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Abstract

The paper investigates the costs of waste disposal and recycling services by using a well-behaved *Composite* cost function model. Our estimates on a unique sample of more than 500 Italian municipalities highlight that the refuse collection technology exhibits constant returns to scale as well as scope economies between disposal and recycling. As far as the size of the municipality increases, scope economies rise up to 14%, but they are accompanied with overall diseconomies of scale. Our findings suggest that, on the one hand, joint management of disposal and recycling should be encouraged, and, on the other hand, that strategies aimed at increasing the share of waste sent for recycling would not imply a considerable increase in total costs.

Keywords: Solid waste, recycling, cost functions.

JEL codes: D24, H42, L33, L99

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Contents

1. Introduction

In local public services such as energy, water, public transport, the attention of policymakers has been devoted, on the one hand, on environmental regulation, and, on the other hand, on the promotion of competition and cost efficiency. As to the second issue, the policies that have been proposed are a mixture of mandatory divestitures, unbundling and competitive tendering, but ownership and corporate governance changes (ranging from privatization or the promotion of private public partnerships to forms of intermunicipal alliances) have been suggested as well.

Municipal Solid Waste (MSW) services, similarly to other network industries, have undergone radical changes in both organizational and market structure. On the basis of EU directives 2006/12 and 2008/98, waste management legislation and policy should be inspired by a principle of hierarchy: prevention, preparing for reuse, recycling, recovery and disposal. This implies an increasing role of separated collection, with a target of 50% of MSW by 2020. In Italy, the legislation has foreseen even more ambitious targets (the share of waste sent for recycling was 15% for 1999, 25% within 2001, and up to 60% in 2011). The reforms introduced by the *Ronchi's decree* (law 22/1997) and by the *Environmental Code* (law 152/2006) aimed at favouring the integrated management of a too much fragmented production process, as well as at promoting competitive tendering procedures for the management of waste collection. Moreover, they introduced a new tariff system that, creating a direct connection between the solid waste generated by households and the amount to be paid for refuse collection, should induce citizens to adopt a more responsible environmental behaviour.

As pointed out by Callan and Thomas (2001), the empirical literature has devoted much more attention to demand-side aspects (i.e., how to discourage land disposal, how to encourage recycling and recovery, how to design and implement an optimal pricing program, and so on)[2] than to supply-side issues such as the cost analysis of the MSW market. The evidence on the costs of waste collection and recycling is even more scant, as pointed out by Bohm et al. (2010):

"*The growth in curbside recycling has presumably evolved independently of costs and, perhaps for this reason, the economics literature is largely silent (with a few important exceptions) on understanding the costs of municipal waste and recycling services. Data limitations may have also hampered investigations into costs*" (Bohm et al., 2010, p. 864).

Since the collection of recycling waste has now reached a quite established share (though, at least for the case of Italy, not fully consistent across the whole national territory), and indeed is a

[2] For example, see Kinnaman (2005 and 2006). In particular, Kinnaman (2005) tries to understand why municipalities are operating cost recycling programs designed to reduce the external costs of garbage disposal. The results pointed towards the presence of altruistic tastes for recycling on the part of households, so that policies aimed at setting specific recycling goals might be expensive but not effective at reaching the required target.

strongly encouraged practice in the planning of public services, an analysis of the costs of joint collection of disposal and recycling waste seems of great relevance.

This paper aims to contribute to the above ongoing debate by analysing the cost structure of a sample of more than 500 Italian municipalities that provided waste collection and disposal services during years 2004-2006. From a methodological point of view, we will take into account, on the one hand, the multi-product nature of the MSW service by allowing for separate outputs for waste simply taken to disposal sites or incinerated and waste sent for recycling, and we will use, on the other hand, a flexible cost function model that is well equipped to measure scope and scale economies at different output levels. The remainder of the paper is organized as follows. In the next section the relevant literature will be briefly reviewed. In section 3 we will present our empirical cost function model. In section 4 we will present our dataset and we will show some first descriptive statistics. Section 5 will show our main results, while section 6 concludes.

2. Literature review

Starting from the seminal works of Hirsch (1965) and Stevens (1978), scholars have analysed the costs of the refuse collection industry by investigating mainly issues such as the optimal scale of operation and the efficiency comparison between private and publicly owned operators.[3]

Overall, albeit there is some variance across studies, the results are pointing towards the existence of scale economies for relatively small communities that are exhausted when the population reaches a certain threshold (50,000 inhabitants according to Stevens, 1978). Another common results is that, rather than the type of ownership itself, the key factor which is more likely to bring cost savings in waste management activities is the organization of competitive tendering procedures.

However, the bulk of the empirical papers have made use of rather *ad hoc* simple cost function models.[4] In a typical study, (average or total) costs are regressed on output (a measure of pick up points or of the quantity of waste collected in a year) and other explanatory variables without taking into consideration the role of input prices, and without respecting some common standard microeconomic theory assumptions (i.e. the so called regularity conditions, such as Shephard's lemma, linear homogeneity with respect to input prices, and so on).

An under explored topic, despite its increasing relevance, is the multi-product nature of the refuse collection service. While in some instances the costs of recycling have been analysed by

[3] See Bohm et al. (2010) for a comprehensive survey on the first issue and Bel et al. (2010) for updated references on the second issue.
[4] Antonioli and Filippini (2002) represents one of the few exceptions.

including the share of waste sent for recycling among the regressors, there are very few papers that jointly consider disposal and recycling.

The present paper aims to contribute to the literature in both respects. From a methodological point of view, we will estimate a *Composite* cost function model, imposing restrictions in order to ensure that estimated costs are originating from a well-behaved cost function specification. In doing so, waste disposed and waste sent for recycling are considered as two separate but interacted outputs, so that it will be possible to infer whether economies of scope are characterising the provision of both services.

2.1 Empirical studies of the costs of recycling

Carroll (1995) focused on recycling costs only and found for a sample of 57 Wisconsin cities observed in 1992 that average recycling costs per household were negatively correlated to a measure of population density and to a variable accounting for in-house provision. Moreover, scale economies were found to be negligible. Most importantly, comparing his results with the ones stemming from the literature investigating garbage collection costs, the author found many similarities between the two technologies characterising waste disposal and waste sent for recycling.[5]

Bel and Fageda (2010) estimated a total cost function on a sample of 65 municipalities in metropolitan areas of the Spanish region of Galicia for year 2005, and included among the regressors a variable accounting for the percentage of the total waste volume that was designated for recycling. Since the coefficient was found to be not significantly different from zero, the authors concluded that: "*the environmental advantages derived from promoting recycling activities do not seem to lead to an important increase in the cost of solid waste collection. Hence, the present results suggest that local government would do well to promote such recycling activities*" (Bel and Fageda, 2010, p. 192).

