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The impact of prices and pricing units on residual and organic waste: Evidence from Wallonia, Belgium

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ABSTRACT

Waste pricing is important to encourage households to reduce their waste and to increase sorting and recycling. Pricing instruments include differentiated prices for the waste fractions collected and the choice of an appropriate pricing unit (volume, weight, levy). In this paper, we test the effectiveness of those different instruments using an original and complete dataset from Wallonia (the southern region of Belgium), covering 10 years and all the 262 municipalities of the region. Our approach is to estimate the sensitivity of households' production of residual and organic waste to prices i.e., the price elasticities, using sophisticated econometric techniques to control for the endogeneity of prices. For residual waste, we show a significant own-price elasticity, which is higher when organic waste is not collected at the curbside. For organic waste, we found an important and significant cross-price elasticity, but a limited own-price elasticity. Hence, the privileged instrument to encourage waste reduction and sorting should be the price of residual waste. Finally, we show that the weight-based pricing system contributes substantially to the reduction of residual waste.

1. Introduction

Reducing waste is a priority for local politicians, regulators, and a concern for environmental activists. The flat tax system that was commonly used to finance the waste services has been replaced by more sophisticated pricing systems, which include non-linear prices, positive and differentiated marginal prices for the different waste fractions and different pricing units to limit the production of residual waste and to encourage reuse, sorting and recycling (Bilitewski, 2008; Morlok et al., 2017). The objective of this paper is to evaluate the effectiveness of different price instruments and, more specifically, the impact of prices and pricing units on the production of residual and organic waste by the household.

To that end, we use data from Wallonia, the southern region of Belgium. An interesting feature of the waste sector in Wallonia is that the general policy framework is designed at the regional level, but the details of implementation are decided at the municipality level and there is a lot of heterogeneity in the practices that the municipalities adopt. Municipalities decide whether to collect organic waste at the curbside or not; they fix the unit price for both the residual and the organic fractions, and they choose the pricing unit (volume or weight) that will be used to calculate the household's bill.

The objective of this paper is to provide new empirical evidence for the effects of the price instruments on the quantities of residential residual and organic waste produced by the households.[1] Our methodological approach is to estimate the price elasticities for residual and organic waste. Elasticities measure the sensitivity of waste production to prices. The direct-price elasticity measures the impact of the price of a given stream (organic or residual) on the volume of that stream; the cross-price elasticity measures the impact on the volume of the other stream. The own-price elasticity is expected to be negative, while the cross-price elasticity is expected to be positive. Price elasticities are useful for waste managers as they provide quantitative estimations of the impact of the pricing policy on the waste production, i.e., they can be used to quantify the impact of a price change on waste production.

We estimate elasticities using a unique aggregate dataset for all the 262 Walloon municipalities covering the period from 2009 to 2018. Our econometric estimations include several control variables at the municipality level that could potentially affect waste production, like income or demographic variables. Our panel-data approach enables us to control for municipalities' unobserved heterogeneity and time-specific effects. In our case, the endogeneity of prices is a concern because the

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[1] In Gautier and Salem (2021), we provide preliminary evidence based on a reduced sample.

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municipalities use the quantities of the current year to fix the prices for the following year. This is known in econometrics as the feedback effect (Wooldridge, 2010), i.e., current prices affect future quantities. This problem is not always correctly addressed in the literature, and consequently, price elasticities are underestimated (Kinnaman and Fullerton, 2000). Appropriate panel data instrumental variable methods should be used to obtain consistent estimators. To control for price endogeneity, we use a two-way fixed effects panel two-step GMM (General Method of Moments). With those sophisticated estimation methods and our large dataset covering 10 years, we can provide robust estimates for the different price elasticities.

2. The importance of pricing in waste management

Over the past few years, many municipalities in many countries have implemented a pay-as-you-throw (PAYT) payment scheme for waste disposal (see Ribas Alzamora and R. Tobias de V. Barros, 2020 for a review of different country experiences). With PAYT, citizens are incentivized to reduce their waste production as producing more waste becomes more costly. However, the implementation of an effective PAYT system is a complex issue (Bilitewski, 2008). What is particularly difficult is the choice of the appropriate unit price for the different waste streams as well as the choice of the pricing unit. Therefore, it is of prime importance to understand how citizens react to pricing. In this section, we review the main contributions from the literature on the subject and we position our paper with respect to that literature.

