets. In the case of metal, this is to a great extent explained by low aluminum packaging recycling. Table 2 also brings to light that recycling levels have increased after the packaging responsibility ordinance was introduced. As was mentioned, packaging materials

[12] This service is sometimes run by the municipalities themselves while others engage waste entrepreneurs.

[13] 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 (Villaägarna, 2006).

[14] Sweden is dived into 290 municipalities.

that are included in a deposit-refund system are excluded from the analysis in this thesis. Still it is interesting to note the quite expected result that recycling levels are higher if the packaging consumers have financial incentives to recycle. None of the packaging types that make use of a deposit-refund system has a recycling level that is below 86 percent. The recycling level of corrugated cardboard is also 86 percent. This is to great extent explained by the fact that the collection of corrugated cardboard from business and industry operations is a profitable activity. 90 percent of the recycled corrugated cardboard comes from these sectors (SEPA, 2004).

Table 2: Recycling and Energy Recovery Rates for Packaging in Selected Years (%)*

Packaging material**	1992	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005
Aluminum, not cans	-	-	-	17	18	24	28	25	22	24	28	27	27
Steel	-	-	-	41	64	40	43	61	71	70	73	67	72
Metal (total), not cans	-	-	-	n.a.	n.a.	n.a.	n.a.	***55***	***63***	***62***	***65***	***60***	***58***
Paper and cardboard	-	10	19	45	34	37	35	35	41	37	38	38	42
Corrugated cardboard	67	74	77	81	84	85	84	84	85	86	85	86	86
Paper (total)	-	-	-	n.a.	n.a.	n.a.	***67***	***68***	***69***	***70***	***70***	***71***	***72***
Plastic, not PET-bottles (brackets includes material recovery)***	-	-	-	***11 (13)***	***13 (21)***	***20 (36)***	***16 (32)***	***15 (33)***	***13 (29)***	***16 (33)***	***18 (69)***	***19 (67)***	***24 (73)***
Glass, not reusable	***55***	***56***	***61***	***72***	***76***	***84***	***84***	***86***	***84***	***88***	***92***	***104***	***95***
Aluminum, cans****	85	91	92	92	91	87	84	86	85	86	85	85	86
PET-bottles****	n.a.	49	73	81	77	80	74	78	78	77	79	80	95
PET-bottles, reusable****	n.a.	n.a.	n.a.	97	98	98	98	98	98	97	97	99	98
Glass, reusable****	n.a.	97	98	96	97	98	99	99	99	99	100	99	n.a.

* The table shows recycling results from both households and producers.

** The outcomes for wood packaging are excluded because the data is not reliable and households are only small consumers of wood packaging. (SEPA, 2006b) Other packaging outcomes are excluded because data were not available. These packaging materials do neither have a collection system for households.

*** The huge increase in energy recovery of plastic packaging in 2003 is due to that soft plastic packaging from the (municipalities) waste stream are collected from that year.

**** The recycling of these packaging materials are organized through a deposit-refund system, and this part of the packaging waste stream is not analyzed in this paper.

Sources: SEPA (2006b), SEPA (2003), and SEPA (2002a).

The annual data from SEPA (Table 2) do not reveal to what extent the household packaging waste (without deposit-refund systems) are recycled. However, SEPA (2002b) reports that the households collect about; 85 percent of the recycled glass packaging waste, 42 percent of the recycled plastic packaging waste, 33 percent of the recycled metal packaging; 52 percent of the recycled paper and cardboard packaging; and 10 percent of the recycled corrugated cardboard. To which degree the packaging waste stems from household or from producers also differs across materials. For instance, about 65 percent of the plastic packaging is used by households, but only a fraction of the corrugated cardboard is used by the same sector. Overall – if considering packaging without deposit-refund systems – the above shows two interesting phenomena. First, households already recycle quite much of their packaging waste without any financial compensation from the packaging waste receiver. Second, in spite of this, there seems to exist a stock of packaging waste that is not recycled.

2.5 The Producer Responsibility Ordinance: The Future

In recent years the issue of convenience in household recycling has gained increased national policy attention. For instance, a Government bill (2002/03:117) states that property-close collection should be the main type of collection of used packaging in multi-family dwellings, and the producers should be economically responsible for setting up this system. The Swedish Waste Council (SEPA, 2006b) also suggests that producers should be responsible for establishing property-close collection schemes as well as curbside recycling whenever this is possible. They also propose that the producers have this responsibility even if the market value of the collected packaging is not sufficient to cover the costs of these systems.[15] The Swedish waste management plan also emphasizes the importance of implementing collection systems that are perceived as convenient by the households (SEPA, 2005a). Furthermore, SEPA (2006a) is doubtful whether today's main system with recycling stations is sufficient for giving the households an adequate amount of service. However, they conclude that the state of knowledge about the effects on private costs, social costs, and environmental effects from curbside collection and property-close collection is not sufficient for giving clear guidance or national regulations on the matter. Hence, more research is clearly needed about these effects and the present thesis could thus play a role in filling this knowledge gap.

Following the above, the *overall purposes* of this thesis are to: (a) analyze the role of norms and economic motivation in explaining household recycling efforts, (b) evaluate and explain the differences in collection rate outcomes across Swedish municipalities, (c) analyze the effectiveness of local policy and waste management facilities that aim to support household recycling, and (d) evaluate the design – and primarily the cost effectiveness – of the Swedish producer responsibility ordinance for packaging materials. Before explaining the scientific methods to be used, a short discussion about the individual motives for recycling as well as about the economic motives for state intervention in the waste sector is needed.

[15] This is not the case today, curbside recycling is mainly financed by the municipalities and the property-close collection is financed by the multi-family dwelling house owners. This situation has been criticized by the government (Government Bill 2002/03:117) and SEPA (2006a), not the least since it is claimed to reduce the producers' incentives for improving the recyclability of their products.

3. Theoretical Consideration on the Household Recycling Decision

3.1 Why do Households Recycle?

The economic man in the standard text books only recycle if the private marginal cost of recycling is below (or equal to) the private marginal revenue of recycling. In the discussion above we noted that there is little to indicate that this condition holds in the case of packaging waste (without a deposit-refund system). The recyclers cost could be quite substantial, she should clean, sort, and store the packaging and many need to transport the packaging waste to the recycling station. For a majority of households it is difficult to see any financial benefits from recycling.

Of course, we could maintain that recycling will give the household other forms of benefits. For instance, recycling could save virgin resources and it will definitely decrease the need of landfills. The answer to this is – according to the standard economic model – that this is certainly correct, but it will not change the recycling behavior of the household. The reason is that the benefits from recycling are characterized by non-rivalry and non-excludability, i.e., recycling activities contributes to the production of public goods. This means that everyone will benefit from the individual's recycling effort and it is impossible to exclude other individuals – that not participate in recycling – from reaping the benefits from individuals' recycling efforts. This will lead to free-rider behavior and, consequently, very few used packaging materials will be recycled. Andreoni (1988) also showed that even if we consider the presence of pure altruism, the contribution to public goods, and hence recycling, should be very small in large economies.

Another objection could be that recycling is mandatory according to the producer responsibility legislation, and that thus non-complying behavior can be enforced through penalties. However, in reality, household participation in recycling schemes is almost never controlled and enforced, so in this respect we may say that the participation is *de facto* voluntary. Hence, according to standard economic theory, it seems like a paradox that households contribute to recycling. Hence, this suggests that we should try to expand the traditional economic models when analyzing the possibility of private provisions to public goods.

Other fields of research have also tried to identify the factors that explain household recycling efforts. For example, psychologists have often focused on the role of personal norms created by internal motivation (moral norms), sociologists stress the importance of norms arising from external approval or disapproval (social norms), while economists typically focus on external motivation through economic incentives in combination with facilitates that lower the time cost for recycling (e.g., Hornik et al., 1995; Guagnano et al., 1995). Still, the success in explaining recycling outcomes for each of these scientific disciplines has been limited. It seems as if each discipline contributes a partial answer; in practice both external and internal motivations matter. Guagnano et al. (1995) also conclude: "Science and policy require a socioeconomic theory of behavior that incorporates both external conditions and internal processes," (p. 700).

In this thesis, it is assumed – following Brekke et al. (2003), Bruvoll (2004), and Nyborg et al. (2006) – that people want to think of themselves as socially responsible persons. In economic terms one could say that they have preferences for maintaining a self-image as a morally responsible person. If the recycler does not confirm to what she belief is the right thing to do – the internal (moral) norm – she will feel guilt and have bad conscience. Hence, if an individual belief that a socially responsible person should recycle she will, at least partly, recycle without any external incentives and pressure. The utility model that is developed in this thesis (see paper 1) is a developed version of the model outlined in Nyborg et al (2006), and it is in line with Schwartz's theory for moral motivation (Schwartz, 1970, 1973, 1977). In the latter studies awareness of consequences and ascription of responsibility are identified as important factors determining moral decisions.

However, in the thesis we also maintain that external motivations will matter for the recycler. The recycler could face external motivation through two different channels. First, it could be the case that relatives, neighbors and friends will influence the recycler through external pressure (social norms). Holländer (1990) suggests that the individual will feel approval (or disapproval) from close ones if she confirms (or does not confirm) to the social norm, in this case recycling. Second, it is assumed that the individual will be influenced by external incentives. In the case of recycling there will mainly be two types of incentives that could matter. One is that the recycler should be stimulated to recycle more if she has some financial incentives to recycle. In section 2 we noted that in Sweden mainly households living in single-family dwellings could face these types of incentives. If they use the recycling stations they could save some money by lowering their waste bill. Another external incentive is that household could be influenced by the presence of convenient recycling facilities that lower the time cost for recycling. The distance to a recycling station and the opportunity to use property-close or curbside collection both represent infrastructure measures that ought to decrease these costs and, thus increase recycling.

In the thesis papers 1, 2 and 3 investigate the above explanations for household recycling. The emphasis in paper 1 is on how moral norms, social norms and external incentives can help in explaining inter-household differences in recycling efforts. Papers 2 and 3 both analyze to what extent local policy measures and regional characteristics influence recycling rates at the municipal level.

3.2 What is the Best Recycling Policy?

Economists distinguish between different efficiency/effectiveness criteria when evaluating policy measures. One central criterion is *social optimality* (or Pareto efficiency), and for our purposes it is present when the social (private plus external) marginal costs of production equal the social 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 associated with measuring what is being exchanged and with 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 social cost-benefit analyses. To what extent the Swedish producer responsibility ordinance for packaging and papers is socially optimal has been evaluated in a few cost benefit analyses (e.g., Radetzki, 2000; Ekvall & Bäckman, 2001; Bäckman et. al. 2001). In the case where the households' efforts are valued (in terms of the opportunity cost of time) all these studies, indicate that the recycling of packaging waste is relatively costly for society, but if households' time is assigned a low value recycling schemes are often found socially beneficial from an economic point of view.

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., the optimal balance between social benefits and costs. Instead they often decide upon "exogenously given" environmental goals such as a quantitative target for recycling. Thus, in the latter case the policy is an outcome of the policy decision process rather than a "technical" issue to be decided using economic cost benefit analysis. Still, in such cases another policy criterion, *cost-effectiveness*, becomes central. Policies that attain a given goal at lowest possible total social costs are cost-effective. It should be clear that cost-effectiveness is a necessary – but not a sufficient – condition for social optimality. It is also important to recognize that all transaction costs should be incorporated in the total cost assessment.

We could also distinguish between different types of cost-effectiveness criteria and in this thesis two of them are analyzed. The first one addressed here is *spatial cost-effectiveness in the collection of packaging waste*. We have spatial cost-effectiveness if the marginal collection costs are equal across all regions. However, it is reasonable to believe that the marginal collection cost for collecting the same amount per capita packaging waste is higher in sparsely populated areas far away from the material recovery firms than in urban areas (e.g. Berglund, 2004). This implies that a spatially cost-effective recycling policy should collect more packaging waste per capita in densely populated areas that, for instance, are close to the recycling industries and collect less in sparsely populated regions far away from the recycling industries.

A cost-effective recycling policy should also give the producers *cost-effective incentives for waste minimization* throughout the entire supply chain. In other words, it is not enough to collect and recycle the packaging waste in a cost-effective way, the producers should also face the correct incentives when they produce and fill the packaging. This implies that producers should face incentives to: (a) decrease the amount of packaging (output effect); (b) use more recycled inputs in packaging production (input substitution effect); and (c) undertake cost-effective changes in packaging design, i.e., improve the packaging recyclability (design for recyclability). The spatial cost-effectiveness of the Swedish producer responsibility is scrutinized in papers 2, 3, and 4, while paper 4 also investigates whether the ordinance provides cost-effective incentives in the production of packaging.

4. The Methodological Approach

In this thesis we apply three different methodological approaches in our evaluation of the Swedish producer responsibility ordinance for packaging materials. These three methods are briefly discussed in this section.

Paper 1 analyzes how moral and social *norms* influence household recycling behavior. The theoretical framework used is a neoclassical utility model that incorporates norm-motivated behavior by assuming that individuals have preferences for being a morally responsible person. The empirical analysis builds on data on self-reported behavior from about 800 households in four Swedish municipalities. Beside questions about moral and social norms the questionnaire includes questions about economic, socio-economic and policy-related variables. The self-reported household behavior is analyzed by employing ordered probit regression.

Papers 2 and 3 analyze the actual *outcome* of the producer responsibility. Actual collection rates of households' plastic packaging collection at the municipal level in Sweden are analyzed, and the main determinants of household collection rates are identified. This is done using econometric analysis based on cross-sectional data. The data include economic, demographic, institutional and policy-related variables. This approach permits us to isolate the factors that determine the differences in collection rates across Swedish municipalities and the papers particularly investigate whether different cost-related factors play an important role in shaping these regional differences as well as whether different policy tools are effective in inducing higher recycling rates. It should also be pointed out that papers 2 and 3 give an opportunity, at least partly, to test the reliability of the results from the analysis of the self-reported behavior in paper 1. The main differences between paper 2 and 3 are: (a) in paper 2 only 252 (out of a total of 290) municipalities are analyzed due to data limitations and in paper 3 almost all (282) municipalities are analyzed; (b) in paper 2 the collection rates in the year 2002 are investigated while in paper 3 the same is done for 2005; (c) in paper 3 more policy variables (the number of recycling stations and the presence of curbside collection) are available and hence possible to evaluate; and (d) in paper 2 an OLS estimation is used while in paper 3 spatial econometric methods that explicitly deal with the incorporation of spatial autocorrelation are used in the econometric analysis.

Finally, in paper 4 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 (e.g., Walls and Palmer, 2001). 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 basis 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 material supply chain, on an analysis of the producer responsibility legislation and the incentives it provides, as well as on 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 for the involved 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 potentially provide a more cost-effective approach to the waste management problem than the present Swedish one.

5. Summaries of the Appended Papers

The thesis consists of this covering preface and four self-contained papers. In this section we briefly summarize the purpose, methodological approach and main results of each of these papers.

Paper 1 Norms and Economic Motivation in Household Recycling: Evidence from Sweden

This paper analyzes the determinants of household packaging recycling efforts in Sweden for four different packaging materials; paper, plastic, glass, and metal. We employ a model that integrates norm-motivated behavior into neoclassical economic theory by assuming that individuals have preferences for maintaining a self-image as morally responsible persons. The empirical analysis is based on a postal survey that was sent out randomly to 2800 households in four different Swedish municipalities. The data on self-reported recycling behavior are analyzed in an ordered probit regression framework.

The study indicates five main findings. First, the analysis indicates that property-close collection in multi-family dwelling houses induces higher recycling outcomes. Second, moral norm activation explains much of the variation across households' recycling efforts, given the current main collection with drop-off stations. This indicates that recycling campaigns could be effective in further increasing recycling efforts, and the results also suggest these campaigns could attempt to influence people's perception of others' (positive) contributions to recycling. Third, the importance of moral norms partly diminishes when the collection infrastructure makes it easier for households to recycle. Fourth, elderly Swedes report that they recycle more packaging material than do younger ones. Fifth, and finally social norms and legal norm cannot explain much of the variation in recycling efforts across households, but in part this can be explained by the fact these types of norms are at least partly mediated through personal (moral) norms.

Paper 2: An Econometric Analysis of Regional Differences in Household Waste Collection: The Case of Plastic Packaging Waste in Sweden (with Patrik Söderholm) (forthcoming in *Waste Management*)

The Swedish producer responsibility ordinance mandates 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 are relatively free to choose the methods needed to fulfill their responsibility. However, SEPA requires that the collection should be nationwide. This paper investigates the main determinants of collection rates of household plastic packaging waste in Swedish municipalities, and this is done by employing regression analysis based on cross-section data for 252 Swedish municipalities.

The results suggest that local polices, geographic/demographic variables, socio-economic 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 single-family dwellings, 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 (fixed) compensation from the material companies to the collection entrepreneurs 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 may be spatially cost-ineffective.

As in paper 1, the analysis also shows that the different actors within the existing waste management regime can affect the collection outcome in various ways. First, 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. However, in this case negative side effects of such fees, such as illegal waste disposal, must also be considered. Second, it also seems as if the material companies and local authorities could increase the collection by improving the information about the packaging collection to new immigrants.

Paper 3 Household Plastic Waste Collection in Swedish Municipalities: A Spatial-Econometric Approach (with Krister Sandberg, Patrik Söderholm and Christer Berglund)

Paper 3 also aims at finding the main determinants of collection rates of household plastic packaging waste in Swedish municipalities. This is done by the use of spatial econometric methods based on cross-sectional data for 282 Swedish municipalities in 2005. Besides attaining data for more municipalities and using more advanced econometric methods, this study also includes variables that address inter-municipality differences in waste management policies and recycling infrastructure in the analysis.

The empirical results suggest that the collection of household plastic packaging (in kg per capita) is positively related to the spatially weighted average of the collection per capita in neighboring municipalities. In other words, the probability that the collection of plastic collection is high increases if the neighboring municipalities also collect high degrees of the household plastic packaging. One explanation for this finding is that the plastic packaging waste sometimes is reported for a group of (normally two) municipalities. However, this result might also be due to, for instance: (a) recycling cooperation over municipalities' borders in regional waste companies; (b) municipalities and waste companies that copy-cat each others' policies and/or collection organizations; or (c) regional pattern in packaging design and usage.

This study clearly strengthens the result from papers 1 and 2, not the least in the sense that local policies are found to be important for recycling results. Just as the results in the paper 2, the results show that municipalities that employ weight-based waste management fees generally experience higher collection rates than those municipalities in which volume-based fees are used. The present study also indicates that the presence of curbside recycling and a high intensity of recycling drop-off stations, both measures that facilitate recycling efforts by creating the infrastructural and logistic mechanisms that enable people to translate their motivation into recycling action, provide important explanations for why some municipalities perform better than others.

Paper 3 also support the findings in paper 2 by finding that a number of important regional cost variables, such as distance to recycling industry, urbanization rate and population density, do not seem to have any significant impact on collection outcomes. A reasonable explanation for this is, again, that the (fixed) monetary compensations from the material companies to the collection entrepreneurs in Sweden vary depending on region and is typically higher in high-cost regions.

This study also shows that municipalities with a high share of newly arrived immigrants collect less plastic packaging than the average municipality, and once again this result does not hold for immigrants as a whole. Hence, overall the results suggest that policy variables rather than geographic/demographic and socio-economic factors are the major drivers behind packaging collection rates in Sweden.

Paper 4: Producer Responsibility for Paper Packaging: An Effective Supply Chain Management Policy? (Published in *Resources, Conservation & Recycling*, 2007)

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 first outlining a model for a cost-effective waste management scheme and comparing the model assumptions and results with empirical facts from a specific supply chain in the Swedish case (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. This 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.

6. Overall Conclusions and Implications

This thesis highlights a number of economic incentives issues in the design of packaging waste management systems, and it focuses in particular on the design and the (observed and self-reported) outcomes of the Swedish producer responsibility for household packaging materials. Below the overall findings from the four self-contained papers are summarized, and in a final sub-section we outline the most important policy implications.

6.1 Overall Findings

Economic theory concludes that the social benefit from household packaging recycling – environmental quality – is a public good and that therefore very few used packaging materials (without deposit-refund system) should be recycled. Still, we find that rather high degrees of the household packaging waste (without deposit-refunds) are recycled in Sweden. Why is this?

6.1.1 Why do Households Recycle?

The results show that *internal motivation* – endogenous *moral activation* – is essential in explaining household recycling decisions, especially when considering the main Swedish collection system with drop-off stations. Recycling outcomes are a positive function of: (a) the felt moral obligation for recycling; (b) the perceived positive external effects generated by recycling; and (c) the beliefs about to what extent other households recycle their packaging waste. The recycling outcome is also a negative function of the extent to which recycling is perceived as a public good by individuals. However, one of these variables stands out. The beliefs about others recycling activities seem to be the most important variable in explaining altruistic behavior in this case.

However, the thesis also points out that *external motivation* as well is very important for understanding individual recycling efforts. There are two different types of *economic incentives* that may matter for households recycling behavior. First, policies that create a *financial benefit* for recycling increase recycling rates. In this case – when deposit-refunds systems are excluded from the analysis – the financial incentives are given by (non-packaging) local waste fees. The results indicate that if the municipality introduces a variable fee for non-packaging waste then recycling increases (because using the recycling stations is without charge). The results also show that weight-based waste fees are much more effective in increasing recycling levels than are volume-based waste fees. Second, the results also show that household packaging recycling levels increase if the recycling infrastructure makes it easier for the household to recycle, hence lowering their *time cost* for recycling. In this thesis we find the following results that confirm that the opportunity cost of time matter for the individual: (a) municipalities that employ curbside recycling scheme for single-family dwellings face higher plastic packaging collection rates than municipalities that rely on the main collection system with recycling drop-off stations; (b) households living in multi-family dwellings with property-close collection state that they recycle more than people that do not have this opportunity; and (c) the household plastic packaging recycling outcome is a positive function of the number of recycling stations per capita in the municipality.

The results indicate that we also have a somewhat complex relationship between internal and external motivation. First, the results indicate that the positive relationship between moral motivations and recycling outcomes weakens if property-close collection – making household recycling efforts very convenient – is introduced. Second, the results indicate that it seems as if the moral norms are internalized social and legal norms, thus making it difficult to empirically distinguish between the different types of norms.

To summarize the above findings, the thesis concludes that there exists an important co-dependence between the norm-motivated man and the economic man. People do feel a moral obligation to participate in recycling schemes but it is only if the appropriate infrastructure and incentives have been introduced that this obligation will be translated into real action.

6.1.2 Is the Producer Responsibility Ordinance Cost-Effective?

The reader should be reminded that the aim of this thesis is not to answer if the existing household recycling policy in Sweden is socially optimal. However, one of the goals has been to investigate if the existing policy gives producers cost-effective incentives to economize with packaging and to analyze if the collection of packaging waste is performed in a spatially cost-effective manner. In other words, the aim was to analyze if it may be possible to fulfill the recycling targets by using less economic resources.

The results show that the latter of these requirements is not fulfilled. Thus, the collection of packaging is performed in a *spatially cost-ineffective* manner. An important explanation for this is that the (fixed) monetary compensations from the material companies to the collection entrepreneurs in Sweden vary depending on region and is typically higher in high-cost regions. This is partly because SEPA prescribe that the collection should be nationwide. The former criterion for cost-effectiveness is partly fulfilled. The results indicate that the existing Swedish producer responsibility ordinance provides cost-effective incentives to decrease the amount of packaging (output effect) and to stimulate the use of recycled packaging on the expense of virgin inputs (input substitution effect). However, the thesis also indicates that the existing ordinance does not appear to encourage enough of design for recyclability.

6.2 Policy Implications

6.2.1 How Can the Recycling Levels be Increased?

Given the political intention to increase household recycling, this thesis delivers good news for policy makers and the recycling industry: It should be possible to increase recycling rates further with the help of different policy measures. However, before commenting on how to achieve this, I want to emphasize that this thesis neither suggests that more recycling is the right thing to do, nor that less recycling is the right thing do.

The results in the thesis indicate that the authorities and the waste management actors could influence recycling efforts considerably if they are able to influence people's moral motivation to recycle through recycling campaigns. The findings also suggest that these campaigns should devote most effort in influencing individuals' perceptions about overall recycling participation. But, of course, the cost for these campaigns should also be considered.

In the former section we also noted that the economic incentives in the household recycling sector are determined by *local waste management policy* and *packaging collection management organization*, and the results points out that these – if properly designed – are quite effective in influencing recycling outcome. Local policy makers that want to increase households' recycling efforts in their municipality could introduce weight-based fees for (non-packaging) waste and/or be more accommodating towards the producer responsibility organization if they want to establish more recycling stations.

The recommendation for the producer responsibility organization is that they should increase the recycling convenience for the households. As was noted above the most effective measure in terms of increasing recycling should be to offer curbside recycling for single-family dwellings, and to offer property-close collection for multi-family dwellings. A maybe cheaper but not as effective measure could be to boost the geographical density of recycling stations. However, before introducing these policies and infrastructure measures, potential undesirable side effects should also be considered. For instance, weight-based waste fees give households incentives for illegal waste disposal and different systems could induce different transformation and transaction (private and social) costs for collection. These potential problems are however not investigated in detail in this thesis.

6.2.2 How Can a Cost-Effective Waste Management Scheme be Attained?

The findings that the collection of packaging waste is spatially cost-ineffective implies that society could be focusing more on the regional collection cost differences and thereby save resources while still collecting the same amount of packaging. Still, purely on the basis of this thesis it is difficult to outline strong policy recommendations. A move towards a more cost-effective collection scheme would have both pros and cons. For instance, the transaction costs for a more cost-effective collection are not considered in this thesis. These may be high and offset any cost savings, but they could also be kept low. The authorities need not necessarily set different collection targets for dense and sparsely populated regions, respectively, and then enforce each of these. It may be enough to reform the producer responsibility organizations compensation scheme and permit these economic incentives determine where collection will be made. We believe instead that one of the major drawbacks of a cost-effective scheme, in which spatial cost differences matter, may lie in the notion that there may exist a trade-off between the cost-effectiveness and the *legitimacy* of the recycling policy. If people as well as politicians feel committed to waste recycling because it is one way of contributing to public environmental goods, they may have a negative attitude towards a policy that encourages spatial variations in collection efforts. The above results also suggest that the policy makers should pay more attention to a policy that stimulates so-called design for recyclability in packaging materials, this issue being one of the weak spots of the Swedish producer responsibility legislation.

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Norms and Economic Motivation in Household Recycling: Evidence from Sweden[1]

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Abstract

This paper analyzes recycling efforts in Swedish households for four different packaging materials: paper, plastic, glass, and metal. The analysis builds on a model that integrates norm-motivated behaviour into neoclassical consumer theory by assuming that the individual has preferences for keeping a self-image as a morally responsible, and thus norm-compliant, person. A postal survey was sent out randomly to 2800 households in four different Swedish municipalities, and self-reported information on recycling efforts at the household level was analyzed in an ordered probit regression framework. The results indicate that property-close collection in multi-family dwelling houses induces higher recycling outcomes, and overall the presence of moral norms explains much of the variation across households. The latter indicates that information campaigns could be effective in increasing recycling efforts, and the results also imply that these campaigns could be made more effective if aimed at influencing individuals' beliefs about others recycling efforts. Moreover, the importance of moral norms in driving recycling efforts partly diminishes if improved collection infrastructure makes it easier for households to recycle. This may also reduce the effectiveness of information campaigns. The results also show that elderly Swedes report that they recycle more packaging material than younger citizens.

Key words: collection rates, recycling, packaging, Sweden, producer responsibility, waste management, ordered probit regression, social norms, moral norms.

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1. Introduction

Environmental policy largely requires people's active involvement, and many obligations are therefore expressed in household-related activities such as sorting of waste and the active purchase of ”green” products and services. In 1994 a producer responsibility ordinance for packaging was introduced in Sweden; this mandates households to sort out packaging waste from other waste, clean the waste, make use of the collection systems that producers provide, and finally sort different packaging materials – paper, plastic, glass, and metal – in different recycling bins (SFS 2006:1273). Household participation is thus mandatory, but in reality participation is almost never controlled and enforced and in practice it may therefore be easy to defect and free-ride on others' contributions. Nevertheless, official statistics show that households in Sweden recycle substantial amounts of packaging materials (SEPA, 2006).

What explains inter-household participation rates in packaging recycling schemes, and are there differences across the different materials? These are the overall research questions addressed in this paper, and in the analysis we pay particular attention to the role of both economic and norm-based motivation as well as the relationship between these motives. Our focus is motivated in part by the fact that waste management policies typically rely on a combination of economic and norm-based policy instruments, thus adhering both to personal moral responsibilities as well as providing the incentives that induce people to translate any felt obligation into recycling action.

In Sweden the ordinance requires that producers of packaging materials should collect used packaging from consumers, and they are therefore obliged to provide a suitable collection system (SFS 2006:1273). The producers have chosen to establish almost 6000 recycling stations (drop-off stations) where households are expected to leave their packaging waste (Funck, 2006). This means that on average about 1500 individuals “share” a station, and since Sweden is a quite sparsely populated country some households may be located far away from their nearest drop-off station. At the same time the local authorities have introduced new waste policies aiming at creating economic incentives for households to lower waste levels and stimulate recycling. For instance, almost all Swedish municipalities have abandoned the flat fee pricing policy for waste collection and introduced either volume-based fees or weight-based waste fees for single-family dwellings (e.g., Hage et al., 2007).[2]

[2] 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 (Villaägarna, 2006). In weight-based programs households pay a certain amount per kg unsorted waste, thus creating a direct incentive to increase recycling efforts at the household level.

