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Cost-efficiency in packaging waste management: The case of Belgium

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ABSTRACT

In order to exploit economies of scale Belgian municipalities regularly cooperate in the provision of waste related services. In particular for the collection and separation of household packaging waste, municipalities appear to seek technical and cost efficiency gains by cooperating via municipal waste joint ventures. Although most Belgian municipal waste joint ventures can present excellent recycling and recovery rates for household packaging waste, their performance in terms of cost-efficiency has never been assessed. Using a unique dataset comprising of the costs for all 35 Belgian municipal waste joint ventures in 2010, this paper present the first assessment of the cost efficiency of household packaging waste collection in Belgium. As we are not sure on the relative importance of the separate cost efficiency scores for the three selectively collected household packaging waste fractions when determining the overall cost efficiency, this paper draws on the Benefit-of-the-Doubt approach. Our results indicate that, despite the substantial cooperation between municipalities, still considerable differences in cost efficiency for household packaging waste collection exist.

1. Introduction

In 2010 the Belgian municipal waste joint ventures received almost 128 million euros from the green dot organization Fost Plus to finance their household packaging waste related activities. Although most waste joint ventures can present excellent recycling and recovery rates for household packaging waste, their performance in terms of cost efficiency has never been assessed. With national recycling targets of packaging waste that exceed the targets defined by the Packaging and Packaging waste directive of the EU (directive 94/62/EC amended by directives 2004/12/EC and 2005/20/EC) and a financial support model for the municipal waste joint ventures that does not explicitly promotes cost efficiency, an analysis of cost efficiency gains could prove very valuable for all stakeholders. In addition cost efficiency estimates of the entire municipal solid waste (MSW) management system for one of the three Belgian regions revealed that municipalities are in general rather cost inefficient in MSW collection and processing services (De Jaeger et al., 2011; Rogge and De Jaeger, 2012, 2013). This paper therefore aims at measuring the cost efficiency for selective collection efforts of household packaging waste in Belgium.

For this purpose we will build on a Benefit-of-the-Doubt (BoD) approach. This non-parametric tool, which is rooted in Data Envelopment Analysis (DEA), allows us to evaluate the comparative overall cost efficiency performance of a set of similar activity units. In our case the method can be used to compute an overall cost efficiency score of the municipal joint ventures' collection efforts based upon the collection cost efficiency scores for the three selectively collected packaging waste fractions. The key feature of the BoD-model is that it uses an endogenous weight selection procedure in the aggregation of the multiple performance indicators. This is an important advantage as we are not sure on the exact importance of the cost efficiency scores of the separate waste fractions when determining the overall cost efficiency. In addition the use of so-called optimistic and pessimistic BoD-based evaluations will allow us to determine the range of cost efficiency values in which the exact cost efficiency score is believed to lie. This range can in turn be used to visualize the level of uncertainty of the collection cost efficiency ranking of each municipal joint venture. Finally an order-*m* version of the BoD model will be used as a robustness check.

Both effectiveness and efficiency of MSW management have received a substantial amount of attention in the international scientific literature during the last decades. A large part of studies focusing on the effectiveness (i.e. the relation between outcomes and desired policy objectives in MSW management), attempt to estimate the impact of unit based pricing on the disposal behavior by households. An overview of the latter studies (up to 2005) in Kinnaman (2006) reveals that the demand for MSW collection services is rather inelastic. However Kinnaman (2006) argues that unit based pricing has a more substantial impact on disposal behavior if households recycle little prior to the introduction of the unit based pricing system. More recently Allers and Hoeben (2010) showed that in the Netherlands user fees reduce the amount of unsorted

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waste and increase the amount of recycling, although the reduction in unsorted waste is much higher than the increase in recycling. For Belgium (Flanders) Gellynck and Verhelst (2007) found that the level of the bag price has a significant negative impact on the amount of unsorted waste collected. However their results also indicate that the ease of recycling (expressed as the number of waste fractions which are collected separately for recycling purposes) has no impact on the amount of unsorted waste collected. Next to the pure policy variables such as the characteristics of the unit based pricing program, many studies also include socioeconomic variables to explain recycling behavior. For instance Jenkins et al. (2003) found a significant effect of variables like age income and education on the intensity of recycling effort for recycling behavior of several materials in the United States. More recently, Kipperberg (2007) shows that socioeconomic characteristics are less important predictors of recycling behavior in Norway compared to the results of Jenkins et al. (2003).

Our paper fist within the second group, where the efficiency of waste management (i.e. to the relation between minimal inputs and maximal outputs of waste management) rather than the effectiveness is assessed. Recent additions to the literature include, amongst others, Chen and Chen (2012), Chen et al. (2010), Marques and Simões (2009), Simões et al. (2010) and Simões and Marques (2012). Efficiency studies of selective collection of packaging waste however, remain relative scarce. An interesting overview can be found in Marques et al. (2012a). The authors distinguish between studies that focus on the optimal percentage of recycling (see for instance Lavee, 2007), the market structure (a recent example is Abrate et al., 2011) and recycling costs. In the last category – the most relevant one in our case – some recent contributions have led to interesting new insights. For instance Cruz et al. (2012) analyzed the cost recovery rate for Portuguese local/regional authorities in the recycling of packaging waste. They found that if the opportunity costs of diverting the packaging waste from landfills are taken into account, the cost coverage is about 127%. Using a similar methodology Marques et al. (2012b) report cost coverage results of 135% for France. However for the local authorities in Romania the authors found a cost coverage of only 87%. Nevertheless adopting an economic perspective (i.e. including the opportunity costs), seems to have a significant impact on the conclusions. This is also confirmed by Larsen et al. (2010), where the authors argue that mainly due to the high cost for incineration avoided, municipal costs for collection and treatment of waste decrease with increasing recycling for the municipality of Aarhus in Denmark.

However the results presented in Cruz et al. (2012) and Marques et al. (2012b) also reveal rather high operational costs for selective collection and sorting for packaging waste compared to the costs of refuse collection and treatment. Similarly for the U.S. Bohm et al. (2010) report that both marginal as average costs are higher for recycling systems compared to waste collection and disposal services. Clearly, those proportional high operational costs can prompt questions about the cost-efficiency of selective collection and treatment of packaging waste. In order to stimulate an efficient packaging waste management system, some countries therefore implemented contribution schemes which depend on the performance of the local authority. For instance in France and Portugal the financial support for the local authorities organizing the collection of packaging waste depends on the per capita collection results (Marques et al., 2012b). The impact of such financial models on cost-efficiency remains a topic for further research, but the possibility to realize cost-efficiency gains has already received some interest from scholars. For instance after analyzing the main determinants of collection rates of household plastic packaging waste in Swedish municipalities, Hage and Söderholm (2008) tentatively conclude that national collection of plastic packaging waste in Sweden could be cost-inefficient. The authors argue that in particular the compensation system for the waste collection operators tends to reduce regional cost differences. Similarly, when applying several non-parametric models to measure the efficiency of Portuguese recycling companies, Marques et al. (2012a) found significant inefficiencies. The authors conclude that a lack of incentives is one of the main reasons for the poor performance and recommend a funding scheme system based on more than the amount of packaging waste collected per capita.