Bohm et al. (2010) analysed both solid waste disposal and recycling activities on a sample of 428 US communities for year 1996. Two quadratic cost functions (one for disposal, one for recycling) were simultaneously estimated using Zellner's SUR model.[6] While the average cost function for disposal was found to be everywhere decreasing, highlighting the presence of

[5] For example, Hirsch (1965), working on a sample of 24 cities and municipalities in the St Louis area in 1960, suggested the presence of constant returns to scale. In a similar vein, Stevens (1978) estimated a Cobb Douglas cost function (including the price of labour among the regressors) on a sample of 340 US public and private firms, and found that, while private operators were better performers, economies of scale were exhausted at population sizes above 50,000 inhabitants.

[6] The authors presented estimates where input prices were included among the regressors, too, but the usual microeconomic theory properties ensuring well-behaved cost functions (i.e. Shephard's lemma, homogeneity of degree one in input prices, concavity) where neither imposed on the estimation nor checked after having estimated the model.

increasing returns to scale, the one for recycling was exhibiting a U shape, suggesting that, after a certain threshold, the costs for recycling were increasing sharply.

Callan and Thomas (2001) are, to the best of our knowledge, the only available study where disposal and recycling are jointly analysed in a context of a multi-product cost function framework. Using a sample of 110 municipalities in Massachusetts observed for years 1996-1997, they estimated two separate cost functions for the two services, each of which was including an interaction term between outputs. By doing so, they were able to measure, together with scale economies, scope effects too. The results suggested the presence of constant returns to scale for disposal and increasing returns to scale for recycling. Most importantly, the coefficients on the interaction terms were both found to be negative, and the computations referring to an hypothetical "average sample firm" revealed the presence of scope economies of the order of 5%.[7]

3. Model specification

As already pointed out, to the best of our knowledge, only Antonioli and Filippini (2002) analysed the technology of the waste collection sector by estimating a well-behaved cost function which satisfies the regularity conditions. Using data on 30 Italian waste and disposal collection firms for years 1991-1995, they estimated a system of equations, including a Translog cost function and the associated cost-share equations, by applying the iterative Zellner's (1962) seemingly unrelated regression (SUR) technique. The results suggest the presence of scale economies for small and medium-sized firms, while the largest firms in the sample were operating in an output region exhibiting diseconomies of scale.

In a similar vein, our proposed research strategy will start with the estimation of a Translog cost function (*TS*):

$$\begin{aligned}\ln C = \alpha_0 + \sum_i \alpha_i \ln Y_i + \frac{1}{2}\sum_i \sum_j \alpha_{ij} \ln Y_i \ \ln Y_j \ + \sum_i \sum_r \delta_{ir} \ln Y_i \ln P_r \\ + \sum_r \beta_r \ln P_r + \frac{1}{2}\sum_r \sum_l \beta_{rl} \ln P_r \ln P_l + \psi_C \end{aligned} \qquad [1]$$

where C refers to the total cost of production, Y_i refers to outputs (in our two-output case i, j = Disposal (D) and Recycling (R)), P_r indicates factor prices (in our three-input case r, l = Labor (L), Capital (K) and Energy (E)), and ψ_C is a random noise having appropriate distributional properties to reflect the stochastic structure of the cost model.

[7] Unfortunately, the authors were not providing estimates of scale and scope economies for different output levels and for different combinations of outputs.

The associated input cost-share equations are obtained by applying the *Shephard's Lemma* to expression [1][8]

$$S_r = \sum_i \delta_{ir} \ln Y_i + \beta_r + \sum_l \beta_{rl} \ln P_l + \psi_r \qquad [2]$$

where ψ_r is the error term relating to the cost-share r.

However, due to its log-additive output structure, the Translog model suffers from the well-known inability to evaluate cost behavior when any output is zero. This has been proved to yield unreasonable and/or very unstable values of the estimates for scope economies. For such a reason, empirical studies based on the Translog specification often rely on measures of *pairwise cost complementarities* for analyzing cost synergies between outputs[9].

To overcome the above problems, Pulley and Braunstein (1992) proposed as an alternative functional form for multi-product technologies the *Composite Specification* (*CS*). The *CS* cost function originates from the combination of the log-quadratic input price structure of the *TS* specification with a quadratic structure for multiple outputs. This makes the model particularly suitable for empirical cost analysis. The quadratic output structure is appropriate to model cost behavior in the range of zero output levels and gives the *CS* specification a clear advantage over the *TS* form as far as the measurement of both economies of scope and product-specific economies of scale are concerned.[10] In addition, the log-quadratic input price structure can be easily constrained to be linearly homogeneous.

The *CS* cost function is written as:

$$\begin{aligned} \ln C = \ln[\alpha_0 + \sum_i \alpha_i Y_i + \frac{1}{2}\sum_i \sum_j \alpha_{ij} Y_i Y_j + \sum_i \sum_r \delta_{ir} Y_i \ln P_r] \\ + \sum_r \beta_r \ln P_r + \frac{1}{2}\sum_r \sum_l \beta_{rl} \ln P_r \ln P_l + \psi_C \end{aligned} \qquad [3]$$

and the corresponding input cost-share equations are

[8] Cost-shares are computed as $S_r = (X_r P_r)/C$. By Shephard's Lemma $X_r = \partial C/\partial P_r$, where X_r is the input demand for the rth input, so that $S_r = \partial \ln C / \partial \ln P_r$.

[9] For a twice continuously differentiable cost function, cost complementarities are present at Y' if

$$CC_{ij}(Y';P) = \frac{\partial^2 C(Y';P)}{\partial Y_i \partial Y_j} < 0, \qquad i \neq j$$

for all $Y' \in [0,Y]$. Cost complementarities between two products imply that the marginal cost of producing one output decreases as the quantity of the other good is increased. Baumol et al. (1982) have shown that a multi-product cost function characterized by weak cost complementarities over the full set of outputs up to the observed level of output exhibits scope economies.