Moreover, in some municipalities infrastructural measures have been undertaken to facilitate households' recycling efforts. For instance, some municipalities offer (for free or sometimes at a rather moderate cost) *curbside recycling* of packaging waste to single-family dwellings. In addition, many of the multi-family dwelling houses buy a related service, *property-close collection*, from recycling entrepreneurs.[3]

In spite of these new facilities and policies, the producer responsibility ordinance still burdens Swedish households, and they are not economically compensated for this work by the producers. The effectiveness of the reformed local waste policies has also been questioned. Empirical research shows, for instance, that volume-based waste pricing tends to be quite ineffective in increasing recycling levels (e.g., Sterner and Bartelings, 1999; Kinnaman and Fullerton, 2000; Jenkins et al., 2003; Hage and Söderholm, 2008). Also the use of weight-based schemes has been questioned on effectiveness grounds (e.g., Ackerman, 1997). This suggests that economic incentives alone cannot contribute to our understanding of recycling outcomes at the household level. Economically household recycling activities contribute to the production of public goods such as improved environmental quality, i.e., goods characterized by non-rivalry and non-excludability in consumption, and for this reason economic theory predicts that such voluntary contributions will be scarce (Bergstrom et al., 1986). This is the typical situation in a so-called social dilemma, i.e., the payoff to each individual of not contributing to the public good is higher than the payoff for voluntary public good provision, but yet overall all individuals receive a lower payoff if all choose to defect than if all contribute. Andreoni (1988) also showed that even in the presence of pure altruism, the contribution to public goods, and hence recycling, would be very small in large economies.

In the social psychology literature it is suggested that the presence of norms – informal rules requiring that one should act in a given way in a given situation – may provide an important reason for a departure from a social dilemma outcome (e.g., Biel and Thogersen, 2007). Thogersen (1996) shows that norm-based motivations are important in understanding of household recycling efforts (see also section 2). Still, Guagnano et al. (1995) conclude that the success in explaining households' contribution to recycling schemes is limited for any single scientific discipline. For this reason the present paper builds on recent developments in integrating norm-motivated behavior into neoclassical economic consumer theory, thus permitting an analysis of the interdependence between norms and economic motivation. The

[3] Some rough estimates suggest that about 45-50 percent of all multi-family dwellings in Sweden can benefit from property-close collection schemes (SEPA, 2006). Mattson et al. (2003) present how some of these schemes are designed and organized.

purpose of this study is to analyze the determinants of inter-household recycling of packaging materials in Sweden, and thus pay particular attention to the presence and the importance of social and moral norms as well as the role of incentive-based policies in the waste management field. The analysis is performed by employing data from a postal survey sent out to 2800 households in four Swedish municipalities. The self-reported data on recycling efforts at the household level are analyzed by using ordered probit regression techniques.

The paper proceeds as follows. In the following section a brief overview of previous research on norm-based motivation and its implication for this study are presented. Section 3 outlines the theoretical model of household recycling behavior used in this study. In the model – which is heavily based on Brekke et al. (2003) and Nyborg et al. (2006) – it is assumed that individuals have a preference for maintaining a self-image as morally responsible – and thus norm-compliant – persons. The survey design and variable definitions are discussed in section 4, and in section 5 the econometric specification of the model is presented. The results are presented and discussed in section 6. Finally, section 7 provides some concluding remarks and implications.

2. Previous Research on Norms and Recycling

As was mentioned above, the presence of both moral and social norms may provide important explanations to voluntary contribution to public goods. A moral norm implies that individuals sanction themselves, while a social norm is enforced by approval and/or disapproval from others. In practice, however, it can be hard empirically to make a clear distinction between these types of norms, especially since it may be asserted that any influence of social norms is mediated through internalized norms (e.g., Schwartz, 1977) (see also section 3).

Numerous studies from different scientific areas find that norms, attitudes and environmental concern are important factors in explaining recycling behavior. See, for example, Hornik et al. (1995), Schultz et al. (1995), and Thogersen (1996) for reviews of this research, as well as the more recent research efforts by Chan (1998), Barr et al. (2003), and Tonglet et al. (2004). The bulk of the recycling literature finds that internalized (moral) norms and attitudes are more important than social norms. However, Tucker (1999) and Barr et al. (2003) conclude that social norms are important in cases where the visibility of recycling behavior was high. Derksen and Gartrell (1993), Guagnano et al. (1995), Ölander and Thogersen (2005) find that the external conditions (e.g., recycling infrastructure) are important for moral recycling decisions, and the last of these studies report results that support the so-called

ABC model of pro-environmental behavior. The ABC model suggests that moral norms do not explain much of recycling behavior when external conditions either strongly support or when they only weakly support recycling behaviour, i.e., if it is very easy or very difficult to recycle, the role of moral norms is limited in explaining differences in recycling efforts across households. Hence, attitudes (A) and norms primarily determine recycling behavior (B) if the external conditions (C) are on an intermediate level.

The above suggests thus that both internal and external motives matter for explaining household recycling levels. Guagnano et al. (1995) support this notion and conclude that ”science and policy require a socioeconomic theory of behavior that incorporates both external conditions and internal processes,” (p. 700). During the last decades economists have tried to achieve just that. For instance, Andreoni (1990), Frey (1997), Brekke et al. (2003), Bruvoll and Nyborg (2004) and Nyborg et al. (2006) attempt to develop neoclassical utility theory by considering moral norms, while Holländer (1990), Lindbeck (1997), Nyborg and Rege (2003) and Rege (2004) do the same by considering social norms. However, there exist few empirical economic studies that employ this new strand of research in the waste management field. Survey results indicate that moral norms seem to be important for understanding recycling behavior in Norway (Bruvoll et al., 2002). Berglund (2006) finds that households' willingness to pay others for sorting the waste is negatively correlated with the existence of strong moral norms for recycling. The present study adds to this limited empirical research by addressing the role of both economic and norm-based motivation as well as the relationship between these rationales.

The above discussion also highlights that policy design could matter for norm activetion, or at least for the extent to which any existing norms will influence recycling behavior. For instance, social norms are triggered by polices that make the individual recycling effort visible to neighbours. Some policies – e.g., recycling infrastructure facilitating the sorting and transporting of waste – could strengthen the role of moral recycling norms. In addition, if moral norms are important it will probably be possible to influence recycling behavior through information campaigns, but these campaigns will presumably be less effective if such behavior is primarily explained by social norms (Nyborg et al., 2006). Social approval and disapproval come from real people, and may be more difficult to influence directly in information campaigns. Clearly, in order to provide appropriate policy recommendations it is crucial to gain a good understanding of household recycling behaviour, its underlying motivations and driving forces.

3. A Simple Model of a Norm-motivated Recycler

The recycler utility model that is presented in this section builds on a model for a morally-motivated green consumer developed by Nyborg et al. (2006),[4] and it is in turn heavily influenced by Schwartz's theory for altruistic behavior (Schwartz, 1970, 1973, 1977).[5] According to Schwartz, social norms regarding moral behavior could be adopted by each of us on a personal level and hence become personal moral norms. When this norm is internalized and activated, no external sanctions are necessary because breaking a moral norm will be sanctioned by oneself. Schwartz (1973, 1977) also stresses that it is not enough to have a personal moral norm to undertake a specific action. People could internalize norms, but may not necessarily act in accordance with them. Nyborg et al. (2006) provide a good explanation for this:

> "Our model is partial; it considers only one type of green consumer good, while there are a nearly unlimited number of other choices to make in everyday life. However, no-one is capable (cognitively or economically) of contributing to every public good in every possible way; there must be some division of labor in society. Hence, in practice, even individuals with a strong preference for considering themselves to be socially responsible will not feel an obligation to contribute to every good cause." (p.354)

Schwartz suggests that to influence behavior a specific norm must be *activated*, and to become activated *problem awareness* and *ascription of responsibility* are important. In the case of recycling, individuals must believe that the waste generated by households really harms the environment and that recycling thus will give rise to positive externalities (and affect others' welfare positively). The individual must also feel a personal responsibility to recycle; they should not believe that it is (solely) some other actors' responsibility to solve waste management problems. In line with Nyborg et al. (2006), we therefore assume that *beliefs about others' recycling effort* will guide people in deciding whether they have a personal responsibility for recycling or not. It seems plausible to assume that individuals who believe that other households take responsibility for recycling will conclude that they also have such a responsibility. Schultz (2002) refers to several studies concluding that there is a positive relationship between recycling and respondents' beliefs about others recycling.

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[4] Their model is in turn a simplified version of a model developed by Brekke et al (2003).

[5] For applications of this theory in the analysis of pro-environmental behavior, see Hopper and Nielsen (1991), Thogersen (1999), Stern et al. (1999), Ek (2005) and Nyborg et al. (2006).

In the remainder of this section we present an economic consumer model that attempts to acknowledge the above findings from the social psychology literature. We assume that society comprises n identical individuals and that the individual utility function is given by:

$$U = u(c, l, E, S) \qquad (1)$$

where c represents the individual's consumption of private goods, l is leisure, and E is environmental quality. The individual also derives utility from maintaining a self-image (S) as a morally responsible – and thus norm-compliant – person. We assume here that recycling is morally superior, and recycling will therefore yield a self-image improvement. The utility function is assumed to be continuous, quasi-concave, and increasing in all variables. To simplify, the individual's labor supply and total income are assumed to be exogenous. This gives the following time constraint for the individual:

$$T = l + a \qquad (2)$$

Hence, the total available amount of time (T) can be allocated to either leisure (l) or to recycling activities (a). This implies that if the individual devotes time to recycling activities she must give up leisure time.

In this simplified model, E is assumed to be a pure public good, and it will depend on the individual's own contribution to an improved environment, e, and the contribution of all other n-1 individuals, E_{-1}. From the perspective of the individual, the latter component is exogenous, something which gives us the following function:

$$E = E_{-1} + e(r) \qquad (3)$$

where e is assumed to be a function of recycling levels at the household level, r (and where e_r is positive). The amount of recycled packaging from households, r, is in turn determined by the time devoted to recycling activities, a, and a collection efficiency parameter, θ. The latter variable reflects technical and institutional differences between different collection systems and it is assumed to be exogenous from the perspective of the individual household:

$$r = r(a, \theta) \qquad (4)$$

where $r(0, \theta) = 0$, and the corresponding derivatives are $r_a > 0$, $r_\theta > 0$, $r_{aa} < 0$, and $r_{\theta\theta} < 0$. It should be clear that although increases in E lead to a higher utility, the public good nature of this good means that the individual's incentive to contribute is likely to be more or less negligible. Thus, such incentives must enter the utility function through other channels.

We assume that by participating in the production of the public good E, individuals will experience an improved self-image, S. Following Nyborg et al. (2006), the change in self-image from recycling is reflected in the personal responsibility the individual feels for the issue. The more willing the individual is to acknowledge his/her own *personal* responsibility to choose "green", the higher is S. However, some individuals may be genuinely uncertain about whether they ought to take the responsibility to buy "green", especially if there does not exist any formal sharing of responsibility through, for instance, laws and regulations. In addition, as was noted above there are many good causes to support and no-one can be expected to contribute to all of these. Following Schultz (2002), among others, Nyborg et al. (2006) suggest that:

> "A natural thing to do, then, is to look around to see who carries this responsibility in practice. If she observes that it is common for people like her to take responsibility (in our case, purchase the green good), it is more likely that she will conclude that she does have some responsibility." (p. 354).

We therefore assume that beliefs about others' behavior have a positive impact on S. Specifically, R is defined as the share of the total population recycling household waste.

Moreover, the impact on S of recycling is also assumed to be affected by the positive environmental externalities arising from the individual's choice (and thus affecting other households), *ne*. If the individual chooses to recycle she will confer an environmental benefit on both itself and on all other households. For our purposes, *ne* represents the individual's *beliefs* about the total positive external effects her recycling behavior gives rise to. The moral – self-image – relevance of purchasing "green" depends positively on *ne*. In other words, individuals who do not believe that recycling activities help improve the environment or (alternatively) that any improvements achieved through recycling are of little value, will not experience any self-image improvements by recycling. In sum, S can be expressed as:

$$S = s(ne, R) \qquad (5)$$

where $s(..)$ is a continuously differentiable function, with the first derivatives, $s_e > 0$, $s_\alpha > 0$.

By maximizing the individual utility in Eq. (1) subject to Eqs. (2-5) we obtain the following first order condition:

$$u_l = e_r r_a (u_e + n u_s s_e) \qquad (6)$$

Eq. (6) implies that, in optimum, the marginal cost of lost leisure should be equal to the marginal utility of recycling activities. The personal benefit from recycling activities consists of two parts; the personal environmental benefit assumed to be virtually zero and the personal benefit from an improved self-image.

Individuals that have a single packaging that they need to get rid of will face two options. They could choose to throw the packaging in the waste bin (option 0) or they could choose to drop the packaging in a recycle station (option 1). The latter option is assumed to give higher self-image but this will only come at a cost. As was noted above, the recycler should clean, sort, store, and transport the packaging, and these efforts result in lost leisure. Thus, the individual will recycle the single packaging if and only if:

$$u(c^0, l^0, E^0, S^0) < u(c^0, l^0 - a, E^1, S^1) \qquad (7)$$

The self-image effect of recycling will be more pronounced the higher the perceived positive environmental externalities are and the more willing the individual is to take personal responsibility, and these self-image effects of recycling packaging waste are also an increasing function of the (perceived) share of other households' choosing to do the same.

So far we have solely considered the case of purely voluntary contributions to the public good. The model employed rests on the assumption that violations of moral norms are sanctioned only, internally by the recycler, i.e., the individual will feel guilt and have a bad conscience if she does not contribute. However, in the case of the Swedish producer responsibility, the ordinance actually mandates households to recycle (although it is not regularly controlled and violations are seldom enforced). This means that in practice legal norms may also play a role in influencing household recycling efforts.

Moreover, there is also a growing literature on how social norms, i.e., norms enforced by sanctions from others, affect individuals' contribution to public goods (e.g., Biel and Thogersen, 2007). Holländer (1990) suggests that the individual will feel approval (or disapproval) from others if she conforms (or does not conform) to the social norm. As was noted above, from an empirical perspective it may be difficult to distinguish between moral and social norms. Schwartz (1977) also suggests that personal moral norms are created from social interaction with others. While our theoretical model focuses solely on the role of moral norms, we also devote attention to both social and legal norms in the empirical investigation.

Guagnano et al. (1995) outline the so-called ABC model. In this model not only the absolute size of moral attitudes (A) and external conditions (C) explain altruistic behavior (B), but so does the relationship between A and C. Marginal improvements in one of these two variables may not be effective if the other variable weakly or strongly supports recycling. For instance, this implies that if external conditions make it very easy to recycle almost all will recycle, and moral attitudes toward recycling will not be important for the recycling outcome. The same will be true if the external conditions make it very hard to recycle. In this case almost none will recycle irrespective of moral norms. This implies that marginal improvements in moral attitudes will have the biggest impact on recycling effort when the external conditions for recycling are on an intermediate level. The ABC model should not to be regarded as a substitute for the neo-classical model that incorporates norms, rather it should be seen as a complement that expand our tool box when analyzing recycling behavior.

Finally, the above distinctions between different types of norms could have important policy implications. If norms are mainly legal, the government could increase recycling levels directly by just increasing the legal norm and inform citizens about the change. If the moral norm is formed endogenously by the individual it is still possible for the authorities to affect the perceptions of the moral norm and hence increase recycling, but now rather by informing about the importance of recycling and hence affect the individuals' perception of the morally ideal recycling level. Information about others' (positive) contributions and information aimed at raising overall problem awareness can also be effective. The ABC model also predicts that recycling campaigns would be most effective in influencing recycling rates if the recycling infrastructure is on an intermediate level. The existence of social and descriptive norms often creates the possibility for multiple equilibriums in the contribution to public goods, one when nobody contributes and one when everyone contributes (e.g., Rege, 2004). In this case, public policy – e.g. economic incentives – could move the society from the former case to the latter one.

4. Survey Design Issues and Variable Definitions

4.1 Survey Reponses and Sample Selection Bias

In May 2006, 2800 questionnaires (see Appendix A) were sent out to randomly drawn household members in four different Swedish municipalities (Piteå, Huddinge, Växjö and Gothenburg). The postal survey formed part of a multidisciplinary research program on environmental sustainability and household activities (see www.sharpprogram.se), mainly financed by the Swedish Environmental Protection Agency (SEPA). Overall the survey collected information about how Swedish households perceive different household activities that can be undertaken to improve the environment quality (recycling and transport choice), as well as the households opinion about a set of policy instruments that can be implemented to stimulate these activities.

The response rate was quite low, 827 respondents (30 percent) returned the postal survey. In addition, about 5 percent of the returned questionnaires had to be excluded in the empirical investigation because some respondents did not answer all questions.[6] The main reason for the low response rate is probably that the questionnaire focused on a large number of different household activities and included questions about related policy instruments. While this makes the study rather unique, enabling, for example, comparative analyses across different household activities as well as investigations of the links between household characteristics, values and attitudes and specific policy instruments, it also implies that the survey was quite demanding to complete for the respondents.

The presence of non-responses can lead to sample selection bias. For example, it is reasonable to expect that respondents that are worried about pollution and resource depletion and/or respondents that believe that recycling efforts and taxes on transports are too high would be more likely to choose to participate in this type of study. In order to evaluate if the respondents are reasonably representative, some of their socio-economic characteristics were compared with the four different populations from which they were drawn (Table 1). Overall Table 1 shows that the different socio-economic characteristics are fairly equal across the two groups. However, it is worth mentioning that respondents between 20 and 64 years are overrepresented in the municipality of Huddinge, while respondents with a university degree are overrepresented in Gothenburg.

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[6] In order to at least in part reduce the problem with missing values some adjustments of the data have been done. For instance, missing data on children in the household have been interpreted as if these households lack children.

Table 1: Socio-Economic Characteristics of the Four Sub-samples

	Piteå		**Gothenburg**		**Växjö**		**Huddinge**	
	Sample	Population	Sample	Population	Sample	Population	Sample	Population
Share of men (%)	52.1	50.0	46.6	49.3	52.4	49.7	46.9	49.7
20-64 years (%)*	78.0	77.0	82.0	81.1	80.0	78.9	***77.3***	***84.0***
Above 65* years (%)	22.0	23.0	18.0	18.9	20.0	21.1	***22.7***	***16.0***
University education for age 25-64 (%)	30.7	30.0	***54.6***	***45.0***	43.4	41	31.4	35.0

* People younger than 20 years are not included in the study.

Note: Characteristics that are over- or underrepresented in the sample using a 95 % confidence interval are written in italic and bold.

Sources: SCB (2006, 2007).

Even if the socio-economic characteristics do not give use reason to suspect the presence of strong sample selection bias, there are still reasons to believe that people with a strong pro-environmental attitude are more likely to participate in this type of study. To check for this problem a short telephone interview, including some of the questions from the original survey, was conducted with 200 of the non-respondents. Most notably, overall this group expressed a significantly lower degree of problem awareness, i.e., they were not as convinced as the responding group that recycling generates environmental benefits.[7]

Furthermore, in the questionnaire the NEP scale (Dunlap et al., 2000)[8] was used for capturing the respondents' environmental orientation. If respondents report significantly higher NEP scores than the total population, we could see this as an indication of sample selection bias. Unfortunately, true NEP scores for the entire population are not available. However, it is possible to compare the respondents NEP scores with the NEP scores from

[7] When confronted with the statement: "Household waste that is not recycled is a very serious problem so immediate action is needed," the responding group reported an average score of 4.26 on the seven grade scale, while the corresponding score for the non-responding group was 3.33. These differences are statistically significant from zero.

[8] In order to identify differences in environmental attitudes, respondents were asked to indicate to what extent they agreed or disagreed to 15 statements known as the modified New Ecological Paradigm (NEP) scale. The modified NEP scale aims to measure five facets of environmental orientation; limits to growth, anti-anthropocentrism, the fragility of the balance of nature, rejection of the idea that humans are exempt from the constraints of nature, and the possibility of an eco-crisis or ecological catastrophe. The response alternatives range between 1 up to 5. Statements that reject the NEP-paradigm are reversed when the total (and average) NEP scores are calculated. Hence, high average scores imply higher pro-environmental orientation.

other empirical studies (with higher response rates). In this study the mean NEP score is 45.4. This value is significantly lower than those reported in many other studies. For instance: Cooper et al. (2004) find that environmental university students had an average NEP score on 58.2 (91 percent response rate). Clark et al. (2003)[9] find that participants in a green electricity program had an average NEP score of 56.8 (95 percent response rate) and that non-participants reported an average NEP score of 50.8 (67 percent response rate). Kotchen and Reiling (2000) find an average NEP score of just above 54 (63.1 percent response rate). Finally, Ek (2005) finds that respondents from the same Swedish municipalities that are investigated in this paper reported an average NEP score of 54.5 (response rate 32 percent). Hence, overall it seems as if selection bias due to pro-environmental preferences in this study may be present but when comparing with other similar studies this problem does not appear to be particularly alarming.

4.2 The Survey Design and Variable Definitions

The dependent variable in this study shows to what extent the households recycle packaging waste (paper, plastic, glass and metal) without refund payment.[10] In the survey, the households had five alternatives to chose between, ranging from "recycle nothing" up to "recycle everything". The independent variables that are included in the econometric analysis are – based on the theoretical discussion in section 3 – divided into four different categories: (a) variables that affect the opportunity cost for recycling (neoclassical variables); (b) factors influencing the extent to witch recycling could give rise to self-image benefits and/or social acceptance (norm variables); (c) socio-economic variables; and (d) separate dummy variables for the different municipalities.

In the case of recycling, the neoclassical variables could in turn be divided into two groups, those that: (a) relate to the time cost of recycling; and (b) measure the presence of financial incentives for household recycling. It is reasonable to believe that policies and situational factors that decrease the time cost of recycling will stimulate household recycling. A number of different questions were included in this study to capture individual differences in time costs. These include distance to recycling stations, access to a car, as well as access to recycling within the borders of the property (i.e., property close collection or curbside

[9] The Clark et al. (2003) study only used ten statements so the average NEP reported in their study has been multiplied with 1.5 for comparison purposes.

[10] In Sweden, recycling of PET-bottles, aluminum cans and a great share of the glass bottles for soft drinks and beer are organized through a deposit-refund system, this part of the packaging waste stream is not analyzed in this study.

recycling).[11] Only the last variable had a statistically significant impact on recycling outcomes so the other two were excluded from the final empirical model. As was mentioned in section 1, all Swedish municipalities have abandoned the flat fees for household waste collection and most rely on some form of volume based waste fees. The remaining ones – 9 percent – have introduced weight-based waste fees. This implies that households – primarily those in single-family dwellings – could lower their waste management fees by increasing recycling efforts (because there are no fees when households use the recycling stations). However, it is still the case that most households in multi-family dwellings will pay for waste collection in the form of a fixed component of the total monthly rent, thus facing no economic incentives to increase recycling levels. None of the investigated municipalities had introduced weight-based waste fees, so in this paper we could only test the null hypothesis that households facing volume-based fees are equally good at recycling their waste as are those that only pay fixed fees.

In section 3 we argued that household recycling is assumed to be positively correlated with increases in the self-image. The perception about the individual responsibility for recycling was measured by two types of questions included in the questionnaire. First, the respondents were confronted with the following statement: "I recognize a moral obligation to recycle". The responses were measured on a seven-point scale with the end-points "disagree entirely" and "agree entirely". Second, individuals' perception about the extent of others' recycling efforts was based on the responses to the following questions: "How much of their household waste do you think that other households in your municipality recycles?". The respondents were then asked to choose between one of five alternatives for each packaging material, ranging from "recycle nothing" up to "recycle everything".

In order to measure the individuals' beliefs about the size of the negative external effect from not recycling, the respondents where confronted with five different statements. These statements and the distributions of the answers are presented in Appendix B. In the econometric analysis, the responses to these statements were used to calculate a "perceived negative externality index" ranging from 1 to 7. We have noted above that recycling implies the provision of a public good in the form of improved environmental quality and it is important that households perceive that increasing their own recycling efforts actually does

[11] Many multi-family dwellings have installed central sorting houses or rooms within the borders of the property. In these facilities the people living in the apartments can leave their packaging waste in different bins for different materials. The term *property-close collection* is used for this form of collection. Moreover, in about 20 municipalities (7 percent) the authorities have organized *curbside collection* for all single-family dwellings. Of course, both these systems lower the time and transport costs for household that choose to recycle. None of the investigated municipalities had introduced curbside recycling, but a substantial share of the respondents live in apartments that can benefit from property-close collection.

contribute to that good. For this reason the households were confronted with the following statement: "It is no idea that I recycle because it will not lead to improved environmental quality". Also here their responses were measured on a seven-point scale with the end-points "disagree entirely" and "agree entirely".

Individual perceptions about the legal norm were measured by the following statement: "The government and the municipality expect me to recycle" (employing the same seven-point scale as above). Some studies (e.g., Tucker, 1999; Barr et al., 2003) find that the presence of social norms could help in explaining recycling behavior if the visibility of the recycling behavior is high. However, in Sweden the recycler is probably rather anonymous, at least for the type of systems that are investigated in this study. When the recycler goes to a recycling station she is not particularly visible to the neighbors, and the same is often valid if she uses a recycling room in a multi-family dwelling house.[12] Considering this, it does not seem very likely that social norms will be a major driver of recycling behaviour in this investigation. Nevertheless, in order to test for this the respondents were confronted with the following statement: "Important persons close to me want me to recycle". The responses were measured on a seven-point scale with the end-points "disagree entirely" and "agree entirely".

On economic grounds there exist no reason to suspect a specific relationship between education and recycling behavior. However, many U.S. studies find that recycling efforts tend to be positively correlated with education (e.g., Schultz et al., 1995; Callan and Thomas, 1997; Jenkins et al., 2003). On the other hand, Hage and Söderholm (2008) find a negative relationship between education and Swedish household plastic packaging recycling. In the present investigation we test for the impact of education level (see Table 2).

Previous research also indicates that the demand for environmental improvements is a "necessary" good (e.g., Kriström and Riera, 1996; Hökby and Söderqvist, 2003). This indicates that households with low incomes allocate relatively more resources to environmental goods than do households with high incomes. Hence, this implies that low income households will recycle higher degrees of the packaging waste. Furthermore, in contrast to the public goods investigated in the above studies,[13] recycling is a time consuming activity for the households. This means that the individual recycling levels will also be influenced by their opportunity cost of time. Clearly, the opportunity cost for recycling will increase with income.

[12] It is possible that curbside recycling programs in single-family dwelling areas create a more visible recycling behavior and hence generate favorable conditions for social norms to be important. However, these systems are not used in the municipalities analyzed in this paper.

[13] Kriström and Riera (1996) analyzed water quality, wetlands, parks and forest preservation while Hökby and Söderqvist (2003) analyzed water quality.

Again, this implies that low income households should recycle higher degrees of packaging. Consequently, this suggests that we have two effects both pointing in the direction that we should have a negative relationship between income and recycling efforts. However, some empirical studies find a positive relationship between income and recycling rates in developed countries (e.g., Callan and Thomas, 1997; Berglund and Söderholm, 2003), but these do typically not control for environmental attitudes.

Households living in single-family dwellings normally have more space to store used packaging. This suggests that it is reasonable to expect that individuals living in single-family dwellings should recycle more than individuals in multi-family dwellings. Hage and Söderholm (2008) also report such a relationship for Swedish plastic packaging recycling in 2002. However, they did not have data on curbside collection (benefiting recycling activities in single-family dwellings) and property close collection (benefiting recycling activities in multi-family dwellings). This result also disappears when Hage et al. (2007) are able to extend the data material and control for single-family dwellings with access to curbside recycling for Swedish plastic packaging collection in 2005. Jenkins et al. (2003) find the same result as Hage and Söderholm (2008) in the cases of U.S. plastic bottles and yard waste (but not for newspaper, glass bottles and aluminum) using individual data and information about curbside recycling and waste fees.

Table 2 summarizes the variables used in the empirical estimation and presents some descriptive statistics. In this study we also tested whether children in the household (dummy variable) and the households' environmental attitudes (according to the NEP scale) influenced self-reported recycling efforts. However, none of these variables had a statistically significant impact on recycling outcomes so they were both excluded from the empirical model.

5. Econometric Specifications

In this study the dependent variable has an ordinal scale. We know that recycling outcome 2 implies more recycling than recycling outcome 1, and that recycling outcome 1 means more recycling than recycling outcome 0. However, the difference between 2 and 1 may differ from that between 1 and 0, because they simply indicate a ranking. This means that linear regression techniques are not appropriate in this case and an ordered response model should instead be used (Greene, 1997). For this reason an ordered probit model is used here.[14]

[14] A common alternative to the ordered probit model is the ordered logit model. According to Greene (1997), choosing between these two models is a trivial choice, and appears to make no difference in empirical work.

