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]

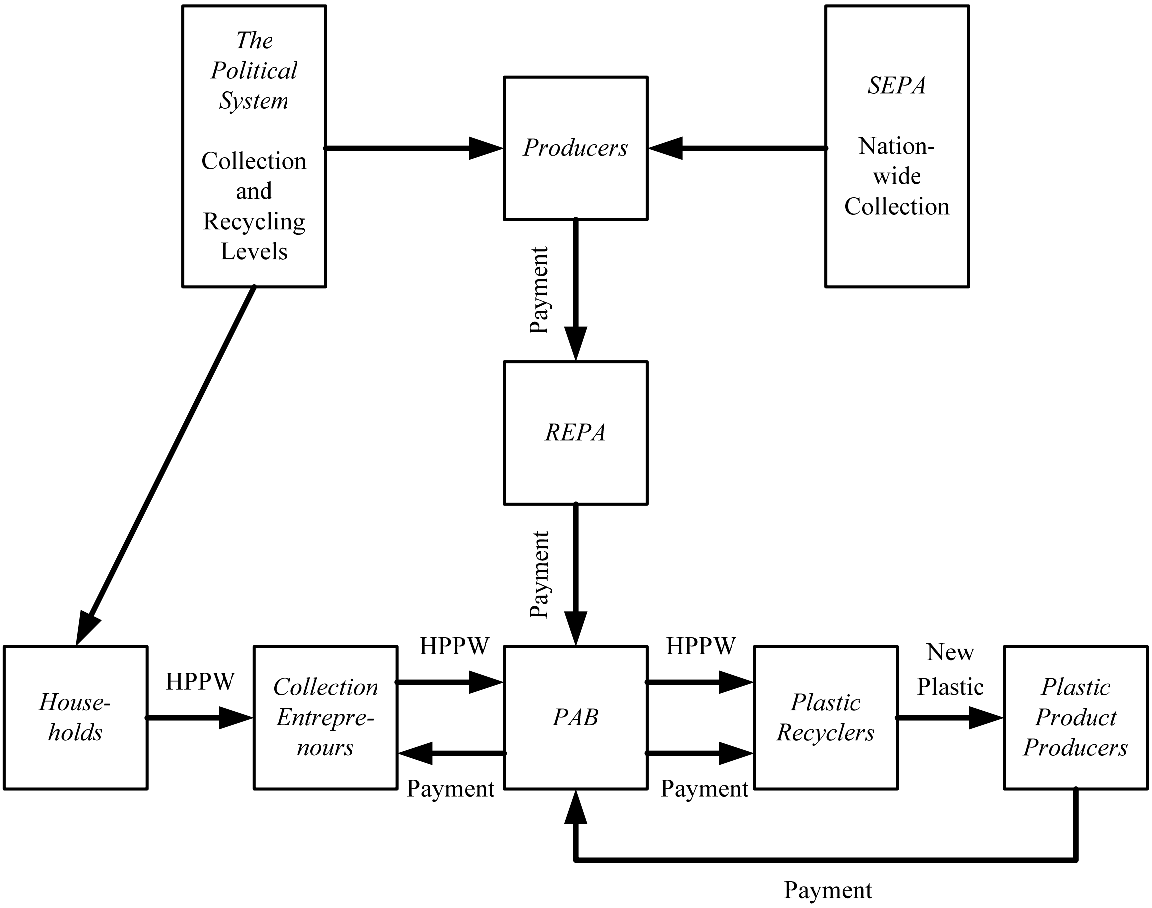

Figure 1: The Structure of Household Plastic Packaging Waste Collection in Sweden

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

[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.