Given the above considerations, we believe that our paper could contribute in several ways. First, the empirical evidence presented in this paper could fuel further discussion on the appropriate funding scheme to promote efficiency gains in selective packaging waste collection. Secondly the results could be highly relevant for both the joint ventures as well as the policy makers on local and regional level. Finally, to our best knowledge, this is the first paper that employs optimistic and pessimistic BoD-based evaluations to analyze the efficiency of packaging waste collection efforts (Section 4).

2. Institutional background

Although packaging waste management in Belgium is an area where essential authority remains with the regional governments, a cooperation agreements effectively aligns packaging waste policy between Flanders, Wallonia and the Brussels-Capital region. An important task hereby is monitoring the so-called extended producer responsibility for packaging waste. This responsibility implies that every company responsible for bringing more than 300 kg of packaging material on the Belgian market must meet the recovery and recycling quotas specified in the cooperation agreement. The quotas include both general targets as minimum recycling targets for a number of specific waste streams (see Table 1 for an overview). The general targets require that minimum 80% of the packaging material should be recycled – i.e. reintroduced in a production process – and minimum 90% of the packaging material should be recovered – i.e. recycled or incinerated with energy recovery. For instance a company responsible for bringing 50 metric tons of packaging material on the Belgian market has to prove that minimum 40 metric tons of the material are recycled and minimum 5 additional metric tons are recovered.

Clearly it would be economically inefficient for each company to organize its own collection and separation program. Therefore the vast majority of the companies conclude an agreement with the officially accredited organization fulfilling the extended producer responsibility objectives for the industry (this is currently the non-profit organization Fost Plus). In terms of quantities of packaging waste, Fost Plus (2011) reports that of the estimated 817,171 metric tons brought onto market in 2010, about 690,828 metric tons were recycled via Fost Plus. In practice the member companies pay a fee, or Green Dot tariff, to Fost Plus. In return Fost Plus promotes, coordinates and finances the selective collection of a number of household packaging waste fractions. These include glass, paper and cardboard and the so-called PMD fraction. The latter fraction consists of plastic bottles and flasks, packaging metals and drinks cartons and is collected on the curbside in a single bag. Next to the collection, Fost Plus also finances the separation of the PMD fraction into, amongst others, steel, aluminum, PET (Polyethylene Terephthalate) and HDPE (High Density Polyethylene). Although Fost Plus uses its revenues to finance household packaging waste management, the responsibility to organize the collection and separation of packaging waste remains, in theory, at the municipal level. However to exploit economies of scale when providing this service, almost all Belgian municipalities engage in inter-municipal cooperation (IMC) with other, often neighboring municipalities via so called municipal waste joint ventures. In 2010, 585 of the 589

Table 1
Recycling targets and results in 2010.

	Target 2004/12/EC (%)	General recycling rate (%)	Target cooperation agr. (%)	Recycling rate Fost Plus (%)	Recycling rate VAL-I-PAC (%)
Glass	60.0	100.0	60.0	100.0	–
Paper and cardboard	60.0	90.2	60.0	100.0	96.1
Drinks cartons	–	77.3	60.0	78.7	–
Metal	50.0	94.7	50.0	100.0	85.1
Plastics	22.5	41.5	30.0	37.9	55.7
Wood	15.0	63.3	15.0	–	64.6
All packaging streams	60.0	79.8	80.0	83.2	81.6

Source: For recycling rates see IPC (2011).

Belgian municipalities organized packaging waste collection and separation via one of the 31 municipal waste joint ventures (Fost Plus, 2012). Only 4 municipalities opt to organize their own collection system. The most common form of IMC for waste management services is the so called 'pure' municipal joint venture, a cooperation format in which the private sector cannot participate directly (see Marques et al. (2012c) for a more detailed overview of the different forms of IMC).

The administrations that organize the collection and separation of packaging waste – i.e. the 31 municipal waste joint ventures and 4 individual municipalities – (for simplicity henceforth called MJV's) receive a contribution from Fost Plus to finance its packaging waste related service. In contrast to for instance Portugal (see Cruz et al., 2012), these contributions are not calculated using the values that correspond to a certain per capita generation of packaging material. Fost Plus rather reimburses the full costs that the MJV's incur for collection and processing of packaging material (IPC, 2008). Since some MJV's contract collection and separation services of packaging materials out to a third party, they are simply reimbursed according to the costs mentioned on the outsourcing contract. As MJV's sometimes differ in the level of service offered to their residents (for instance the number of collection rounds or the number of pick-up points), Fost Plus only reimburses the costs incurred for a "standard" level of service. For example the standard level of service includes PMD collection twice a month at curbside, while additional collection rounds should be financed by the MJV (a detailed description of the services included in the standard level can be found in IPC (2008)). MJV's that collect and sort packaging waste themselves are also reimbursed, but they must provide a proof of the full costs incurred in case they spent more than a reference cost outlined by Fost Plus and approved by IVCIE (IPC, 2008).

3. The data

Before turning to the data used in our analysis, we give a brief overview of waste figures in Belgium for 2010. Total municipal solid waste generation in Belgium is with its 466 kg/capita in 2010 well below the EU-27 average[1] (Eurostat, 2012). Nevertheless, 12 of the 27 EU countries can present lower per capita waste generation figures. In terms of recycling rates of packaging waste however, Belgium is clearly at the forefront of the EU. With a recycling rate of almost 80% for household and industrial packaging waste in 2010, Belgium is only outperformed by Denmark (Eurostat, 2012). With this result Belgium exceeds the general packaging waste recycling target set by the Packaging and Packaging Waste Directive 94/62/CE (and the reviewed directive 2004/12/EC). In addition, the recycling targets for specific packaging material streams defined in the Packaging and Packaging Waste Directive are all easily met (see the first column in Table 1 for the recycling targets of the directive and the second column for the actual recycling rates obtained in Belgium). As mentioned in the previous section, the cooperation agreement also specifies general and specific recycling targets (see the third column of Table 1 for an overview). Although the targets and actual recycling results apply to the same packaging waste streams as in the Packaging and Packaging Waste Directive, the calculation method differs somewhat. For instance, in contrast to the directive, the cooperation agreement omits reusable or non-declared packaging waste when calculating the obtained recycling results. Therefore the recycling rates for Fost Plus (household packaging waste) and VAL-I-PAC (industrial packaging waste) presented in the two last columns of Table 1, are not entirely comparable to the general recycling results.

In order to evaluate the cost efficiency of household packaging waste collection, we use a unique dataset comprising of the household packaging waste quantities collected separately and the corresponding contributions each MJV received from Fost Plus to finance its packaging waste related services in 2010. As the contributions are the equivalent of the real and proven cost for providing the standard level of service (see previous section) we can use the data as an input in our cost efficiency analysis without worrying about data comparability issues.[2] The quantities correspond to the actual amounts that were separately collected by the MJV's in 2010. Note that our analysis omits any industrial packaging waste. As we focus on the cost efficiency of household packaging waste collection efforts, all costs linked to separation activities are also excluded from our analysis. The dataset was provided to us by Fost Plus, but due to the confidential nature of the data, we cannot display the detailed figures for individual MJV's in this paper. Therefore we opt for a combination of choropleth maps (see Fig. 1) and descriptive statistics (see Table 2) to present the data.