Table 2: Variables Included in the Econometric Analysis

Variables	Coding	Mean	Std. dev	Min	Max
Dependent Variables					
Paper recycling	0 for none up to 4 for everything	3.26	1.29	0	4
Plastic recycling	0 for none up to 4 for everything	3.27	1.30	0	4
Glass recycling	0 for none up to 4 for everything	3.41	1.23	0	4
Metal recycling	0 for none up to 4 for everything	3.49	1.09	0	4
Neoclassical Variables					
Property-close collection	1 if yes, and 0 if no	0.34	0.47	0	1
Volume-based waste fees	1 if yes, and 0 if no	0.51	0.50	0	1
Norm Variables					
Moral obligation	1 for disagree entirely, and 7 for agree entirely	5.51	1.67	1	7
Others recycling of paper	1 for none up to 5 for everything	3.45	0.97	1	5
Others recycling of plastic	1 for none up to 5 for everything	3.39	0.94	1	5
Others recycling of glass	1 for none up to 5 for everything	3.54	0.97	1	5
Others recycling of metal	1 for none up to 5 for everything	3.59	0.95	1	5
Negative externality	Index based on responses presented in Appendix B	4.65	1.46	1	7
No idea	1 for disagree entirely, and 7 for agree entirely	2.39	1.55	1	7
ABC	Constructed by multiplying the scores for the moral norm with the property-close collection dummy	1.88	2.79	0	7
Legal norm	1 for disagree entirely, and 7 for agree entirely	5.54	1.52	1	7
Social norm	1 for disagree entirely, and 7 for agree entirely	3.78	1.97	1	7
Social norm for property-close collection	Constructed by multiplying the scores for the social norm with the property-close collection dummy	1.33	2.18	0	7
Socio-Economic Variables					
Gender	1 if male and 0 if female	0.50	0.50	0	1
Age	Age in years	49.58	14.53	19	75
Education	1 if elementary school; 2 if senior high school; and 3 if university education	2.12	0.77	1	3
Income	1 if below SEK 10 000 per month, up to 11 for above SEK 100 000*	4.09	2.07	1	11
Multi-family dwelling	1 if multi-family dwelling and 0 if single-family dwelling	0.43	0.50	0	1
Municipalities Dummies					
Piteå	1 if Piteå, 0 if other municipality	0.31	0.46	0	1
Gothenburg	1 if Gothenburg, 0 if other municipality	0.23	0.42	0	1
Växjö	1 if Växjö, 0 if other municipality	0.20	0.40	0	1
Huddinge	1 if Huddinge, 0 if other municipality	0.25	0.43	0	1

* The average exchange rate for USD 1 was SEK 7.30 as of May 2006.
Note: Variables that were excluded in the final empirical analysis are not presented in this table.

As was noted in section 4, a respondent i could choose between five different recycling outcomes y for four different packaging materials j. The ordered probit regression will thus estimate the following relationship:

$$y_{ij}^{*} = \beta_j' \; x_i + \varepsilon_{ji} \tag{8}$$

where y^* is the unobserved willingness to recycle. The vector x_i contains the neo-classical variables, the self-image variables, the socio-economic variables, and the municipality dummy variables. β is a vector of coefficients that are estimated by means of maximum likelihood methods. The error term ε_{ij} is assumed to be normally distributed, and we normalize the mean of ε to zero and the variance to 1. The probabilities P for each recycling outcome are:

$$\begin{aligned}
&P(y=0) = \Phi(-\beta' x),\\
&P(y=1) = \Phi(\mu_1 - \beta' x) - \Phi(-\beta' x),\\
&P(y=2) = \Phi(\mu_2 - \beta' x) - \Phi(\mu_1 - \beta' x),\\
&P(y=3) = \Phi(\mu_3 - \beta' x) - \Phi(\mu_2 - \beta' x),\\
&P(y=4) = 1 - \Phi(\mu_3 - \beta' x).
\end{aligned} \tag{9}$$

where μ is an unknown parameter that is estimated jointly with β. These estimations are performed by using the software Limdep 8.0.

A potential problem using a five-grade scale is that the intermediate alternatives (i.e., outcomes 1-3) could mean different things for different respondents. In order to test if this is a problem in this analysis; the data for these three categories was aggregated into one intermediate category. In this alternative specification the dependent variable has three recycling outcomes: nothing, intermediate and everything. The results from this alternative specification are briefly discussed in the next section and the parameter estimates and the corresponding marginal effects are presented in Appendix C.

6. Empirical Results and Discussion

In reporting about their recycling efforts, households could choose between five options, ranging from “recycle nothing” (alternative 0) up to “recycle everything” (alternative 4). The percentage distributions of the reported answers are presented in Table 3. These results are consistent with previous observations that Swedes generally participate in recycling programs. Well above two thirds of the respondents report that they recycle all of their packaging waste, and in particular in the cases of metal and glass.

Table 3: Self-reported Recycling Efforts (Percentage Distribution)

Packaging materials	0 Recycle nothing	1	2	3	4 Recycle everything	Total
Paper	9.9	2.3	6.9	13.7	67.2	100
Plastic	10.1	3.0	5.8	12.5	68.6	100
Glass	8.4	2.5	5.3	7.8	76.0	100
Metal	6.0	2.0	5.6	9.3	77.1	100

The parameter results of the econometric estimation for the ordered probit recycling models are presented in Table 4.[15] When interpreting these results we should consider the nature of an ordered probit regression. The *size* of the coefficients will be a probability and it will therefore not provide a good indicator of the magnitude of the effect on recycling efforts in the case of a change in any of the independent variables. However, the *signs* will have an important economic interpretation. For instance, when the estimated coefficients have a positive sign this implies that an increase in the corresponding independent variable (x_i) in question will increase the probability for the individual to recycle everything ($y = 4$) and that the probability to recycle nothing ($y = 0$) will decrease. On the other hand, the impacts on the intermediate options (i.e., $y = 1$, 2 and 3) are ambiguous. Of course, a negative coefficient will imply the opposite relationship (Greene, 1997).

In order to find out the magnitudes of the impacts as well as to gain information about the intermediate answers, the so-called marginal effects are calculated. These are presented for the statistically significant variables in Tables 5a-d. The marginal effects of the continuous variables are partial derivatives of the probability function evaluated at the sample mean values of the independent variables. Hence, the marginal effect could be interpreted as the marginal change in the probability of each recycling outcome if there is a unit increase in the investigated independent variable. When analyzing these variables it is important to note that different variables could have different scales (see Table 2). In the case of dummy variables, the marginal effects are calculated by comparing the probabilities that result when the dummy variable takes its two different values (0 and 1) and all other independent values are held at their mean values. So in this case, the marginal effect could be interpreted as the marginal change in the probability of each recycling outcome if the dummy goes from “off” (0) to “on” (1).

[15] The corresponding results for the alternative ordered probit specification estimation are presented in Appendix C. Overall these results show few differences compared to the estimations presented in this section. Thus, these results give no reason to question the robustness of the results presented in Table 4.

Table 4: Parameter Estimates for the Ordered Probit Model

Variables	Expected sign	Paper	Plastic	Glass	Metal
Constant	?	-1.392***	-2.446***	-2.330***	-2.577***
		(-2.94)	(-5.04)	(-4.30)	(-4.84)
Neoclassical Variables					
Property-close collection	+	0.891**	0.165	1.331***	1.061***
		(2.44)	(0.44)	(3.27)	(2.67)
Volume-based waste fees	+	0.403**	0.195	0.262	0.307
		(2.36)	(1.15)	(1.37)	(1.62)
Norm Variables					
Moral obligation	+	0.145***	0.100**	0.179***	0.170***
		(3.30)	(2.20)	(3.70)	(3.54)
Others' recycling	+	0.475***	0.556***	0.443***	0.463***
		(8.02)	(9.34)	(6.98)	(7.47)
Negative externality	+	0.111***	0.169***	0.143***	0.163***
		(2.77)	(4.08)	(3.07)	(3.59)
No idea	-	-0.095***	-0.147***	-0.095**	-0.104***
		(-2.59)	(-3.92)	(-2.30)	(-2.56)
ABC	-	-0.110*	0.058	-0.145**	-0.048
		(-1.66)	(0.86)	(-1.98)	(-0.68)
Legal norm	+	-0.013	-0.004	0.034	-0.010
		(-0.34)	(-0.11)	(0.83)	(-0.24)
Social norm	+	-0.022	-0.013	-0.009	0.031
		(-0.63)	(-0.36)	(-0.22)	(0.74)
Social norm for property-close collection	+	0.026	-0.057	-0.039	-0.086
		(0.44)	(-0.94)	(-0.60)	(-1.31)
Socio-Economic Variables					
Gender	?	-0.017	0.088	0.081	0.029
		(-0.16)	(0.84)	(0.68)	(0.25)
Age	?	0.014***	0.019***	0.020***	0.022***
		(3.61)	(4.93)	(4.66)	(5.16)
Education	?	-0.158**	0.006	0.004	0.126
		(-2.12)	(0.08)	(0.04)	(1.52)
Income	-	-0.003	-0.013	-0.016	-0.028
		(-0.11)	(-0.52)	(-0.57)	(-0.99)
Multi-family dwellings	-	0.071	-0.052	-0.242	-0.401*
		(0.37)	(-0.27)	(-1.13)	(-1.91)
Municipality Dummies					
Piteå	?	0.061	0.318**	0.255	0.132
		(0.40)	(2.20)	(1.47)	(0.82)
Gothenburg	?	-0.558***	0.674***	-0.445***	0.382**
		(-3.86)	(4.31)	(-2.82)	(2.26)
Växjö	?	-0.092	-0.077	0.093	-0.268*
		(-0.59)	(-0.53)	(0.53)	(-1.68)
Number of observations		689	692	695	689
Log likelihood		-599	-588	-472	-470
Restricted log likelihood		-723	-713	-592	-579
Chi squared		247***	249***	241***	218***

Note: *t*-statistics are given in brackets. *, **, and *** indicate statistical significance at the ten, five, and one percent level.

Table 5a: Marginal Effects for Paper Packaging Model

Variables	0 (Recycle nothing)	1	2	3	4 (Recycle everything)
Neoclassical variables					
Property-close collection	-0.067***	-0.024***	-0.070	-0.115**	0.276***
Volume-based waste fees	-0.037***	-0.013*	-0.036	-0.053	0.138*
Norm variables					
Moral obligation	-0.013***	-0.005***	-0.013	-0.019	0.050***
Others' recycling	-0.043***	-0.015***	-0.042	-0.064	0.164***
Negative externality	-0.010***	-0.004***	-0.010	-0.015	0.039**
No idea	0.008**	0.003***	0.008	0.013	-0.033**
ABC	0.010*	0.004*	0.010	0.015	-0.038*
Socio-Economic Variables					
Age	-0.001***	-0.001***	-0.001	-0.002	0.005***
Education	0.014**	0.005**	0.014	0.021	-0.055**

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

Table 5b: Marginal Effects for Plastic Packaging Model

Variables	0 (Recycle nothing)	1	2	3	4 (Recycle everything)
Neoclassical variables					
Property-close collection	-0.015*	-0.006	-0.012	-0.021	0.054
Volume-based waste fees	-0.019*	-0.008	-0.014	-0.025	0.066
Norm variables					
Moral obligation	-0.010**	-0.004**	-0.007	-0.013	0.033**
Others' recycling	-0.053***	-0.021***	-0.042	-0.070	0.186***
Negative externality	-0.016***	-0.006***	-0.013	-0.021	0.056***
No idea	0.014***	0.006***	0.011	0.019	-0.049***
ABC	-0.006	-0.002	-0.004	-0.007	0.019
Socio-Economic Variables					
Age	-0.002***	-0.001***	-0.001	-0.002	0.006***
Education	-0.001	-0.000	-0.000	-0.001	0.002

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

Table 5c: Marginal Effects for Glass Packaging Model

Variables	0 (Recycle nothing)	1	2	3	4 (Recycle everything)
Neoclassical variables					
Property-close collection	-0.068*	-0.028	-0.072	-0.101	0.270***
Volume-based waste fees	-0.167	-0.007	-0.018	-0.024	0.066
Norm variables					
Moral obligation	-0.011***	-0.005	-0.012	-0.016	0.045***
Others' recycling	-0.028***	-0.012	-0.030	-0.041	0.111***
Negative externality	-0.009***	-0.004	-0.010	-0.013	0.036**
No idea	0.006**	0.003	0.006	0.009	-0.024*
ABC	0.009**	0.004	0.010	0.013	-0.036***
Socio-Economic Variables					
Age	-0.001***	-0.001	-0.001	-0.002	0.005***
Education	-0.000	-0.000	-0.000	-0.001	0.001

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

Table 5d: Marginal Effects for Metal Packaging

Variables	0 (Recycle nothing)	1	2	3	4 (Recycle everything)
Neoclassical variables					
Property-close collection	-0.042***	-0.017***	-0.064	-0.108	0.232***
Volume-based waste fees	-0.015*	-0.006	-0.023	-0.036	0.078
Norm variables					
Moral obligation	-0.008***	-0.003	-0.012	-0.012	0.044***
Others' recycling	-0.023***	0.009***	-0.034	-0.054	0.120***
Negative externality	-0.008***	-0.003**	-0.012	-0.019	0.042***
No idea	0.005**	0.002***	0.008	0.012	-0.027
ABC	0.002	0.001	0.004	0.006	-0.013
Socio-Economic Variables					
Age	-0.001***	-0.001***	-0.002	-0.003	0.006***
Education	-0.006	-0.002	-0.009	-0.015	0.032

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

Before discussing the parameter estimates for each variable a short comment on overall model performance is useful. The chi-square for each model is (at least) above 218, indicating that for all four models the hypothesis that all coefficients equal zero can be rejected at the one percent significance level.

The coefficients representing the *neoclassical variables* show rather expected results. In particular, the coefficient for *property-close* collection is positive as for all four packaging material and statistically significant at the one percent level for glass and metal and at the five percent level for paper. This highlights the fact that recycling based on drop-off stations imply storage, time and transport costs for the individual and if we lower these costs, recycling will increase. One reason for the insignificant result in the case of plastic packaging could in part be due to confusion about the question posed. Plastic packaging (without deposit-refunds) can be divided into two different sub groups, hard plastic and soft plastic, and the question in the survey only refers to plastic packaging materials in general. This is a potential problem since all soft plastic is usually thrown in the waste bin (and is ultimately energy recovered). A second reason for this result could be that plastic packaging have low weight and it is perhaps easier to transport to a recycling station and, hence, not as influenced by perceived transport costs.

Investigating the marginal effects from introducing property-close collection confirm the above results. For instance, the probability that an individual recycles all paper packaging increases by 27.6 percentage points if she has the opportunity to leave her paper packaging within the borders of the property. At the same time, the probabilities for all other recycling outcomes decrease. The results for metal and glass packaging are very similar.

Volume-based waste fees also seem to matter for the individual, at least to some degree. Once again, the estimated coefficient is positive for all four packaging materials. However, only the coefficient for paper is statistically significant at the five percent level. It is somewhat difficult to explain why paper packaging seems to be particularly influenced by the use of volume-based fees. Normally it should be quite easy to compress the paper packaging if you want to put it in the waste bin. This overall weak relationship between volume-based fees and recycling efforts is also in line with earlier research. For instance, Sterner and Bartelings (1999), Dahlén et al. (2007), Hage and Söderholm (2008), Hage et al. (2007) find that weight-based waste fees are often more effective in stimulating recycling. There are several potential explanations for the ineffectiveness of volume-based waste fees. The household could change the size of the waste bin and hence pay less but there are at least two problems associated with this strategy. First, the flexibility in the size of the waste bin is not high. Second, households need to notify the local authorities in advance when they want to change bin size. Consequently, a household that wants to be sure that it could manage weekly variation in the waste generation, has an incentive to choose a waste bin that can cope with more than their average trash production. Hence, in an average week there will be few incentives to reduce waste generation by recycling more. An additional explanation is that under a volume-based waste fee you could also use your neighbors' waste bins if they have free space. None of these problems will exist if the household need to pay based on the weight of the waste generated.

The coefficients for the *moral norm variables* show interesting results. All proxies for personal moral norm activation appear to be very important for explaining recycling behavior. The sign of the coefficient for *moral obligation* is as expected positive and statistically significant at the one percent level for paper, glass and metal and at the five percent level for plastic. This supports the hypothesis that individuals will recycle more if they feel a personal responsibility to recycle. The results in Tables 4 and 5 strongly confirm that the perception of *others' recycling* efforts is an important driver of own recycling efforts. The coefficients for beliefs about others' recycling behavior are as expected positive and statistically significant at the one percent level for all four packaging materials. When considering the marginal effects it is obvious that the perception about others behavior is important. The results also show that this variable seems to be more important for plastic and paper packaging recycling decision.

Tables 4 and 5 also highlight the importance of awareness of consequences as a determinant for explaining recycling behavior. The coefficients for the *negative externality* variable, which is a proxy for awareness of consequences, are positive and statistically significant at the one percent level for all packaging materials. This indicates that individuals that believe that refraining from recycling will lead to large negative external effects are more likely to recycle their packaging waste. The magnitudes of the effects are also quite similar when compared across materials. However, it seems as if the perception of the negative externality is more important for people that recycle high degrees of plastic packaging and less important for people that recycle high degree of paper packaging. This is maybe not surprising since plastic is made from non-renewable fossil resources and paper is made of renewable forest resources that are abundant in Sweden.

The negative signs of the coefficients for the *no idea* variables, also proxies for awareness of consequences, are as expected, and also statistically significant at the one or five percent levels for all packaging materials. This implies that individuals that do not see the relationship between their choice to either recycle or throw away packaging waste on the one hand and the environment on the other will be less willing to recycle. Once again, this seems to be a more important variable for plastic recyclers.

The ABC model suggests that moral norms are most important if recycling facilities are at an intermediate level while these norms should not explain much if recycling facilities weakly or strongly support recycling. Hence, the above results suggest that overall the main collection system in Sweden with drop-off stations could be a system that support recycling at an intermediate level. This implies that it should be possible to attract individuals with lower moral norms for recycling, and hence to achieve a sustainable increase in the recycling outcome by making it easier for households to recycle (and/or by recycling campaigns that try to influence their moral norms for recycling). According to the ABC model, moral norms could become less important in explaining household recycling when households have access to property close-collection. In order to test for this hypothesis an interactive slope variable (*ABC*) was created. This variable was constructed by multiplying the respondent's moral norm for recycling with the property-close collection dummy.

The coefficient for these variables is negative for paper, glass and metal (but not for plastic), well in line with the ABC hypothesis. Furthermore, the coefficient is statistically significant at the five percent level for glass and at the ten percent level for paper. These results indicate that moral norms are less important for explaining recycling behavior when the recyclers have the opportunity to use property-close recycling, at least for glass and paper. It is reasonable to suspect that this tendency is even more obvious if individuals have the opportunity to use curbside recycling because that system should be even more convenient for households.

The legal norm variable and the social norm variable appear to have limited influences on individual recycling behavior. The coefficients for the *legal norm* actually have unexpected signs for paper, plastic and metal, but none of these are statistically significant. The coefficients for the variable *social norm* show similar results. Hence, Swedish household recyclers do not seem to be influenced by friends, family and other important persons. In order to test if property-close recycling alters the role of social norms, an interactive slope variable – *social norm for property-close collection* – was created. This variable was constructed by multiplying the scores for the social norm with the dummy variable for the property-close collection. Once again, three of the coefficients have unexpected signs and none of them are statistically significant. Thus, it seems like none of the investigated recycling management system is supported by social norms.

Still, it is important to understand that this does not necessarily imply that legal rules and social norms do not matter at all for packaging recycling in Sweden. First of all, before the producer responsibility ordinance was introduced in 1994 glass was the only packaging material (if we disregard from packaging materials with a deposit-refund system) that was possible to recycle for Swedish households. Of course, it is difficult to know what had happen if the ordinance never had been introduced, but it does not seem likely that the private sector had established a non-profitable collection of used packaging without a legal regulation. Second, according to Schwartz (1977) moral norms may start with social interaction and/or legal interventions and over time these legal and social norms become internalized and thus personal (or moral).

The above suggests that it can be difficult to separate the impacts of moral and social norms in the empirical analysis. In order to test for this, other model specifications were tested. First, the social norm and the legal norm variables were deleted from the model. Overall this test show almost the same results as the estimations presented in Tables 4 and 5.[16] Second, in order to test if the impact of the beliefs about others' recycling effort could be substituted with the social norm variable ("important persons close to me wants me to recycle"), the *others' recycling* variable was excluded from the model. In this case the overall model fit decreased but the parameter estimates were almost the same as before.[17] Finally, all variables that are used as proxies for the endogenous moral norm formation were excluded. The results from this estimation are presented in Appendix D, and they indicate two main findings. The model fit decreases considerably and the coefficients for the social norm and the

[16] The results of this estimation are not presented in this paper, but are available from the author upon request.
[17] The results of this estimation are not presented in this paper, but are available from the author upon request.

legal norm variables become positive and statistically significant. Thus, this provides some support for the hypothesis that the legal and social norms are mediated through internalized moral norms.

The *socio-economic variables* also show some interesting results. In the cases of *gender* and *income* there are, however, no statistically significant results. The findings that gender do not matter for recycling behavior is in line with earlier research (e.g., Schultz et al., 1995; Hage and Söderholm, 2008; Hage et al., 2007). The result that income and recycling outcome are not correlated contradicts some international studies (e.g., Callan and Thomas, 1997; Berglund and Söderholm, 2003), but is in line with research on Swedish conditions (Hage and Söderholm, 2008; Hage et al., 2007). The coefficients for *education* show some contradicting results, the ones for plastic, glass and metal are all positive but not statistically significant. However, the coefficient for paper is negative and statistically significant. These results contradict some U.S. studies (e.g., Schultz et al., 1995; Callan and Thomas, 1997; Jenkins et al., 2003), while the results for paper packaging support the findings in Hage and Söderholm (2008).

Furthermore, the coefficient for *age* is positive and statistically significant at the one percent level for all four packaging materials. The magnitudes of these impacts are also quite similar for all packaging materials. These results imply that recycling efforts increase with age. One possible explanation for this could be that older people are retired and, thus, have lower opportunity cost of time for recycling (also after having controlled for income).

The coefficient for the *multi-family dwellings* variable is as expected negative and statistically significant for metal, but statistically insignificant for all other packaging materials. This suggests that people living in apartments will recycle less metal. Hage and Söderholm (2008) find the same relationship for plastic packaging. As was noted in section 4, a possible explanation for this is that single-family dwelling households have more space for storing used packaging. Another question is why only metal recycling is affected by the type of living. One reason could be that metal packaging is rather difficult to compress and hence need more space than paper and plastic packaging.

Finally, the municipality dummies also seem to explain some of the variance in the household packaging recycling. It is difficult to form any *ex ante* expectations about the signs of these coefficients. The differences in results across municipalities could be explained by the presence of municipality-specific characteristics such as differences in collection productivity in the recycling industry and/or information to household. As was mentioned in section 1, the producers have contracted different collection entrepreneurs in different municipalities for running the recycling stations and of course this means that management skills could explain differences in collection productivity between municipalities. For instance, a rather common problem is that some entrepreneurs are ineffective in terms of emptying recycling containers and generally maintaining the quality and the cleanliness of the drop-off stations (e.g. SEPA, 2006). Another problem could be that the recycling stations are not located with respect to households daily movements, e.g., in the adjacent to shopping centres (Ibid). However, the political attitude of the municipality is also important for the collection productivity and the extent to which they embrace recycling goals. The municipalities could rent sites at subsidized rates, and be flexible when providing building permits for the recycling stations. After 2005, the municipalities are also responsible for informing citizens about the packaging recycling schemes. Hage et al. (2007) find that the number of recycling stations per resident differs substantially across the different municipalities, and this is an important determinant for explaining the observed variance in household plastic packaging levels in Sweden.

7. Concluding Remarks and Implications

The purpose of this study has been to investigate important determinants of the collection of Swedish household packaging waste and analyze if these differs for different packaging materials. Emphasis was put on the roles and the interaction between economic and norm-based motivation. Overall the results show that both neo-classical economic variables and personal moral norms are important in explaining recycling behavior in Swedish households.

The results clearly show that increased service to households will increase the recycling outcome, not the least property-close collection in multi-family dwellings. However, before introducing these systems, the negative effects should also be weighted against the benefit of increased recycling levels and lower costs for the households. The private cost for the waste management operators will probably be higher in these system and the environmental effects from transports should also be considered.

The findings also show that moral norm activation is important in explaining household recycling efforts for all packaging materials, at least when considering the main Swedish packaging collection system with drop-off stations. The felt moral obligation for the issue, the beliefs about others behaviour, the perceived positive external effect from recycling, and the extent to which recycling generates environmental public goods are all factors that have the expected signs and are strongly statistical significant. However, one of these variables stands out. The beliefs about others' recycling efforts seem to be the most important variable in explaining altruistic recycling behavior. These results indicate that the authorities and the recycling industries could influence recycling efforts with recycling campaigns. The study also shows that the importance of moral norms decreases when property-close collection is introduced. This implies that the effectiveness of these information campaigns will decrease if the policy and the external conditions make it easier for households to recycle. The results are also fairly similar for different packaging materials. However, the results for plastic packaging differ somewhat. The beliefs about others' behavior and proxies for awareness of consequences are more important for explaining plastic packaging outcome while moral obligation and the neo-classical variables are comparatively less important.

The remaining variables do not seem to add much information about Swedish households' recycling behavior. The results suggest that neither social nor legal norms matter for the individual recycler. Nevertheless, we should be careful when interpreting these results, and in recommending policy design. First, without the ordinance (legal norm) there had probably not been any nation-wide collection system for household packaging. If this was the case, we also know from the ABC model that moral norms hardly matter for recycling rates. Second, in practice it is difficult to clearly distinguish between moral and social norms. For instance, the importance of the beliefs about others' recycling effort could also stem from social norms. Schwartz (1977) maintains that moral norms start through social interaction between individuals. Hence, the moral norms that support recycling in Sweden today had not existed in the absence of legal and social norms in the introduction phase. An alternative model specification also suggests that the social and legal norm variables matter also for present efforts if we exclude the moral norm variables. Hence, it seems like that social and legal norms are mediated through personal moral norms. Finally, almost all socio-economic variables do a poor job in explaining recycling behavior. However, age is one exception since the recycling effort clearly increases with age.

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Appendix A: Cover Letter and Selected Parts of the Questionnaire in English

Households in Huddinge*

At present, a joint research program is in progress at the universities in Luleå, Linköping and Umeå. The purpose of the project is to increase the knowledge about households' attitude towards policy measures that aim at improving the environment. The Swedish Environmental Protection Agency is funding the project and the research is carried out in cooperation with your municipality.

The investigation is addressed to people aged between 20 and 75 years in four Swedish municipalities. You are one of the 700 randomly chosen participants in Huddinge*. In a scientific study as this it is important that people with different opinions have the possibility to participate, including people that do not have an interest in environmental questions. To ensure the reliability of the results of the survey it is very important that all participants answer the questionnaire. Your response can not be replaced by any one else's. Your answers will only be presented in statistically processed forms and your anonymity will be guaranteed.

Please answer the questions as soon as possible and return the completed questionnaire to us in the pre-stamped envelope. We are grateful if you do not skip any questions.

The code number on the first page of the survey makes it possible for us to note that you have answered the questionnaire so we do not have to bother you with reminders. Thereafter, the link between the code number and your name will be removed.

If you have any questions about the survey, you are welcome to contact us by phone or mail.

Kind regards,

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Addresses have been obtained from the local register in your municipality.

* The cover letter and the questionnaire that were sent out to the households in the other municipalities were exactly the same (with the exception of the name of the municipality).

QUESTIONS ABOUT YOU AND YOUR HOUSEHOLD

1. I am ☐ Female ☐ Male

2. I am ______ years old

3. I am ☐ Living alone ☐ Married/cohabiting

4. My highest level of education is:

☐ Elementary school or equivalent
☐ Senior high school or equivalent
☐ College or university
☐ Other:______________________

5. My primary occupation is:

☐ Wage earning, full time
☐ Wage earning, part time
☐ Studying
☐ Job-seeking
☐ Home working or on parental leave
☐ Retiree or on disablement pension
☐ Other:______________________

6. I/we have ______ (number) children at home aged ________________________

7. Total household income per month before taxes is.

☐ 10000 SEK or less
☐ 10001-20000 SEK
☐ 20001-30000 SEK
☐ 30001-40000 SEK
☐ 40001-50000 SEK
☐ 50001-60000 SEK
☐ 60001-70000 SEK
☐ 70001-80000 SEK
☐ 80001-90000 SEK
☐ 90001-100000 SEK
☐ More than 100000 SEK

8a. I live in:

☐ A multi-family dwelling
☐ A row house
☐ A single-family dwelling
☐ Other:______________

8b. For you that live in an multi-family dwelling: Do you have access to property-close collection for your household waste?