[10] See Piacenza and Vannoni (2004) and Piacenza et al. (2010), for more details on *CS*-type models and for some applications to the cost analysis of multi-product firms.

$$S_r = \left(\sum_i \delta_{ir} Y_i\right)\left[\alpha_0 + \sum_i \alpha_i Y_i + \frac{1}{2}\sum_i \sum_j \alpha_{ij} Y_i Y_j + \sum_i \sum_r \delta_{ir} Y_i \ln P_r\right]^{-1} + \beta_r + \sum_l \beta_{rl} \ln P_l + \psi_r \quad [4]$$

Given the regularity conditions ensuring duality between the production function and the cost function, the *CS* specification does not impose a priori restrictions on the characteristics of the underlying technology. Thus, it is a flexible form in the sense of Diewert (1974). To be consistent with cost minimization, [1]-[2] and [3]-[4] must satisfy symmetry ($\alpha_{ij} = \alpha_{ji}$ and $\beta_{rl} = \beta_{lr}$ for all couples i, j and r, l) as well as the following properties: *a*) non-negative fitted costs; *b*) non-negative fitted marginal costs with respect to outputs; *c*) homogeneity of degree one of the cost function in input prices ($\Sigma_r\beta_r = 1$ and $\Sigma_l\beta_{rl} = 0$ for all r, and $\Sigma_r\delta_{ir} = 0$ for all i); *d*) non-decreasing fitted costs in input prices; *e*) concavity of the cost function in input prices. Symmetry and linear homogeneity in input prices are imposed *a priori* during estimation, whilst the other regularity conditions are checked ex-post.

Therefore, as a second step of our analysis, we will estimate the Composite cost function system [3]-[4] and we will compare the results with the one stemming from the Translog system [1]-[2].

4. Data Description

Our dataset refers to a balanced panel of 529 Italian municipalities providing waste disposal and recycling services over the period 2004-2006, for a total of 1587 *pooled* observations.

The sample composition by geographical area, ownership form and output mix is presented in Table 1. 39% of observations refer to municipalities localized in Northern and Southern Italy, respectively, while the remaining 22% are localized in the central regions of the country.

As to the organizational form chosen to provide the service, in-house provision form accounts for 10% of the total sample, and is mostly concentrated in the South. A similar pattern can be observed for intermunicipal partnership, which accounts for only 8% of the municipalities (with a prevalence in the South). Finally, the limited responsibility company is by far the most popular juridical form chosen to organize the refuse collection service (82% of the entire sample and 94% of municipalities in the North).[11]

[11] Unfortunately, data limitations prevent us to disentangle corporations which are owned by private operators from limited companies whose shares are still in the hands of the local governments. Therefore, in the subsequent cost analysis we will not be able to separate the effects of *corporatization* (i.e. the transformation of the juridical form without implying a change in the ownership) from the ones stemming from privatization. See Cambini et al (2011) for an attempt to measure the impact of *corporatization* on the costs of a sample of Italian local public transport firms.

Turning now towards our main variable of interest, i.e. recycling activities, Table 1 shows that the share of the total waste volume designated for recycling is 20%. However, this average value is heavily dependent on the more virtuous Northern municipalities (where waste sent for recycling accounts for 37% of the total), while the shares of recyclable waste collected in Southern and Central regions of the country are rather limited (7% and 13%, respectively).

On the whole, our sample can be considered as fairly representative of the entire population. In fact, official data (see Chiades and Torrini, 2008) show statistics which are very similar to the ones reported above.[12]

Data on costs and output quantities are obtained from annual MUDs (i.e. annual declarations concerning municipal solid waste collection) which have been provided by Ecocerved. Input prices have been computed by integrating the information available in the MUDs with additional information drawn from questionnaires sent to the firms (or organizational structures) managing the service in the municipalities. Total cost (C) is the sum of labor, capital, and energy costs of the municipalities.[13] The two output categories are tons of MSW disposed (Y_D) and tons of MSW recycled (Y_R). Productive factors are labor, capital and energy. The price of labor (P_L) is given by the ratio of total salary expenses to the number of employees. Capital price (P_K) is obtained by dividing depreciation costs by the capital stock.[14] Summary statistics on outputs, input prices and shares as well as other demographic and urban variables are provided in Table 2.

5. Estimation and Results

Both the *TS* and *CS* specifications of the multi-product cost function are estimated jointly with their associated input cost-share equations. In order to ensure that the cost functions are linearly homogeneous in input prices we normalize total cost and input prices by the price of energy. Because the three share equations sum to unity, to avoid singularity of the covariance matrix only the labor and capital equations (S_L and S_K, respectively) are included in the systems [1]-[2] and [3]-[4]. Before the estimation, all the right-hand side variables were standardized on their respective sample average values. Parameter estimates were obtained via a non-linear GLS estimation

[12] For example, official data report that, in 2005, 11% of population (22% in the South) was receiving refusal collection services from municipalities by means of in-house arrangements. In the same year, the share of recycling over total refuse collection was 24% (38% in the North, 19% in the Central Regions and 9% in the South).

[13] Consistently with the large majority of empirical papers in this field, we rely on municipal data. It must be acknowledged that the reported cost data might in principle overstate the actual costs in the case in which the local public administrations are contracting with private firms for the provision of the service. Stevens (1978, p.441) tackles this issue and argues that the cost approach can be relied on. See also Carroll (1995, p.219) and Hirsch (1965, p. 91). This issue should however be mitigated by the completion of the reform geared to the exclusive assignment of the service through competitive tendering procedures.

[14] Following Antonioli and Filippini (2002), we assume that the price of fuel is the same for all municipalities in the sample.

(NLSUR), which is the non-linear counterpart of the Zellner's iterated seemingly unrelated regression technique. This procedure ensures estimated coefficients to be invariant with respect to the omitted share equation (Zellner, 1962).