Although some packaging waste streams are further separated after collection, we focus on the three fractions – cardboard and paper, PMD and glass – which are collected separately at the source. Table 2 shows that paper and cardboard is by far the largest selective collected packaging waste stream in terms of weight. However, only a share of this waste stream actually consists of discarded packaging material (the lion's share comprises of newspapers, advertising leaflets, magazines, etc.). Therefore 75% of paper and cardboard is not brought into the equation when calculating the official overall recycling rate of packaging waste in Belgium. As we wish to evaluate cost efficiency of the collection activities and it is difficult to allocate the exact collection cost to the packaging fraction of this waste stream, we will include 100% of collected paper

[1] Eurostat defines municipal waste generated as waste collected by or on behalf of municipal authorities. Although Municipal waste is mainly produced by households, similar wastes from sources such as commerce, offices and public institutions are included (see http://epp.eurostat.ec.europa.eu/cache/ITY_SDDS/en/env_wasmun_esms.htm for the metadata for municipal waste).

[2] As we focus on collection cost, sorting costs were excluded from the dataset. Any additional fees paid by Fost Plus, such as a fee which can be used for increasing the level of service for glass collection are also excluded from our analysis, as we which to evaluate the cost efficiency for the standard level of service.

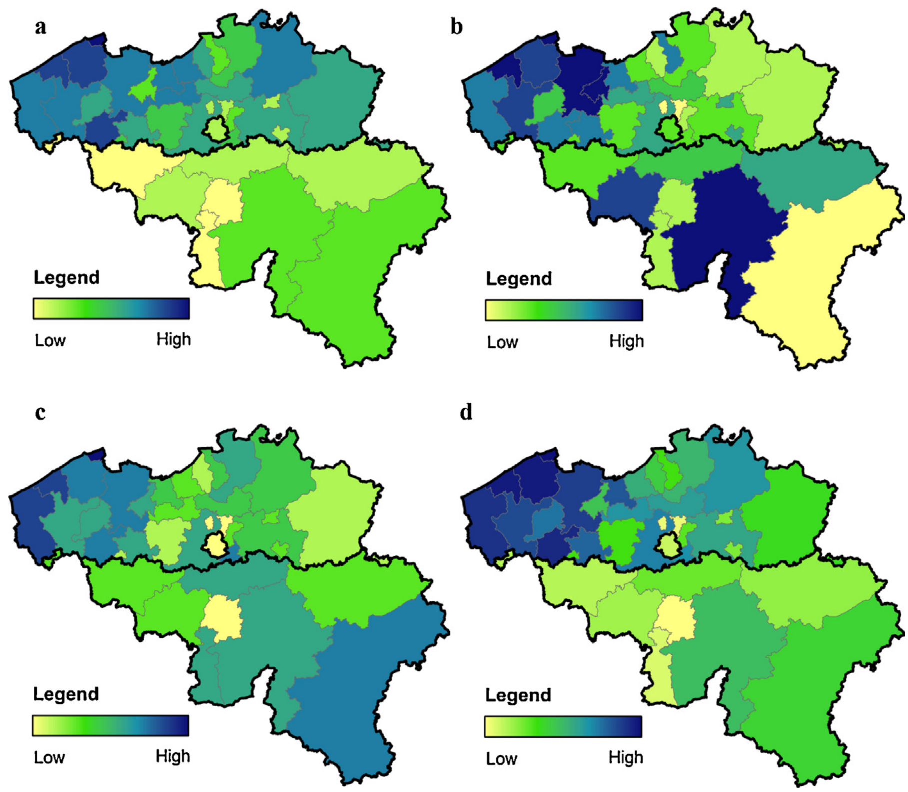

Fig. 1. Amount of household packaging waste collected in Belgium in 2010. (a) Amount of paper and cardboard collected per capita. (b) Amount of PMD collected per capita. (c) Amount of glass collected per capita. (d) Total amount of packaging waste collected per capita. Note: PMD collection for the most southern MJV (AIVE) is due to its different scenario difficult to compare with the other MJV's.

Table 2
Descriptive statistics.

	Mean	Std. Dev.	Min	Max
kg/cap separately collected via the Green dot system				
Packaging waste total	113.16	20.04	84.44	196.20
Paper and cardboard	68.67	13.51	48.30	115.50
PMD	15.00	1.83	10.34	18.13
Glass	29.49	7.00	22.92	62.69
Cost/ton for separate collection via the Green dot system				
Packaging waste total	71.84	12.88	49.64	105.50
Paper and cardboard	53.34	10.24	31.36	75.50
PMD	199.62	42.52	135.13	362.59
Glass	49.94	12.14	24.58	89.73

and cardboard in our analysis. Also note that the amount of paper and cardboard reported in this table exceeds the amount that is brought onto the Belgian market. According to the Interregional Packaging Commission (IPC, 2012) this is due particularly to the fact that households also hand in paper and cardboard that is considered as industrial waste. The second packaging waste fraction, PMD, is the most heterogeneous one, as it comprises of plastics bottles and flasks (PET and HDPE), metal (steel and aluminum) and drinks cartons. Although the average weight collected per capita seems rather low, the recycling rates depicted in Table 1 indicate that the majority of this PMD brought onto the market is collected separately. For the final packaging waste fraction, glass, the standard deviation (std. dev. = 7.00) is relative high compared to the average amount collected per capita (=29.49).[3] This indicates that the average collection results differ considerably between the MJV's. Comparable to the case of paper and cardboard, the total amount of selective collected glass is higher than the estimated amount brought onto the market. Fost Plus argues that parallel imports, estimated at 30,000 metric tons, could explain this difference (Fost Plus, 2012).

Fig. 1d reveals a considerable degree of geographic clustering for selective collection of packaging waste in terms of kg per capita. Most MJV's in the Flemish region achieve higher collection rates per capita compared to the Walloon and Brussels region. When looking at the geographic dispersion of the three packaging waste streams separately, the picture seems less clear (see Fig. 1a–c). In particular for the PMD and glass fractions, the collected quantities are more heterogeneously distributed over the different regions, indicating that the relative heavy paper and cardboard fraction is one of the main drivers of geographic clustering of total packaging waste.

The cost figures in Table 2 are based on the contributions paid by Fost Plus to finance the full and real cost for the standard level of service. The figures reveal that the average cost to collect one metric ton of PMD is considerably higher compared to the other two fractions. This is not surprising considering that the standard level of service includes a twice-per-month PMD curbside collection for a waste fraction which has a relative high volume/weight ratio. For glass, which is collected via bottle banks, and paper cardboard, which is collected once a month at the curbside (if the standard level of service is applied), the collection costs per metric ton are roughly a quarter of the costs for PMD. Note that the relative high standard errors for all average cost figures are a first indication that the cost-efficiency might differ significantly between the MJV's. In addition the choropleth maps in Fig. 2 reveal a high degree of geographic clustering, showing that regional differences in the collection cost per metric ton of packaging waste might play a role.

[3] This high correlation is for a large part driven by outlying quantities collected in the tourist regions near the coast.