2.1. Pricing level

There is a vast literature on the price elasticity of residential waste, with the common objective of estimating how agents react to prices. The question has been addressed using different types of data, aggregate municipal data or household level data, panel or cross-sectional (see Bel and Gradus (2016) for a meta-analysis). The price elasticities reported in the literature are quite heterogeneous. This heterogeneity could be attributed to differences in the period, region, or the econometric approach used for the estimation. One potential problem in the previous studies is the composition of waste used as a dependent variable in the estimations. Some studies aggregate both organic and non-organic waste (Usui and Takeuchi, 2014), others exclude the organic part (Allers and Hoeben, 2010; Dijgraaf and Gradus, 2004; Linderhof et al., 2001), and others use total residential waste (Van Houtven and Morris, 1999; Huang et al., 2011). The use of different waste streams makes the comparisons complicated. In our study, we can provide different estimations for the elasticities depending on whether the municipality collects organic waste at the curb or not. It is known that a separate collection of the organics reduces the residual waste and encourages recycling (Andersson and Stage, 2018; Best and Kneip, 2019). We additionally show that the own-price elasticity for residual waste is lower when organics are collected. In other words, the collection of organics reduces the residual waste but also the price sensitivity, i.e., the effectiveness of the price instrument.

The literature on the price elasticity of organic waste is less developed. Using Dutch data, Linderhof et al. (2001), Dijgraaf and Gradus (2004), and Allers and Hoeben (2010) all found a negative own-price elasticity that differs in magnitude when pricing units are considered. Dijgraaf and Gradus (2004) computed a positive cross-price elasticity, whereas Allers and Hoeben (2010) found a non-significant one. Park (2018) estimated that an increase in the specific waste bag price leads to more recycling, especially in regions with more pro-environmental citizens. For the Walloon municipalities that collect organic waste, we found a positive and significant cross-price elasticity and a negative but non-significant own-price elasticity. This means that a higher price for the residual waste reduces the production of residual waste and increases the production of organic waste, i.e., it encourages the sorting of the organic fraction. A lower price for the organic fraction is found to be less effective, as our estimations show a non-significant impact on the quantities of organic waste.

2.2. Pricing units

The choice of an appropriate pricing unit is of equal importance. The weight-based pricing system is more effective compared to the bag (volume-based pricing). A bag is often prone to the problem of stuffing. When a bag with a certain volume is bought, the household discards as much waste as the bag can handle. In the literature, this phenomenon is known as the "Seattle Stomp". Unlike the bag, the weight-based system is more accurate to quantify the actual amount of waste, hence it is a more sensitive pricing system that better translates the polluter-payer (pay-as-you-throw or PAYT) principle. Kinnaman and Fullerton (1996) found that pricing waste by the bag (or can) had little effect on the weight, even though there was a significant reduction in the volume. Dijkgraaf and Gradus (2004), De jaeger and Eyckmans (2015) found a significant effect for the introduction of weight-based pricing on the quantities of residual waste. Sasao et al. (2021) show that the continuous participation of weight-based pricing significantly reduces waste by almost 10.4 %. Park and Lah (2015) found that volume-based waste fees had a positive influence on recycling rates. In Wallonia, a high proportion of the municipalities (almost 42 %) introduced weight-based pricing. This allows us to give new insights about the adoption of this pricing system. We estimate the effect of weight-based pricing on both the residual and the organic fractions, and we show that weight-based pricing significantly contributes to the reduction of residual waste.

Finally, if price instruments are important to reduce waste and improve recycling, the literature has also pointed out the importance of non-price instruments, including behavioural and socio-economic characteristics like education (Reschovsky and Stone, 1994; Kinnaman and Fullerton, 1997; Okonta and Mohlalifi, 2020), age (Williams and Kelly, 2003), homeownership (Kinnaman, 1994) and information campaigns (Callan and Thomas, 1997; Mickaël, 2014).