☐ Yes ☐ No

9. The household fee for the waste management:

☐ is based on the size of the garbage bin and/or garbage collection intervals (volume based fee).
☐ is only based on the garbage weight (weight based fee).
☐ is based on a combination of volume and weight fees.
☐ is included in the rent. (flat fee)
☐ do not know

10. Our household has:

☐ 0 cars
☐ 1 cars
☐ 2 cars
☐ 3 cars or more

11. How far is the distance between your household and the closest drop-off station for packaging waste?

______________ km

ATTITUDE TOWARDS THE ENVIRONMENT

18. Below follow a number of statements about the environment. State to what extent you agree or disagree with each of the items. (Mark the number that best corresponds with your opinion for each statement)

	Disagree entirely	Disagree partly	Unsure	Agree partly	Agree entirely
The balance of nature is strong enough to cope with the impacts of modern industrial nations.	1	2	3	4	5
Humans have a right to modify the natural environment to suit their needs	1	2	3	4	5
Plants and animals have as much right as humans to exist	1	2	3	4	5
Despite our special abilities, humans are still subject to the laws of nature.	1	2	3	4	5
Humans were meant to rule over the rest of nature.	1	2	3	4	5
The earth is like a spaceship with very limited room and resources.	1	2	3	4	5
When humans interfere with nature it often produces disastrous consequences.	1	2	3	4	5
We are approaching the limit of the number of people the earth can support.	1	2	3	4	5
The earth has plenty of natural resources if we just learn to develop them.	1	2	3	4	5
Humans are severely abusing the environment.	1	2	3	4	5
Humans' ingenuity will insure that we do not make the earth unlivable.	1	2	3	4	5
If things continue on their present course, we will soon experience a major ecological catastrophe.	1	2	3	4	5
Humans will eventually learn enough about how nature works to be able to control it.	1	2	3	4	5
The so-called "ecological crisis" facing human kind has been greatly exaggerated.	1	2	3	4	5
The balance of nature is very delicate and easily upset.	1	2	3	4	5

QUESTIONS ABOUT RECYCLING BEHAVIOR

27. To what extent do you and your family recycle the following household packaging waste? *(Mark the box that best corresponds to your recycling behavior for each packaging waste fraction)*

	Nothing				Everything
Paper packaging	☐	☐	☐	☐	☐
Plastic packaging (without deposit refund))	☐	☐	☐	☐	☐
Glass packaging (without deposit refund)	☐	☐	☐	☐	☐
Metal packaging (without deposit refund)	☐	☐	☐	☐	☐

28. How much of their household waste do you think that other households in your municipality recycles?

	Nothing				Everything
Paper packaging	☐	☐	☐	☐	☐
Plastic packaging (without deposit refund)	☐	☐	☐	☐	☐
Glass packaging (without deposit refund)	☐	☐	☐	☐	☐
Metal packaging (without deposit refund)	☐	☐	☐	☐	☐

29. Approximately, how many minutes on average during one week does your household allocate to:

- Sorting and cleaning the waste that you recycle? __________minutes
- Transporting the waste to a drop-off station? ___________minutes

30. Do you transport the waste with the sole purpose of leaving your waste at the drop-off stations? *(Mark the number that best corresponds to your opinion)*

No, never						Yes, always
1	2	3	4	5	6	7

31. How much sacrifice (in terms of time, money, and feelings of discomfort) is required to recycle the following packaging waste fractions? *(Mark the box that best corresponds to your opinion for each packaging waste fraction)*

	No sacrifice at all				Very much sacrifice	Do not know
Paper packaging	☐	☐	☐	☐	☐	☐
Plastic packaging (without deposit refund)	☐	☐	☐	☐	☐	☐
Glass packaging (without deposit refund)	☐	☐	☐	☐	☐	☐
Metal packaging (without deposit refund)	☐	☐	☐	☐	☐	☐

32. To what extent do you agree or disagree with the following statements about the effects from not recycling the household waste? (Mark the number that best corresponds to your opinion for each statement)

	Entirely disagree			Unsure			Agree entirely
Household waste that not is recycled is a threat to humans and the environment world-wide	1	2	3	4	5	6	7
Household waste that not is recycled is a threat to humans and the environment in Sweden.	1	2	3	4	5	6	7
Household waste that not is recycled is a threat to humans and the environment in the municipality where I live	1	2	3	4	5	6	7
Household waste that not is recycled is a threat to me and my family's health and wellbeing.	1	2	3	4	5	6	7
Household waste that not is recycled is a very serious problem so immediate action is needed.	1	2	3	4	5	6	7

33. To what extent do you agree or disagree with the following statements about household recycling? (Mark the number that best corresponds to your opinion for each statement)

	Entirely disagree			Unsure			Agree entirely
My household waste gives rise to negative effects on the environment.	1	2	3	4	5	6	7
It is no idea that I recycle because it will not lead to a better environmental quality.	1	2	3	4	5	6	7
I feel a moral obligation to recycle.	1	2	3	4	5	6	7
I get bad conscience if I do not recycle.	1	2	3	4	5	6	7
I recycle because I believe that I should do what I want others to do.	1	2	3	4	5	6	7
I recycle because I want to think about myself as a responsible person.	1	2	3	4	5	6	7
The government and the municipality expect me to recycle.	1	2	3	4	5	6	7
Important persons close to me want me to recycle.	1	2	3	4	5	6	7
I observe many people in my municipality that recycle.	1	2	3	4	5	6	7
There exist many obstacles that prevent me from recycling more.	1	2	3	4	5	6	7
If I want it is easy for me to recycle more.	1	2	3	4	5	6	7
I am willing to recycle more to reduce the negative effects on the environment.	1	2	3	4	5	6	7
Recycling is good.	1	2	3	4	5	6	7

THANK YOU FOR YOUR COOPERATION!

Appendix A: Cover Letter and Selected Parts of the Questionnaire in Swedish

Till hushåll i Huddinge* kommun

Just nu pågår ett gemensamt forskningsprojekt vid universiteten i Luleå, Linköping och Umeå. Syftet med projektet är bl.a. att öka kunskapen om hur Huddinges* innevånare ställer sig till åtgärder som syftar till att förbättra miljön. Naturvårdsverket finansierar projektet och undersökningen utförs i nära samarbete med Huddinge kommun.

Undersökningen vänder sig till personer i åldrarna 20-75 år, och Du är en av 700 slumpmässigt utvalda deltagare. I en vetenskaplig undersökning som denna är det viktigt att människor med olika uppfattning får tillfälle att delta, även de som kanske inte har ett intresse i miljöfrågor. Värdet av undersökningens resultat är beroende av att så många som möjligt besvarar frågeformuläret. Ditt svar kan inte ersättas av någon annans. Dina svar kommer endast att redovisas i statistiskt bearbetad form och varje deltagares anonymitet är garanterad.

Besvara frågorna så fort som möjligt och skicka det ifyllda formuläret till oss i det bifogade portofria svarskuvertet. Vi är tacksamma om Du inte hoppar över någon fråga.

Det kodnummer som finns på formulärets första sida gör det möjligt för oss att notera att just Du har svarat så att vi inte behöver besvära Dig med påminnelser. Därefter kommer kopplingen mellan kodnummer och namn att tas bort.

Har Du några frågor angående undersökningen kan Du kontakta oss på nedanstående telefonnummer eller via e-post.

Med vänlig hälsning

Forskningsassistent
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Luleå tekniska universitet
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Adressuppgifterna har erhållits från Huddinges* kommunregister.

* Exakt samma enkät och följebrev skickades ut till övriga tre kommuner (med undantag för själva kommunnamnet).

BAKGRUNDSFRÅGOR

1. Jag är ☐ Kvinna ☐ Man

2. Jag är ______ år

3. Jag är ☐ Ensamstående ☐ Gift/sammanboende

4. Min högsta genomförda utbildning är:

☐ Grundskola/folkskola
☐ Gymnasieskola/folkhögskola
☐ Universitet/högskola
☐ Annat:______________________

5. Min huvudsakliga sysselsättning är:

☐ Förvärvsarbete, heltid
☐ Förvärvsarbete, deltid
☐ Studerande
☐ Arbetssökande
☐ Hemarbetande eller föräldraledig
☐ Pensionär eller sjukpensionär
☐ Annat:______________________

6. Jag/vi har ______ (antal) hemmavarande barn i åldrarna ________________________

7. Ungefär hur hög är Ditt <u>hushålls</u> sammanlagda inkomst <u>per månad före skatt</u> (inkludera alla slags inkomster, till exempel eventuell sjukpenning, föräldrapenning, studiemedel, arbetslöshetsersättning etc.).

☐ 10000 kronor eller mindre ☐ 40001-50000 kronor ☐ 80001-90000 kronor

☐ 10001-20000 kronor ☐ 50001-60000 kronor ☐ 90001-100000 kronor

☐ 20001-30000 kronor ☐ 60001-70000 kronor ☐ Mer än 100000 kronor

☐ 30001-40000 kronor ☐ 70001-80000 kronor

8a. Jag bor i:

☐ lägenhet
☐ radhus
☐ fristående villa
☐ annat:______________

8b. För Dig som bor i <u>lägenhet eller radhus</u>, finns det behållare för källsortering i den fastighet där Du bor eller i direkt anslutning till fastigheten?

☐ Ja ☐ Nej

9. Mitt hushålls kostnad för sophantering:

☐ baseras på sopkärlets storlek och/eller hur ofta sopkärlet töms
☐ baseras endast på hur mycket soporna väger
☐ baseras dels på en avgift (t ex beroende på sopkärlsstorlek), dels på hur mycket soporna väger
☐ ingår i hyran (fast avgift)
☐ vet ej

10. I hushållet finns:

☐ 0 bilar
☐ 1 bil
☐ 2 bilar
☐ 3 bilar eller fler

11. Hur långt är det mellan Din bostad och närmaste återvinningsstation i kilometer:

________________km

INSTÄLLNING TILL MILJÖN

18. Nu följer ett antal påståenden om miljön. Ange i vilken utsträckning Du instämmer i eller tar avstånd från vart och ett av påståendena. *(Ringa in den siffra som bäst svarar mot Din uppfattning)*

	Tar helt avstånd ifrån	Tar delvis avstånd ifrån	Osäker	Instämmer delvis	Instämmer helt
Balansen i naturen är tillräckligt stark för att klara av de moderna industrinationernas påverkan.	1	2	3	4	5
Människan har rätt att förändra naturen efter sina behov.	1	2	3	4	5
Växter och djur har lika stor rätt att existera som människor.	1	2	3	4	5
Trots våra speciella förmågor lyder vi människor Fortfarande under naturens lagar.	1	2	3	4	5
Människorna är ämnade att härska över naturen.	1	2	3	4	5
Jorden kan liknas vid en rymdfarkost med mycket Begränsade utrymmen och resurser.	1	2	3	4	5
När människan ingriper i naturens förlopp får det ofta katastrofala följder.	1	2	3	4	5
Vi närmar oss gränsen för den folkmängd jorden kan föda.	1	2	3	4	5
Jorden har gott om naturresurser bara vi lär oss hur vi ska använda dem.	1	2	3	4	5
Människan förgriper sig allvarligt på naturen.	1	2	3	4	5
Människans uppfinningsrikedom kommer att garantera att vi *inte* gör jorden obeboelig.	1	2	3	4	5
Om utvecklingen fortsätter som hittills kommer vi snart att få uppleva en stor ekologisk katastrof.	1	2	3	4	5
Så småningom kommer människan att lära sig tillräckligt om hur naturen fungerar för att kunna kontrollera den.	1	2	3	4	5
Den så kallade ”ekologiska krisen” som mänskligheten står inför har kraftigt överdrivits.	1	2	3	4	5
Balansen i naturen är väldigt känslig och rubbas lätt.	1	2	3	4	5

FRÅGOR OM HUSHÅLLSAVFALL OCH KÄLLSORTERING

27. Hur stor del av hushållsavfallet källsorterar Du och Din familj? *(Kryssa för det svarsalternativ som bäst svarar mot Din uppfattning för varje typ av avfall)*

	Inget				Allt
Pappersförpackningar	☐	☐	☐	☐	☐
Plastförpackningar (utan pant)	☐	☐	☐	☐	☐
Glasförpackningar (utan pant)	☐	☐	☐	☐	☐
Metallförpackningar (utan pant)	☐	☐	☐	☐	☐

28. Hur stor del av det totala hushållsavfallet tror Du att andra hushåll i Huddinge kommun källsorterar?

	Inget				Allt
Pappersförpackningar	☐	☐	☐	☐	☐
Plastförpackningar (utan pant)	☐	☐	☐	☐	☐
Glasförpackningar (utan pant)	☐	☐	☐	☐	☐
Metallförpackningar (utan pant)	☐	☐	☐	☐	☐

29. Ungefär hur många <u>minuter</u> spenderar Ert hushåll i genomsnitt <u>under en vecka</u> för:

- Att källsortera och rengöra avfall i hemmet? __________minuter
- Att frakta källsorterat avfall till återvinningsstationer? __________minuter

30. Gör Ni extra/enkom resor till återvinningsstationen (för papper, glas, plast etc.) för att lämna Ert hushållsavfall? *(Ringa in den siffra som bäst svarar mot Din uppfattning)*

Nej, aldrig						Ja, alltid
1	2	3	4	5	6	7

31. Hur stora uppoffringar (i form av t. ex. tid, pengar och känslor av obehag) tycker Du att det krävs för att källsortera nedanstående avfallstyper? *(Kryssa för det svarsalternativ som bäst svarar mot Din uppfattning för varje typ av avfall)*

	Inga upp-offringar alls				Mycket stora uppoffringar	Vet ej
Pappersförpackningar	☐	☐	☐	☐	☐	☐
Plastförpackningar (utan pant)	☐	☐	☐	☐	☐	☐
Glasförpackningar (utan pant)	☐	☐	☐	☐	☐	☐
Metallförpackningar (utan pant)	☐	☐	☐	☐	☐	☐

32. Nedan följer ett antal påståenden om källsortering. Ange i vilken utsträckning Du instämmer i eller tar avstånd från vart och ett av påståendena. *(Ringa in den siffra som bäst svarar mot Din uppfattning)*

	Tar helt avstånd ifrån			Osäker			Instämmer helt
Hushållsavfall som inte källsorteras är ett hot mot människor och miljö i hela världen.	1	2	3	4	5	6	7
Hushållsavfall som inte källsorteras är ett hot mot människor och miljö i Sverige.	1	2	3	4	5	6	7
Hushållsavfall som inte källsorteras är ett hot mot människor och miljö i den kommun där jag bor.	1	2	3	4	5	6	7
Hushållsavfall som inte källsorteras är ett hot mot min och min familjs hälsa och välbefinnande.	1	2	3	4	5	6	7
Hushållsavfall som inte källsorteras är ett så allvarligt problem att åtgärder måste sättas in omedelbart.	1	2	3	4	5	6	7

33. Nedan följer ett antal påståenden om källsortering. Ange i vilken utsträckning Du instämmer i eller tar avstånd från vart och ett av påståendena. *(Ringa in den siffra som bäst svarar mot Din uppfattning)*

	Tar helt avstånd ifrån			Osäker			Instämmer helt
Mitt hushållsavfall leder till negativa effekter på miljön.	1	2	3	4	5	6	7
Det är ingen idé att jag källsorterar eftersom det inte för med sig några positiva miljöeffekter.	1	2	3	4	5	6	7
Jag känner en moralisk skyldighet att källsortera.	1	2	3	4	5	6	7
Jag får dåligt samvete om jag inte källsorterar.	1	2	3	4	5	6	7
Jag källsorterar för jag anser att jag själv bör göra sådant som jag förväntar mig att andra ska göra.	1	2	3	4	5	6	7
Jag källsorterar för jag vill se mig själv som en ansvarsfull person.	1	2	3	4	5	6	7
Stat och kommun vill att jag ska källsortera för att reducera de negativa effekterna på miljön.	1	2	3	4	5	6	7
Viktiga personer i min närhet vill att jag ska källsortera.	1	2	3	4	5	6	7
Jag ser många personer i Huddinge kommun som källsorterar.	1	2	3	4	5	6	7
Det finns många faktorer i min omgivning som hindrar mig från att källsortera mer.	1	2	3	4	5	6	7
Om jag vill är det enkelt för mig att källsortera mer.	1	2	3	4	5	6	7
Jag är villig att källsortera mer för att reducera de negativa effekterna på miljön.	1	2	3	4	5	6	7
Källsortering är bra.	1	2	3	4	5	6	7

TACK FÖR DIN MEDVERKAN!

Appendix B: Perceptions of the Externalities that Arise from Not Recycling (Percentage Distribution)

Statements	*Disagree entirely*			*Unsure*			*Agree entirely*
Household waste that is not recycled is a threat to humans and environmental quality world-wide.	4.4	5.4	8.3	28.3	18.9	16.4	18.3
Household waste that is not recycled is a threat to humans and environmental quality in Sweden.	3.4	5.4	5.7	26.5	22.9	17.8	18.4
Household waste that is not recycled is a threat to humans and environmental quality in the municipality where I live.	3.1	5.6	6.5	27.8	21.2	17.3	18.3
Household waste that is not recycled is a threat to me and my family's health and wellbeing.	5.3	7.1	8.5	31.3	18.1	14.8	14.9
Household waste that is not recycled is a very serious problem so immediate act-ion is needed.	6.9	9.8	8.0	37.1	15.8	9.8	12.6

Appendix C: Aggregation of the Intermediate Recycling Outcomes

Table C1: Parameter Estimates for the Ordered Probit Model

Variables	Expected sign	Paper	Plastic	Glass	Metal
Constant	?	-1.265***	-2.090***	-2.214***	-2.462***
		(-2.66)	(-4.34)	(-4.10)	(-4.66)
Neoclassical Variables					
Property-close collection	+	0.965***	0.062	1.325***	1.151***
		(2.61)	(0.17)	(3.25)	(2.89)
Volume-based waste fees	+	0.372**	0.135	0.258	0.288
		(2.18)	(0.81)	(1.36)	(1.51)
Norm Variables					
Moral obligation	+	0.148***	0.110**	0.172***	0.172***
		(3.34)	(2.45)	(3.58)	(3.59)
Others' recycling	+	0.445***	0.465***	0.398***	0.421***
		(7.51)	(8.09)	(6.37)	(6.87)
Negative externality	+	0.098**	0.137***	0.134***	0.166***
		(2.43)	(3.37)	(2.92)	(3.68)
No idea	-	-0.087**	-0.115***	-0.078*	-0.087**
		(-2.38)	(-3.11)	(-1.89)	(-2.15)
ABC	-	-0.157**	0.044	-0.176**	-0.087
		(-2.37)	(0.68)	(-2.44)	(-1.24)
Legal norm	+	-0.016	0.002	0.033	-0.010
		(-0.43)	(0.05)	(0.82)	(-0.25)
Social norm	+	-0.033	-0.009	0.001	0.026
		(-0.91)	(-0.26)	(0.02)	(0.62)
Social norm for property-close collection	+	0.069	-0.018	-0.014	-0.065
		(1.17)	(-0.31)	(-0.22)	(-1.00)
Socio-Economic Variables					
Gender	?	-0.050	0.071	0.020	-0.023
		(-0.47)	(0.68)	(0.17)	(0.84)
Age	?	0.014***	0.017***	0.020***	0.021***
		(3.75)	(4.40)	(4.71)	(4.89)
Education	?	-0.106	0.027	0.041	0.140*
		(-1.42)	(0.37)	(0.62)	(1.69)
Income	-	-0.017	-0.022	-0.024	-0.029
		(-0.65)	(-0.86)	(-0.85)	(-1.03)
Multi-family dwellings	-	0.022	-0.208	-0.270	-0.425**
		(-0.12)	(-1.09)	(-1.26)	(-2.02)
Municipalities Dummies					
Piteå	?	-0.077	0.238*	0.145	0.098
		(-0.51)	(1.67)	(0.85)	(0.61)
Gothenburg	?	-0.561***	0.669***	-0.479***	0.317*
		(-3.83)	(4.32)	(-3.02)	(1.89)
Växjö	?	-0.201	-0.127	-0.042	-0.315**
		(-0.59)	(-0.88)	(0.81)	(-1.98)
Number of observations		695	700	700	694
Log likelihood		-481	-485	-389	-383
Restricted log likelihood		-590	-591	-499	-483
Chi squared		217***	211***	220***	201***

Note: *t*-statistics are given in brackets. *, **, and *** indicate statistical significance at the ten, five, and one percent levels.

Table C2a: Marginal Effects for Paper Packaging

Variables	0 (Recycle nothing)	1 - 3	4 (Recycle everything)
Neoclassical variables			
Property close collection	-0.090***	-0.212***	0.302*
Volume based waste fees	-0.042**	-0.089***	0.130
Norm variables			
Moral obligation	-0.017***	-0.035***	0.052***
Others' recycling	-0.050***	-0.106***	0.156***
Negative externality	-0.011**	-0.023**	0.034**
No idea	0.010**	0.021**	-0.031*
ABC	0.018**	0.037**	-0.055**
Socio-Economic Variables			
Age	-0.002***	-0.003***	0.005***
Education	0.012	0.025	-0.037

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

Table C2b: Marginal Effects for Plastic Packaging

Variables	0 (Recycle nothing)	1 - 3	4 (Recycle everything)
Neoclassical variables			
Property close collection	-0.008	-0.013	0.021
Volume based waste fees	-0.017	-0.029	0.046
Norm variables			
Moral obligation	-0.014**	-0.024**	0.038**
Others' recycling	-0.058***	-0.101***	0.159***
Negative externality	-0.017***	-0.030***	0.047***
No idea	0.014***	0.025***	-0.039**
ABC	0.005	0.010	-0.015
Socio-Economic Variables			
Age	-0.002***	-0.004***	0.006***
Education	-0.003	-0.006	-0.009

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

Table C2c: Marginal Effects for Glass Packaging

Variables	0 (Recycle nothing)	1 - 3	4 (Recycle everything)
Neoclassical variables			
Property close collection	-0.086***	-0.195***	0.281
Volume based waste fees	-0.021*	-0.046**	0.067
Norm variables			
Moral obligation	-0.014***	-0.031***	0.045*
Others' recycling	-0.032***	-0.072***	0.104**
Negative externality	-0.011***	-0.024***	0.035**
No idea	0.006*	0.014*	-0.020
ABC	0.014**	0.032**	-0.046*
Socio-Economic Variables			
Age	-0.002***	-0.004***	0.006**
Education	0.003	0.007	-0.010

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

Table C2d: Marginal Effects for Metal Packaging

Variables	0 (Recycle nothing)	1 - 3	4 (Recycle everything)
Neoclassical variables			
Property close collection	-0.059***	-0.198***	0.257
Volume based waste fees	-0.018*	-0.058**	0.077
Norm variables			
Moral obligation	-0.011***	-0.035***	0.046**
Others' recycling	-0.026***	-0.086***	0.112**
Negative externality	-0.010***	-0.034***	0.044**
No idea	0.005**	0.018**	-0.023
ABC	0.005	0.018	-0.023
Socio-Economic Variables			
Age	-0.001***	-0.004***	0.005**
Education	0.009*	0.028*	-0.037*

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

Appendix D: Excluding the Moral Norm Variables from the Model

Table D1: Parameter Estimates for the Ordered Probit Model

Variables	Expected sign	Paper	Plastic	Glass	Metal
Constant	?	0.126	-0.661*	-0.398	-0.807**
		(0.35)	(-1.82)	(-1.00)	(-2.03)
Neoclassical Variables					
Property-close collection	+	0.477**	0.388*	0.633**	0.849***
		(2.12)	(1.68)	(2.53)	(3.42)
Volume-based waste fees	+	0.404**	0.246	0.270	0.348**
		(2.53)	(1.58)	(1.52)	(1.97)
Norm Variables					
Legal norm	+	0.088***	0.122***	0.143***	0.118***
		(2.70)	(3.72)	(4.05)	(3.37)
Social norm	+	0.076**	0.060*	0.080**	0.116***
		(2.44)	(1.81)	(2.18)	(3.23)
Social norm for property-close collection	+	-0.038	-0.028	-0.076	-0.110*
		(-0.72)	(-0.51)	(-1.29)	(-1.89)
Socio-Economic Variables					
Gender	?	-0.127	-0.067	-0.754	-0.091
		(-1.30)	(-0.69)	(-0.70)	(-0.86)
Age	?	0.012***	0.015***	0.017***	0.020***
		(3.32)	(4.27)	(4.37)	(5.17)
Education	?	-0.134*	0.011	-0.015	0.097
		(-1.91)	(0.16)	(-0.19)	(1.26)
Income	-	0.014	-0.014	-0.003	-0.009
		(0.54)	(-0.61)	(-0.10)	(-0.33)
Multi-family dwellings	-	0.192	-0.006	-0.086	-0.285**
		(1.08)	(-0.04)	(-0.44)	(-1.48)
Municipalities Dummies					
Piteå	?	-0.269*	0.410***	0.328**	0.267*
		(1.94)	(3.14)	(2.09)	(1.82)
Gothenburg	?	-0.728***	0.629***	-0.647***	0.483***
		(-5.47)	(4.41)	(-4.55)	(3.12)
Växjö	?	0.036	0.013	0.180	-0.194
		(-0.25)	(0.10)	(1.12)	(-1.33)
Number of observations		724	722	725	724
Log likelihood		-689	-708	-549	-551
Restricted log likelihood		-760	-751	-622	-606
Chi squared		141***	87***	145***	110***

Note: *t*-statistics are given in brackets. *, **, and *** indicate statistical significance at the ten, five, and one percent level.

Table D2a: Marginal Effects for Paper Packaging

Variables	0 (Recycle nothing)	1	2	3	4 (Recycle everything)
Neoclassical variables					
Property close collection	-0.056***	-0.014*	-0.038	-0.054*	0.162**
Volume based waste fees	-0.054***	-0.012*	-0.033	-0.044*	0.143**
Norm variables					
Legal norm	-0.011***	-0.003***	-0.007	-0.010	0.031***
Social norm	-0.010**	-0.002**	-0.006	-0.008	0.027**
Socio-Economic Variables					
Age	-0.001***	-0.001***	-0.001	-0.001	0.004***
Education	0.018*	0.004*	0.011	0.015	-0.048**

Table D2b: Marginal Effects for Plastic Packaging

Variables	0 (Recycle nothing)	1	2	3	4 (Recycle everything)
Neoclassical variables					
Property close collection	-0.054*	-0.015**	-0.024	-0.039	0.132*
Volume based waste fees	-0.038	-0.010*	-0.015	-0.024	0.087
Norm variables					
Legal norm	-0.019***	-0.005***	-0.008	-0.012	0.043***
Social norm	-0.008*	-0.002*	-0.004	-0.006	0.020*
Socio-Economic Variables					
Age	-0.002***	-0.001***	-0.001	-0.001	0.005***
Education	0.002	0.000	0.001	0.001	-0.004

Table D2c: Marginal Effects for Glass Packaging

Variables	0 (Recycle nothing)	1	2	3	4 (Recycle everything)
Neoclassical variables					
Property close collection	-0.056***	-0.018*	-0.037	-0.051	0.162**
Volume based waste fees	-0.028**	-0.009*	-0.017	-0.022	0.076
Norm variables					
Legal norm	-0.015***	-0.004***	-0.009	-0.012	0.040***
Social norm	-0.008**	-0.002**	-0.005	-0.007	0.024**
Socio-Economic Variables					
Age	-0.002***	-0.001***	-0.001	-0.001	0.005***
Education	0.001	0.001	0.001	0.001	-0.004

Table D2d: Marginal Effects for Metal Packaging

Variables	0 (Recycle nothing)	1	2	3	4 (Recycle everything)
Neoclassical variables					
Property close collection	-0.061***	-0.018*	-0.054	-0.080	0.213***
Volume based waste fees	-0.031***	-0.009	-0.025	-0.035	0.100
Norm variables					
Legal norm	-0.010***	-0.003***	-0.009	-0.012	0.034***
Social norm	-0.010***	-0.003***	-0.008	-0.012	0.033***
Socio-Economic Variables					
Age	-0.002***	-0.001***	-0.001	-0.002	0.006***
Education	0.008	0.003	0.007	0.010	-0.028

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

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An econometric analysis of regional differences in household waste collection: The case of plastic packaging waste 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 the use of a regression analysis based on cross-sectional data for 252 Swedish municipalities. The results suggest that local policies, geographic/demographic variables, socio-economic factors and 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, to be both statistically and economically insignificant. A reasonable explanation for this is that the monetary compensation from the material companies to the collection entrepreneurs vary depending on region and is typically higher in high-cost regions. This implies that the plastic packaging collection in Sweden may be cost ineffective. Finally, the analysis also shows that municipalities that employ weight-based waste management fees generally experience higher collection rates than those municipalities in which flat and/or volume-based fees are used.

1. Introduction

During the last two decades, environmental policy has largely shifted focus from production processes to finished products. One part of finished products, the packaging, has even become the symbol of ''unsustainable'' consumption in industrialized countries. As a result, government policies that encourage source reduction as well as waste diversion through recycling have become increasingly common, not the least in the packaging material sector. Recycling policy goals – e.g., collection and utilization rates – are often formulated at the national level, but the primary responsibility for policy implementation and evaluation are typically assigned to local actors (e.g., Callan and Thomas, 1997).

The Swedish producer responsibility ordinance for packaging (introduced in 1994) is a good example of this policy hierarchy. The legislation implies that the producers have the physical and the economic responsibility for the packaging waste, i.e., they are obliged to provide suitable systems for the collection and the recycling of packaging waste and inform households and firms about these systems or engage different collection entrepreneurs to perform the necessary tasks. The producers must also consult with the municipalities about the recycling systems, and overall the municipalities in Sweden take a very active role in running and/or supervising the collection schemes (see also Section 2). Households are obliged to clean and sort out packaging waste from other waste, and transport used packaging materials to assigned drop-off stations. Policies aimed at encouraging the recycling efforts of households, including waste management fees, infrastructural measures etc., are designed and implemented at the municipal level. This implies that recycling initiatives and outcomes are not uniform across the country. In part, any regional differences in collection rates may also be the result of geographic, socio-economic and demographic factors, which are less amenable to change.