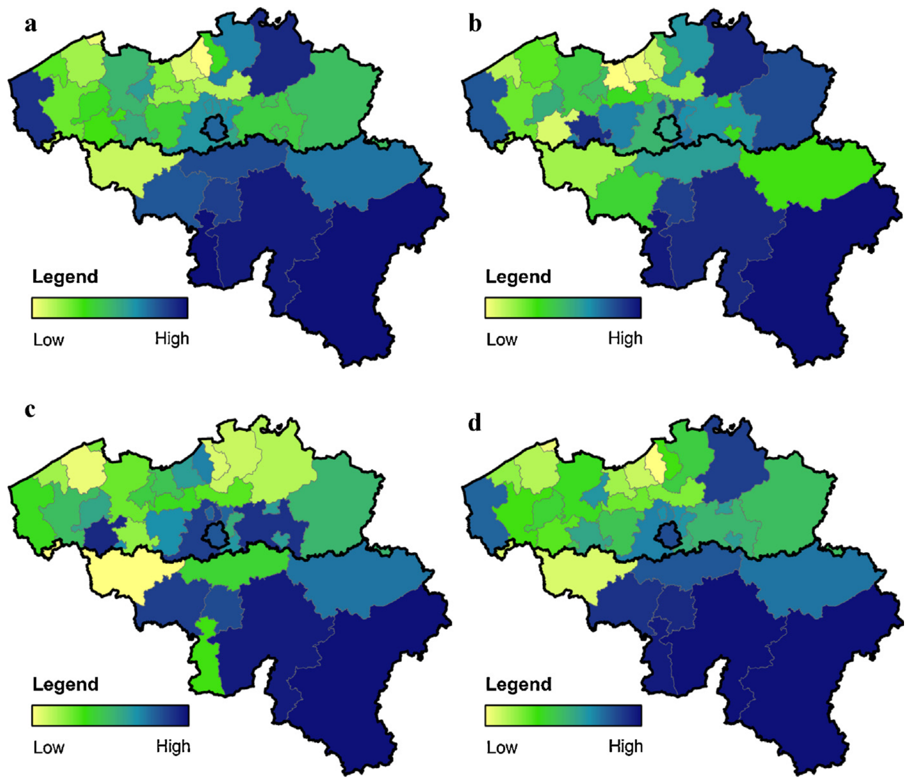

Fig. 2. Cost per ton for household packaging waste collected in Belgium for the baseline scenario. (a) Cost per ton for paper and cardboard collection services. (b) Cost per ton for PMD collection services. (c) Cost per ton for glass collection services. (d) Cost per ton for all packaging waste collection services. Note: PMD collection for the most southern MJV (AIVE) is due to its different scenario difficult to compare with the other MJV's.

4. Methodology

As our dataset comprises of the collection costs for each of the three packaging waste fractions (for the standard scenario), calculating cost efficiency scores for each fraction separately would yield no additional insight compared to the cost per metric ton statistics displayed in Fig. 2. Although these figures can generate some interesting insights for the stakeholders, a more relevant question is how municipal joint ventures perform in terms of overall cost efficiency for collection the three packaging waste fractions.

To compute the overall cost efficiency of the MJV's, we aggregate the cost efficiency scores on the separate packaging waste fractions using two different versions of the BoD-model. The BoD-methodology is rooted in DEA, an efficiency measurement technique originally developed by Farrell (1957) and put into practice by Charnes et al. (1978) in the Management Science and Operations Research literature. In essence, the BoD-model is a non-parametric tool for evaluating the comparative overall efficiency performance of a set of similar activity units (e.g. companies, production units, individuals, countries, MJV's, etc.) given data observations on an array of individual performance indicators that measure multi-dimensional concepts, and often no precise understanding on the true importance weights.[4] In the current application of evaluating the overall cost efficiency of the Belgian MJV's in collecting multiple packaging waste fractions, this means that the BoD-model can be used to compute an overall cost efficiency of the MJV's collection efforts based upon the cost efficiency in the collection for the three packaging waste fractions. The key feature of the BoD-model is that it uses an endogenous weight selection procedure in the aggregation of the multiple performance indicators. Stated differently, in the absence of detailed knowledge on the correct importance weights for the different waste fractions, the BoD-model retrieves information on the importance weights from the cost efficiency scores realized by the MJV's in the collection of the separate waste fractions (i.e. a posteriori letting the performance data speak by themselves). In particular, the basic idea is to put the cost efficiency scores realized by the MJV in the collection of the three waste fractions in a relative perspective to the cost efficiency scores realized by the other MJV's in the sample set and look for the waste fractions where the MJV's do relatively well compared to the others and/or waste fractions where they perform relatively poorly. It is based on these relative comparisons that the BoD-model determines the weights. For a step-wise presentation of the BoD-model as well as an extensive discussion of the relation between the DEA-model and the BoD-model (also suited for non-specialist audience), we refer the interested reader to Cherchye et al. (2007).

We employ an optimistic and pessimistic version of the BoD-model as discussed by, among others, Zhou et al. (2007) and Rogge (2012). The optimistic version of the BoD-model as in (1) is the traditional version of the BoD-model. It evaluates each MJV under the most favorable evaluation conditions (i.e. the most favorable importance weights). More specifically, this

[4] Note that the traditional DEA-model as well as alternative versions of the DEA-model (e.g. slack-based efficiency model) have been used by Marques and Simões (2009) and Marques et al. (2012a,b) in efficiency analyses of recycling systems. Their approach differs in that they consider multiple inputs and multiple outputs whereas in the current paper, by using the BoD-model, only multiple outputs are considered in the analysis (i.e., the cost efficiency realized by the municipalities in the collection of the three different fractions). More generally, in comparison to traditional DEA-setting, the only difference is that the BoD-model only looks at achievements or outputs without explicitly taking into account the input dimension. For a more elaborate discussion, see among others Melyn and Moesen (1991) and Cherchye et al. (2007).

version of the BoD-model looks for the set of weights that maximize the impact of collection cost efficiency scores of relative strength and minimize the influence of collection cost efficiency scores of relative weakness. Stated otherwise, the optimistic version of the BoD-model assigns for each MJV venture high weights to the waste fractions in which it performs well in terms of collection cost efficiency and low weights to the waste fractions in which it realizes more moderate or even poor collection cost efficiency scores. By consequence, the estimated e_k^g-values are optimistic approximations of the true e_k. The pessimistic version of the BoD-model as in (2) takes the opposite stance in the evaluations of the overall collection cost efficiency of the Belgian MJV's. This version of the model looks at how well MJV's perform vis-à-vis each other under the least favorable evaluation conditions (i.e. using the least favorable importance weights).[5] In particular, are MJV's able to keep up their good evaluation score relative to other MJV's under the 'worst-case' evaluation scenario? Contrary to the optimistic counterpart, the pessimistic BoD-model assigns high weights to the waste fractions in which the evaluated MJV performs relatively poor in terms of collection cost efficiency relative to the other MJV's and low weights to the waste fractions in which it realizes high collection cost efficiency scores. So, in essence, the pessimistic variant of the weighting model evaluates how close each MJV is from the worst performing MJV(s) in the data set under the least favorable evaluation conditions (i.e. worst possible weights). Formally, the optimistic and pessimistic version of the BoD-model are computed by the following linear programming models:Best-case evaluation scenario

$$\underset{w_{k,r}^g}{\text{Max}} \quad e_k^g = \sum_{r=1}^{s} w_{k,r}^g y_{k,r} \tag{1}$$

$s.t.$

$$\sum_{r=1}^{s} w_{k,r}^g y_{j,r} \leq 1 \quad \forall j = 1, \ldots, n$$

$$w_{k,r}^g \geq \varepsilon > 0 \quad \forall r = 1, \ldots, s$$

Worst-case evaluation scenario

$$\underset{w_{k,r}^b}{\text{Min}} \quad e_k^b = \sum_{r=1}^{s} w_{k,r}^b y_{k,r} \tag{2}$$