3. The organization of the waste sector in Wallonia

Belgium is a federal state composed of three regions: Brussels, Flanders, and Wallonia. Wallonia is the French-speaking southern region. It is subdivided into 262 municipalities with a total population of 3.6 million inhabitants. In line with the European Waste Framework Directive (2008/98/EC), each region is responsible for regulating the waste sector within its jurisdiction and issues its specific set of rules. Wallonia adopted its first legislative package in 2008.[2] In addition to the legislative package, the regional administration issued two master plans, in 2010 and 2018,[3] that provide orientations for the waste policy at the regional level. Though the region issues the general guiding rules, it is up to the municipalities to determine the details of implementation.

3.1. The collection of residential waste

Municipalities have the obligation to offer services for the disposal of waste to all their residents. In particular, they have to guarantee the services of collection for 16 waste streams either at the curbside or at drop-off facilities as specified in the legislation. Curbside collection is

[2] Arrêté du Gouvernement wallon relatif à la gestion des déchets issus de l'activité usuelle des ménages et à la couverture des coûts y afférents (M.B. 17.04.2008). Available on the administration's website https://environnement.wallonie.be/legis/dechets/degen028.htm.

[3] Plan Wallon des déchets - horizon 2010 (PWD). Available on the administration's website https://environnement.wallonie.be/rapports/owd/pwd/index.htmPlan Wallon Déchets Ressources (PWD-R). Available on the administration's website https://environnement.wallonie.be/rapports/owd/pwd/PWDR_3.pdf.

compulsory for residual waste and most of the municipalities provide additional curbside collection for plastic and packaging (hereafter P&P), paper and cardboard (hereafter P&C), and increasingly, for organic waste. For the fractions of waste that are not collected at the curbside, except for the organic part, municipalities have to provide drop-off facilities (bottle banks and recycle parks) freely accessible to households.

3.1.1. Residual waste

Residual waste is the fraction of waste that cannot be reused or recycled. The collection of this fraction is carried out in all the municipalities, at least once a week, as stipulated in the legislation. Municipalities decide to collect residual waste either in a specific plastic bag or in a container. They fix the price of the bags, and they are sold in agreed sales' points like supermarkets. In case containers are used, they are rented to the households. When organic waste is not collected separately at the curb, households have to dispose of this fraction with the residual one (alternatively, it can be home composted). If the collection of organic waste is provided, households can sort and dispose separately of the organic and the residual parts.

The composition of the residual waste stream is therefore different in municipalities when a specific collection of organics is provided. We will use the terminology "residual unsorted waste" (hereafter RUW) to designate the residual fraction when organics are not collected at the curb and the terminology "residual solid waste" (hereafter RSW) to designate the residual fraction when the organics are collected apart.

These two fractions (RUW and RSW) cannot be reused or recovered and are sent for disposal to be either incinerated or land-filled. Incineration and land-filling are the least preferred waste treatment options in the European waste management hierarchy,[4] hence the importance to find policies and tools to reduce those fractions.

It is important to distinguish between the two streams (RUW and RSW) in the analysis for three main reasons. First, the definition of residual waste differs, and pooling all municipalities together will make interpretations problematic. Second, it would enable us to evaluate the price effect on the residual solid part which, in all cases, is always sent for disposal. Finally, we would expect that the price effect differs when a separate collection of organic waste at the curb is available.

3.1.2. Organic waste

Organic waste (or biodegradable waste) is the fraction of waste that is fermentable. It is constituted mainly of putrescible kitchen waste (meal leftovers, peelings, perished food, unconsumed bread), small garden waste (such as small herbs or flowers) and may also include all kinds of other biodegradable waste (like tissues, paper towels, kitchen towels, …). The organic waste collected is treated by anaerobic digestion. This process transforms them into biogas, which is a green energy source used to produce gas, electricity, and heat. Additionally, the produced digestat can be used as an organic fertilizer in agriculture. It can also be home composted, and the resulting compost is used as fertilizer in home gardens. According to the EU waste hierarchy, organic waste can thus be classified as reusable and recyclable waste.

In Wallonia, the collection of organic waste at the curb is not yet mandatory. In 1997, a leading municipality started collecting the organic fraction at the curb. The number of municipalities providing the service has increased steadily since then to reach 163 municipalities in 2018. The updated legislative package of 2018 makes the separation of organic waste an obligation by 2025. The separate disposal of organic waste is not compulsory, and households have the option to dispose of their organic waste with the residual fraction. The collection of organic is not free and households have to pay a per-unit price similarly to the residual part. Bernad-Beltrán et al. (2014) show that this can lower participation.