The summary results of the NLSUR estimations for the *TS* and *CS* models are presented in Table 3.[15] The first rows present the estimates of cost elasticities with respect to outputs and factor prices for the 'average' municipality.[16] The latter are very easy to recover from *TS* model, in that $\varepsilon_{CY_i} = \alpha_i$, while S_r is simply the estimate of β_r (see equations [1] and [2]). In the *CS* model the computation of output and factor-price cost elasticities is a little bit more cumbersome:

$$\varepsilon_{CY_i} = \frac{\alpha_i + \sum_j \alpha_{ij}}{\alpha_0 + \sum_i \alpha_i + \frac{1}{2}\sum_i \sum_j \alpha_{ij}} \quad [5]$$

$$S_r = \beta_r + \frac{\sum_i \delta_{ir}}{\alpha_0 + \sum_i \alpha_i + \frac{1}{2}\sum_i \sum_j \alpha_{ij}} \quad [6]$$

By looking at the figures reported in Table 3, it appears that the two estimated cost function models are performing in a rather similar way: the estimates of labour (S_L) and capital (S_K) price elasticities are around 0.45 and 0.06 in both cases, and the same pattern applies to the estimates of the output elasticities: ε_{CY_D} (ε_{CY_R}) is 0.78 (0.23) for the *TS* model and 0.76 (0.25) for the *CS* model. The summary statistics are quite similar, too. The R^2 for the cost function is 0.93 in both cases, while the R^2 for the labor-share and capital share equations are higher for the *CS* specification.[17] McElroy's (1977) R^2 can be used as a measure of the goodness of fit for the NLSUR system. The results suggest that the fit is roughly the same for both specifications. More rigorously, the Vuong's (1989) statistics for selection among non-nested models (VLR test), that consists in normalizing the standard LR test in order to account for the fact that the models to be compared are not nested, is significantly different from zero. We must therefore conclude that the *CS* model does a better job in describing observed data. Moreover, it enjoys a clear advantage over the *TS* model as far as the measurement of scope and scale economies is concerned, as it has been discussed in section 3. Scale economies (*SE*) can be measured by computing the inverse of the sum of output cost elasticities,

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[15] The Translog model is estimated with NLSUR so that it is straightforward to make comparisons with the *Composite* model. However, we estimated also the *TS* model using iterated GLS as well as maximum likelihood estimators. As expected, the results are virtually unchanged across the three different estimation procedures.

[16] The *average* municipality (the point of normalization) corresponds to an hypothetical council operating at an average level of production for each output and facing average values of the input price variables.

[17] A similar pattern can be observed by comparing the estimated sums of squared errors (SSE) of the cost and input-share equations.

while scope economies (*SCOPE*) are computed by comparing the costs of specialized production with the costs of jointly providing Y_D and Y_R:

$$SE = 1/(\varepsilon_{CY_D} + \varepsilon_{CY_R}) \tag{7}$$

$$SCOPE = \frac{\mathrm{C}(\mathrm{Y_D}, 0) + \mathrm{C}(0, \mathrm{Y_R})}{\mathrm{C}(\mathrm{Y_D}, \mathrm{Y_R})} - 1 \tag{8}$$

Table 3 highlights that the average municipality, which collects 17,122 tons of Y_D and 3,770 tons of Y_R, and provides refuse collection services for a population of about 42,500 inhabitants, exhibits constant returns to scale and enjoys scope economies of the order of 2%. This means that, by doubling the amount of both disposal and recycling, costs will double as well. Moreover, consistently with the results found by Callan and Thomas (2001), there is an incentive to jointly provide both services.[18]

Fully exploiting the potential of our *CS* flexible cost function model, we can evaluate if and how scale and scope economies are changing if the size of the municipality and/or the output mix changes. Moving along row A of Table 4 it is possible to simulate how much waste disposal costs increase with size (assuming that recycling services are not provided). It appears that there are constant returns to scale up to λ =1 (i.e. up to Y_D = 17,122 tons), and decreasing returns to scale in correspondence of larger output levels. Similarly, the figures reported in row B suggest that the same pattern applies to recycling activities.[19] However, diseconomies of scale are found to be larger for recycling than for disposal.[20]

The costs of joint production $C(Y_D, Y_R)$ – row C – are always lower than the sum of the costs of specialised production ($C(Y_D, 0) + C(0, Y_R)$). This is suggestive of the fact that the cost function exhibits scope economies at all simulated output levels, thus justifying the choice to assign the two services through a single tender. However, scope economies are rather limited (and not significantly different from zero) up to λ =1, and become more important at higher output levels (7% for λ = 4 and 14% for λ = 8).

The results for aggregate scale economies summarize the patterns reported above. The figures reported in the last row of Table 4 imply that, by doubling the amounts of refuse collection

[18] The results of the Translog specification show the presence of cost complementarities, since the coefficient on the parameter α_{DR} of equation [1] is negative (-0.12) and statistically significant.
[19] The presence of constant returns to scale for relatively small municipalities is consistent with Carroll (1995), who was using a sample of municipalities of an average population size of 26,284 inhabitants. In addition, the finding of scale diseconomies in correspondence with higher output levels is consistent with the analysis conducted by Antonioli and Filippini (2002), as far as disposal is concerned, and with the outcomes obtained by Bohm et al. (2010), as far as recycling is concerned.
[20] For example, moving from λ =1 to λ =8, costs increase by a factor of 9 for disposal and by a factor of 9.5 for recycling.

(both disposal and recycling), costs are doubling up to λ =1. At municipality sizes above the sample mean, however, overall diseconomies of scale appear, but the presence of scope economies counterbalances the effect of decreasing returns to scale for both recycling and disposal activities. Therefore, the resulting estimates of aggregate scale diseconomies are found to be not very large.[21] In spite of the fact that Table 4 shows estimates relative to six different hypothetical municipalities, the figures reported are quite plausible. For example, the 2004 Report on waste collection in the province of Milan indicated that the per capita average cost was increasing with the size of the towns, passing from 83.3 (110) euros for municipalities with less than 5,000 (more than 30,000) inhabitants up to 151 euros for the city of Milan. By dividing $C(Y_D, Y_R)$ by the population size (first row), we obtain a very similar pattern, confirming that our model fits the data quite well.

Since our paper mostly focuses on recycling, we want now to investigate to what extent different shares of recyclable waste collection are affecting the level of costs. Figures 1a) and 1b) plot the behaviour of costs (on the vertical axis) for different percentage values of the ratio $Share_R = Y_R/(Y_D+Y_R)$. Each curve corresponds to a specified level of the total quantity Y_D+Y_R. Similarly to what has been done in Table 4, the municipality size has been scaled up and down by multiplying and dividing the average sample quantities by the parameter λ. The shapes of the "isoquant" curves offer some very interesting insights. As expected, costs increase in correspondence with higher shares of recycling, but this happens especially at higher percentages and for municipalities with more than 100,000 inhabitants.