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The above suggests that any meaningful analysis of the impact of recycling policies and measures in Sweden must rely on local observations, and to assure the validity in any generalizations made, also control for region-specific characteristics. In this paper the focus lies on plastic packaging waste collection in Sweden, which shows significant differences among the 290 municipalities (see Section 2). What explains these differences in collection rates, and what role can be ascribed to important cost elements (e.g., population density) in the respective municipalities? What is the role of the differing municipal waste management policies in determining collection outcomes? These are some of the central questions to be addressed and analyzed in this paper. The purpose of the paper is therefore to analyze the main determinants of household collection rates of plastic packaging waste in Swedish municipalities. This is done using an econometric model based on cross-sectional data from 2002, which include economic, demographic, institutional and policy-related variables at the local level.

This research undertaking will permit an evaluation of the effectiveness of different policy initiatives at the municipal level, and thus generate results that can point towards local strategies to improve national collection rates. In addition, the analysis can indicate to what extent inter-municipal differences are due to important cost differences across regions. It is reasonable to assume that both private and environmental costs for different waste management schemes differ across municipalities. For example, the external costs arising from landfilling and burning are probably lower in sparsely populated areas than in urban areas, while marginal collection costs are likely to be relatively high in the former regions (e.g., Berglund, 2004). The responsible Swedish authorities do not appear to acknowledge this cost-heterogeneity in their instructions to the producers. Basically the only instruction is that the packaging collection should be nationwide, and in many cases entrepreneurs that are active in "high-cost" regions tend to obtain comparably high monetary compensations (Pihl, 2002; SEPA, 1996; Hage, 2007a). If our empirical analysis indicates that important region-specific collection cost elements have an insignificant impact on plastic packaging collection rates, this could be interpreted as support for the hypothesis that the spatial cost-effectiveness of the current policy scheme is low.

Previous research on the determinants of recycling levels has largely used case studies of one or more municipalities to analyze the determinants of recycling efforts (e.g., Duggal et al., 1991; Sterner and Bartelings, 1999; Thomas, 2001; Lyas et al., 2005; Tonglet et al., 2004; Dahlén et al., 2007). The few studies that make use of more comprehensive databases of either state- or country-wide observations tend to focus on either US conditions (e.g., Callan and Thomas, 1997; Jenkins et al., 2003) or on inter-country differences in recovery rates (e.g., Berglund and Söderholm, 2003; Van Beukering and Bouman, 2001). The present study focuses solely on the Swedish situation in a large number of municipalities 8 years after the producer responsibility ordinance was introduced. The fairly low level of geographic aggregation permits an analysis of the impact of municipality policies, geography and socio-economic factors on plastic packaging collection rates. As was noted above, by drawing on the results from the cross-section analysis, we pay particular attention to the spatial cost-effectiveness of the national Swedish policy design. This paper illustrates the fact that analyzing collection rates for household waste is not simply a matter of understanding household waste sorting behavior, but also of addressing the behavior of local governments and the companies responsible for the collection. These latter actors provide the infrastructure and the incentives, which in turn determine households' willingness to participate in recycling schemes (e.g., Ölander and Thogersen, 2005). Most importantly, and as noted above, one of the main contributions of the paper is to analyze how the design of – and the incentives provided by – the producer responsibility ordinance affect the possibility to attain the set policy goals in a cost-effective manner. Clearly, such knowledge must – in order to form the basis of policy recommendations – also be complemented by more in-depth studies of household behavior, including both qualitative and quantitative studies. For instance, the impact of moral and social norms is important for understanding inter-household recycling behavior (e.g., Bruvoll and Nyborg, 2004; Hage, 2007b; Thogersen, 1996). In our study on differences across municipalities, the importance of norms cannot be properly addressed. Different types of analysis (e.g., qualitative and quantitative) are complements, not substitutes, and the present paper therefore adds some – but not an entirely comprehensive – understanding of the driving forces behind the rate of plastics packaging waste collection in Sweden.

Section 2 of the paper provides a background to packaging recycling in Sweden with special emphasis on the collection of household plastic packaging waste. In Section 3 an econometric model of household plastic packaging collection intensity is presented as are a number of important data and model estimation issues. The empirical results are outlined and discussed in Section 4, while Section 5 provides some concluding remarks and implications.

2. Packaging waste collection in Sweden: policy scheme and outcome

The Swedish producer responsibility ordinance implies that producers should collect, remove and recover the packaging waste from consumers. However, they are not required to take care of *all* packaging waste; the ordinance regulates to what extent packaging waste should be collected and for what purpose the collected packaging waste should be used (recycling and/or energy recovery). In the case of plastic packaging, the producers are required to collect at least 70% of all plastic packaging waste (in terms of packaging weight). At least 30% of the plastic packaging should be recycled, and hence used as input in new plastic products. The rest of the collected packaging, 40% of all plastic packaging, is not allowed to end up in landfills but can instead be used for energy recovery purposes (Parliamentary Auditors, 1999).

Overall the Swedish producer responsibility is an ordinance with few detailed instructions. It obliges producers to provide suitable systems for collecting packaging waste and inform households about these systems. The Swedish Environmental Protection Agency (SEPA) – that has the authority to outline instructions for the producers – has required that the collection should be nation-wide (SEPA, 1996). Municipalities are responsible for supervising the collection within their own borders. Households have the responsibility to clean and sort the packaging waste and transport it to drop-off recycling stations. Although producers have the economic responsibility for the packaging waste, households do not receive any economic compensation for their effort.

In order to comply with the producer responsibility, retailers and producers have founded four joint material companies that administrate the collection and recycling of packaging waste. One of these, Platskretsen AB (PAB),[1] administers plastic packaging waste (with the exception of PET-bottles).[2] All material companies form the service organizations Svenska Förpackningsinsamlingen AB (SFAB) and Reparegistret AB (REPA). SFAB's task is to coordinate the different responsibilities of the material companies. For instance, they establish and operate recycling stations and inform packaging consumers about the collection and recycling system (SFAB, 2000). Through REPA the material companies can offer nation-wide coverage of packaging waste collection. Individual producers can fulfill their producer responsibility if they join REPA; they then pay a packaging fee to REPA based on the weight of their packaging. In the case of plastic packaging materials; in 2004 this fee amounted to SEK 2.70 (USD 0.4) per kg (REPA, 2004). This is redistributed to the material companies to cover the costs of collection. 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. In 2002, 10,000 firms had joined REPA and together they represented about 90% of all packaging materials used in Sweden (SEPA, 2002).

Fig. 1 summarizes the system for collection and recycling of household plastic packaging waste in Sweden. The arrows marked HPPW indicate the physical flow of hard plastic packaging waste,[3] and the ''payment'' arrows show which actors are compensated for their work. In order to facilitate the collection of plastic packaging waste from recycling stations, PAB has engaged different collection entrepreneurs.[4] These entrepreneurs put out and empty the containers at the recycling stations and transport the plastic waste to plastic recyclers. The recyclers are also engaged by PAB. They clean and process the plastic waste into new plastic materials. PAB owns the plastic up to this stage and compensates both collection entrepreneurs and plastic recyclers. PAB sells the new plastic material to plastic product producers. Overall PAB's operations are to be 90% financed by the packaging fees, while the remaining 10% of the revenues come from the sales of new plastic material (PAB, 2004; SEPA, 2004b).

Table 1 shows how well PAB have fulfilled the policy targets of at least 70% recycling, including 30% material recycling and 40% energy recovery. The figures indicate that in 2004 PAB collected 37% of total plastic packaging material consumed. However, only 19% of total plastic packaging consumption is recycled into new material, and PAB thus experiences problems in fulfilling the 30% target. Still, the recycling levels show a slowly increasing trend, and PAB almost fulfills the goal for total recovery (19% + 18% + 30% = 67%). PAB has reported to SEPA that one important reason for burning a large part of the collected plastic packaging waste relates to different quality problems (SEPA, 2002). For instance, non-marked plastic packaging and especially plastic laminate is not only uneconomical to recycle but also technically impossible. Other problems relate to contaminated and poorly sorted plastic waste.

Given the focus on household plastic packaging collection in this paper, it is worth noting that in 2001 Swedish households consumed about 95,000 tons of plastic packaging and that the business sector (firms) consumed about 50,000 tons (SEPA, 2002). Nevertheless, 42% of all collected plastic packaging waste was collected from households, while as much as 57% was collected from the producers. Consequently, in 2001 only 19% of the household plastic packaging waste was collected by PAB. 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 Swedish households. Fig. 2 indicates the

[1] The other three include Svensk Kartongåtervinning AB (SKAB) (paper and cardboard packaging), Svenska Metalkretsen AB (SMAB) (metal packaging), and RWA Returwell AB (RWAB) (corrugated cardboard packaging). In 2006 SKAB and RWAB merged to form the new entity Returkartong AB.

[2] The recycling of returnable PET-bottles is organized by Svenska Returpack through a deposit-refund system, and this part of the plastic packaging waste stream is not analyzed in this paper. We focus only on plastic packaging waste for which neither advance disposal fees nor any refund payment are involved.

[3] Due to production constraints, PAB is only interested in rigid plastic packaging (e.g., bottles and containers) for material recycling purposes. Flexible plastic packaging waste (e.g., bags and film) is typically recovered through energy recovery.

[4] 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. Finally, 22 entrepreneurs only act in one municipality.

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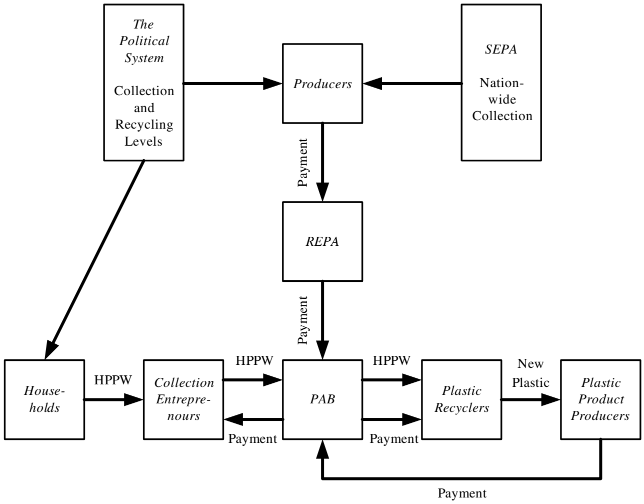

Fig. 1. The structure of household plastic packaging waste collection in Sweden.

Table 1
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	2004
Material recycling from PAB	11	13	20	16	15	13	16	18	19
Energy recovery from PAB	2	8	16	16	17	15	17	18	18
Energy recovery from household waste[a]	n.a.	n.a.	n.a.	(27)	(34)	(32)	(31)	32	30

Sources: SEPA (2001b, 2002, 2003, 2004a, 2006).
[a] Data collected by the municipalities. The data for the period 1999–2002 are not included in the official packaging collection result.

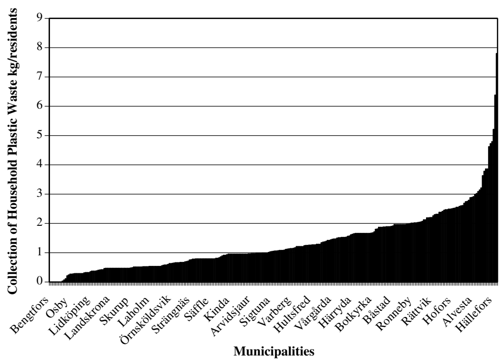

Fig. 2. Collection of household plastic packaging waste in 289 Swedish municipalities, 2002 (kg/resident) (*source*: SFAB (2004)).

amount (in kg) of household plastic packaging waste per resident that was collected in Swedish municipalities in 2002. The average municipality collected 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., Bengtsfors) did not collect any household plastic packaging waste, and Älvdalen was the municipality that collected the most, 7.8 kg per resident. These observed differences across municipalities form the basis of the empirical investigation in this paper.

The discussion above shows that this paper's focus on household plastic packaging is motivated for a number of reasons. The collection of plastic packaging displays large differences across Swedish municipalities, and overall the plastic recycling scheme has not fulfilled the targets of the producer responsibility ordinance. Furthermore, Swedish households consume more plastic packaging than the business sector, and an important strategy to improve recycling rates could be to increase collection from households. If this is to be achieved, however, a proper understanding of the main drivers of plastic packaging collection from the household sector is needed.

3. The econometric model and data

We model the household plastic packaging collection rate of a municipality as the annual collection in terms of kg used plastic packaging per inhabitant. The determinants of the collection rate are assumed to include local policies, geographic/demographic factors, socio-economic characteristics, and environmental preferences, as well as the features of the entrepreneurs engaged in each municipality. There exist no *a priori* theoretical reasons to specify a certain functional form for the regression equation to be estimated, and for this reason we follow previous studies (e.g., Callan and Thomas, 1997) and specify a linear econometric model of the following form[5]:

$$\begin{aligned}\mathrm{CR}_i = {} & \alpha_0 + \alpha_1\mathrm{FEEVOL}_i + \alpha_2\mathrm{FEEWE}_i + \alpha_3\mathrm{DIST}_i \\ & + \alpha_4\mathrm{URB}_i + \alpha_5\mathrm{POP}_i + \alpha_6\mathrm{BIG\ CITY}_i \\ & + \alpha_7\mathrm{AGE}_i + \alpha_8\mathrm{INC}_i + \alpha_9\mathrm{EDU}_i \\ & + \alpha_{10}\mathrm{WOMEN}_i + \alpha_{11}\mathrm{UNEMP}_i + \alpha_{12}\mathrm{HOUSE}_i \\ & + \alpha_{13}\mathrm{IM}_i + \alpha_{14}\mathrm{NEWIM}_i + \alpha_{15}\mathrm{ENVM}_i \\ & + \alpha_{16}\mathrm{ENVH}_i + \sum_{n=1}^{17}\beta_n D_n \end{aligned} \tag{1}$$

where CR_i equals the plastic packaging collection rate in kg per inhabitant in municipality i (excluding thus the deposit-driven PET-bottle collection system). Cross-section data for 252 Swedish municipalities for the year 2002 are used in the estimations. In 2002 there were 290 municipalities in Sweden, but due to data limitations 38 municipalities had to be excluded from the sample.[6] Table 2 summarizes the independent variables used in the econometric model. Information on some of the independent variables was not available for 2002, and in these cases we had to revert to information for adjacent years. Overall the discussion below indicates that this should not be a major problem since most of the variables used – including the policy-related ones (e.g., waste management fees) – have tended to be very stable over time.

The selection of independent variables has been heavily influenced by a review of the existing literature, including both quantitative and qualitative studies as well as meta-studies (e.g., Hornik et al., 1995). We first note that Swedish municipalities have implemented different types of waste management fees. Some use so-called *volume-based pricing* programs, while others have implemented *weight-based fees* for household waste collection. A third group of municipalities employ flat fees, indicating that no explicit economic incentive exists to leave packaging waste at the recycling station. In the weight-based programs, households pay a certain amount per kg unsorted waste. The volume-based fees include: (a) the opportunity to choose longer garbage collection intervals and hence pay less; (b) the opportunity to share a garbage container and the garbage fee with neighbours; and (c) the opportunity to pay based on the size of the garbage container (Villaägarnas riksförbund, 2004). The results from previous studies suggest that weight-based schemes may be more effective in reducing waste than other regimes, at least if the problem of illegal dumping can be avoided (e.g., Sterner and Bartelings, 1999; Dahlén et al., 2007). In 2002, 20 Swedish municipalities, 7%, had introduced weight-based fees, in particular for private house owners (SEPA, 2001a). The results on the impact of volume-based fees are more mixed. Fullerton and Kinnaman (1996) find that volume-based waste pricing decreases garbage volumes but not the garbage weight, and Jenkins et al. (2003) conclude that volume-based waste pricing does not appear to have a significant effect on the rate of household recycling.

The *distance* between the municipality center and the plastic recycling 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 was noted above, in the Swedish case this cost disadvantage may often be neutralized by higher monetary compensation levels for the collected household plastic packaging waste. Pihl (2002) confirms that entrepreneurs operating far away from recycling industries and in sparely populated areas get higher compensation for their collection of packaging waste (compared to those operating in densely populated areas). This is clearly a violation of the cost-effectiveness principle. However, the plastic packaging waste compensation is decided in secret negotiations 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 are not publicly available we cannot test this hypothesis empirically.

If it is the case that collection in high-cost municipalities is compensated through higher compensation, it is also reasonable to presume that other cost factors as well will only have minor impacts on reported collection rates. In addition, since we are uncertain about the exact shape of the collection cost function at the municipal level, it is useful to test for the impact of several types of cost indicators. High *urbanization rates* and *densely populated* municipalities imply shorter distances for households and material companies. For this reason these variables should lower the transport cost for both households and material companies. Still, high population urbanization rates and densely populated areas could also drive up land prices and hence the material companies' costs for establishing recycling stations. This implies the presence of one positive transport cost effect and one negative land cost effect associated with high urbanization rates and population densities. Which of them dominates in practice

[5] We also tested a log-linear specification, and the overall results were fairly similar to the ones reported here.

[6] The plastic packaging waste is sometimes reported for a group of (normally two) municipalities. In these cases, the total collection rates from these greater areas have been allocated to the respective municipalities based on the total population in each single municipality (SFAB, 2004). Clearly this causes some error in the data used, but overall the size of this error should not be significant as this procedure was only employed in a limited number of cases.

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Table 2
Variable definitions and sources

Variables	Definitions and units	Sources
Dependent variable		
CR	The amount of plastic packaging collected per resident in 2002 (kg)	SFAB (2004)
Policy variables		
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)
Geographic and demographic variables		
DIST	Distance between each municipality and the closest plastic recycling industry (km)	SRA (1999)
URB	Urbanization rate, i.e., the share of the population living in densely populated areas in 2001 (%). A densely populated area is defined as a group of buildings not more than 200 m apart from each other and having at least 200 inhabitants	SCB (2002a)
POP	The population density, i.e., the total population divided by the total land area of the municipality (in km^2) in 2002	SCB (2003)
BIG CITY	Dummy (1) for municipalities with at least 800 residents per km^{2}in 2002, and (0) if less than 800	SCB (2003)
Socio-economic variables		
AGE	Average age of the population (year) in 2002	SCB (2004)
INC	Average income for people between 20 and 64 years in 2000 (SEK)	SCB (2003)
EDU	People with at least 3 years at university divided by total population In 2003 (%)	KFAKTA (2003)
WOMEN	Women as a share of total population in 2002 (%)	SCB (2003)
UNEMP	Open unemployment rate for people between 16 and 64 years in 2002, annual average (%)	AMS (2004)
HOUSE	The share of private houses in each municipality in 1995 (%)	KFAKTA (2003)
IM	Immigrants, foreign-born outside the Nordic countries as a share of total population in 2001 (%)	KFAKTA (2003)
NEWIM	New immigrants, foreign citizens with 0-4 years in Sweden as a share of total population in 2002 (%)	SCB (2003)
Environmental preferences		
ENVM	Dummy for "environmental preferences" in the local government, 1 if the Green party was represented in the local 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 national election (%)	SCB (2002b)
Collection entrepreneur dummies		
D	Dummies for selected packaging collection entrepreneurs (see main text)	SFAB (2004)

remains thus an empirical question.[7] One hypothesis is that the relationship between population density and/or urbanization rate on the one hand and collection costs on the other hand 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 possess land that could be rented to the material companies at relatively favorable charges, but that cheap land is much scarcer in very dense cities. *Second*, small- and medium-sized cities in Sweden generally have relatively small city centers. Hence, here it is possible for the material companies to establish their recycling centers just outside the city center but still avoid 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 in congested areas. *Fourth*, congested cities often have problems with the traffic situation. The above suggests that there could well be a positive urbanization/population density effect but there could also be a negative *big city* effect. This notion is also supported by an ongoing debate in Stockholm, the capital of Sweden, about who should pay for household waste collection within the borders of the property (e.g., Fastighetstidningen, 2004).[8] However, as suggested earlier, the impact of these regional cost differences in collection can be offset by PAB's pricing policy. An essentially

[7] Berglund and Söderholm (2003) find, using country data, that increased urbanization and population density rates generally imply higher waste paper recovery rates, but that these effects are much weaker in developed compared to developing countries. It should also be noted that in our data sample of Swedish municipalities the urbanisation rate and the population intensity variables are not highly correlated (the correlation coefficient equals 0.4).

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

non-economic explanation for a negative big city effect may also be that the impact of social norms, i.e., norms sanctioned (directly or indirectly) by other people, is less pronounced in the more anonymous big cities.

A number of socio-economic variables are also included in the empirical investigation. After consulting a number of previous studies, Schultz et al. (1995) report that the relationship between *age* and US household recycling efforts appears to be ambiguous. 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 is positive but less than one. This implies that households with higher *income* will not necessarily be more willing to allocate more resources for environmental improvements and hence collect more packaging than low-income households. The opportunity cost for the households must also be considered. The collection of packaging is a time-consuming activity for the household so the opportunity cost for packaging collection should also increase with income. Consequently, this means that the impact of income on collection behavior will be determined by a income effect and a negative opportunity cost effect. Which of these two effects will dominate is an empirical question. However, as noted above, many of the earlier empirical studies have found that household recycling efforts in developed countries tend to be positively correlated with income (e.g., Callan and Thomas, 1997; Berglund and Söderholm, 2003). A number of US studies also present evidence in support of a positive relationship between *education* and household recycling efforts (e.g., Schultz et al., 1995; Callan and Thomas, 1997).

Schultz et al. (1995) find no relationship between *gender* and household recycling outcomes, and we test whether this conclusion also holds in the Swedish case. The rate of *unemployment* could also matter. One possible explanation for this is that the opportunity cost of the time spent on waste packaging sorting is likely to be lower for unemployed people, and one can therefore expect that they will (ceteris paribus) spend relatively more time on waste sorting activities. The Swedish Consumer Agency (2001) also concludes that perception about the opportunity cost of time is one of the most important determinants of recycling behavior in Sweden. Clearly, other factors may also play a role in explaining any relationship between unemployment and waste collection rates. The *type of housing* may be an important determinant of recycling efforts. It is worth noting that very few single family households in Sweden can benefit from curbside recycling services. However, it is reasonable to expect that private house owners have more space for storing used packaging, and they are more likely to own a car and also to have easy access to the car. This suggests that collection could, ceteris paribus, be higher in areas with a large share of private house owners. However, the fact that some apartment houses have packaging collection within the borders of the property (i.e., so-called property-close collection) may offset this impact (e.g., Mattsson et al., 2003). According to SEPA (2003), in 2002 about 25% of all apartment houses had packaging waste collection within the property, while the remaining households in this category had to transport the waste to collection stations.[9] The data on the share of private houses are only reported for the situation in 1995. Still, since housing construction activities in Sweden were unusually low during the 1990s, this should not imply major problems for the investigation. Finally, *immigrants*, especially newly arrived immigrants from outside the Nordic countries, are not used to Swedish laws and 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 who have lived in Sweden for a long time. The empirical literature lacks tests of this hypothesis, and in this paper we make a distinction between the share of immigrants in general and the share of newly arrived immigrants.

As has been noted above, concern for the environment is likely to influence plastic packaging collection rates. In the empirical investigation we add two independent variables that explicitly attempt to address the strength of *environmental preferences* in the respective municipalities. The more emphasis the local government puts on environmental issues, the more likely it is that it will attempt to facilitate packaging collection. There exist a number of ways through which this can be achieved. The material companies often inform packaging consumers through the municipalities' waste information; more effective waste information should naturally increase the packaging collection levels. The municipalities also rent sites for the recycling stations and provide building permits.[10] In the empirical analysis we use the influence of the Green party in the local government as a proxy for the "environmental preferences" in the policy arena. It is also reasonable to believe that households that are concerned about the environment should be motivated to sort packaging waste (e.g., Schultz et al., 1995; Hornik et al., 1995). We test this plausible hypothesis by employing the share of votes on the Green party in the 2002 national government election. Clearly this is only a rough proxy for environmental concern. Still, one should also note that strong support for the Green party may indicate the presence of strong social norms in the household recycling sector, i.e., people (including those that vote on other parties) feel that other households expect them to perform waste sorting activities (e.g., Bruvoll and Nyborg, 2004).

Finally, we add intercept dummy variables, D_n, for those 17 entrepreneurs that collect plastic packaging waste in

[9] Hage (2007b) uses survey data from Swedish households to explain recycling efforts at the household level, and he concludes that access to property-close collection has a significant positive impact on the recycling of household packaging waste.

[10] There exists also an important economic reason for the municipalities to "support" the 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; SOU, 2001).

more than one municipality. These variables are mainly to be regarded as control variables that may, for instance, capture the presence of firm-specific characteristics such as differences in collection productivity, strategies and/or negotiation skills. As will be suggested below, in some cases they will also tend to reflect regional differences that are not entirely addressed by the other independent variables in the econometric investigation.

All variables that are hypothesized to influence household plastic packaging collection are likely to be exogenously given. This implies that we can estimate the econometric model using ordinary least square (OLS) techniques. In order to implement the model in Eq. (1) empirically we add a error term, ε_i, which is assumed to be normally distributed with zero mean and constant variance. Since the presence of heteroscedasticity is a common problem in cross-sectional data (e.g., Greene, 1997) this was tested for using the Breusch–Pagan–Godfrey test. Based on this test the null hypothesis of homoscedasticity was rejected at the 1% level. The presented t-statistics and significance levels have therefore been calculated using the White estimator for the heteroscedasticity-consistent covariance matrix.

Different specifications of the model were tested. Following the hypothesis of a non-linear relationship between, for instance, population density and collection rates, it was tested if a quadratic form of the population density variable could explain inter-municipality differences better. Similar tests were performed for the urbanization rate variable. Different variables were also used to measure environmental preferences in the municipality, e.g., an environmental municipality ranking (see Miljöeko, 2001). However, none of these specifications added any new information.

4. Empirical results and discussion

The parameter estimates for the household packaging waste collection model are presented in Table 3. The goodness-of-fit measure, R^2-adjusted, for the regression model is 0.22. This means that 78% of the inter-municipality differences were 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 Greene (1997), low R^2-adjusted estimates are very common when using cross-sectional data. It is also obvious that all five categories of variables help explain parts of the observed inter-municipality differences in collection rates.

The parameter estimates for the *policy variables* show some interesting results. The coefficient for weight-based fee is, as expected, positive and it is statistically significant at the 5% level. A municipality that has introduced a weight-based fee has, ceteris paribus, on average 372 g more plastic packaging waste collected per resident than municipalities in which other types of waste management fees are used. This is a noteworthy difference considering the fact that in 2002 an average Swedish municipality collected 1.33 kg per resident. The coefficient for the volume-based fee variable is statistically insignificant, indicating that these types of fees are less effective in encouraging plastic packaging collection than are weight-based fees. This result can partly be explained by the fact that it is generally relatively easy for households to compress plastic bottles and containers (Sterner and Bartelings, 1999; Jenkins et al., 2003).

Overall the *geographic and demographic variables*, all proxies for the marginal collection costs, appear to have limited influences on collection rates. First, the distance between the municipality and the recycling industry does not seem to matter for collection levels. One 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 are 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, ceteris paribus, lower recycling levels than the municipalities covered by the reference entrepreneurs.[11] Second, out of the three coefficients representing urbanisation rate, population density and "big city", respectively, only the coefficient for big city is statistically significant (at the 7% level). The coefficient for the variable "big city" implies that there may be a negative congestion effect for the collection of household plastic packaging as was hypothesized in Section 3, but the relative lack of social norms may also play a role in explaining the low collection in the big cities. The results indicate that households in a congested city collect on average 530 g less plastic packing per resident than do households in smaller cities. The insignificant results for the urbanization rate and population density variables contradict the findings from earlier research focusing on inter-country differences (Berglund and Söderholm, 2003). It is also worth noting that the inclusion of entrepreneur dummies does not seem to affect these latter results; both coefficients remain statistically insignificant even after removing these dummy variables.

The above suggests that overall the costs of plastic packaging collection do not seem to matter much for the collection outcome, implying that the collection of household plastic packaging waste in Sweden may not be performed in a cost-effective manner. One plausible explanation for this is, as noted above, the pricing negotiations between PAB and the entrepreneurs. Practical experience suggests that entrepreneurs that collect plastic packaging in "high-cost" municipalities obtain a higher monetary compensation for their collection activities compared to entrepreneurs that are active in municipalities that score high on urbanization rate and population density (Pihl, 2002). There exists thus no built-in incentive to lower collection efforts in sparsely populated areas.

[11] The coefficient for distance actually becomes statistically significant at the 5% level if all entrepreneur dummies are excluded from the econometric estimation.