The quantities of the fractions of residual and organic residential waste are presented in Table 1. We observe a decrease in the residual fraction and an increase over time in the organic part, though the capture rate of organics is low. On average, when the service is provided, municipalities collect thirty-four kilos of organic waste per inhabitant while the food waste is estimated to be equal to 105 kilos per inhabitant (Favoino and Giavini, 2020). The region wishes to increase the quantities collected of this fraction of waste. The 2018 plan sets an objective of forty-three kilos per inhabitant by 2025, an increase of almost 32 %.

3.2. The cost of residential waste

Since the reform of 2008, municipalities can no longer use their general budget to pay the costs of the waste service. As stipulated by the legislation, the *polluter-payer* and the *true-cost* should be the guiding principles for determining the prices that households pay for their waste. Both criteria guarantee that citizens will pay the actual economic cost of the waste they generate. Accordingly, municipalities have to pass the full cost of residential waste to their residents. The revenues should cover the entire costs[5] associated with the services of collection, reuse, recycling, and disposal of waste.

The price is fixed according to the following process: at the end of the year *t*, each municipality should do projections for its costs and revenues for the next year, and based on these forecasts, the revenues should be set to entirely cover the costs. The cost of the service may differ from one municipality to the other depending on the service quality (additional doorstep collection), the topography of the municipality, and the organizational choice made by the municipality.[6]

3.2.1. Price structure and pricing units

The municipalities should provide incentives for households to improve sorting and to reduce their waste production. To that end, municipalities use a two-part tariff: the household's bill consists of a flat tax and a variable fee. The flat tax is paid annually, and it depends on the household size. It covers the costs of collection and treatment of a predefined quantity of waste, as well as the costs of managing the drop-off facilities.

Municipalities use different pricing units for the variable fee: the volume, the weight, the frequency of collection, or a combination of them. There are five unit-based pricing (UBP) schemes in Wallonia. When the bag (*Bag*) is used, households have to buy the (official) bag with a specific volume, they usually come in two sizes either 30 or 60 L. In rare cases (*Bag&tag*), households can use their own bag (with a predefined volume) but must attach a costly tag to it. In both cases, the pricing unit is the volume (litre). When containers have no weighing chip ($Container_{volume}$), households pay a price per levy each time the container is presented at the collection point. Given that containers have a given volume, 40 or 140 L as chosen by the household, a price per levy is equivalent to a pricing based on the volume of waste. In case containers are equipped with a weighing chip ($Container_{weight}$), the pricing unit is the price paid per kilo of waste. The latter can also be combined with a price per levy ($Container_{weight\&volume}$), thus the pricing unit

Table 1
Quantities of waste.

Kilo per inhabitant	2009	2018	Percentage change
Residual unsorted waste (RUW)	162.09	155.67	−3.96 %
Residual solid waste (RSW)	112.92	98.381	−12.88 %
Organic waste	30.48	33.92	11.28 %

[4] The European waste hierarchy refers to the five steps in its Waste Framework Directive: prevention, reuse, recycling, recovery, and disposal.

[5] Net of the subsidies provided by the region for the treatment and the recycling facilities.

[6] See Gautier and Reginster (2013) for empirical evidence on Wallonia.

becomes a combination of both the weight and the volume. Table 2 shows the frequencies of each UBP for the period from 2009 to 2018.

Progressively, more municipalities are abandoning the bag in favour of the container with a price per kilo and per levy. The statistics provided in Table 3 indicate that this shift contributes to the reduction of residual waste, as the average quantities of both RUW and RSW are lower for $Container_{weight\&volume}$ than for *Bag*.

Similarly, organic waste is collected either in bags or in containers, but a municipality does not necessarily use the same system for the two streams. Many municipalities use a "duobac", a container with two compartments, one for the residual solid fraction and the other for organic waste. In such a case, there is a single weighting chip, and the marginal price is the same for both fractions. Despite that, households have incentives to sort their waste, otherwise, they will fill one compartment only and will have to present their container at the collection point more often, which is costly when there is a price per levy ($Container_{weight\&volume}$).