The joint interplay of scope economies and decreasing returns to scale for the recycling technology implies that:

a) It is not very costly to increase the percentages of recycling up to 30%-35% at all municipalities' sizes. For example, increasing recycling shares from 10% to 20% would imply that total costs increase by about 4% in correspondence of all estimated sizes (i.e. for λ ranging from 0.25 to 8). Moreover, for municipalities of a population size of about 300.000 inhabitants (λ =*8*), the increase of $Share_R$ from zero up to 10%-15% implies a slight reduction of total costs;[22]
b) It is not very costly to increase even further the percentages of recycling for relatively small municipalities;
c) It is indeed very costly to increase the ratio $Share_R$ beyond certain levels for large municipalities. For example, when λ = 8, costs increase by 32.5% if $Share_R$ increases from 20% to 40%.[23]

[21] The results of the Translog specification are remarkably similar also with respect to the estimates of scale economies for municipalities larger or smaller than the sample average.
[22] This is due to the fact that the effect of economies of scope is still dominating over the effect of decreasing returns to scale for recycling.
[23] Notice that for municipalites with population above 100,000 inhabitants, $Share_R$ is on average 18% with a maximum value of 42%. Therefore, one should use particular caution when interpreting results for large councils (i.e. when λ = 4 and λ = 8), because the curves in Figure 1b partially rely on out of the sample simulations.

The above results can be partially reconciled with some of the results summarized in section 2.1. Bel and Fageda (2010) are working on a sample of a much smaller size as compared to our sample of Italian municipalities. Even if we are using a different methodology, results a) and b) are consistent with the absence of significant effects of recycling shares on total costs found for Galician municipalities. However, since our flexible functional form allows us to investigate the shape of the cost function at all output levels, we can better qualify their findings. Our estimates suggest that, for municipalities of a population size above 50,000 inhabitants, the impact of $Share_R$ is not negligible anymore, and becomes very strong in correspondence with high values of λ.

Bohm et al. (2010) report increasing returns to scale for municipalities that recycle up to 13,200 tons and decreasing returns to scale for larger quantities. Our sample of Italian municipalities exhibits decreasing returns to scale, too, but they appear at lower output regions (at about 4,000 tons).

We believe that our analysis can be useful for policymakers who are interested in pursuing strategies aimed at increasing the volume of recycling services. As already pointed out, recycling shares are still rather low in Italy, especially in the Southern regions. Our findings suggest that, keeping constant the total amount of waste collected, it is worth to expand recycling programs where the recycling shares are very low (irrespective of the size of the municipality), and, in the case of higher starting levels of $Share_R$, where the population size is below 150,000-200,000 persons. Moreover, as argued by Bohm et al. (2010), the extra costs reported in Figures 1a) and 1b) are not taking into account possible revenues stemming from the sale of recyclable materials, as well as possible savings in the total waste collected due to a more responsible and environment friendly behaviour on the part of households.[24] Both arguments should have the effect of flattering the shape of the cost functions plotted in Figure 1, thereby reinforcing our arguments in favour of the improving of recycling activities.

5.1 Extended model

As a first robustness check, we have split the sample and run separate regressions for small, medium size and large municipalities, as well as for the three different geographical areas. The results are very similar across sub-samples. However, our baseline model is, admittedly, very parsimonious, in that it only considers output quantities and input prices as right hand side variables. Therefore, we have enriched our specification by adding other explanatory variables that

[24] As stated by the authors: "*Household source reduction efforts presumably complement recycling practices. Households that increase recycling may simultaneously seek ways to reduce the use of shopping bags and beverage containers*" (Bohm et al., 2010, p.867).

have been usually considered in the literature. Table 5 shows the results of the estimates of our extended *TS* model[25], where a time trend *t*, size dummies, geographical dummies, density, and organizational form dummies have been included among the regressors. In particular, *small* and *medium* are dummy variables which identify municipalities where inhabitants are less than 20,000 or included in the 20,000-50,000 range, respectively. *Density* is measured by the number of persons per square km.[26] *In house* and *Intermun* take the value of 1 in the cases in which waste is directly collected by the local authority or through cooperation between different municipalities. The remaining category, *Corp,* identifies the cases in which the service is provided by a company, which may be a private firm, a State-owned firm, or a private public partnership.[27]

The second column of Table 5 reports the estimates of the extended model. The coefficient of *t* is not statistically significant, so that, in the three year period under investigation, there has not been a significant technological progress. Small and medium sized cities appear to be characterized by lower collection costs as compared to municipalities that serve more than 50,000 citizens (the omitted category). Moreover, the costs are estimated to be lower in the Northern and Central regions of the country, confirming our *a priori* expectations.

The results on ownership type and *Density* are intriguing and deserve more discussion.

The negative and significant coefficient on *In house* suggests that, as compared to the omitted category (*Corp*), in house arrangements are characterized by lower costs. Albeit the results reached by the literature are rather mixed (Bel et al., 2010), our finding is somewhat contrary to expectations. We have two possible explanations for it. First, it is possible that for refusal collection services which are directly provided by the municipality, some costs categories (depreciation, interests on debts) are not fully reported, so that costs result to be underestimated. As a robustness check, we have run regressions (of both systems [1]-[2] and [3]-[4]) after having deleted the 159 observations where the dummy variable *In house* was equal to one. The results concerning scale economies and the impact on costs of the share of recycling are virtually unchanged. Secondly, by looking at the descriptive statistics reported in Table 1, it is easy to realize that 38 of the 53 municipalities with in house arrangements are localized in the South, while in the other two regions such an ownership form is clearly marginal. Considering that, as reported by Chiades and Torrini

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[25] The estimates of the *CS* model, which are available upon request, are very similar. We decided to present the results of the *TS* model for ease of exposition, since the estimated coefficients can be straightforwardly interpreted by the reader.

[26] We have used also the number of homes per square km, or the number of buildings per square km, as alternative measure, obtaining identical results.

[27] While data limitation prevent us to disentangle the three subcategories of *Corp*, official data report that in 2005 11.1% of the Italian population was served by municipalities through in house arrangements, 58.5% by State owned firms, and only 30.4% by private operators. Therefore, a large part of municipalities classified as *Corp* (66%) organize garbage collection by relying on publicly owned firms.

(2008)[28], the share of in house arrangements in Italy has reduced from 34% in 1996 to 11% in 2005, it might be the case that municipalities that have decided to keep a direct management of the waste collection service are relatively more virtuous than the ones that have decided (or have been forced) to change their organizational form. Pursuing this line of investigation, we have run separate regressions for the three geographical areas. Quite interestingly, while the coefficient on *In house* remains significant and shows a larger effect (-0.12) for municipalities in the South (where the population served with in house arrangements has reduced from 11 million persons to 4 million persons from 1998 to 2007), it becomes positive and significant (0.17) for municipalities in the North. Finally, it must be considered that municipalities classified as *In house* are of a relatively smaller size. By running regression on the sub-sample of municipalities with less than 20,000 inhabitants, we found that *In house* keeps its positive sign but loses significance.