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Table 3
Parameter estimates for plastic packaging collection rate model

Variables	Coefficient	Expected signs	*t*-Statistics	Statistical significance
Constant	−1.493	?	−0.265	0.791
Policy variables				
Weight-based fee (FEEWE)	0.372**	+	2.007	0.046
Volume-based fee (FEEVOL)	−0.412	?	−1.596	0.112
Geographic and demographic variables				
Distance to recycling industry (DIST)	−0.000	−	−0.177	0.860
Urbanization rate (URB)	0.001	+	0.121	0.904
Population density (POP)	0.000	+	1.284	0.200
Big city (BIG CITY)	−0.530*	−	−1.835	0.068
Socio-economic variables				
Average age (AGE)	0.033	?	0.999	0.319
Average income level (INC)	0.003	?	1.074	0.284
Education level (EDU)	−0.027**	+	−2.156	0.032
Women share (WOMEN)	−0.023	?	−0.212	0.832
Unemployment rate (UNEMP)	0.233**	+	2.382	0.018
Private house (HOUSE)	0.016*	?	1.959	0.051
New Immigrants (NEWIM)	−0.183***	−	−3.083	0.002
Immigrants (IM)	0.075***	?	3.292	0.001
Environmental preferences				
Local government (ENVM)	0.065	+	0.492	0.623
Households (ENVH)	0.201***	+	3.510	0.001
Collection entrepreneur dummies (D)				
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: Parameter estimates adjusted for heteroscedasticity. *, **, ***Indicate statistical significance at the 10%, 5%, and 1% levels, respectively.

Overall the *socio-economic variables* add quite a lot to our understanding of plastic packaging collection rates. In the cases of age, income and gender there are, however, no statistically significant results. The coefficient for the education level is not only unexpected (negative sign), but it is also statistically significant at the 5% level. Still, it is worth noting that education and income are relatively highly correlated with a correlation coefficient of 0.69. The coefficient for unemployment is statistically significant at the 5% level. A 1% increase in the open unemployment rate yields a 233 g increase in household plastic waste collection per resident. Overall the above supports the hypothesis – much in line with Bruvoll et al. (2002) – that the opportunity cost of time for households seems to influence their collection activities as these activities become less intense with higher income/education and employment rates. The impacts of these variables, however, merit further investigation.

We further find that the coefficient for "share of private house" is positive and statistically significant at the 5% level. The model suggests that a 1% increase in the share of private houses in a municipality induces an increase in household plastic packaging waste collection by 16 g per resident. As was noted above, plausible explanations for this are that private house owners tend to have more space for storing used packaging materials, and they also gener-

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ally have easier access to a car. The coefficient for share of new immigrants has the expected negative sign and it is statistically significant at the 1% level. However, the coefficient for immigrants as a whole is positive and also statistically significant at the 1% 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 the Swedish language very well. This has a negative impact on waste sorting activities. However, over time they learn the language and pick up existing social norms of behavior, and the results suggest thus that immigrants as a group sort out more plastic packaging waste than the average Swedish citizen.

The fourth category of variables, *environmental preferences*, helps in explaining collection outcomes. Both the coefficient for Green party representation in the local government, and the coefficient for Green party support among households have the expected signs. The former coefficient is however not statistically significant, while the latter coefficient is statistically significant at the 1% level. A 1% increase in the share of Green party supporters implies an increase in the collection of household plastic packaging by 193 g per residents. Overall this suggests that environmental concern among citizens tend to matter more for recycling outcomes than do a strong position for green politicians in the local government.

The *entrepreneur dummies* also seem to explain some of the variance in household plastic packaging waste collection. It is difficult to form any *a priori* expectations about the signs of these coefficients. The results show that the coefficients for Sita, Kangos, Merab, Östgötafrakt and Karskoga are all negative and statistically significant at either the 1%, 5% or 10% levels. The coefficients for Vafab and NSR are positive and statistically significant at the 1% and 5% levels, respectively. The inter-entrepreneur differences could be explained by the presence of firm-specific characteristics such as collection productivity and/or negation skills. For instance, in municipalities where NSR is active, quite a lot of effort has been put on developing property-close collection at the household level. Still, it must also be acknowledged that the entrepreneurs do not possess very large opportunities to directly affect collection rates. Thus, in practice firm-specific attributes are unlikely to explain a great deal of the observed variance. It is equally plausible that these results simply are due to differences in municipality characteristics that are not entirely accounted for in the model used. For example, Sita, one of the three nation-wide entrepreneurs, tends to operate in major Swedish cities. As has been noted above, this is likely to reduce household plastic waste collection due to high land prices, and our ''big city'' variable may not fully reflect these congestion impacts. The other entrepreneurs, for whom statistically significant coefficients are reported, are all regional companies that operate within a constrained area (e.g., a specific county). For example, Kangos operates 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 different geographic and demographic variables. Again, however, it is possible that these could not fully address these impacts in a comprehensive way.[12]

Finally, in order to provide a rough test of the robustness of our empirical results, we have looked in more detail at the characteristics of the top-10 performing municipalities in terms of plastic packaging waste collection. First it is worth noting that Älvdalen, the municipality reporting by far the highest collection rate, stands out in the sense that it is the only municipality in Sweden in which (to our knowledge) households are required to bring *all* of their waste to drop-off recycling stations. This clearly makes it harder – and less meaningful – to throw plastic materials in the regular waste bin. Apart from this it is worth noting that municipalities using weight-based fees are clearly overrepresented in the top-10 group, and a vast majority of these 10 municipalities are also fairly small (around 10,000 inhabitants). These results are consistent with our previous findings concerning the positive influence of weight-based fees and the negative big city impact. Finally, municipalities using either NSR or Vafab as entrepreneurs are strongly overrepresented in the top-10 group, a result that is also evident in our empirical analysis (see Table 3). This strengthens the case that the choice of entrepreneurs may matter, as some of these (such as NSR) may be more prone to adopt property-close collection schemes.

5. Concluding remarks and implications

The purpose of this paper has been to analyze the determinants of inter-municipality differences in the collection of household plastic packaging waste in Sweden. Overall the results suggest that policy, geographic/demographic factors, socio-economic variables and environmental preferences all influence collection rates to various degrees. Some particularly interesting findings and implications are worth noting.

We first note that the different proxies for the marginal costs of plastic packaging collection in the respective municipalities do not appear to have a significant effect on collection outcomes. A reasonable explanation for this is that the compensation from the material companies varies depending on region and this tends to reduce regional cost differences in collection. This suggests that the national collection is cost ineffective, i.e., it should be possible to collect the same amount of household plastic pack-

[12] One example is the urbanization rate variable. According to the definition of an urban area in Sweden it is enough if a village has 200 residents if there are not more than 200 m between the buildings. The municipalities are also fairly large in terms of land area 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 an urban area but the collection cost for household plastic packaging collection could still be high.

aging waste at a lower total cost and/or increase collection rates without imposing a higher cost. Interestingly the ‘‘big city’’ variable has a statistically and an economically significant impact on collection, suggesting that there exists some critical level for the population density, at least within the existing system, at which marginal collection costs start to increase with increases in the density (due to, for instance, congestion effects and high land prices).

The above indicates that the Swedish society *could* save resources by paying more attention to regional cost differences. Still, purely on the basis of our study it is difficult and even inappropriate to outline strong policy recommendations. A move to a more cost-effective collection scheme would have both pros and cons. First, we have not considered the transaction costs involved, that is the costs of administering, monitoring and enforcing a new system. These may be high and offset any cost savings, but we still believe that they potentially could be kept low. The authorities need not necessarily set different collection targets for densely and sparsely populated regions, respectively, and then enforce each of these. It may be enough to reform the compensation scheme and let these economic incentives determine where collection will be made. We believe instead that one of the major drawbacks of a cost-effective scheme, in which spatial cost differences matter, may lie in the notion that there may exist a trade-off between the cost-effectiveness and the legitimacy of the policy. If people as well as politicians feel committed to waste recycling because it is one way of contributing to the public environmental good, they may have a negative attitude towards a policy that encourages spatial differences in collection efforts.

The empirical results also suggest that the actors within the existing waste management regime can affect the collection outcome in various ways. *First*, the municipalities can increase collection by increasing the reliance on weight-based waste fees. However, even though this seems to be an effective method for increasing the collection of packaging materials, undesirable side-effects of such fees must also be acknowledged. A weight-based waste fee will give households an incentive for illegal waste disposal; empirical research in the US provides some support for this notion (Fullerton and Kinnaman, 1996). *Second*, it also seems as if the material companies and the municipalities could increase the collection of packaging collection by spending more resources on informing new immigrants about the collection system. However, the costs for such information campaigns should of course also be considered.

This study also indicates that several issues could form the focus of future research efforts. *First*, it is important to investigate the determinants of the collection of packaging materials other than plastic. It is possible that the effects of different policy schemes differ across materials. Jenkins et al. (2003) confirm this notion for US recycling behavior. *Second*, US 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, and our data material has not permitted an explicit test of this hypothesis. *Third*, this study shows – in line with many other investigations – that weight-based fees are effective in promoting waste collection. However, it is important to obtain more knowledge about the costs of this system, including the risks for illegal disposal. *Finally*, although this paper has focused primarily on the cost-effectiveness of recycling policies, one should also acknowledge that other policy criteria are important as well. In brief, household recycling is, as noted above, both a matter of economics and morals (e.g., Thogersen, 1996), and this must be acknowledged when implementing policies that target the waste sorting efforts of households. Economically efficient policies may prove inefficient in the longer-run if they undermine overall policy legitimacy. Clearly these issues also merit further investigations.

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III

Household Plastic Waste Collection in Swedish Municipalities: A Spatial-Econometric Approach*

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Abstract

This paper investigates the main determinants of collection rates of household plastic packaging waste in Swedish municipalities. This is done by the use of spatial econometric methods based on cross-sectional data for 282 Swedish municipalities in 2005. The empirical results suggest that the collection of plastic packaging is positively related to collection in neighboring municipalities. The analysis also shows that municipalities that employ weight-based waste management fees generally experience higher collection rates than those municipalities in which volume-based fees are used. The presence of curbside recycling and a high intensity of recycling drop-off stations, both measures that facilitate recycling efforts by creating the infrastructural and logistic mechanisms that enable people to translate their motivation into recycling action, provide important explanations for why some municipalities perform better than others. Overall the impacts on collection outcomes of a number of important regional cost variables, such as distance to recycling industry, urbanization rate and population density, turn out, though, both statistically and economically insignificant. An important explanation for this is that the (fixed) monetary compensations from the material companies to the collection entrepreneurs in Sweden vary depending on region and is typically higher in high-cost regions. This implies that the plastic packaging collection in Sweden may be performed in a cost ineffective manner.

Key words: collection rates, recycling, plastic packaging, regional differences, Sweden, producer responsibility, cost effectiveness, waste management, spatial econometrics.

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1. Introduction

Environmental policy and its related objectives are often formulated at the national level, but the primary responsibility for policy implementation and monitoring is typically assigned to local actors (e.g., Callan and Thomas, 1997). The latter could include, for instance, local authorities, private entrepreneurs, and individual households. This implies in turn that the effectiveness of many environmental policies is influenced by a sometimes rather complex network of actors, the organization of their respective activities, and the incentives they face.

The 1994 Swedish producer responsibility ordinance for packaging waste is a good example of this policy set-up. The legislation outlines national recycling goals and states that the producers have the physical and the economic responsibility for the packaging waste, i.e., they are obliged to provide suitable systems for the collection and the recycling of packaging waste. The producers must also consult with the municipalities about the recycling systems. According to the ordinance, the municipalities in Sweden should also take an active role in informing households about the collection systems as well as in supervising the collection schemes. Households are obliged to clean and sort out packaging waste from other waste, and transport used packaging materials to assigned drop-off stations. Policies aimed at encouraging households' recycling efforts, including waste management fees, infrastructural measures etc., are designed and implemented at the municipal level. This implies that recycling initiatives and outcomes are not uniform across the country. In part any regional differences in collection rates are also the result of geographical, socio-economic and demographic factors, which only to a limited extent may be amenable to change.

The above suggests that any meaningful analysis of the impact of recycling policies and measures must rely on local observations, and to assure the validity in any generalizations made, also control for region-specific characteristics. In this paper the focus lies on plastic packaging waste collection in Sweden, which shows significant differences across the country's 290 municipalities (see also section 3). What explains these differences in collection rates, and what roles can be ascribed to important regional cost elements and policy measures in the respective municipalities? This is the central research question to be addressed and analyzed in this paper.

The purpose of the paper is to provide a spatial econometric analysis of the main determinants of household plastic waste collection in Swedish municipalities. This is done using cross-sectional data set from 2005, which include economic, demographic, institutional and policy-related variables at the municipal level. When using sample data collected with references to location, there are reasons to suspect spatial dependence between the observations, i.e., one observation associated with location i depends on other observations at locations $i \neq j$ (e.g., Anselin, 1988; LeSage, 1999). In the household waste case there may be several reasons why the recycling outcome in one municipality is influenced by policies and behavior in neighboring municipalities. For instance, in some regions neighboring municipalities have started a jointly owned waste company. This implies that information, collection systems and perhaps even other policies may be standardized in the greater region. It is also reasonable to suspect that neighboring municipalities will meet and exchange experiences and in this way influence each other's policies and collection rates. If such spatial interactions exist ordinary least square (OLS) methods produce parameter estimates that are biased and inefficient. For this reason we use spatial econometric methods that explicitly deal with the incorporation of spatial autocorrelation in the econometric estimations.

The research undertaking in this paper is important for at least two main reasons. *First*, it permits an evaluation of the effectiveness of different policy initiatives at the municipal level, and thus generates results that can point towards local strategies to improve national collection rates. The current waste management policy of Sweden is focused on facilitating households' recycling activities, not the least through increased reliance on so-called property-close waste collection (SEPA, 2005b). This includes, for instance, easier access to drop-off stations and containers as well as more intense use of curbside recycling. Moreover, an increasing number of municipalities have implemented weight-based waste management fees, and thereby introduced an explicit economic incentive for households to undertake waste sorting activities. It is however so far unclear if the above policy measures have had the desired impacts on recycling outcomes, but our data set permits explicit empirical tests of these impacts. *Second*, the analysis can indicate to what extent inter-municipal differences in collection rates are due to important cost differences across regions. It is reasonable to assume that both the private and the environmental costs for different waste management schemes differ across municipalities. For instance, the external costs arising from landfill and burning are probably lower in sparsely populated areas than in urban areas, while marginal collection costs are likely to be relatively high in the former regions (e.g., Berglund, 2004). The Swedish legislation does not appear to acknowledge this cost-heterogeneity in their instructions to producers. Basically the only instruction is that the packaging waste collection should be nationwide, and in practice entrepreneurs that are active in "high-cost" regions tend to obtain comparably high monetary compensations (Pihl, 2002; SEPA, 1996; Forselius, 2007; Hage, 2007a). If our empirical analysis indicates that important region-specific collection cost elements have an insignificant impact on plastic packaging collection rates, this could be interpreted as support for the hypothesis that the spatial cost-effectiveness of the current policy scheme is low.

Previous research on the determinants of recycling levels has largely used case studies of one or more municipalities to analyze the determinants of recycling efforts (e.g., Duggal et al., 1991; Sterner and Bartelings, 1999; Thomas, 2001; Tonglet et al., 2004; Lyas et al., 2005; Dahlén et al., 2007). Some aggregate studies focus on household behavior (e.g., Kipperberg, 2006), but these studies do not address the incentives facing collection entrepreneurs and the producers of, for instance, packaging waste. The few studies that make use of more comprehensive databases of either state- or country-wide observations – thus taking into account not only household-specific decisions – tend to focus on either U.S. conditions (e.g., Callan and Thomas, 1997; Jenkins et. al. 2003) or on inter-country differences in recovery rates (e.g., Berglund and Söderholm, 2003; Van Beukering and Bouman, 2001). The present study focuses solely on the Swedish situation in a large number of municipalities eleven years after the producer responsibility ordinance was introduced. The fairly low level of geographical aggregation permits an analysis of the impact of local policies, geography and socio-economic factors on plastic packaging collection rates. As was noted above, by drawing on the results from the cross-section analysis – and controlling for spatial interactions – we pay particular attention to the effectiveness and the spatial cost-effectiveness of the Swedish policy design. By achieving the above this paper illustrates that analyzing collection rates for household waste is not simply a matter of understanding household waste sorting behavior, but also of addressing the behavior of local governments and the companies responsible for the collection. These latter actors provide the infrastructure and the incentives, which in turn determine households' willingness to participate in recycling schemes (e.g., Ölander and Thogersen, 2005). Given that Sweden has been one of the international forerunners in the promotion of household waste recycling, the results presented below could also provide important lessons for other countries.

Section 2 provides a background to the Swedish producer responsibility for packaging with special emphasis on the collection of household plastic packaging waste. In section 3 an econometric model of household plastic waste collection is presented as are a number of important data definitions. Section 4 discusses selected model estimation issues, not the least those related to the spatial characteristics of the data employed in the investigation. The empirical results are outlined and discussed in section 5, while section 6 provides some concluding remarks and implications.

2. Packaging Waste Collection in Sweden: Policy Scheme and Outcome

The Swedish producer responsibility ordinance implies that producers should collect, remove and recover the packaging waste from consumers. However, producers are not required to take care of *all* packaging waste; the ordinance regulates to what extent packaging waste should be collected and for what purpose the collected packaging waste should be used (i.e., recycled and/or energy recovered). In the case of plastic packaging, the producers are required to collect at least 70 percent of all plastic packaging waste (in terms of the packaging weight). At least 30 percent of the plastic packaging should be recycled, hence used as input in new plastic products. The remainder of the collected packaging, 40 percent of the total, is not allowed to end up on landfills but can instead be used for energy recovery purposes (Parliamentary Auditors, 1999).

The Swedish producer responsibility is an ordinance with few detailed instructions concerning policy implementation and enforcement. It obliges producers to provide suitable systems for collecting packaging waste, and the Swedish Environmental Protection Agency (SEPA) – that has the authority to outline instructions for the producers – requires that the collection should be nationwide (SEPA, 1996). The municipalities are responsible for informing households about the collection system as well as supervising the collection within their own borders. Households have the responsibility to clean and sort the packaging waste and transport it to drop-off recycling stations. Although producers have the economic responsibility for the packaging waste, the households do not receive any economic compensation for their efforts.

In order to comply with the producer responsibility, the retailers and the producers have established four joint material companies that administrate the collection and recycling of packaging waste. One of these, Platskretsen AB (PAB),[1] administers plastic packaging waste (with the exception of PET-bottles).[2] All material companies form the service organizations Förpacknings- och tidningsinsamlingen AB (FTI)[3] and Reparegistret AB (REPA). FTI's task is to coordinate the different responsibilities of the material companies. For instance, they

[1] The other three include Svensk Kartongåtervinning AB (SKAB) (paper and cardboard packaging), Svenska Metalkretsen AB (SMAB) (metal packaging), and RWA Returwell AB (RWAB) (corrugated cardboard packaging). In 2006 SKAB and RWAB merged to form the new entity Returkartong AB (RAB).

[2] The recycling of returnable PET-bottles is organized by Svenska Returpack through a deposit-refund system, and this part of the plastic packaging waste stream is not analyzed in this paper. We focus only on plastic packaging waste for which neither advanced disposal fees nor any refund payments are involved.

[3] In August 2007 the organization of the producer responsibility was partly altered. REPA moved all of their operations to FTI. Furthermore, FTI will also take the responsibility for the household collection from PAB, SMAB, and RAB.

establish and operate recycling stations and inform packaging consumers about the collection and recycling system (FTI, 2006). 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; they then pay a packaging fee to REPA based on the weight of their packaging. In the case of plastic packaging materials, in 2005 this fee amounted to SEK 2.0 (USD 0.27) per kg (REPA, 2005). The revenues generated by this fee are redistributed to the material companies to cover the costs of collection. As a rule, the fees are paid by the packaging filler, packer or re-packer for products made in Sweden, and by the importer in the case of foreign products. In 2002, about 10000 firms had joined REPA and these represented about 90 percent of all packaging materials used in Sweden (SEPA, 2002).

Figure 1 summarizes the system for collection and recycling of household plastic packaging waste in Sweden. The arrows marked HPPW indicate the physical flow of hard plastic packaging waste, and the "payment" arrows show the actors that are compensated for their work. In order to facilitate the collection of plastic packaging waste from recycling stations, PAB has engaged different collection entrepreneurs.[4] These entrepreneurs put out and empty the containers at the recycling stations and transport the plastic waste to plastic recyclers. The recyclers are also engaged by PAB. They clean and process the plastic waste into new plastic materials. PAB owns the plastic up to this stage and provides compensation to both collection entrepreneurs and plastic recyclers. PAB sells the new plastic material to plastic product producers.[5] Overall PAB's operations are to 90 percent financed by the packaging fees, while the remaining 10 percent of the revenues stem from the sales of new plastic materials (SEPA, 2004b; Schyllander, 2007).

In Sweden there exist about 6000 recycling drop-off stations for a total Swedish population of roughly 9 million (Funck, 2006). This means that on average about 1500 individuals "share" a station, and since Sweden is a quite sparsely populated country some households may be located far away from their nearest drop-off station. Only households are allowed to use these drop-off stations, and since the data we employ in this paper are derived from the amounts collected at these drop-off stations, the analysis ought not to be affected by the plastic packaging waste generated by private firms. Firms that need to get rid of plastic

[4] These entrepreneurs can be divided into three categories. 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. Finally, 22 entrepreneurs only act in one municipality. (PAB, 2007; Schyllander, 2007)

[5] Since 2006, PAB only owns the plastic packaging waste during the collection process and then sells the plastic packaging waste to the recyclers (Schyllander, 2007).

packaging waste should instead contract a waste entrepreneur for collecting their plastic packaging waste.[6]

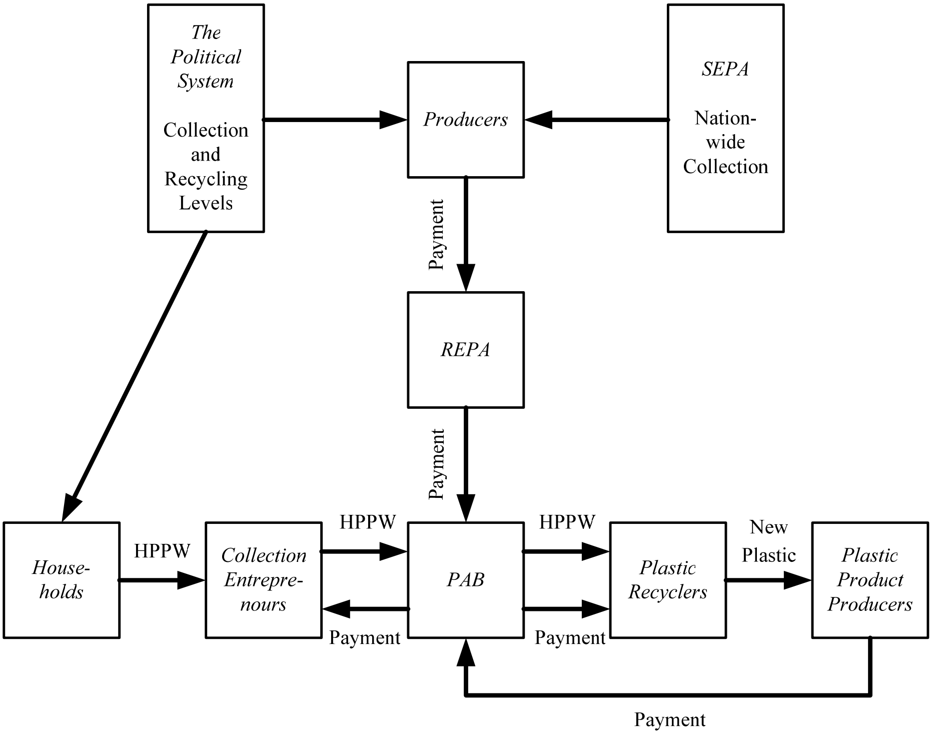

Figure 1: The Structure of Household Plastic Packaging Waste Collection in Sweden

In recent years the issue of convenience in household recycling has gained increased policy attention.[7] For instance, in a Government bill (2002/03:117) it is stated that recycling within the borders of the property – so called property-close collection – should be the main type of collection of used packaging in multi-family dwellings and that the producers should

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[6] If the firm generates significant amounts of relatively clean plastic packaging waste this could actually be profitable because there is a positive market value for the plastic waste. However, for companies that generate small amounts of plastic packaging waste the collection cost will often outweigh the waste value. Hence, for these latter companies there exist incentives to make use of the household collection system. Nevertheless, according to Georgsson (2007) very few companies (illegibly) use the drop-off stations that are assigned for household. An additional reason for this is that the recycling bins at these stations are designed to make it difficult to drop-off big plastic packaging waste.

[7] In Sweden there are two different forms of collection achieving this aim. First, many multi-family dwellings have installed central sorting houses or rooms within the borders of the property. Here people living in apartments can leave their plastic packaging waste in specially assigned bins. This system is called *property-close collection*. Second, in about 15 municipalities (in 2005) the local authorities have organized *curbside collection* of packaging for single-family dwellings.

be economically responsible for setting up this system. In late 2004, the Swedish Waste Council also suggested that producers should be responsible for establishing property-close collection schemes as well as curbside recycling whenever this is possible (SEPA, 2006b). They also propose that the producers have this responsibility even if the market value of the collected packaging is not sufficient to cover the costs of these systems.[8] The Swedish waste management plan also emphasizes the importance of implementing collection systems that are perceived as convenient by households (SEPA, 2005b). Furthermore, SEPA (2006b) concludes that the current state of knowledge about the effects on private costs, social costs, and environmental effects from curbside collection and property-close collection is insufficient for giving clear guidance or national regulations on the matter. Hence, more research is clearly needed and the present paper could thus play a role in filling this knowledge gap.

Table 1 shows how well PAB has fulfilled the national policy targets of at least 70 percent recycling, including 30 percent material recycling and 40 percent energy recovery. The figures indicate that in 2005 PAB collected 43 percent of the total plastic packaging material consumed. However, only 24 percent of total plastic packaging consumption was recycled into new material; PAB thus experiences problems in fulfilling the 30 percent target. Still, the recycling levels show a slowly increasing trend, and PAB fulfills the goal for total recovery (24+19+30=73 percent). PAB has reported to SEPA that one important reason for burning a large part of the collected plastic packaging waste relates to different quality problems (SEPA, 2002). For instance, non-marked plastic packaging and especially plastic laminate is not only uneconomical to recycle but also technically impossible. Other problems relate to contaminated and poorly sorted plastic waste.

Table 1: 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	2004	2005
Material recycling from PAB	11	13	20	16	15	13	16	18	19	24
Energy Recovery from PAB	2	8	16	16	17	15	17	18	18	19
Energy Recovery from household waste*	n.a.	n.a.	n.a.	(27)	(34)	(32)	(31)	32	30	30

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

Sources: SEPA (2001b, 2002, 2003, 2004a, 2005a, 2006a).

[8] This is not the case today, curbside recycling schemes are mainly financed by the municipalities and the property-close collection is financed by the multi-family dwelling house owners. This situation has been criticized by the government (Government Bill 2002/03:117) and SEPA (2006b) since it is claimed to reduce the producers' incentives for improving the recyclability of their products.

In 2001 Swedish households consumed about 95000 tons of plastic packaging and the business sector (firms) consumed about 50000 tons (SEPA, 2002). Nevertheless, 42 percent of all plastic packaging waste were collected from households but as much as 57 percent were collected from the producers. Consequently, in this year only 19 percent of the household plastic packaging waste were collected by PAB. Hence, in the efforts to increase collection rates, there appears to exist a rather large reserve of non-collected plastic packaging waste in Swedish households. Figure 2 indicates the amount (in kg) of household plastic packaging waste per resident that was collected in Swedish municipalities in 2005. In 2005 the average municipality collected about 1.97 kg of plastic packaging waste per resident. However, the collection rates differ significantly across municipalities with a minimum value of 0.04 kg per resident, and a maximum value of 5.71 kg per resident. These observed differences across municipalities form the basis of our empirical investigation.

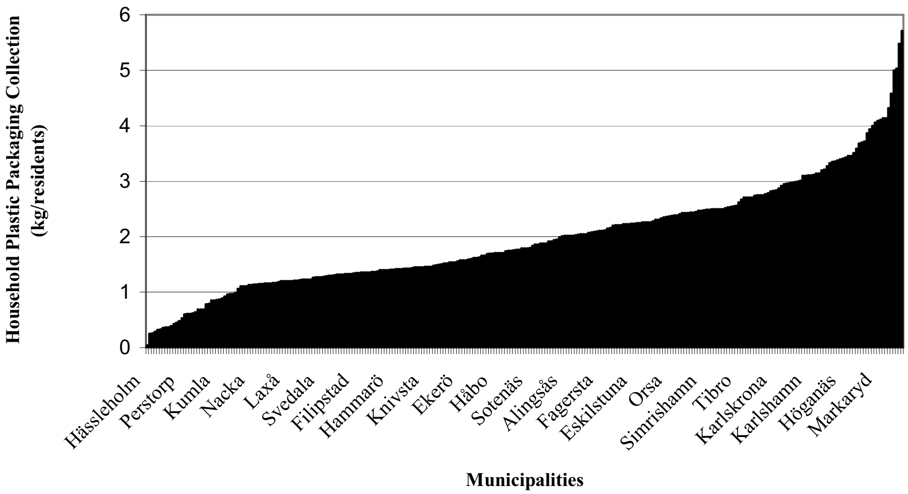

Figure 2: Collection of Household Plastic Packaging Waste in 283 Swedish Municipalities, 2005 (kg/resident)

Note: Seven municipalities have been excluded because they did not deliver their plastic packaging waste to PAB (e.g. Gotland), and for this reason their collection results are no attainable.

Source: Staaf (2006).

The map of Sweden displayed in Figure 3 indicates how plastic packaging collection rates in different municipalities are related to the weighted average collection rate in the neighbouring municipalities.