As explained above, municipalities adopt different pricing units. The regional Waste Management Infrastructure Department evaluates one kilo of waste to be equivalent to a volume of 6.5 L. Because we need to have the same unit to compute the price elasticity, we will use this rate to transform a price per litre to a price per kilo[7] for all municipalities that use a volume-based pricing system. Two remarks need to be made. First, when different sizes of the bags or containers are available, we use the price of the largest, which is 60 L for bags and 140 L for containers. For example, a bag of size 60 L that costs 1€ will be equivalent to 0.108€ per kilo after the conversion. A container of 140 L with a price of 0.07€ per kilo and 0.65€ per levy, its final price will be 0.10€ per kilo after converting the price per levy to its equivalent of 0.03€ per kilo. Second, most of the municipalities that use weight-based pricing have unit-prices that increase in blocks based on the quantities of waste generated. Because the bounds differ from one municipality to the other, we use the price of the first block in our computations.

Table 4 shows the average marginal price for the years 2009 and 2018 as well as the percentage change during the period for each system after they have been converted to a single unit. Prices are corrected for inflation and expressed in 2009 euros. We can observe that there are large differences between the different UBP systems, the weight-based systems being costlier. We can also see that there are large variations during the period, both overall and within each UBP system, making it possible to use panel data methods in our empirical analysis.

4. Data and methodology

4.1. Data sources and description

We use ten years' data for the period from 2009 to 2018 for all 262 Walloon municipalities. Data on waste quantities and prices are publicly available.[8] It was provided directly by the administration for the period from 2009 to 2015 and collected from the administration's website for the period from 2016 to 2018. Quantities of waste are expressed in kilo per inhabitant. Marginal prices are expressed in euro per kilo and are corrected for inflation.

We construct dummy variables for the different UBP systems defined previously for both the residual and the organic fractions.

In line with the current literature, we collect demographic and socioeconomic variables that can have an impact on the quantities of waste. The data was collected from the website of the Walloon statistical office (Walstat[9]). We include the density (inhabitants per km^2) because households living in small areas will be less likely to participate in sorting organic waste. We also add its quadratic term to test for illegal dumping within the municipality. As with both a very low (rural areas with remote spots for dumping) and a very high density (urban areas with commercial dumpsters) the garbage left can hardly be attributed to a specific household (Kinnaman and Fullerton, 2000). The opportunity for illegal dumping will have an impact on the quantities of residential waste. If households have a higher propensity to illegally dump, they might react to an increase in unit-pricing by illegally dumping their waste (Seguino et al., 1995) and in such cases, the reported quantities will decrease. This means that the illegal dumping variable (the proxy variable) and the price variable are positively correlated. Failure to control for illegal dumping leads to an upward bias of the parameter on the price variable, and the price elasticity will be over-estimated.

We control for the average income (in euro). On one hand, a higher income increases consumption and therefore waste production. On the other hand, highly educated people have a higher income, and they tend to have higher environmental awareness which increases recycling activities. We include the proportions of inhabitants under 20 years of age and those above 60 as proxies for consumption patterns, as waste production may not be the same for all age categories. Finally, household size is incorporated to account for the presence of (dis)economies of scale. Descriptive statistics for the main variables are provided in Table 5. All monetary variables are adjusted for inflation (base year is 2009) using the Belgian consumer price index.

4.2. Econometric approach

4.2.1. Specification of the econometric model

Our objective is to estimate the price elasticity for different waste fractions, i.e., how quantities are affected by prices, controlling for the other variables that could affect quantities. We use aggregate data at the municipality level for the following fractions of waste: (1) residual unsorted waste (RUW), (2) residual solid waste (RSW), and (3) organic waste (ORG). The econometric specification is a panel double log model with fixed effects at the municipality level that takes the following form:

$$\ln q_{it} = \gamma_f \ln p_{r_{it}} + \vartheta_f \ln p_{org_{it}} + \varphi_f UBP_{it} + X\hat{A}_{it}\beta + \alpha_i + \lambda_t + \varepsilon_{it} \quad (1)$$

The dependent variable q_{it} is the quantity of waste in municipality i at year t; p_{rit} is the price of one kilo of residential residual (unsorted/solid) waste; p_{orgit} is the price of one kilo of organic waste. Quantities and prices are in logarithms, and the price coefficients (γ_f and θ_f) can be directly interpreted as price elasticities. We include a set UBP_{it} of dummies for the UBP systems.[10] The vector X_{it} is the vector of demographic variables. We include a municipality fixed effect (α_i) to control for unobserved time-fixed factors at the municipality level that have an impact on the quantities of waste, and a time fixed effect (λ_t) to capture all the commons shocks that affect municipalities in a given year. Finally, ε_{it} is the idiosyncratic disturbance term that is assumed to be independently and identically distributed over i and serially correlated over t.