The coefficient of *Intermun* is positive but not significant. While Bel and Mur (2009) and Sorensen (2007) offer arguments and some empirical results in favor or against intermunicipal agreements as a way to reduce costs, our results, which show no significant effect, are inconclusive with respect to this important issue.[29] As a final remark, we must recall that our omitted category, *Corp*, is including mostly publicly owned firms (see footnotes 27 and 11), so that we are not able to examine the effect of full (or partial) privatization on costs. This must be considered if one wants to correctly interpret and appreciate our findings for the variables *Intermun* and *In house*.

The coefficient on *Density* is found to be positive and significant. It is not rare in this field of studies to interpret the sign and magnitude of such a coefficient as evidence of the existence of economies\diseconomies of density.[30] However, we think that in the case of the refuse collection industry, given the high correlation existing between municipality size and degree of urbanization as proxied by a density measure, it is not appropriate to make such an inference. A positive coefficient could indicate, as suggested by Bohm et al. (2010), that high-density municipalities may incur high costs to transport waste due to the inability to operate vehicles in densely populated urban areas,[31] as well as to the need to drive towards remote landfills for disposal. In order to

[28] The authors found, for a sample of Italian municipalities, a negative impact of *In house* arrangements on costs, too.

[29] Bel and Fageda (2009), working on Spanish data, argue that intermunicipal agreements can be used as a way to reach scale economies for relatively small municipalities, while Sorensen (2007), working on Norwegian data, underlines the difficulties of managing the service when the ownership is very dispersed, as in the case of intermunicipal joint ventures. Consistently with Sorensen's analysis, Garrone et al. (2010) found for a sample of Italian utilities operating in gas, water, electricity and refuse collection in the years 1997-2006, a positive and significant impact of a proxy of *Intermun* on total costs.

[30] Compare, for example, the comments offered by Bohm et al. (2010) and Carroll and Thomas (2001), who both found a positive coefficient on *Density* (measured as persons per square mile and number of homes per square mile, respectively).

[31] For instance, the presence of narrow streets may reduce the ability to use large, specialized equipment. In addition, the extent of on-street parking may involve difficulties in using some automated machinery, with the consequence that operators are forced to use more manual labor.

elaborate more on this, we have split *Density* into two variables, used as proxies for horizontal and vertical degrees of urbanization:

$$Density = \frac{Population}{Km^2} = \frac{Population}{Buildings} \times \frac{Buildings}{Km^2} = Urb_{HOR} \text{ x } Urb_{VER} \qquad [9]$$

The last column of Table 5 highlights that both coefficients are positive and significantly different from zero, but the impact of Urb_{HOR} is much higher. Therefore, the results are suggestive of the fact that congestion problems are more serious when the population is spread over several buildings with fewer floors than in the case with higher vertical development of buildings insisting on a given surface.

The traditional approach to measure density economies in network industries (Caves et al., 1985), requires to enrich the models [1]-[2] and [4]-[5] by including a proxy for the size of network (*N*) as an additional "output" (i, j) and to measure density economies as:

$$DE = 1/(\varepsilon_{CY_D} + \varepsilon_{CY_R}) \qquad [10]$$

and scale economies as:

$$SE=1/(\varepsilon_{CY_D} + \varepsilon_{CY_R} + \varepsilon_{CY_N}) \qquad [11]$$

The cost elasticity with respect to *N* (ε_{CY_N}) measures how much costs increase when the network size becomes bigger, keeping constant the amount of waste (both disposal and recycling) collected. Accordingly, *DE* is a measure of how costs are rising when both outputs Y_D and Y_R increase, keeping constant the size of the network. When including *N* (measured by the number of homes) and the corresponding interactions with output and input prices in the system [1]-[2], we get, for the average sample firm, the following estimates: ε_{CY_D} = 0.63 (standard error = 0.02), ε_{CY_R} = 0.19 (s.e. = 0.01) and ε_{CY_N} = 0.20 (s.e. = 0.03). Therefore, the refuse collection industry in Italy is characterized by the presence of density economies (*DE* = 1.22). This suggests that, in order to organize the service, franchised monopolies at the municipality level have to be preferred over side by side competition. In addition, the presence of scope economies points towards the organization of single tenders for both garbage and recycling activities. Finally, overall scale economies are found to be constant (*SE* = 0.99), confirming our previous results for the baseline model.[32] Therefore, our results suggest that aggregating nearby councils (which implies to increase simultaneously both waste collection and *N*) could not bring savings in total costs.

[32] Notice than in the last two columns of Table 5 the coefficient on Y_D is lower than the estimates reported for the baseline model (incidentally, when adding a proxy for density among the regressors, Bohm et al. (2010) experience a similar contraction, too). Coupling our discussion about the role of the variable *Density* with our results for density economies (*DE*), we do not think that the estimates of the extended models should be considered as supportive of the presence of increasing aggregate returns to scale for the average firm. Our interpretation is that the variable *Density* is capturing part of the magnitude of the output elasticity ε_{CY_D}.

6. Summary and conclusions

Despite the importance of the refuse collection service and the rising worries about the impact of waste disposal activities on the environment, the empirical literature on the costs of garbage collection and disposal is rather limited. The available empirical works mostly concentrate on the US, and, most importantly, recycling activities are rarely included into the analysis.

Our paper provides fresh evidence on the above issues by analysing a sample of Italian municipalities which are observed in the years 2004-2006. From a methodological point of view, we jointly consider waste taken to disposal sites or incinerated and waste sent for recycling in a multi-product framework. Moreover, we will estimate cost function models which are consistent with the duality assumptions of microeconomic theory.

Our results suggest that, for a municipality of a size of about 42,500 inhabitants, the refuse collection industry exhibits aggregate constant returns to scale, while moderate economies of scope can be enjoyed by simultaneously providing disposal and recycling services. While scope economies are increasing with the size of the council (up to 14% when inhabitants are about 300,000), decreasing returns in the collection of both garbage and waste sent for recycling are such that moderate overall diseconomies of scale appear for large municipalities.

Our simulations suggest that it is worth to devote efforts to increase the share of recycling activities up to 30%-35%, since the total costs of refuse collection would not increase too much, and this is especially true for relatively small municipalities.