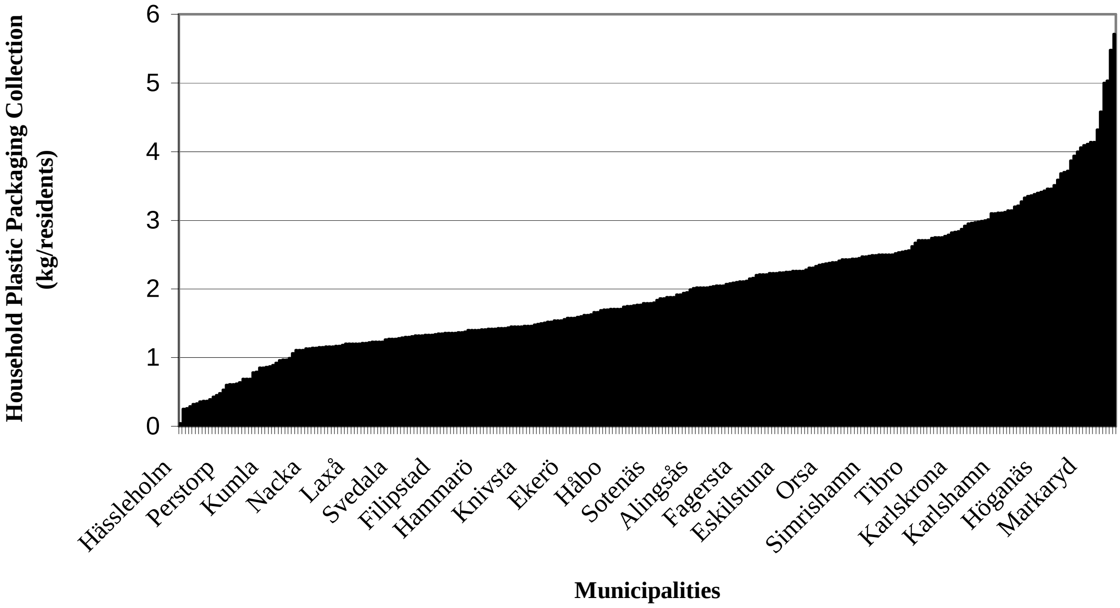

Figure 2: Collection of Household Plastic Packaging Waste in 283 Swedish Municipalities, 2005 (kg/resident)

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.

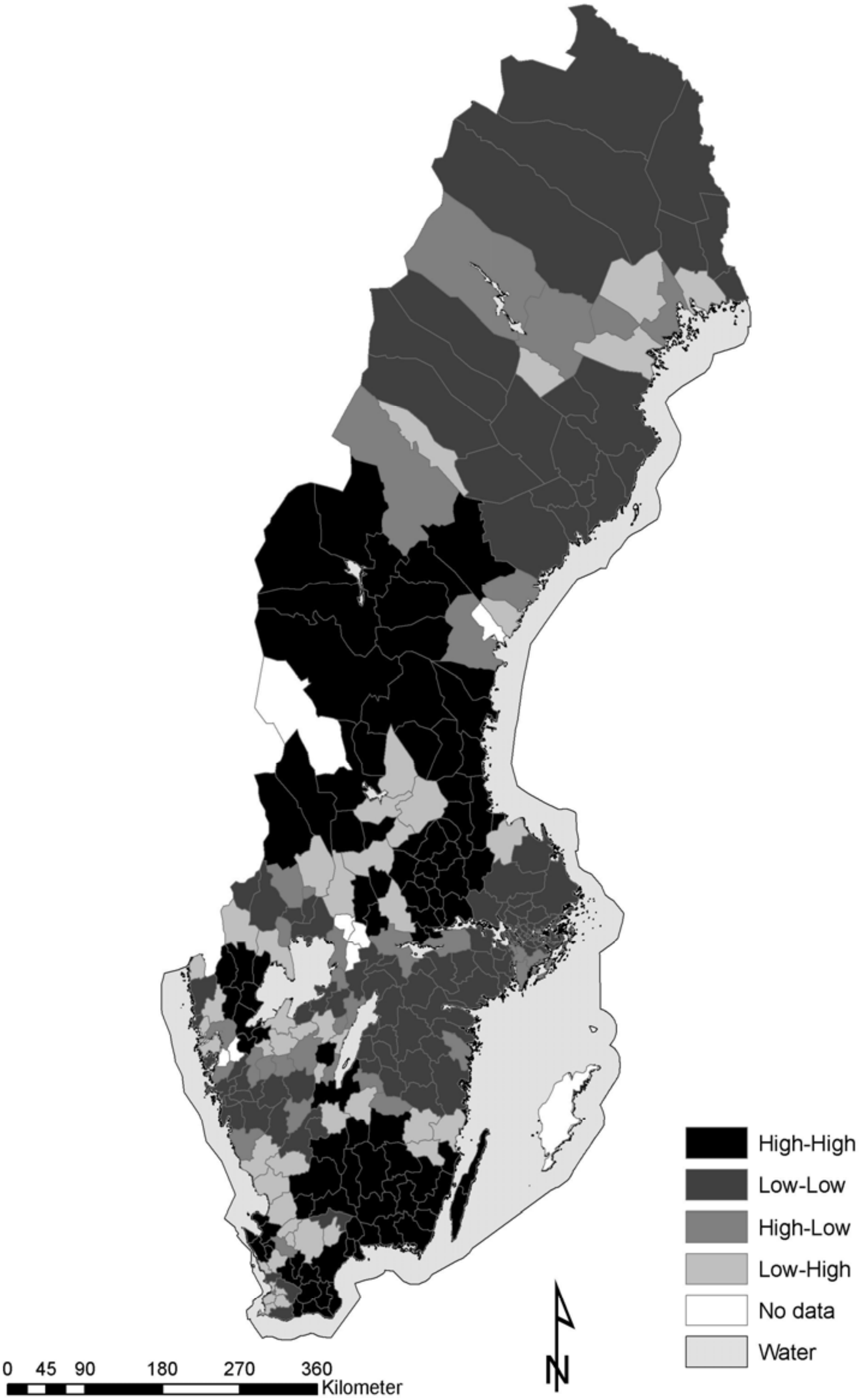

Figure 3: Moran Scatterplot Map for Plastic Packaging Collection in Sweden

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 munici-pality'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;

$$y = X\beta + \varepsilon$$
$$\varepsilon = \lambda W\varepsilon + \xi \tag{2}$$

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:

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

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 statistic-cally 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.