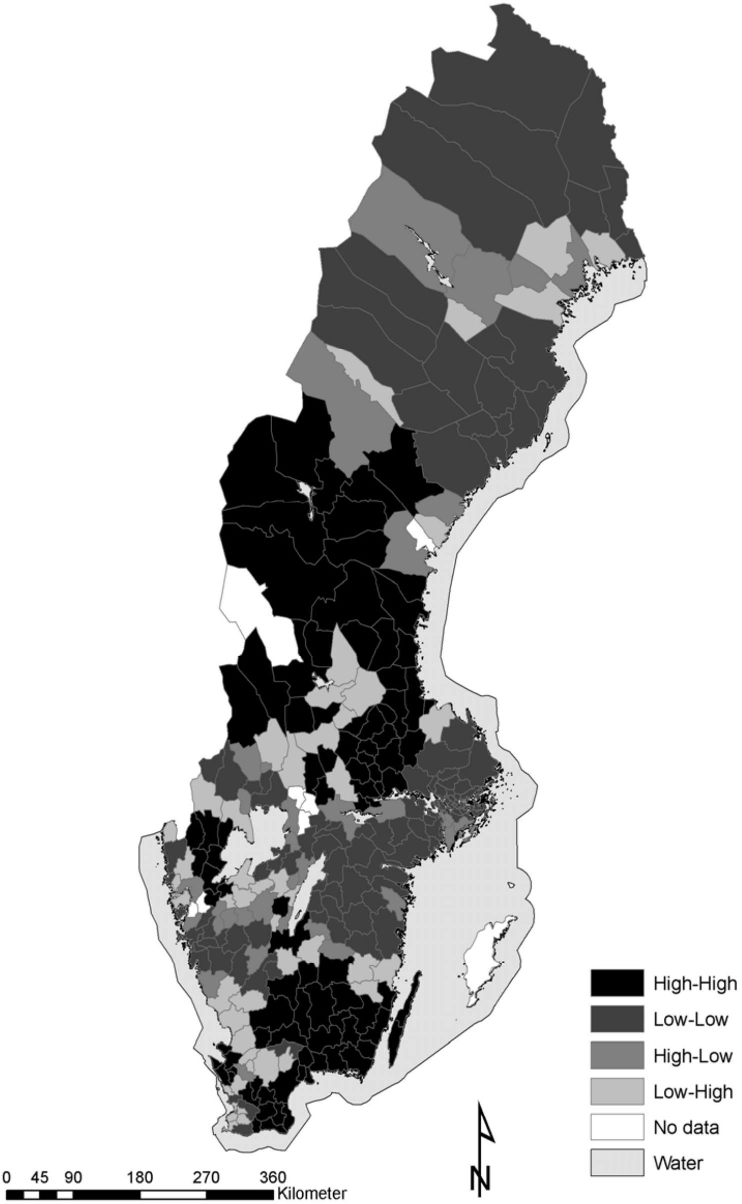

Figure 3: Moran Scatterplot Map for Plastic Packaging Collection in Sweden

Note: The spatial weight matrix used when constructing this map is defined as a row-standardized inversed squared distance matrix, with a distance cut-off at the samples first quartile (182 km). The coloured boxes indicate the plastic collection rate in a chosen municipality (first word) and in their neighbours (second word).

For example, the low-high label specifies that a given municipality collects relatively low amounts of plastic packaging waste (per person) but their neighbours report relatively high collection rates. Figure 3 illustrates that Sweden appears to have several plastic packaging collection "clusters", some with low and some with high rates of collection. Thus, this strengthens our a priori suspicion that there exists a spatial dependence in the plastic packaging collection. Still, the map does not give us any information if these observations are statistically significant, thus motivating the use of spatial econometric methods.

In sum, the discussion above shows that this paper's focus on household plastic waste is motivated for a number of reasons. The collection of plastic packaging displays large differences across Swedish municipalities, there appear to exist spatial relations in collection, and overall the plastic recycling scheme has not fulfilled the targets of the producer responsibility ordinance. Furthermore, Swedish households consume more plastic packaging than the business sector, and an important strategy to improve recycling rates could be to increase collection from households. If this is to be achieved, however, a proper understanding of the main drivers of plastic packaging collection from the household sector – including the role of different policy measures – is needed.

3. The Econometric Model and Variable Definitions

We model the household plastic packaging collection rate of a municipality as the annual collection in terms of used plastic packaging in kg per inhabitant. The collection rate determinants are assumed to include local policies, geographic/demographic factors, socio-economic characteristics, environmental preferences as well as the nature of the entrepreneurs engaged in each municipality. There exist no *a priori* theoretical reasons to specify a certain functional form for the regression equation to be estimated, and for this reason we follow previous studies (e.g., Callan and Thomas, 1997) and specify a linear econometric model of the following form:[9]

$$\begin{aligned}
&Plastic_i = \alpha_0 + \alpha_1 Feewe_i + \alpha_2 Curbside_i + \alpha_3 Drop_i + \alpha_4 Dist_i + \alpha_5 Urb_i \\
&+ \alpha_6 Popden_i + \alpha_7 Bigcity_i + \alpha_8 Age_i + \alpha_9 Inc_i + \alpha_{10} Edu_i + \alpha_{11} Unemp_i \\
&+ + \alpha_{12} SFD_i + \alpha_{13} Newim_i + \alpha_{14} Tim + \alpha_{15} Envm_i + \alpha_{16} Envh_i + \sum_{n=1}^{3} \beta_n D_n
\end{aligned} \tag{1}$$

[9] We also tested a log-linear specification, and the overall results were fairly similar to the ones reported here. Moreover, employing a logarithmic form of the dependent variable did not lead to fewer problems connected with non-normally distributed residuals.

where $Plastic_i$ equals the plastic packaging collection rate in kg per inhabitant in municipality i (excluding thus the deposit-driven PET-bottle collection system). Cross section data for 282 Swedish municipalities for the year 2005 are used in the estimations. In 2005 there were 290 municipalities in Sweden, but due to data limitations eight municipalities had to be excluded from the sample.[10] Table 2 summarizes the independent variables used in the econometric model, and in Appendix B some descriptive statistics for these variables are presented. Information on some of the independent variables was not available for 2005, and in these cases we had to revert to information for adjacent years. Overall the discussion below indicates that this should not be a major problem since most of the variables used – including the policy-related ones (e.g., waste management fees) – have been very stable over time.

The selection of independent variables has been heavily influenced by a review of the existing literature, including both quantitative and qualitative studies as well as meta-studies (e.g., Hornik et al., 1995). We first note that Swedish municipalities have implemented different types of waste management fees. A large majority of the municipalities use so-called volume-based pricing programs while the others have implemented *weight-based fees* for household waste collection. In the weight-based programs households pay a certain amount per kg unsorted waste. The volume-based fees include the opportunities to: (a) choose longer garbage collection intervals and hence pay less; (b) share garbage container and the garbage fee with neighbours; and (c) pay for the size of the garbage container (Villaägarna, 2006). The results from previous studies suggest that weight-based schemes may be more effective in reducing waste than other regimes, at least if the problem of illegal dumping can be avoided (e.g., Sterner and Bartelings, 1999; Dahlén et al., 2007). In 2005, 25 Swedish municipalities, 9 percent, had introduced weight-based fees, in particular for private house owners (Villaägarna, 2006). Previous research on the impact of volume-based fees presents more mixed results. Fullerton and Kinnaman (1996) find that volume-based waste pricing decrease garbage volumes but not the garbage weight, and Jenkins et al. (2003) conclude that volume-based waste pricing does not appear to have a significant effect on the rate of household recycling. In the empirical investigation we include a dummy variable which equals one (1) for those municipalities that employ weight-based fees (and zero otherwise).

[10] The plastic packaging waste is sometimes reported for a group of (normally two) municipalities. In these cases, the total collection rates from these greater areas have been allocated to the respective municipalities based on the total population in each single municipality (FTI, 2006). Clearly this causes some error in the data used, but overall the size of this error should not be significant as this procedure was only employed in a limited number of cases. Furthermore, this particular limitation of the data used provides an additional reason for employing an econometric estimation technique that explicitly acknowledges the presence of spatial interactions.

Table 2: Variable Definitions and Sources

Variables	Description and units	Source
Dependent variable		
Plastic	The amount of household plastic packaging collected per resident in 2005 (kg).	Staaf (2006)
Policy variables		
Feewe	Dummy for weight-based waste fees in 2005, 1 if yes and 0 if no.	Villaägarna (2006)
Curbside	Dummy for curbside collection of plastic packaging in private houses (single-family dwellings) in 2005, 1 if yes and 0 if no.	Villaägarna (2006), and Mattson (2006)
Drop	The number of household plastic packaging recycling stations divided by the municipality's total land area measured in km^2 for 2005 (and controlling for the urbanization rate).	Funck (2006), SCB (2005a), and own calculations
Geographic and demographic variables		
Dist	Distance between each municipality and the nearest recycling industry (km) in 2005.	SRA (1999), and Schyllander (2007)
Urb	Urbanization rate, i.e., the share of the population living in densely populated areas as of December, 31, 2004. 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.	KFAKTA (2006)
PopDen	Population density, i.e., total population divided by the municipality's land area measured in km^2 as of December, 31, 2004.	SCB (2005a)
Big City	Dummy (1) for municipalities with 800 residents per km^2 or more as of December, 31, 2004, and 0 if less than 800.	SCB (2005a)
Socio-economic variables		
Age	Average age of the population as of December, 31, 2004.	SCB (2005a)
Inc	Average income for people between 20 and 64 years as of December 31, 2003 (kSEK).	KFAKTA (2006)
Edu	People with at least three-years university degree divided by total population (%) as of December, 31, 2003.	KFAKTA (2006)
Unemp	Open unemployment rate for people between 16 and 64 years in 2005, annual average (%).	AMS (2006)
SFD	The share of single-family dwellings in 2005 (%).	SCB (2005b)
Tim	Total immigrants, foreign born outside the Nordic countries as a share of total population (%) as of December, 31, 2004.	KFAKTA (2006)
Newim	New immigrants, foreign citizens with 0-4 years in Sweden as a share of total population as of December, 31, 2004 (%).	SCB (2005a)
Environmental preferences		
Envm	Dummy for environmental preferences in the municipality government, 1 if green party was represented in the municipality government in 2003, and 0 if not.	KFAKTA (2006)
Envh	"Environmental preferences" in households, measured by the share of votes on the Green party in the 2002 parliamentary election (%).	SCB (2002)
Collection entrepreneur dummies (D)		
PNE	Dummy for private-owned packaging entrepreneurs with a nation-wide collection in 2005, 1 if yes and 0 if no.	PAB (2007)
PRE	Dummy for private-owned packaging entrepreneurs with a region-wide collection in 2005, 1 if yes and 0 if no.	PAB (2007)
MRE	Dummy for municipality-owned packaging entrepreneurs with a region-wide collection in 2005, 1 if yes and 0 if no.	PAB (2007)
MLE	Dummy for municipality-owned packaging entrepreneurs with a collection only in one municipality in 2005, 1 if yes and 0 if no.	PAB (2007)

We also use a dummy variable to examine the influence of the presence of *curbside recycling* for plastic packaging in the case of single-family dwellings. Clearly, such arrangements ought to (ceteris paribus) have a positive impact on collection rates. Moreover, we have gathered data on the total number of recycling drop-off stations for household plastic waste in each municipality, and by dividing these numbers with the respective land areas (in square kilometers) and controlling for the urbanization rate (see also below), we obtain a measure of the relative intensity of *drop-off stations* in each municipality. The higher this intensity is, the higher collection rate would be expected.

The *distance* between the municipality center and the plastic recycling 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 household waste. However, as was noted above, in the Swedish case this cost disadvantage may often be neutralized by higher monetary compensation levels for the collected household plastic packaging waste. Pihl (2002) and Forselius (2007) confirm that entrepreneurs operating far away from recycling industries and in sparely populated areas obtain a higher fixed compensation for their collection of packaging waste (compared to those operating in densely populated areas).[11] This is clearly a violation of the cost-effectiveness principle, and suggests that the values of the fixed compensations for plastic waste collection in different municipalities are probably important for explaining differences in collection rates. However, these fixed values are determined in secret negotiations between PAB and the respective entrepreneurs so we cannot explicitly test this hypothesis in the empirical investigation.

If it is the case that collection in high-cost municipalities is compensated through higher monetary compensation, it is also reasonable to presume that other cost factors as well will only have minor impacts on reported collection rates. In addition, since we are uncertain about the exact shape of the collection cost function at the municipal level, it is useful to test for the impact of several types of cost indicators. High *urbanization rates* and *densely populated* municipalities imply shorter distances for households and material companies. For this reason these variables should lower the transport cost for both households and material companies. Still, high population urbanization rates and densely populated areas could also drive up land prices and hence the material companies' costs for establishing recycling stations. This implies the presence of one positive transport cost effect and one negative land

[11] The compensation to the entrepreneurs consists of one variable and one fixed component. The variable compensation is official and equal for all companies that sell plastic packaging waste to PAB, while the fixed component thus varies across different municipalities (Forselius, 2007)

cost effect associated with high urbanization rates and population densities. Which of them dominates in practice remains thus an empirical question.[12] One hypothesis is that the relationship between population density and/or 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 possess land that could be rented to the material companies at relatively favorable charges, but such cheap land is much scarcer in dense cities. *Second*, small- and medium-sized cities in Sweden generally have relatively small city centers. Hence, here it is possible for the material companies to establish their recycling centers just outside the city center but still avoid 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 in congested areas. *Fourth*, congested cities often have problems with the traffic situation. The above suggests that there could well be a positive urbanization/population density effect but there is also a negative *big city* effect. This notion is supported by an ongoing debate in Stockholm, the capital of Sweden, about who should pay for household waste collection within the borders of the property (e.g., Fastighetstidningen, 2004).[13] However, as suggested earlier, the impact of these regional cost differences in collection can be offset by PAB's pricing policy. An essentially non-economic explanation for a negative big city effect may also be that the impact of social norms, i.e., norms sanctioned (directly or indirectly) by other people (e.g., Thogersen, 1996), is less pronounced in the more anonymous big cities.

A number of socio-economic variables are also included in the empirical investigation. After consulting a number of previous studies, Schultz et al. (1995) report that the relationship between *age* and U.S. household recycling efforts is ambiguous. Kriström and Riera (1996) as well as Hökby and Söderqvist (2003) find that the demand for environmental improvements is

[12] Berglund and Söderholm (2003) find, using country data, that increased urbanization and population density rates generally imply higher waste paper recovery rates, but that these effects are much weaker in developed compared to developing countries. It should also be noted that in our data sample of Swedish municipalities the urbanisation rate and the population intensity variables are not highly correlated (the correlation coefficient equals 0.4).

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

a “necessary” good; hence the income elasticity for environmental improvements is positive but less than one. This indicates that households with low incomes will allocate relatively more resources to environmental goods than households with high incomes. Consequently, this implies that low income households may recycle higher degrees of the packaging waste. Furthermore, recycling is also a time consuming activity for the households. This means that the individual recycling levels will also be influenced by their opportunity cost of the time spent on recycling efforts. Of course, the opportunity cost for recycling will increase with income. Again, this implies that low income households should recycle higher degrees of packaging. Consequently, this suggests that we have two effects that both conclude that we should have a negative relationship between income and recycling rates. However, some empirical studies find a positive relationship between income and recycling rates in developed countries (e.g., Callan and Thomas, 1997; Berglund and Söderholm, 2003). Clearly, these contradictory findings need more attention. A number of U.S. studies also present evidence in support of a positive relationship between *education* and household recycling efforts (e.g., Schultz et al., 1995; Callan and Thomas, 1997).

Schultz et al. (1995) find no relationship between *gender* and household recycling outcomes, and we test whether this conclusion also holds in the Swedish case. The rate of *unemployment* could also matter. One possible explanation for this is that the opportunity cost of the time spent on waste packaging sorting is likely to be lower for unemployed people, and one can therefore expect that these will (ceteris paribus) spend relatively more time on waste sorting activities. The Swedish Consumer Agency (2001) concludes that perception about the opportunity cost of time is an important determinant of recycling behavior in Sweden.

The type of housing may be an important determinant of recycling efforts. It is worth noting that very few single-family dwellings (*SFD*) in Sweden can benefit from curbside recycling services (and we also control for this service by using the above-mentioned dummy variable). However, it is reasonable to expect that people living in these types of houses have more space for storing used packaging, and they are more likely to own a car and also to have easy access to the car compared to people living in multi-family dwellings. This suggests that collection could, ceteris paribus, be higher in areas with a large share of single-family dwellings. However, the fact that rather many multi-family dwellings have access to property-close collection schemes may offset this impact (e.g., Mattsson et al., 2003).[14] According to

[14] Hage (2007b) uses survey data from Swedish households to explain recycling efforts at the household level, and he concludes that access to property-close collection has a significant positive impact on the recycling of household packaging waste.

SEPA (2006b), about 46 percent of all multi-family dwellings had packaging waste collection within the property in 2006, while the remaining households in this category had to transport the packaging waste to drop-off stations. The data on the share of single-family dwellings are reported for the 2005 situation.

Finally on socio-economic determinants, we note that *immigrants*, especially newly-arrived immigrants from outside the Nordic countries, may not be acquainted with Swedish laws, regulations and may have difficulties in understanding the language. This makes it reasonable to expect that their participation in packaging collection programs are generally lower than for people who have lived in Sweden for a long time. The empirical literature lacks tests of this hypothesis, and in this paper we make a distinction between the shares of immigrants in the municipalities in general and the share of newly arrived immigrants.

As was noted above, concern for the environment is likely to influence plastic packaging collection rates. In the empirical investigation we add two independent variables that explicitly attempt to address the strength of *environmental preferences* in the respective municipalities. The more emphasis the local government puts on environmental issues the more likely it is that it will attempt to facilitate packaging collection. There exist a number of ways through which this can be achieved. For instance, more effective waste information should naturally increase the packaging collection levels. The municipalities also rent sites for the recycling stations and provide building permits.[15] In the empirical analysis we use the influence of the Green party in the local government as a proxy for the "environmental preferences" in the policy arena. It is also reasonable to believe that households that are concerned about the environment should be motivated to sort packaging waste (e.g., Schultz et al., 1995; Hornik et al., 1995). We test this hypothesis by employing the share of votes on the Green party in the 2002 central government election. Clearly this is only a rough proxy for environmental concern. Still, one should also note that strong support for the Green party may indicate the presence of strong social norms in household recycling, i.e., people (including those that vote on other parties) feel that other households expect them to perform waste sorting activities (e.g., Bruvoll and Nyborg, 2004).

Finally, we add intercept dummy variables, D_n (n = 1…3), for three types of collection entrepreneurs (and a fourth one, *MLE*, is used as a reference category). These variables are mainly to be regarded as control variables that may, for instance, capture the presence of

[15] There exists also an important economic incentive for the municipalities to support the 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; SOU 2001:102).

differences in collection productivity, incentive structure, strategies and/or negotiation skills. As will be suggested below, in some cases they will also tend to reflect regional differences that are not entirely addressed by other independent variables in the econometric analysis.

4. A Spatial-Econometric Specification

As has been noted above, it is reasonable to suspect that municipalities and collection entrepreneurs are influenced by the performance and the policy measures of neighboring municipalities when making their own decisions. A central aspect of most economic markets is spatial interaction, externalities, spill-overs, copy-catting, etc., and these phenomena are no less likely to occur in the waste management field. When economic behaviour is modelled, spatial dependence of this type calls for spatial econometric methods. In a spatial context the dependence among two municipalities may, and usually does, operate in both directions. Initially, dependence due to measurable relations such as distance, barriers, and congestions are easily controlled for. However, there may still be signs of dependence due to omitted variables or tacit relations. One common simplification is first to assume equally strong dependence in both directions. A spatial weights matrix, W, is thereafter constructed to proxy for these multiple dependencies between observations that are to be included in the estimation. This matrix is assumed to be a matrix of known elements and in which all elements on the main diagonal equal zero. There are various ways to construct this matrix; most common is a binary approach based on unit contiguity or a matrix based on some distance decay function. The spatial weights matrix used in this paper is defined as a row-standardized inversed squared distance matrix, with a distance cut-off at the samples first quartile (182.3 km).[16] This fits theoretically well with the gravity model approach that applies Newton's law stating that the attractive force between two bodies is directly related to their size and inversely related to the distance between them.

Exploratory data analysis is a good starting point in order to test for spatial dependence (spatial autocorrelation). In this way we may confirm or reject the hypothesis that objects of similar values are more clustered than by pure chance. At our disposal are a couple of global tests for spatial autocorrelation, such as Moran's I and Geary's C (Moran 1948; Geary 1954; Cliff and Ord 1973, 1981). The notion of global tests refers to the fact that they consider the overall data pattern and only return a single value which either confirm or rejects the

[16] This weight matrix was also tested by using the inverse distance. Additional cut-off distances (2 and 3 quartiles) by using inverse distance and inverse squared distance, and 5 respectively 10 nearest neighbours' weight matrices, were also tested (all row-standardized). See also below.

hypothesis. No specific information is given about the prevailing pattern. When this is of interest, local tests may be used (Getis and Ord, 1992; Ord and Getis, 1995, 2001; and Anselin, 1995).

For our purposes the Moran's I for the tested matrices are presented in Appendix A. According to the results, the chosen matrix (the row-standardized inversed squared distance matrix, with a distance cut-off at the samples first quartile) and the row-standardized nearest neighbors' matrices detect the highest spatial autocorrelation. The choice of the former matrix is motivated by the regression diagnostic. Regression results using the five nearest municipalities weight matrix are discussed in section 5. More detailed results when using these matrices are available from the authors on request.

The next step is typically to solve for spatial dependence in a regression analysis. Two kinds of spatial dependencies are commonly assumed to potentially contaminate the analysis. The first arises when variables of adjacent observations move together due to common or correlated unobservable variables, i.e., lack of stochastic independence between observations. This dependence leads to inefficient estimates if left unsolved. The problem is discussed at length in Cliff and Ord (1972, 1973). We add an error term, ε, to equation (1), and a partition of the error term into two parts, together with a given spatial weights matrix *W*, solves this spatial dependence problem. The model is known as the Spatial Error Model;

$$
\begin{aligned}
y &= X\beta + \varepsilon \\
\varepsilon &= \lambda W \varepsilon + \xi \qquad (2)
\end{aligned}
$$

where λ is the spatial autocorrelation coefficient and ξ is a vector of independently and identically distributed errors (i.i.d.) errors. In our case this could be reflected in the fact that the collection rate in one municipality is a function of municipality-specific characteristics but also of omitted variables in neighbouring municipalities. Estimates based on ordinary least squares (OLS) methods would in this case remain unbiased but would lose the efficiency property.

The second and more serious problem of spatial dependence is present when spatial correlation in the dependent variables between observations exists. Such dependence leads to both biased and inefficient estimates (Anselin, 1988). This problem may be solved for by including the dependent variable of the other observations on the right hand side of the equation lagged by a spatial weights matrix. This model is known as the Spatial Lag Model:

$$y = \rho Wy + X\beta + \varepsilon \qquad (3)$$

where y is the dependent variable (i.e., *plastic* in equation (1)), ρ is a spatial autoregressive coefficient, X is a vector of independent variables, and ε is a vector of i.i.d. errors. As usual, additional problems during estimation, such as heteroskedasticity may occur. These problems can be solved similarly as by standard econometric methods.

The classical estimation routine towards a proper model specification under the potential influence of spatial dependence is, for instance, given in Florax et al. (2003). The initial model is estimated by means of OLS. The residuals are then used to test the hypothesis of no spatial dependence caused by an omitted spatial lag or by spatially autoregressive errors by use of two Lagrange Multiplier tests, the LM-lag test and the LM-error test (e.g., Anselin, 1988; Burridge, 1980). When this null hypothesis cannot be rejected (no spatial dependence is at hand) the results based on OLS may be used. However, in the event that the hypothesis is rejected, a new model should be estimated. The proper model is indicated by the most significant LM test. In case that only the LM-lag test is significant, the next step would be to estimate a Spatial Lag Model and a Spatial Error Model if the opposite results are indicated.

5. Empirical Results and Discussion

The regression results are presented in Table 3 below.[17] As suggested above, we begin to estimate the model by OLS, and the results from this estimation are presented in column 2. The null hypothesis of homoskedasticity is not rejected by the Koenker-Basset test but the assumption of normally distributed residuals is rejected by the Jarque-Bera test.[18] This calls for caution since the tests for spatial dependence are sensitive towards non-normally distributed residuals. The value for Moran's I is statistically significant at the 1 percent level, hence we could reject the null hypothesis of no spatial dependence. And, despite the lack of normality, the LM-tests indicate that spatial lag dependence is present.[19] Also the observation

[17] These estimations were performed using SPACESTAT version 1.91.

[18] The problem with non-normally distributed residuals can sometimes be solved by using the logarithm of the dependent variable in the regression. However, when testing for this, the Jarque-Bera in this case becomes even more statistically significant.

[19] The standard LM-tests for spatial lag and error are both highly significant, thus confirming the problem with spatial autocorrelation. As in this case, when both standard LM-tests are significant, we should consider the robust LM-tests for choosing the appropriate specification of the model. However, none of these are highly significant (the robust LM-test is significant at the 12 percent level). This implies that we should return to the standard LM-test and chose the most significant test. When doing so we could see that the standard test for spatial lag is more significant (9.64 > 7.33). As a result, we will use a spatial lag model in the estimation.

that collection sometimes is reported for a group of (normally two) municipalities implies that we have spatial lag correlation. We therefore continue and estimate the spatial lag model by Maximum Likelihood (ML). This is done despite its weakness in connection with non-normally distributed residuals. However, the benefit is that it provides the most insight in the form of tests available on our way towards a final model. The results from the ML estimation are given in column 3 of Table 3. The spatial lag parameter ρ is positive and highly statisticcally significant. This means that the collection of plastic packaging in one municipality is positively influenced by the amount that is collected in nearby municipalities. The last two tests in the ML column verify that we have indeed done a correct specification by including a spatial lag and not solved the spatial dependence problem by a spatial error model. On the other hand, the null hypothesis of homoskedasticity is also now rejected, something which may be due to the problem of non-normally distributed residuals.

To solve the problem of heteroskedasticity and non-normally distributed residuals a viable solution is to use a robust IV estimation (2SLS). The instruments used to estimate this model was spatially lagged exogenous variables as suggested by Kelejian and Robinson (1992). The estimation results are given in column 4. We note that the estimate for the spatial lag parameter (ρ) increases remarkably compared to the ML estimate. It is also noteworthy, that the coefficients for *Inc*, *Newim*, and *PRE* are no longer significant in the robust IV estimation. In column 5 we instead present a model estimated by bootstrap[20] with 999 permutations, another good alternative to ML in situations where heteroskedasticity may be present and the normality assumption is possibly invalid (Freedman and Peters, 1984a, 1984b; Anselin, 1988, 1990). When comparing these results to the results of the robust IV estimation we find that the parameter estimates for *Newim* and *PRE* become significant again. Otherwise, the results are very similar so we may conclude that the results are fairly robust for different model specifications.[21]

[20] The bootstrap is a robust estimator that uses random resampling technique for statistical inference. The procedure in this bootstrap is based on residuals and is suggested by Freedman and Peters (1984a, 1984b). First, an IV-estimation is done and the vector of estimated residuals *(e)* is calculated. Second, we use ***e*** for generating a vector of pseudo residuals ***(e*)*** by drawing them random with replacement; in our case this is done 282 times. Third, pseudo data for the vector of independent variable ***(y*)*** is calculated by using the vector of exogenous variables ***(X)***, the estimated parameters from the IV-estimation, and the ***e****. Fourth, new parameter estimates are now obtained by using the IV estimation on ***y**** and ***X***. This is the first permutation, and this step is repeated 998 times. Finally, the bootstrap parameter estimate is then calculated by the mean values from these permutations.

[21] This is also valid if estimating the model by using the row standardized 5 nearest municipalities weight matrix and bootstrap technique. For example, all the signs for parameter estimate are unchanged and the size for the parameter estimate is quite similar. However, this matrix will generate changes in the significance level for some parameter estimates. For instance, the parameter estimates for *PopDen* and *Inc* become significant at the 10 percent level, and the parameter estimate for *Newim* becomes significant at the 5 percent level.