$s.t.$

$$\sum_{r=1}^{s} w_{k,r}^b y_{j,r} \geq 1 \quad \forall j = 1, \ldots, n$$

$$w_{k,r}^b \geq \varepsilon > 0 \quad \forall r = 1, \ldots, s$$

In both versions of the BoD-model, there are n MJV's ($j = 1, \ldots, n$) that collect s different packaging waste fractions (in our current application, it holds that $s = 3$). The $y_{j,r}(r = 1, \ldots, s)$ are the cost efficiencies realized by the MJV's in the collection of these s waste fractions. The BoD-model is computed n times, one computation per MJV in the data set. That is to say, each MJV is singled out once and evaluated relative to all the municipalities in the data set. This MJV under evaluation is referred to as MJV 'k'. The waste fractions, the associated importance weights for the waste fractions, and the overall collection cost efficiency scores of the evaluated MJV k are denoted by respectively $y_k = (y_{k,1}, \ldots, y_{k,s})$, $w_k^g = (w_{k,1}^g, \ldots, w_{k,s}^g)$ and e_k^g in the best-case evaluation scenario and $y_k = (y_{k,1}, \ldots, y_{k,s})$, $w_k^b = (w_{k,1}^b, \ldots, w_{k,s}^b)$ and e_k^b in the worst-case evaluation scenario.

Note that in both the optimistic and pessimistic version of the endogenous BoD-model, there are two minor constraints. The second constraint is the non-negativity constraint. This constraint limits weights to be non-negative. The first constraint is a normalization constraint. In the optimistic version of the BoD-model, this constraint enforces that the overall collection cost efficiency scores of all MJV's in the dataset, as computed with the optimal weights of the assessed MJV k, can at most be one (or, equivalently, 100%). In the pessimistic version of BoD, the normalization constraint imposes that the overall collection cost efficiency scores of all MJV's computed using the least favorable weights of MJV k, should be at minimum one. By consequence, it holds that $0 \leq e_k^g \leq 1$ and $1 \leq e_k^b$. The normalization constraints also highlight the benchmarking idea of the BoD-models: the most (least) favorable weights of the each assessed MJV are applied to the performances of all other MJV's. One is in that way, effectively looking for which of the other MJV's performances are worse, similar or better. In the interpretation of e_k^g, a value of one indicates that the evaluated MJV receives the highest possible collection cost efficiency score being assessed optimally. The MJV acts in that case as its own benchmark. Values of e_k^g below one reveal that there is at least one other MJV that performs better than the MJV under evaluation, even when applying the latter's most favorable weights in the evaluations. The value $1 - e_k^g$ then indicates the room for cost efficiency improvement in the MJV k's overall collection performance. In the interpretation of e_k^b, high values denote that the MJV does relatively well in terms of collection cost efficiency (in the sense that there are no waste fractions on which the municipal joint venture is moderately or very cost inefficient compared to the other MJV's). On the other hand, values of e_k^b close to one show that the MJV is performing very cost inefficient on at least one (and probably more) waste fractions compared to the other MJV's.

As argued by Zhou et al. (2007), the optimistic and pessimistic BoD-based evaluations mark out a range of overall cost efficiency values (specific for each evaluated MJV) in which the exact cost efficiency score is believed to lie. Rogge (2012) demonstrated how these scores also provide other interesting information. In particular, a brief comparison of the two 'boundary' cost efficiency scores (and the corresponding rankings) provides an important indication of whether or not undesirable specialization is present in the collection performances (in the sense that MJV's realize high cost efficiencies in collecting one or multiple waste fractions and very low cost efficiencies in collecting other waste fractions).

To conclude, note that by computing a range of possible overall cost efficiency estimates instead of a point estimate (as in traditional DEA and BoD), the optimistic vs. pessimistic BoD-approach already to a large extent accounts for potential robustness issues. However, by using insights of Cazals et al. (2002), one could also opt to estimate a so called order-m version of the BoD-model. Basically, the order-m method no longer builds on the assumption that all observations should be considered in the computation of the efficiency scores. Instead the MJV under evaluation is B times evaluated relative to a subsample of m observations drawn from the original dataset. The final order-m efficiency score e_k^m is the arithmetic mean of the estimates resulting from each of the B evaluation rounds. Note that the order-m efficiency scores e_k^m can be above or below one. An e_k^m value higher (lower) than 1 indicates that the MJV under evaluation is doing better (worse) in terms of collection cost efficiency compared to the average MJV included in the subsample of m observations. To determine the level of m we estimate, as in for instance De Witte and Rogge (2011), the m-value for which the percentage of order-m efficiency scores above unity

[5] This model is largely based on the minimum efficiency concept introduced by Zhu (2004) and applied by, among others, Takamura and Tone (2003) and Wang et al. (2007) in the DEA-context.

Table 3
Overall recycling efficiency of the Belgian municipal joint ventures.

Municipal joint venture (MJV)	e_k^g	e_k^b	e_k^m
MJV 8	1 (1)	1.6868 (10)	1.1740 (2)
MJV 12	1 (1)	1.7563 (6)	1.0679 (1)
MJV 21	1 (1)	1.8625 (2)	1.2185 (4)
MJV 34	0.9909 (4)	1.7360 (7)	1.0499 (5)
MJV 15	0.9458 (5)	1.4838 (19)	0.9961 (7)
MJV 25	0.9303 (6)	1.8211 (4)	1.0844 (3)
MJV 29	0.9107 (7)	1.6993 (9)	0.9956 (8)
MJV 31	0.9040 (8)	1.9793 (1)	1.0217 (6)
MJV 24	0.8767 (9)	1.8228 (3)	0.9723 (9)
MJV 3	0.8305 (10)	1.5988 (12)	0.9632 (10)
MJV 33	0.8245 (11)	1.7082 (8)	0.9024 (13)
MJV 35	0.8114 (12)	1.7583 (5)	0.9136 (11)
MJV 28	0.7890 (13)	1.4939 (17)	0.8717 (14)
MJV 1	0.7745 (14)	1.5346 (15)	0.8430 (18)
MJV 13	0.7740 (15)	1.3800 (25)	0.9108 (12)
MJV 19	0.7589 (16)	1.3600 (26)	0.8159 (21)
MJV 23	0.7555 (17)	1.4269 (21)	0.8253 (19)
MJV 26	0.7513 (18)	1.6037 (11)	0.8440 (16)
MJV 7	0.7393 (19)	1.3080 (28)	0.7929 (24)
MJV 27	0.7371 (20)	1.4866 (18)	0.8433 (17)
MJV 14	0.7289 (21)	1.5805 (13)	0.8220 (20)
MJV 6	0.7258 (22)	1.4079 (23)	0.7848 (27)
MJV 9	0.7252 (23)	1.2692 (29)	0.8024 (22)
MJV 5	0.7238 (24)	1.5399 (14)	0.8014 (23)
MJV 20	0.7203 (25)	1.1464 (32)	0.8604 (15)
MJV 18	0.7154 (26)	1.3908 (24)	0.7835 (28)
MJV 17	0.7137 (27)	1.4431 (20)	0.7868 (26)
MJV 2	0.7121 (28)	1.3429 (27)	0.7693 (30)
MJV 32	0.7058 (29)	1.5021 (16)	0.7922 (25)
MJV 30	0.6973 (30)	1.2138 (31)	0.7792 (29)
MJV 16	0.6972 (31)	1.4137 (22)	0.7671 (31)
MJV 10	0.6636 (32)	1.2524 (30)	0.7222 (32)
MJV 22	0.5613 (33)	1.0065 (34)	0.6825 (33)
MJV 4	0.5557 (34)	1.0267 (33)	0.5961 (34)
MJV 11	0.4442 (35)	1 (35)	0.5044 (35)

decreases only marginally. The so called elbow effect (i.e. a sharp decline, followed by a stable slope) occurs at $m = 15$, indicating that including 15 observations in the subsample should result in both meaningful and robust estimates. The advantage of the order-m approach is that the obtained efficiency scores e_k^m are less affected by outlying observations, extreme values and even measurement error in the data. We will therefore include the order-m estimations as a robustness check in the results section.[6]