The estimates of the extended model add new important insights. First, refuse collection costs are found to be lower in the Northern regions of the country and for municipalities with a population below 20,000 inhabitants. Second, urban areas face higher congestion costs especially due to horizontal urbanization effect there is clear evidence of the existence of density economies. Finally, councils that are relying on intermunicipal joint-ventures as organization forms to provide the service are not exhibiting lower costs.

From a policy point of view, we think that the above set of results provide some useful insights. Our computations suggest that recycling programs should be strongly encouraged, since total costs are not likely to increase sharply. This is particularly important in a country like Italy where, as reported in our descriptive statistics, the share of recycling activities is somewhat limited, especially in the South. The presence of density economies suggests that franchised monopolies could be the better form to provide the service, while the existence of scope economies suggests that tender procedures should be organized so as to consider disposal and recycling activities as a single bundle. However, since we found constant returns to scale up to 21,000 tons of waste collection (i.e. up to a service area of about 45,000 inhabitants), we cannot provide support for the arguments in favour of the consolidation of the service for small municipalities,

Finally, our results provide useful insights for managers in charge of the planning and management of the refuse collection services. In fact, managers must have a precise idea of the costs of garbage collection and on the impact of recycling activities on total costs when they must decide whether and to what extent participating to tendering procedures or simply when they are required to compute the budget plans for the waste management activity.

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Tables and figures

Table 1. Sample breakdown by geographical area

	North	Centre	South	*Total*
Number of municipalities	204	118	207	529
- In house	2%	9%	18%	10%
- Intermunicipal partnership	4%	4%	15%	8%
- Corporation	94%	87%	67%	82%
Share of Recycling	37%	13%	7%	20%

Table 2. Summary Statistics

	Mean	Std. dev.	Min	Max
Total Cost (10^3 euro)	5,436	23,965	46	48,065
Output				
Waste Disposed (tons)	17,122	71,196	118.44	1,462,128
Waste Recycled (tons)	3,770	13,044	8.86	210,211
Input prices				
Price of capital	0.102	0.021	0.040	0.160
Price of labor (euro)	36,607	5,735	22,663	62,613
Cost shares				
Capital share (%)	5.71	3.90	1.00	17.90
Labor share (%)	44.90	12.01	18.91	73.02
Other variables				
Density	902.8	1,242	21.83	9,441
Population	41,058	142,272	993	2,711,491
Number of homes	19,336	67,165	430	1,150,547
Number of buildings	4,960	7,309	353	127,713
Share of recycling (%)	19.8	17.7	0.1	76.5

Table 3. NLSUR estimation: Translog (TS) and Composite (CS) cost function models [a]

	TS MODEL	CS MODEL
Output and factor price elasticities [b]		
ε_{CY_D}	0.7812*** (0.0120)	0.7601*** (0.0082)
ε_{CY_R}	0.2305*** (0.0112)	0.2466*** (0.0099)
S_L	0.4480*** (0.0030)	0.4487*** (0.0028)
S_K	0.0604*** (0.0019)	0.0546*** (0.0257)
Scale and Scope Economies [b]		
SE	0.9884 (0.0088)	0.9933 (0.0057)
SCOPE	-	0.0209** (0.0089)
Cost complementarities CC	-0.1196*** (0.0070)	-
Cost function		
R^2	0.9308	0.9298
SSE [c]	133.48	135.34
Labor share equation		
R^2	0.1261	0.2342
SSE	20.01	17.53
Capital share equation		
R^2	0.1784	0.2060
SSE	1.99	1.93
System log-likelihood	4083.09	4175.86
Goodness of fit [d]	0.8237	0.8262
VLR test statistic [e]	CS vs. TS: VLR = 8.03***	

[a] Estimated asymptotic standard errors in parentheses.
[b] The values are computed at average values of output and input price variables. The coefficient subscripts are *D* = disposal, *R* = recycling, *K* = capital, *L* = labor.
[c] Sum of squared errors.
[d] The goodness-of-fit measure for the NLSUR systems is McElroy's (1977) R^2.
[e] See Vuong (1989). The VLR statistic is distributed as a N (0,1).
.*** Significant at 1 percent level in a two-tailed test. ** Significant at 5 percent level * Significant at 10 percent level. For the *SE* index the null hypothesis is that it is not significantly different from one

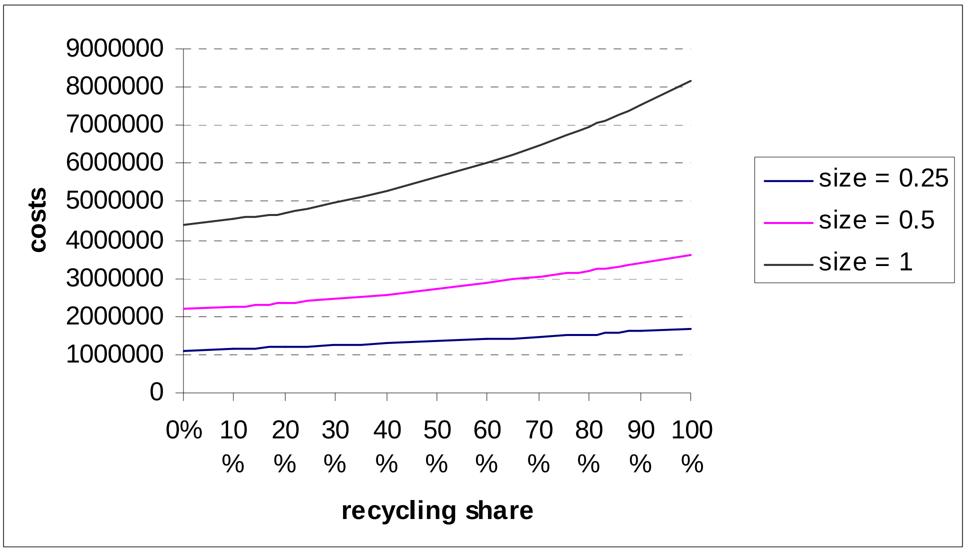
Table 4. Estimated costs (*CS* model) for disposal and recycling at different output levels