Because, municipalities do not necessarily introduce the collection of organic waste in the beginning year and since we have no detailed data for the month of introduction, we control for this by adding a dummy variable $change_{org}$ in the regressions of both residual solid waste and organic waste. The variable takes the value 1 in the year curbside collection of organic waste was introduced and zero otherwise.

[7] Previous studies used other conversion rates: Allers and Hoeben (2010) used 1 kilo = 7.5 L for the residual solid waste and 3.8 L for the organic waste; Dijgraaf and Gradus (2004) used 1 kilo = 5 L for both streams. We performed robustness checks using other conversion rates (1 kilo= 5 or 7.5 L) and we found similar results.

[8] The data for the entire period 2009–2018 are publicly available on https://environnement.wallonie.be/.

[9] https://walstat.iweps.be/walstat-accueil.php#.

[10] We drop the two systems *bag&tag* and $container_{weight}$ due to the small number of observations and the low variability.

Table 2
Frequencies of UBP systems.

Year	Bag	Bag &tag	Container weight & volume	Container weight	Container volume	no UBP	Total
2009	176	8	56	8	8	6	262
2010	165	4	73	6	13	1	262
2011	158	4	81	7	11	1	262
2012	156	4	83	7	11	1	262
2013	155	4	84	7	11	1	262
2014	149	1	93	7	12	0	262
2015	145	1	97	7	12	0	262
2016	140	1	102	7	12	0	262
2017	139	1	103	7	12	0	262
2018	139	1	103	7	12	0	262

Table 3
Quantities of residual waste and organic waste per UBP system.

Kilo per inhabitant	Bag&tag	Bag	Container weight& volume	Container weight	Container volume	Average
Residual unsorted	165.73	163.93	113.59			161.46
Residual solid		117.09	91.88	105.77	111.25	101.86
Organic		32.52	34.45	27.01	57.82	34.07

Table 4
Marginal price of residual waste and organic waste per UBP system (in 2009 euros).

Euro/kilo*	Bag&tag	Bag	Container weight&volume	Container weight	Container volume	Average
Residual	0.119	0.107	0.207	0.243	0.109	0.114
%Δ2009-2018	147.83	16.03	34.93	38.12	55.69	43.97
Organic		0.109	0.123	0.167	0.110	0.116
%Δ2009-2018		8.22	32.23	67.51	47.79	24.95

*Corrected for inflation.

4.2.2. Price endogeneity

As already mentioned, the Walloon municipalities are required to do projections for the expected costs associated with the collection and treatment of residual waste, as well as for the revenues that will cover those costs. In practice, municipalities use past quantities to forecast their revenues and adapt subsequently the marginal price. The prices of year *t* are communicated to the households at the end of the previous year (*t-1*), and they react according to this new price. In this respect, it is likely that the prices depend on past quantities: p_{it} is a function of $q_{i,\ t-1}$. This causes a problem of endogeneity since we have a feedback effect from q_{it} to future values of p_{it} (Wooldridge, 2010),[11] and in this case the estimated parameters on the price variables will be biased.

To address this problem of price endogeneity, we use an instrumental variable approach. Given that past quantities affect future prices, we can assume sequential exogeneity for prices: past prices do not affect the future quantities, stated more formally we have E ($p_{is}\ \varepsilon_{it}$) = 0, $s \leq t$. This assumption leads to additional moment conditions that can be used in a general method of moments (GMM) setting to consistently estimate our parameters of interest. This is similar to the Arellano-Bond estimator, except that in the general procedure as in here, no dynamics are included in the model (Cameron and Trivedi, 2005).

In the first-difference (FD) of equation (1) we have: E ($p_{is}^{\prime}\Delta\varepsilon_{it}$) = 0, s = 1,..., t-1; t = 1,..., T. Therefore, our endogenous variable p_{it} is instrumented by $z_{it} = \{p_{i1}, p_{i2}, \ldots\ldots, p_{i,\ t-1}\}$, t = 2,..., T; where in period *t*, all lags of the endogenous variable can be used as instruments.