5. Results

The estimations of the overall collection cost efficiencies realized by the Belgian MJV's as computed by the optimistic, pessimistic and order-m version of the BoD-model are displayed in Table 3. The second and third column of Table 3 show the overall collection cost efficiencies and their associated rank values as evaluated by optimistic and pessimistic BoD, respectively. The final column shows the overall collection cost efficiencies and their associated rank values as computed by the order-m method. The 35 MJV's are listed based on the optimistic e_k^g-values. The optimistic approximations show that three MJV's ('MJV 8′, 'MJV 12′, and 'MJV 21′) are overall perfectly cost efficient in collection attaining the highest possible e_k^g-values of one. 'MJV 34′ is with an overall cost efficiency score of 0.9909 close to attaining perfect collection cost efficiency. However, the results also reveal that there are also MJV's which are overall

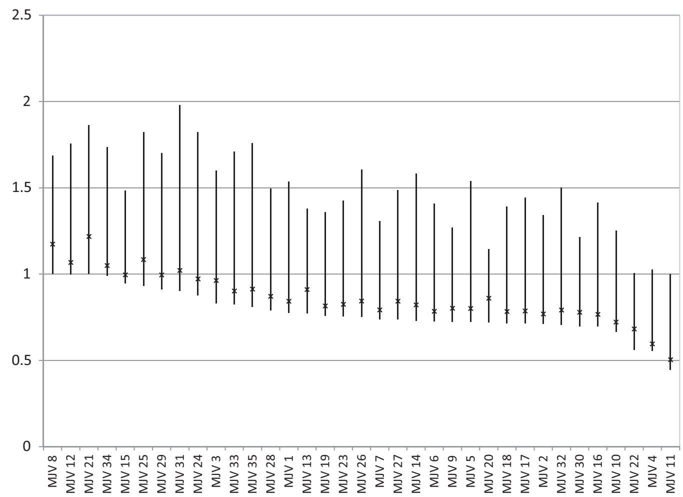

Fig. 3. Optimistic, pessimistic and order-m evaluations of the overall recycling efficiency (scores).

rather cost inefficient in collecting the three waste fractions, even when being evaluated under the most favorable evaluation conditions or when the order-m version of the BoD-model is used. 'MJV 11′ is the worst performer in the dataset. It obtains the lowest overall cost efficiency scores under the three evaluation conditions (i.e. $e_k^g = 0.4442$, $e_k^b = 1$ and $e_k^m = 0.5044$). This implies that 'MJV 11′ performs cost inefficiently in collecting all waste fractions.

In general, the outcomes of the optimistic, pessimistic and order-m BoD-based collection evaluations are largely similar. MJV's which are evaluated as being cost efficient by the optimistic version of the BoD-model are commonly evaluated as good performers by the pessimistic version of the BoD-model. The same reasoning applies to the cost inefficient MJV's (being evaluated with low e_k^g- and e_k^m-values and e_k^b-values of close to one). Stated differently, irrespective of being evaluated under the most favorable or the most unfavorable evaluations, cost efficient (inefficient) MJV's receive generally high (low) cost efficiency scores. The congruence of the results also appears from the high Spearman's rank correlation (with values between 0.79 and 0.96 in absolute values) computed between the 3 BoD-based cost efficiency evaluation outcomes (both the scores and ranks). Moreover, note that our finding of considerable inefficiencies for the average MJV is in line with previous findings of, for instance, Marques et al. (2012a) who found significant inefficiencies for Portuguese recycling companies. That is, as found in other studies, also for the Belgian MJV's it holds that there is considerable potential for cost efficiency improvements.

The BoD-based collection cost efficiency scores and ranks for the Belgian MJV's are visualized in Figs. 3 and 4. Fig. 3 shows the range of cost efficiency values as marked out by the optimistic and pessimistic BoD-based evaluations. As noted above, the exact overall cost efficiency score is believed to lie in this range. The ranges can be interpreted in two parts. The part below the value of one shows the measure of collection cost inefficiency (i.e. $1 - e_k^g$) as indicated by the optimistic version of BoD. Higher distances between the lower bound of the ranges and the value of one thus imply more cost inefficiency. The part of the figure above the value of one shows the degree of cost efficiency as estimated by pessimistic BoD. Higher distances between the upper bound of the ranges and the value of one indicate more cost efficiency. So, based on the above, ideally, the lower bound of the range should be near the value of one and the upper bound of the range should be as high as possible. 'MJV 8′, 'MJV 12′, 'MJV 21′ and 'MJV34′ satisfy this pattern. The opposite pattern is found for the MJV's at the lower end of the ranking ('MJV 22′, 'MJV 4′ and 'MJV 11′): respectively high distances and distances near to

[6] One could also opt to estimate order-m versions of the optimistic and pessimistic BoD model. This would yield confidence intervals to the optimistic and pessimistic overall cost efficiency estimates and, hence, on the range of possible overall cost efficiency estimates. The results of the robust order-m estimations of the optimistic and pessimistic BoD-approach are available from the authors upon request.

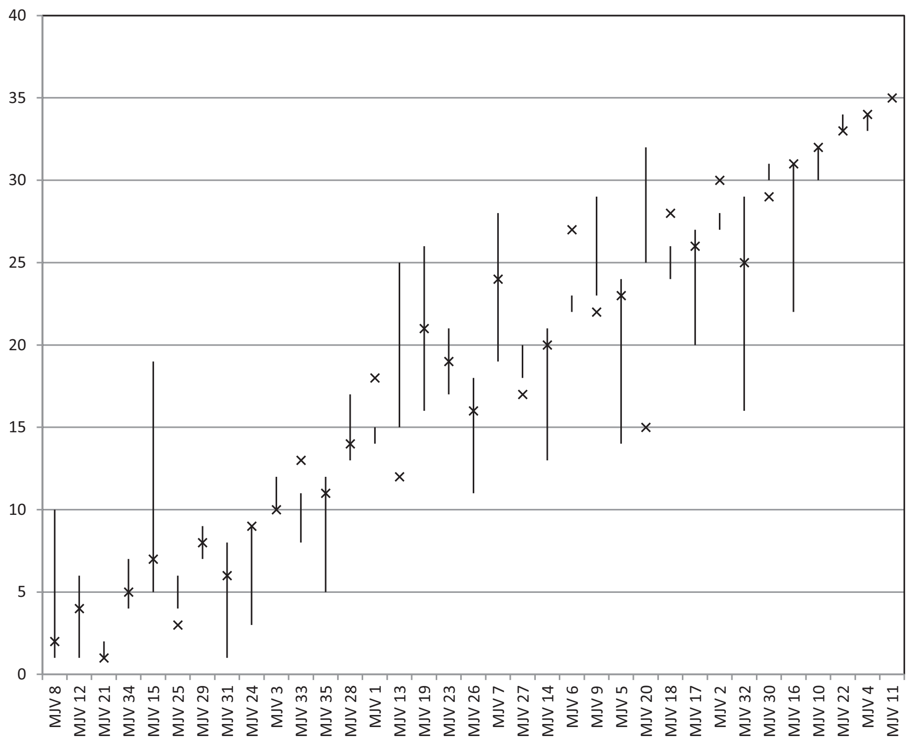

Fig. 4. Optimistic, and pessimistic and order-*m* evaluations of the overall recycling efficiency (ranks).