Table 3: Parameter Estimates for Plastic Packaging Collection Rate Model

Variables	OLS		ML		IV-robust		Bootstrap	
	Estimate	t-value	Estimate	z-value	Estimate	z-value	Estimate	z-value
Rho			***0.379	3.51	***0.756	3.28	***0.744	5.08
Constant	2.22	0.94	2.05	0.91	0.821	0.45	1.67	0.77
Policy variables								
Feewe (+)	**0.377	1.96	**0.356	1.95	**0.388	2.33	**0.334	1.96
Curbside (+)	***0.786	2.93	**0.636	2.51	**0.680	2.28	**0.533	2.16
Drop (+)	**0.616	2.41	***0.645	2.68	**0.556	2.12	***0.661	2.89
Geographic and demographic variables								
Dist (-)	-0.0006	-1.23	-0.0003	-0.72	0.0006	-1.58	-0.0000	-0.25
Urb (+)	-0.0007	-0.12	-0.0035	-0.65	-0.0051	-1.19	-0.0056	-1.08
PopDen (+)	*-0.0004	-1.73	-0.0003	-1.62	-0.0002	-1.61	-0.0003	-1.60
Big City (-)	0.367	0.73	0.400	0.84	0.491	1.03	0.448	0.98
Socio-economic variables								
Age (?)	0.044	1.10	0.026	0.69	0.026	0.83	0.011	0.28
Inc (-)	**-0.010	-2.27	*-0.007	-1.78	-0.004	-0.96	-0.005	-1.20
Edu (+)	0.009	0.53	0.006	0.36	-0.006	-0.49	-0.0004	-0.02
Unemp (+)	-0.019	-0.34	-0.022	-0.40	-0.006	-0.12	-0.026	-0.49
SFD (?)	-0.002	-0.26	-0.004	-0.73	-0.006	-1.17	-0.007	-1.14
Newim (-)	*-0.119	-1.83	*-0.111	-1.82	-0.074	-1.40	*-0.102	-1.78
Tim (?)	0.038	1.31	0.038	1.40	0.022	0.86	0.038	1.54
Environmental preferences								
Envm (+)	-0.101	-0.76	-0.096	-0.76	-0.058	-0.58	-0.085	-0.66
Envh (+)	-0.022	-0.36	-0.026	-0.45	-0.020	-0.43	-0.013	-0.25
Collection entrepreneurs								
PNE (?)	0.018	0.08	0.034	0.16	0.002	0.01	0.055	0.26
PRE (?)	***-0.813	-2.85	**-0.647	-2.37	-0.443	-1.59	*-0.503	-1.87
MRE (?)	0.337	1.35	0.310	1.32	0.312	1.31	0.282	1.19
Diagnostics								
R^2	0.244		0.258		0.328		0.305	
R^2-adj	0.189							
F-test	***4.45							
Sq.corr			0.278		0.274		0.201	
LIK	-353.04		-348.42					
AIC	746.08		738.85					
SC	818.92		815.33					
Jarque-Bera	***102.47							
Koenker-Basset	22.53							
Breusch-Pagan			***57.75					
Moran's I	***4.00							
LM error	***7.33							
LM lag	***9.64							
Robust LM error	0.05							
Robust LM lag	2.36							
L-ratio lag			***9.24					
LM error			0.11					

Note: *, **, and *** indicate statistical significance at the ten, five, and one percent levels, respectively. The expected values of the respective parameter estimates are displayed in brackets in column 1.

The parameter estimates for the *policy variables* show some interesting results, and all three of them are statistically significant. The coefficient for weight-based fee is as expected positive and statistically significant (at least) at the 5 percent level. A municipality that has introduced a weight-based fee has on average, ceteris paribus, approximately 350 gram more plastic packaging waste collected per resident than municipalities in which volume-based fees are used. Furthermore, and in line with our initial conjecture, the coefficients for the curbside collection variable and the variable for the density of plastic packaging recycling stations in each municipality are both positive and statistically significant at the five percent level. Thus, these results suggest that municipalities that make use of a curbside collection system will, ceteris paribus, on average collect more than 500 grams additional plastic packaging waste per resident than municipalities without such schemes. These impacts are also economically significant considering the fact that in 2005 an average Swedish municipality collected about 2 kg per resident. The results therefore also suggest that measures to facilitate recycling efforts by creating the infrastructural and logistic mechanisms that enable people to translate any environmental motivation into recycling action may be effective. Thus, although social and moral norms also are important determinants of household recycling behavior (Thogersen, 1996; Hage 2007b), people also respond significantly to economic incentives in the waste management field. Indeed, any existing norms may well be strengthened as these types of incentives are introduced (Thogersen, 2003).

Overall the *geographic and demographic variables*, all proxies for the marginal collection costs at the municipal level, appear to have limited influences on collection rates. First, the distance between the municipality and the recycling industry does not seem to matter for collection levels. As commented on below, one explanation for this result could be that the coefficient for the dummy variables for some entrepreneurs may capture some of the differences in plastic packaging collection that are due to differences in distance. Second, none of the three coefficients representing urbanization rate, population density and "big city", respectively, are statistically significant. These results clearly contradict the findings from earlier research focusing on inter-country differences (Berglund and Söderholm, 2003).

The above suggests that overall the regional-specific costs of plastic packaging collection do not seem to matter much for the collection outcome, implying that the collection of household plastic packaging waste in Sweden may not be performed in a cost-effective manner. One plausible explanation for this is – as noted above – the pricing negotiations between PAB and the entrepreneurs. Practical experience suggests that entrepreneurs that collect plastic packaging in "high cost" municipalities obtain a higher fixed monetary compensation for their collection activities compared to entrepreneurs that are active in municipalities that score high on urbanization rate and population density (Pihl, 2002; Forselius, 2007). There exists thus no built-in incentive to perform lower collection efforts in sparsely populated areas.

The *socio-economic variables* overall add little to our understanding of plastic packaging collection rates, and to some extent this is probably due to the fact that these variables show limited variation across the different municipalities. In the cases of age, unemployment and education there are no statistically significant results. The coefficient for income is negative in the ML estimation (and statistically significant at the ten percent level) but not significant in neither the robust IV nor the Bootstrap estimation, indicating that it is hard to draw any conclusions about the impact of income on plastic packaging collection rates in Swedish municipalities. We further find that the coefficient for "share of single-family dwellings" is not statistically significant, a result that contradicts the anticipation that these single-family dwellings owners should contribute more to the recycling of plastic waste as they tend to have more space for storing used packaging materials, and they also generally have easier access to a car. The coefficient for share of new immigrants has the expected negative sign and it is statistically significant at the ten percent level in all estimations with one exception, the robust IV estimation. However, the coefficient for the share of immigrants as a whole is not statistically significant. 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. This has a negative impact on waste sorting activities. However, over time they learn the language and pick up existing social norms of behavior, and the results suggest thus that immigrants as a group are just as good recyclers of household plastic waste as Swedes in general.

The fourth category of variables, *environmental preferences*, does not help in explaining collection outcomes. The coefficients for Green party representation in the local government and for Green party support among households are both statistically insignificant. Lastly, for our *collection entrepreneur dummies*, only one seems to explain some of the variance in household plastic packaging waste collection, namely the dummy for private regionally-based entrepreneurs. There are five such entrepreneurs in Sweden and the result that we (ceteris paribus) should expect lower collection rates for municipalities in which this category of entrepreneurs are active can partly be attributed to one of these five, namely Kangos. The average collection rate in those eight municipalities where this entrepreneur operates is only 0.39 kg per person (compared to the national average of about 2 kg person). Kangos operates in sparsely populated areas in the most northern parts of Sweden (located far away from the recycling industries). This should decrease collection rates, and in the model we attempt to capture these impacts using different geographical and demographic variables. Again, however, it is possible that these could not fully address these impacts in a comprehensive way. Nevertheless, the operations of Kangos have recently been criticized in local media; it is claimed that the enterprise is ineffective in terms of emptying recycling containers and generally maintaining the quality and cleanliness of the drop-off stations. Moreover, households have claimed that Kangos seldom locate their drop-off stations in the adjacent to, for instance, shopping centres. The latter increases the need for sole-purpose travels to drop-off household waste and reduces recycling convenience.

6. Concluding Remarks and Implications

The purpose of this paper has been to analyze the determinants of inter-municipality differences in the collection of household plastic packaging waste in Sweden. We use spatial econometric methods and the results reveal that spatial interaction is present in the data used, and when we control for this and attempt to address the presence of heteroskedasticity we obtain results that are fairly robust across different model specifications.

The spatial lag parameter suggests that the collection of household plastic packaging per capita is positively related to the spatially weighted average of the collection per capita in neighboring municipalities. In other words, the probability that the collection of plastic collection is high increases if the neighboring municipalities collect high degrees of the household plastic packaging. This may be due to, for instance, cross-municipality interaction and cooperation, or simply because municipalities and waste companies copy-cat each others' policies and/or collection organizations. An additional reason is that the amounts of plastic packaging waste collected could be reported for a group of (normally two) municipalities.

Overall the results suggest that policy variables rather than geographic/demographic and socio-economic factors are the major drivers of collection rates. *First*, the municipalities can positively affect collection rates by increasing the reliance on weight-based waste fees. However, even though this seems to be an effective method for increasing the collection of packaging materials, undesirable side-effects of such fees must also be acknowledged. A weight-based waste fee can give households an incentive for illegal waste disposal. Empirical research suggests that such negative outcomes cannot be neglected (Fullerton and Kinnaman, 1996; Dahlén et al., 2007). It is also important to analyze the administrative costs of introducing such a system if economic efficiency is to be ensured. *Second*, making household recycling of plastic packaging easier by introducing curbside recycling and/or increasing the density of recycling centers would imply higher collection rates. Thus, we find evidence that facilitating means of this kind increase the efforts undertaken by citizens. Yet again it is imperative to stress the importance of weighing the administrative costs of operating, for instance, curbside recycling against the social benefits of having such schemes in place before supporting wide-spread adoption of these means (e.g., Kinnaman, 2006).

We do find that the different proxies for the marginal costs of plastic packaging collection in the respective municipalities do not exert a significant effect on collection outcomes. A reasonable explanation for this is that the compensation from the material companies varies depending on region and this tends to reduce regional cost differences in collection. This indicates that the Swedish society *could* save economic resources by paying more attention to regional cost differences. Still, purely on the basis of our study it is difficult to outline strong policy recommendations. A move to a more cost-effective collection scheme would have both pros and cons. First, we have not considered the transaction costs involved, that is the costs of administering, monitoring and enforcing a new system. These may be high and offset any cost savings, but we still believe that they potentially could be kept low. The authorities need not necessarily set different collection targets for dense and sparsely populated regions, respectively, and then enforce each of these. It may be enough to reform the compensation scheme, implement uniform compensation levels, and permit these economic incentives determine where collection will be made. We believe instead that one of the major drawbacks of a cost-effective scheme in which spatial cost differences matter may lie in the notion that there could be a trade-off between the cost-effectiveness and the legitimacy of the policy. If people as well as politicians feel committed to waste recycling because it is one way of contributing to public environmental goods, they may have a negative attitude towards a policy that encourages spatial differences in collection efforts.

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Appendix A: Moran's I Test of Spatial Correlation in Plastic Packaging Collection

Weight Matrix	Moran's I
The row-standardized inversed distance matrix, with a distance cut-off at the samples first quartile.	***0.10
The row-standardized inversed squared distance matrix, with a distance cut-off at the samples first quartile.	****0.18*
The row-standardized inversed distance matrix, with a distance cut-off at the samples second quartile.	***0.06
The row-standardized inversed squared distance matrix, with a distance cut-off at the samples second quartile.	***0.15
The row-standardized inversed distance matrix, with a distance cut-off at the samples third quartile.	***0.05
The row-standardized inversed squared distance matrix, with a distance cut-off at the samples third quartile.	***0.15
The row-standardized 5 nearest municipalities matrix.	***0.25
The row-standardized 10 nearest 10 municipalities matrix.	***0.16

Note: The chosen matrix is written in italics, and *** indicate statistical significance at the one percent level.

Appendix B: Descriptive Statistics

Variables	Mean	Std. dev	Min	Max
Dependent variable				
Plastic	1.97	0.97	0.04	5.71
Policy variables				
Feewe (D)	0.09	0.28	0.00	1.00
Curbside (D)	0.05	0.22	0.00	1.00
Drop	0.57	0.26	0.09	1.86
Geographic and demographic variables				
Dist	178	151	10.0	940
Urb	73.6	15.5	31.0	100
PopDen	129	429	0.20	4075
BigCity (D)	0.04	0.20	0.00	1.00
Socio-economic variables				
Age	42.0	2.27	36.2	47.3
Inc	216	23.2	178	393
Edu	12.3	6.14	5.00	48.0
Unemp	4.28	1.26	1.80	8.60
SFD	62.3	15.3	2.66	93.1
Tim	5.64	3.79	1.20	27.6
Newim	1.61	1.05	0.20	7.60
Environmental Preferences				
Envm (D)	0.24	0.43	0.00	1.00
Envh	3.89	1.16	0.90	8.80
Collection entrepreneur dummies				
PNE (D)	0.65	0.48	0.00	1.00
PRE (D)	0.12	0.32	0.00	1.00
MRE (D)	0.16	0.37	0.00	1.00
MLE (D)	0.07	0.25	0.00	1.00

Note: *D* denotes the use of (1/0) dummy variables.

IV

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The Swedish producer responsibility for paper packaging: An effective waste 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 induce 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. This implies a violation of the cost effectiveness principle. 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 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 "upstream combination tax/subsidy" scheme (UCTS system),[1] 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 analyses, and there 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.

[1] The UCTS system combines an output tax on produced intermediate goods with a subsidy granted to collectors of recyclables.

A proper understanding of the producer responsibility legislation and its incentive structure is important for the evaluation of environmental policy as are comparisons with alternative waste regimes. In Sweden today only a selected number of materials 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, in Sweden the Swedish Ecocycle Commission (SEC) (1997) has suggested that the producer responsibility should be extended to all goods in Sweden. Also the Swedish Environmental Protection Agency (e.g., SEPA, 1999) has been 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. Hence, all producers, not only the 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 legislation, in particular the ability of the system to induce 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 to empirically illustrate the ways in which the systems works in practice. 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% of the packaging market (SEPA, 2003) and (c) most of the packaging waste has a very low economic value, and Dafgård's lasagna packaging is a representative case also in this sense. 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 constitutes the basis of 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 for Karin's lasagna 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 reviews 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 a 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 (SOU, 1991, 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. That 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 ensuring that these are 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 concerning the means of compliance, but instead leaves a significant amount of discretion on the part of the producers to organize the system effectively.

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§).

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 should 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 collection systems and they should also gather data on the outcome of the collection, recycling and energy recovering activities and report these 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 assigned recycling stations. The producers are not obliged to pay the consumers for this 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).

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

Table 1
Swedish targets in the producer responsibility for packaging (% by weight)

Type of *packaging* until 29 June 2001 and type of *packaging waste* after 30 June 2001	Recycling and reuse after 1 January 1997 (%)[a]	Recycling or energy recovering after 30 June 2001 (%)[b]
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).

[a] Reuse in parenthesis.

[b] Energy recovery in parenthesis.

Table 2
Packaging fee for different packaging materials (as of January 2002)

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

Source: REPA (2002).

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 recovery 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% of all cardboard packaging. At least 40% of all this packaging should be recycled, hence used as input in new products. The rest of the collected packaging, 30% 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 have included Svensk Kartongåtervinning AB (SKAB) (cardboard), Plastkretsen AB (PAB) (plastic), Svenska Metalkretsen AB (SMAB) (metal) and RWA Returwell AB (RWAB) (corrugated board).[2] Together they all form the service organizations Svenska Förpackningsinsamlingen 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 cooperates 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 instance, 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. This latter fee is based on weight and it varies across different materials (see Table 2).

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

[2] In 2006, SKAB and RWAB merged into one single material company, Returkartong AB.

Table 3
Reuse and/or recycling of packaging (percent of production for selected years)[a]

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[b]	–	–	–	–	13	19	34	43	13	16
PET-bottles[c]	–	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.

[a] The figures include reuse and recycling before 30 June 2001 and thereafter only recycling. *Sources*: SEPA (2003), SEPA (2002a), SEPA (2002b) and SEPA (2001b).

[b] 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.

[c] 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.

by the packaging filler, packer or re-packer for products made in Sweden, and in the case of foreign products by the importer. 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 fixed 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 they will 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% 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 systems through design and product development. Since this goal is difficult to evaluate, the authorities have set the recycling and energy recovery targets for packaging waste that were outlined in Table 1. Table 3 displays to what extent the producers (and material companies) have fulfilled these recycling targets.

In 2002 the producers of corrugated cardboard, steel plate and glass all fulfilled their responsibility. They even have a history of almost always attaining the required targets. Producers of aluminum cans and paper and cardboard 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.

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 less heavy 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 weights for these commodities halved during these 5 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%. In 1991 the total amount of packaging was estimated at 1,141,000 tonnes and in 2000 it had decreased to 950,000 tonnes (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 tonnes of packaging waste went to recycling in 2000. This represents approximately 70% 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; at that time the recycling amounted to 250,000 tonnes of packaging waste. Consequently, this indicates that there exists evidence also of an *input substitution effect* in the sense that recycled inputs have replaced virgin inputs after the producer responsibility was introduced. However, 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% during the last 70 years and the Swedish soft drink and beer can has become 25% lighter between 1983 and 1995 (Sandström and Soutokorva, 2001). According to SEC (1998) and

Sandström and Soutokorva (2001) the main reasons for the recent weight decreases cannot be attributed to any fundamental technical developments in packaging construction. Instead it is mainly due to small constant improvements within existing technologies and packaging materials. 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 play an important role. A survey also shows that in the early 1990s there was a trend towards "green" products in Sweden (SEPA, 1998). In addition, there has been other new legislation in the waste area during the last decade. For instance, a new Environmental Code was introduced in 1999. In 2000 and 2002, respectively, a tax on waste disposal and a prohibition to deposit burnable waste were introduced (SEPA, 2001a; 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%) have introduced weight-based fees for the garbage, primarily for private house owners (SEPA, 2001a).

In sum, it is still unclear to what extent the Swedish 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.

3. Producer responsibility in supply chain management: the case of Karin's lasagna[3]

In this section the supply chain for the packaging for Karin's lasagna is briefly presented, and we use this case as an example of how the producer responsibility affects single firms in practice. Fig. 1 provides an overview of the most important components of this specific 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 supply chain starts when Stora Enso harvests or buys forest *raw material* (Engström, 2002). 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 placed in a cardboard box produced by Ekmans. Both CC pack and Ekmans buy their cardboard from the *material producer* Stora Enso (Lindgren, 2002; Olbrich, 2002). Smurfit Dalwell produces the transport packaging of corrugated cardboard. 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.

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

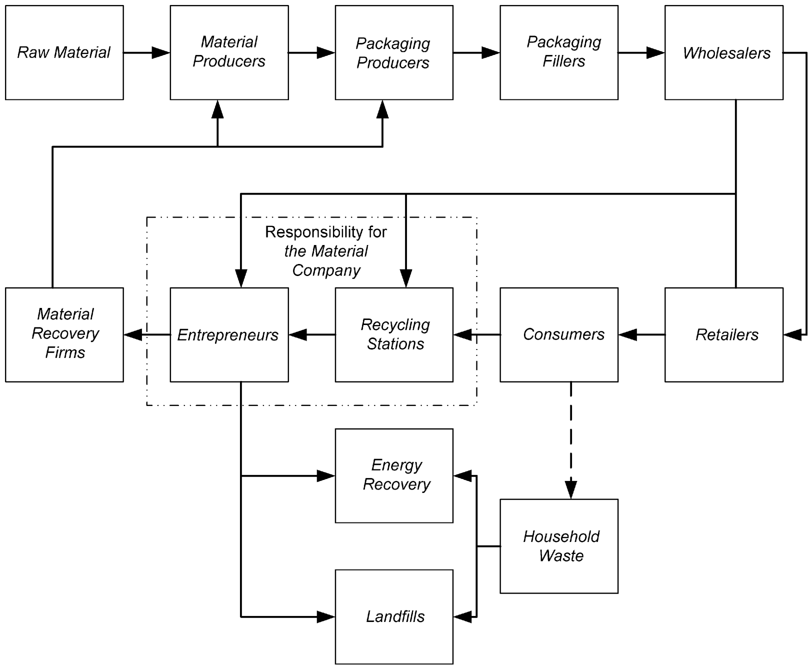

Fig. 1. The supply chain for paper packaging.

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 these 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*,[4] 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

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

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* (there are approximately 7000 of these in Sweden), and put in the containers for 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 five municipalities the municipal authorities 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 waste. The recycled cardboard from Fiskeby board mill is sold to the packaging producers and will eventually become new packaging.[5]

In the empirical section of the paper we occasionally revert to the lasagna case to illustrate important issues that may affect the effectiveness of the producer responsibility scheme in Sweden.

4. The economics of packaging waste management

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 policy 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 benefits

[5] 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.

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 may motivate 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 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. 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. Production based on virgin materials, especially in the case of an exhaustible (non-renewable) material, denies future generations these materials and consequently implies an external cost.[6] 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 lump-sum fees. 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 can decide upon an "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 is mainly concerned with cost effectiveness. Thus, the analysis in Section 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 Pigo-

[6] 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.

vian taxes,[7] subsidizes and tradable pollution permits. These incentive-based instruments have several advantages over "command-and-control" policies that involve technology or emission standards. An economic instrument promotes the *least cost way* of achieving any pre-determined environmental goal. For instance, a uniform tax on waste deposition will raise the cost of this activity and consequently producers with low marginal costs for recycling will recycle more than a producer with high recycling costs. Furthermore, economic instruments induce so-called *dynamic efficiency* effects, i.e., 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 (e.g., the weight) of waste 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, 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 can lead to problems with illegal waste disposal and high transaction costs. Fullerton and Kinnaman (1996) provide some empirical support for this notion.

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 that unit-based pricing only creates incentives for "design for environment" in the case where 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 systems (Palmer and Walls, 1999).

A special case of the deposit-refund system is the (UCTS) system. This method utilizes an output tax on the producer of goods and combines it with a subsidy to producers that recycle the waste. This method can, it is argued, impose lower transactions cost than the

[7] Pigovian taxes are named after the English economist A.C. Pigou that pioneered welfare economics. To be a true Pigovian tax, the tax rate must equal the marginal environmental costs at the social optimum.

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 the UCTS system 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) reach the same conclusion using a 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.

Palmer et al. (1997) use a theoretical model calibrated with elasticity estimates for demand and supply to compare the UCTS-policy with single policy instruments such as an output tax and a recycling subsidy in the American waste markets for paper, glass, plastic and steel. They conclude that the UCTS system is more cost effective than the other two policy measures. 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 outlines a model that can be used as the basis for an evaluation of the effectiveness of the Swedish producer responsibility ordinance and a comparison with 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 displays the 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 acknowledges 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 their full social costs (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 firms' choices of packaging and 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 ends up in a landfill or in the form of recycling r which ends up in new products. The household technology will determine the generation of g so that:

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

where $g(\ldots)$ 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 $r(\ldots)$ 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 ($u_q < 0$). For the above reasons the u_q could both be positive or negative. 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 returns 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 Eqs. (4)–(6) and the waste generation technologies Eq. (1) and (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 parenthesis in every equation is defined as the marginal social cost per unit garbage (MSC_g). 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 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 the 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 as well as enforcement problems. The household budget is altered as follows when the government is allowed to provide different tax incentives:

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

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:

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

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:

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

$$p_{\mathrm{r}}=\frac{f_{\mathrm{r}}}{f_{\mathrm{k}}} \tag{12}$$

$$\frac{\partial p_{\mathrm{q}}}{\partial \rho}q=qt_{\rho}-\frac{f_{\rho}}{f_{\mathrm{k}}} \tag{13}$$

Consequently, the price p_{q} just about 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 $(p_{\mathrm{g}}g-p_{\mathrm{k}}k_{\mathrm{g}})$, where $g=\gamma k_{\mathrm{g}}$ and $p_{\mathrm{k}}=1$, so $p_{\mathrm{g}}=1/\gamma$. Consequently, the price of garbage 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 Eqs. (11)–(13) above to replace each price with the corresponding cost of production gives:

$$\frac{u_{\mathrm{q}}}{u_{\mathrm{h}}}=\frac{1}{f_{\mathrm{k}}}+\rho t_{\rho}+t_{\mathrm{q}}+\left(\frac{1}{\gamma}+t_{\mathrm{g}}\right)g_{\mathrm{q}}+\left(\frac{f_{\mathrm{r}}}{f_{\mathrm{k}}}+t_{\mathrm{r}}\right)r_{\mathrm{q}} \tag{14a}$$

$$qt_{\rho}-\frac{f_{\rho}}{f_{\mathrm{k}}}+\left(\frac{1}{\gamma}+t_{\mathrm{g}}\right)g_{\rho}+\left(\frac{f_{\mathrm{r}}}{f_{\mathrm{k}}}+t_{\mathrm{r}}\right)r_{\rho}=0 \tag{14b}$$

$$qt_{\rho}-\frac{f_{\rho}}{f_{\mathrm{k}}}=0 \tag{14c}$$

Eqs. (14a)–(14c) shows the general equilibrium condition in which 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.

4.3.3. 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 Eqs. (14a)–(14c) to equal the social optimum conditions in Eqs. (8a) and (8b).

4.3.3.1. 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 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 shifts from the household to the firm. Hence, the household budget constraint will change as follows:

$$k-k_{\mathrm{h}}=(p_{\mathrm{g}}+t_{\mathrm{g}})q \tag{15}$$

In addition, the firm profit function becomes:

$$\pi = p_q q - k_q - (p_g + t_g)g - t_r r - q\rho t_\rho \tag{16}$$

It can be shown that this system will lead to a socially 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.

4.3.3.2. 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 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 follows:

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

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

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 would imply 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 defined 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 include the labor and capital costs 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 costs. This is not the same as minimizing the transaction costs. It is possible that a system with low transaction costs 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 discussed in Section 5.2.4.

4.5. Brief summarizing remarks

The general equilibrium model outlined above 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. 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 are derived from the above model discussions, 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 induce 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 implements and proceeds with activities in all three areas. 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 level of 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 an 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 above 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% 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. Little in the Swedish scheme induces this type of behavior at the individual firm level, and this is of course a shortcoming of the Swedish producer responsibility system. Our interviews with firm representatives along the Karin's lasagna supply chain confirm this conclusion (Asp, 2002; Pihl, 2002; Vretman, 2002).

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.

Nevertheless, 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 a packaging fee, something which de facto means that they hand over the responsibility to REPA and to SKAB. The total 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*, 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. This means that the Swedish producer responsibility has rather large similarities with this UCTS system. The Swedish packaging fee functions 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 sub-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 our interviews with industry representatives along the supply chain for Karin's lasagna, 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 (e.g., Vretman, 2002). Hence, this packaging fee will essentially work as a fiscal 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 this can of course be low, not the least if firms mentally view this as a fiscal tax that is implemented solely to finance other actors' activities to fulfill the policy goals. It is also obvious when talking to industry representatives that this fee represents one – out of many – economic reasons to decrease the amount of packaging for the studied lasagna cardboard (see, in particular, Lindgren, 2002; Olbrich, 2002; Vretman, 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 output goals regardless of the system chosen. For instance, if the fee is too 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 across 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. The packaging producers in the Karin's lasagna supply chain (e.g., CC Pack, Ekmans, etc.) are all of the opinion that the packaging fee has increased customer demand for lighter packaging as well as increased the use of recycled inputs (e.g., Lindgren, 2002; Olbrich, 2002). This implies that the packaging fee – paid by Dafgård – has an impact upstream on producer behavior in this supply chain.

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. 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 (Berglund and Söderholm, 2003). 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 instructions 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 that 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 necessarily be rational for the individual firm to consider recyclability in their production decision. The industry representatives along the supply chain of Karin's lasagna confirm that the producer responsibility has not influenced packaging design in the sense that the packaging has become easier to recycle (Olbrich, 2002; Vretman, 2002). Lindgren (2002) even argues that the demand for lighter packaging in this sector has stimulated the use of plastic laminate packaging (which implies less recyclability). If true, this indicates that by differentiating the packaging fee according to weight we may achieve a rather weak proportionality to environmental damage done.

The fact that the ordinance assigns households extensive responsibilities (to sort, clean and transport waste) may 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"). Those households that fulfill their responsibility can be expected to 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% of the paper and cardboard boxes were recycled in 2002 (SEPA, 2003). This means that quite a few households 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 reasons to believe that many consumers do not care – or simple lack the knowledge to decide – which type of packaging is easy to recover for firms. Consequently, it is probably fair to conclude that the ordinance provides too few incentives for the packaging producer to increase the packaging recyclability. This problem is also difficult to resolve as the number of consumers – as illustrated in the Karin's lasagna supply chain (see Section 3) – are plentiful (e.g., ICA, Axfood, final consumers, etc.), and each of these faces weak private incentives to demand packaging designed for increased 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 calculating 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. As a consequence, the consumer will not have enough 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) argue 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 even in the absence of household waste sorting, 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 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 24 h time period a "fine" is imposed. *Second*, households and companies sometimes 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 this amounts to about 50 million SEK per year for the material companies. The five material companies (including SGAB) share this cost (Pihl, 2002; SFAB, 2003b).

As was mentioned in 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 any 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; SFAB, 2003a). Furthermore, SFAB and each material company use advertising campaigns in newspapers and on television to inform packaging consumers. Pihl (2002) claims that these campaigns give results but they are also expensive. Finally, each material company must report the collection and recycling outcomes 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 only has 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% of the administration cost for REPA and SFAB should burden SKAB (corresponding to about 6–7 full-time 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 burden the material recovery companies. For instance, they need to control the quality of 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 as suggested by Walls and Palmer (2001).

The material companies are unnecessary in the UCTS system discussed by Palmer and Walls (1999), but of course someone 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 their 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% 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% 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 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 4.3 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 tonne recycled. Ekvall and Bäckman (2001) estimate the average cost at a minimum of SEK 410 per tonne and a maximum of SEK 6950 per ton. The 410 figure is based on the assumption that leisure has no value. According to SEPA (2003), in 2002 74,882 tonnes 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 2434 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, other costs 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 become more unpleasant if the packaging is severely 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 material use. 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 enough 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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