	Scaled outputs [a]					
	$\lambda = 0.25$ Y_D=4,281 Y_R =943	$\lambda = 0.5$ Y_D=8,561 Y_R =1,865	$\lambda = 1$ Y_D=17,122 Y_R =3,770	$\lambda = 2$ Y_D=34,244 Y_R =7,540	$\lambda = 4$ Y_D=68,488 Y_R =15,080	$\lambda = 8$ Y_D=136,976 Y_R =30,160
Population Size	*13,500*	*25,000*	*42,500*	*92,000*	*163,000*	*290,000*
Product-specific estimated costs:						
A) $C(Y_D, 0)$	892	1,784	3,592	7,308	15,136	32,382
B) $C(0, Y_R)$	292	583	1,190	2,497	5,491	12,994
Total (A+B)	*1,184*	*2,367*	*4,782*	*9,805*	*20,627*	*45,376*
Multi-product estimated costs						
C) $C(Y_D, Y_R)$	*1,170*	*2,336*	*4,684*	*9,439*	*19,189*	*39,649*
Scope Economies [b] [(A+B)/C]-1	0.01 (0.01)	0.01 (0.01)	0.02** (0.01)	0.04** (0.02)	0.07*** (0.03)	0.14*** (0.05)
Overall Scale Economies (SE) [b]	0.99 (0.01)	1.00 (0.00)	0.99 (0.01)	0.98 (0.01)	0.97* (0.02)	0.94* (0.04)

[a] λ =1 indicates a municipality that collects average quantities of Y_D and Y_R. The parameter λ is used to scale up and down the outputs of the "average municipality". Costs are measured in thousands of euros.
[b] Standard errors in parenthesis.*** Significant at 1 percent level in a two-tailed test. ** Significant at 5 percent level * Significant at 10 percent level. For the *SE* index the null hypothesis is that it is not significantly different from one.

Figure 1. Costs and Recycling Shares

1 a) Municipalities below 50,000 inhabitants.



1 b) Municipalities above 50,000 inhabitants.

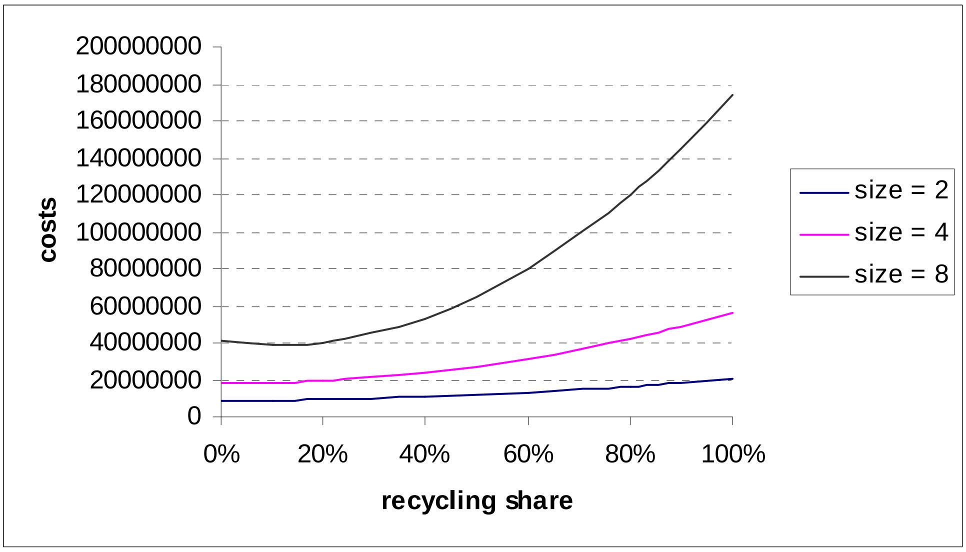


Table 5. Translog estimates of the baseline and extended cost function models [1]-[2]

REGRESSORS [a]	BASELINE MODEL		EXTENDED MODEL I		EXTENDED MODEL II	
	Estimates	s.e.	Estimates	s.e.	Estimates	s.e.
Constant	15.402***	(0.012)	15.576***	(0.035)	15.154***	(0.057)
$\ln Y_D$	0.781***	(0.012)	0.712***	(0.018)	0.707***	(0.018)
$\ln Y_R$	0.231***	(0.011)	0.230***	(0.016)	0.229***	(0.016)
$\ln Y_D{}^2$	0.367***	(0.027)	0.299***	(0.028)	0.301***	(0.027)
$\ln Y_R{}^2$	0.178***	(0.012)	0.156***	(0.014)	0.161***	(0.014)
$\ln Y_D \ln Y_R$	-0.120***	(0.007)	-0.099***	(0.007)	-0.101***	(0.007)
$\ln P_L$	0.448***	(0.003)	0.448***	(0.003)	0.448***	(0.003)
$\ln P_K$	0.060***	(0.002)	0.060***	(0.002)	0.060***	(0.002)
$\ln P_L{}^2$	-0.254***	(0.018)	-0.255***	(0.018)	-0.254***	(0.018)
$\ln P_K{}^2$	0.034***	(0.006)	0.040***	(0.006)	0.040***	(0.006)
$\ln P_L \ln P_K$	0.056***	(0.009)	0.060***	(0.009)	0.060***	(0.009)
$\ln Y_D \ln P_L$	0.005***	(0.001)	0.005***	(0.001)	0.005***	(0.001)
$\ln Y_R \ln P_L$	-0.002**	(0.001)	-0.002**	(0.001)	-0.002**	(0.001)
$\ln Y_D \ln P_K$	0.148***	(0.041)	0.123***	(0.042)	0.133***	(0.041)
$\ln Y_R \ln P_K$	-0.013	(0.029)	-0.012	(0.030)	-0.013	(0.030)
t	-	-	0.001	(0.008)	0.002	(0.008)
ln *Density*	-	-	0.060***	(0.007)	-	-
ln Urb_{HOR}	-	-	-	-	0.118***	(0.013)
ln Urb_{VER}	-	-	-	-	0.037***	(0.008)
North	-	-	- 0.129***	(0.027)	- 0.151***	(0.027)
Center	-	-	-0.101***	(0.022)	-0.121***	(0.022)
Medium	-	-	-0.099***	(0.034)	-0.084**	(0.034)
Small	-	-	-0.166***	(0.040)	-0.150***	(0.040)
In house	-	-	-0.063***	(0.024)	-0.062***	(0.023)
Intermun	-	-	0.025	(0.026)	0.021	(0.026)
Goodness of Fit [b]	0.8237		0.8361		0.8379	
System Log-Lik.	4083.09		4156.22		4167.91	

[a] Estimated asymptotic standard errors in parentheses. [b] The goodness-of-fit measure systems is McElroy's (1977) R^2. The coefficient subscripts are D = disposal, R = recycling, K = capital, L = labor. *** Significant at 1 percent level in a two-tailed test. ** Significant at 5 percent level in a two-tailed test. All regressors, except from dummies and *t*, have been normalized on their respective sample mean values.