zero between the lower bounds and upper bounds of the ranges and the value of one. The MJV's in the middle of the ranking show moderate distance values between the lower and upper bounds of their cost efficiency ranges and the value of one. The order-*m* cost efficiency scores for each MJV are visualized in the same figure by an '*x*'. Although the exact value of the order order-*m* efficiency scores is difficult to compare with the value of the optimistic and pessimistic scores (MJV's are evaluated relative to a smaller reference group when the order-*m* method is used), both methods reveal a similar picture. In addition all order-*m* efficiency scores are situated within upper and lower bounds as marked out by the optimistic and pessimistic BoD-based evaluations. The order-*m* efficiency scores seem to confirm the prior finding that 'MJV 8′, 'MJV 12′, 'MJV 21′ and 'MJV34′ are among the best performers. However the order-*m* method adds 'MJV 25′ and 'MJV 31′ as top performers. Finally the 3 MJV's which are situated at the lower end of the ranking according to the range determined by optimistic and pessimistic BoD-based evaluations (i.e. 'MJV 22′, 'MJV 4′ and 'MJV 11′), also have the lowest efficiency score when the order-*m* method is used.

Fig. 4 shows the range of cost efficiency rank values as marked out by the optimistic and pessimistic BoD-based evaluations. The MJV's are listed based upon their collection cost efficiency estimates by optimistic BoD, with the most cost efficient being displayed at the left of the plot and the least cost efficient at the right part of the plot. Bigger ranges imply more difference between the ranking as computed by optimistic and pessimistic BoD (so more uncertainty on the true collection cost efficiency ranking). For instance, 'MJV 8′ is ranked first with a small range. This denotes that 'MJV 8′ is evaluated as among the top performers irrespective of the evaluation conditions (hence, with a high certainty). On the other hand, the big range for 'MJV 15′ shows that 'MJV 15′ is evaluated among the top performers under the most favorable evaluation conditions, yet, among the middle performers under the least favorable evaluation conditions. Hence, there is more uncertainty on the true collection cost efficiency ranking of 'MJV 15′. Fig. 4 also reveals that for 24 of the 35 MJV's, the order-*m* ranks (indicated by the *x*'s on the figure), lie within the range of the ranks obtained via optimistic and pessimistic BoD-based evaluations.

Finally it is important to understand the possible causes for the differences in efficiency scores between the MJV's. An important influential factor hereby could be linked to the contracting out decisions. Although this issue already received an extensive level of attention by scholars, debate is still ongoing. Based on a comprehensive review of the literature Bel and Warner (2008) conclude that the majority of studies find no difference between public and private provision of waste related services. More recently Simões and Marques (2012) found higher efficiencies for privately managed recycling companies in Portugal, while Bel and Fageda (2010) report no major cost differences between private and public service delivery in a sample of Galician municipalities in Spain. Although endogeneity issues might play a role, our results seem to be more in line with the findings of Simões and Marques (2012). When comparing the average efficiency scores of MJV's which contract out most of their municipal packaging waste collection to MJV's which mainly organize their own collection, results reveal that both e_k^g and e_k^b-values are higher for MJV's outsourcing their collection activities. However an independent sample *t*-test reveals that this difference is only statistically significant for the pessimistic e_k^b-values (2-tailed significance = 0.035).

A second factor that may influence the relation between the costs and the collection results is the population size and/or density or, closely related, the number or concentration of pick-up points. The impact of population related variables on the collection costs has also been recognized in the international literature. Although generally a positive relation between the population size and the cost efficiency scores is identified (see for instance Lawarree (1986) or García-Sánchez (2008)), the impact of the population density or density of the pick-up points is less clear. In particular the trade-off between the distance to be covered by the collection trucks and potential traffic congestion costs complicates an a priory assessment of the relation between density and cost efficiency (see Rogge and De Jaeger (2013) for an overview of the literature). Our results reveal no significant correlation between population density and the optimistic and pessimistic BoD-based efficiency scores. However for the total population we find, somewhat unexpectedly, a negative correlation with the pessimistic e_k^b-values which is significant the 10% level (Pearson correlation = −0.290 with a 2-talied significance of 0.092). This indicates that MJV's serving a bigger population are generally evaluated as less cost efficient under the least favorable evaluation conditions.

6. Concluding remarks

Engaging in inter-municipal cooperation to exploit benefits of scale when providing waste related services, demonstrates to a certain extent that cost efficiency is an important decision variable for municipalities. However local authorities – and hence their MJV's – typically focus on more than just cost efficiency. The level of service provided to the residents, recycling efforts by households (and environmental concerns in general) or even employment of the residents in the waste sector might play a role when organizing local waste management. Not surprisingly, these policy choices will, at least to some degree, be reflected in the cost efficiency. Our results indeed reveal important variations in both costs per metric ton collected as overall cost efficiency of collection activities. Even when both the optimistic version and the pessimist variant of the BoD-model are used to determine the boundaries of the overall cost efficiency scores, important differences between the best and worst performing MJV's persist. Nevertheless, these results should be treated carefully, as several variables might influence the cost structure of the MJV's. Specifications in the funding scheme for separate collection of household packaging waste might partially explain the higher costs for public provision. In particular the fact that the reference contributions, which are based on the market prices, can be exceeded for MJV's that collect and sort packaging waste themselves (if they can present a proof of the costs incurred), hardly promotes cost efficiency in the public provision of the service. Next to contracting out decisions we only tested for the impact of the total population and the population density (due to data limitations). Nevertheless other variables such as the average distance to the processors, the level of competition on the local market for waste collection services or the household size could play a role. Therefore our findings should serve as a starting point for further research. Note that an analysis based on detailed data would benefit from a more elaborate approach than basic methods such as ANOVA analysis, Tobit regressions or the procedures used in this paper (i.e. *t*-tests such and Pearson correlations). The use of these methods can be criticized for the fact that two distinct steps are required (i.e. a DEA or BoD analysis and a Tobit regression, ANOVA or *t*-test) which implicitly assumes separability. In particular it assumes there is no direct link between set of attainable outputs and the background or management variables (for more details see for instance De Witte and Rogge (2010) and De Witte and Rogge (2011)). Alternative approaches, such as the conditional DEA-framework as proposed by, among others, Daraio and Simar (2007) could provide a solution (see Rogge and De Jaeger (2013), for an application of this method to waste management in Flanders).

References

Abrate G, Erbetta F, Fraquelli G, Vannoni D. The costs of disposal and recycling: an application to Italian municipal solid waste services. HERMES: Working Paper 2011:2.

Allers MA, Hoeben C. Effects of unit-based garbage pricing: a differences-in-differences approach. Environmental and Resource Economics 2010;45:405–28.

Bel G, Fageda X. Empirical analysis of solid management waste costs: some evidence from Galicia, Spain. Resources, Conservation and Recycling 2010;54:187–93.

Bel G, Warner M. Does privatization of solid waste and water services reduce costs? A review of empirical studies. Resources, Conservation and Recycling 2008;50:1337–48.

Bohm A, Folz DH, Kinnaman TC, Podolsky MJ. The costs of municipal waste and recycling programs. Resources, Conservation and Recycling 2010;54: 864–71.

Cazals C, Florens JP, Simar L. Nonparametric frontier estimation: a robust approach. Journal of Econometrics 2002;106(1):1–25.

Charnes A, Cooper WW, Rhodes E. Measuring the efficiency of decision making units. European Journal of Operational Research 1978;2:429–44.

Chen H-W, Chang N-B, Chen J-C, Tsai S-J. Environmental performance evaluation of large-scale municipal solid waste incinerators using data envelopment analysis. Waste Management 2010;30:1371–81.

Chen Y, Chen C. The privatization effect of MSW incineration services by using data envelopment analysis. Waste Management 2012;32:595–602.

Cherchye L, Moesen W, Rogge N, Van Puyenbroeck T. An introduction to 'benefit of the doubt' composite indicators. Social Indicators Research 2007;82: 111–45.

Cruz NF, Simões P, Marques R. Economic cost recovery in the recycling of packaging waste: the case of Portugal. Journal of Cleaner Production 2012;37:8–18.

Daraio C, Simar L. Advanced robust and nonparametric methods in efficiency analysis: methodology and applications. In: Studies in Productivity and Efficiency. Springer; 2007.

De Jaeger S, Eyckmans J, Rogge N, Van Puyenbroeck T. Wasteful waste-reducing policies? The impact of waste reduction policy instruments on collection and processing costs of municipal solid waste. Waste Management 2011;31:1429–40.

De Witte K, Rogge N. To publish or not to publish? On the aggregation and drivers of research performance. Scientometrics 2010;85:657–80.

De Witte K, Rogge N. Accounting for exogenous influences in performance evaluations of teachers. Economics of Education Review 2011;30:641–53.

Eurostat. Municipal waste generation and treatment, by type of treatment method (tsdpc240); 2012, http://epp.eurostat.ec.europa.eu/ [accessed on 01.12.12].

Farrell MJ. The measurement of productive efficiency. Journal of the Royal Statistical Society 1957:253–90 [Series A CXX Part 3].

Fost Plus. Jaarverslag 2010 – Recyclage, een succes dankzij de betrokkenheid van iedereen. Brussels: Fost Plus; 2011.

Fost Plus. Intercommunales in België; 2012, http://www.fostplus.be/Partners/Intermunicipal_authorities [accessed on 24.11.12].

García-Sánchez IM. The performance of Spanish solid waste collection. Waste Management Research 2008;26:327–36.

Gellynck X, Verhelst P. Assessing instruments for mixed household solid waste collection services in the Flemish region of Belgium. Resources, Conservation and Recycling 2007;49:372–87.

Hage O, Söderholm P. An econometric analysis of regional differences in household waste collection: the case of plastic packaging waste in Sweden. Waste Management 2008;28:1720–31.

IPC. Besluit van de Interregionale Verpakkingscommissie van 18 december 2008 tot erkenning van de vereniging zonder winstoogmerk Fost Plus, Martinus V Straat 40, 1200 Brussel als organisme voor verpakkingsafval. Brussels: Interregionale Verpakkingscommissie; 2008.

IPC. Activiteitenverslag 2011. Brussels: Interregionale Verpakkingscommissie; 2012.

Jenkins RR, Martinez SA, Palmer K, Podolsky MJ. The determinants of household recycling: a material-specific analysis of recycling program features and unit pricing. Journal of Environmental Economics and Management 2003;45: 294–318.

Kinnaman TC. Examining the Justification for Residential Recycling. Journal of Economic Perspectives 2006;20:219–32.

Kipperberg G. A comparison of household recycling behaviors in Norway and the United States. Environmental and Resource Economics 2007;36:215–35.

Larsen AW, Merrild H, Møller J, Christensen TH. Waste collection systems for recyclables: an environmental and economic assessment for the municipality of Aarhus (Denmark). Waste Management 2010;30:744–54.

Lavee D. Is municipal solid waste recycling economically efficient? Environmental Management 2007;40:926–43.

Lawarree J. Une comparaison empirique des performancesdes secteurs privé et public: le cas de la collecte d'immondices en Belgique. Cahiers Economiques de Bruxelles 1986;109:3–31.

Marques RC, Cruz NF, Carvalho P. Assessing and exploring (in)efficiency in Portuguese recycling systems using non-parametric methods. Resources, Conservation and Recycling 2012a;67:34–43.

Marques RC, Ferreira SF, Cruz NF, Pereira MC, Simões P. Comparing the recycling systems of Portugal, France, Germany, Romania and the UK, research report for EIMPack – Economic Impact of the Packaging and Packaging Waste Directive. Lisbon: EIMPack; 2012b.

Marques RC, Ferreira SF, Cruz NF, Pereira MC, Simões P, De Jaeger S. Framework and evolution of the packaging sector in Belgium – Report 0. Research report for EIMPack – Economic Impact of the Packaging and Packaging Waste Directive. Lisbon: EIMPack; 2012c.

Marques RC, Simões P. Incentive regulation and performance measurement of the Portuguese solid waste management services. Waste Management & Research 2009;27:188–96.

Melyn W, Moesen W. Towards a synthetic indicator of macroeconomic performance: unequal weighting when limited information is available. Public Economics Research Paper 1991:17.

Rogge N. Undesirable specialization in the construction of composite policy indicators: the environmental performance index. Ecological Indicators 2012;23:143–54.

Rogge N, De Jaeger S. Evaluating the efficiency of municipalities in collecting and processing municipal solid waste: a shared input DEA-model. Waste Management 2012;32:1968–78.
Rogge N, De Jaeger S. Measuring and explaining the cost efficiency of municipal solid waste collection and processing services. Omega 2013;41:653–64.
Simões P, De Witte K, Marques RC. Regulatory structures and operational environment in the Portuguese waste sector. Waste Management 2010;30: 1130–7.
Simões P, Marques RC. Influence of regulation on the productivity of waste utilities. What can we learn with the Portuguese experience? Waste Management 2012;32:1266–75.
Takamura Y, Tone KA. comparative site evaluation study for relocating Japanese government agencies out of Tokyo. Socio-Economic Planning Sciences 2003;37:85–102.
Wang YM, Chin KS, Yang JB. Measuring the performances of decision-making units using geometric average efficiency. The Journal of the Operational Research Society 2007;58:929–37.
Zhou P, Ang BW, Poh KL. A mathematical programming approach to constructing composite indicators. Ecological Economics 2007;62:291–7.
Zhu JA. buyer-seller game model for selection and negotiation of purchasing bids: extensions and new models. European Journal of Operational Research 2004;154:150–6.