The sequential exogeneity assumption implies that the number of valid instruments increases with *t*. However, the use of too many identifying restrictions leads to poor finite sample properties of GMM (Wooldridge, 2010). Accordingly, we define two criteria for determining the number of instruments used. First, the threshold level of the first-stage *F*-statistic should be above 10 so that the maximal bias in comparison to OLS is no more than 10 % (Cameron and Trivedi, 2005). Second, we set a 10 % significance level for the over-identifying restrictions test (Hensen *J*-test). We apply panel two-step GMM with clustering at the municipality level, obtaining standard errors that are robust for both heteroscedasticity and serial correlation over time. The results are presented in the following section.

5. Results and discussion

In this section, we report our estimations of Eq. (1) for the three categories of waste: residual unsorted waste (RUW), residual solid waste (RSW) and organic waste. We report several specifications for comparison purposes, but our preferred specification is the one that treats the price as endogenous.[12] By comparing the different specifications, we can see that the price elasticity is systematically underestimated when the

[11] In the literature, a potential problem with cross-sectional data is that the exogeneity assumption may not be satisfied due to the presence of unobserved factors that affect simultaneously the prices and the waste quantities. In this case, an instrumental variable approach should be applied to correct for any omitted variable bias. Kinnaman and Fullerton (2000) used a two-stage least squares (2SLS) estimation and showed that the estimates are quite different compared to the OLS results and that ignoring the endogeneity problem underestimates the effect of prices. The use of panel data methods can mitigate this issue if the unobserved factors are constant over time. There are few panel data studies but most of them did not include fixed effects at the municipalities' level (Dijgraaf and Gradus, 2004). Moreover, the strict exogeneity assumption will not be satisfied if there are time-varying factors that have an impact on the marginal prices. Only Allers and Hoeben (2010) deal specifically with both issues, testing for price endogeneity and applying a corrective IV method.

[12] When the price is considered endogenous, we report the *p*-values of the Hansen *J*-test statistic for over-identifying restrictions, which shows that the null hypothesis that the instruments are exogenous is not rejected. We also report the first-stage regression *F*-statistic and the total number of instruments included in the estimations.

Table 5
Descriptive statistics.

	N	Mean	S.D.	Min	Max
Waste quantities (kilo per inhabitant)					
Residual unsorted waste (RUW)	1,180	161.46	25.7	88.15	220.19
Residual solid waste (RSW)	1,440	101.82	26.07	50.54	220.88
Organic waste	1,440	34.07	16.17	1.34	105.86
Marginal price* (euro per kilo)					
Residual waste					
Bag	1,520	0.108	0.032	0.015	0.215
Bag&tag	29	0.119	0.061	0.049	0.264
$\text{Container}_{\text{volume}}$	114	0.109	0.039	0.022	0.23
$\text{Container}_{\text{weight}}$	70	0.243	0.089	0.07	0.396
$\text{Container}_{\text{weight \& volume}}$	877	0.207	0.109	0.089	0.922
All municipalities	2,620	0.144	0.086	0	0.922
Organic waste					
Bag	819	0.109	0.077	0	0.624
$\text{Container}_{\text{volume}}$	60	0.110	0.049	0.023	0.226
$\text{Container}_{\text{weight}}$	50	0.167	0.089	0.056	0.39
$\text{Container}_{\text{weight \& volume}}$	511	0.123	0.061	0	0.387
All municipalities	1,440	0.116	0.072	0	0.624
Demographics					
Population size	2,620	13,654.29	21,022.82	1,382	20,3871
Density (inhabitants per km^2)	2,620	318.63	439.78	24	3,531
Household size	2,620	2.384	0.14	1.86	2.81
Average income per inhabitant*	2,620	16,266	2,294	10,398	26,649
% of the population under 20 years	2,620	24.27	1.85	18.8	31.21
% of the population above 60 years	2,620	22.74	2.68	14.9	32.7
*Adjusted for inflation					

Table 6
Own-price elasticity for residual unsorted waste (RUW).

	(1)	(2)	(3)
Residual unsorted waste			
p endogenous	no	no	yes
ln $p_{residual}$	−0.767***	−0.789***	−1.204***
	(0.160)	(0.160)	(0.133)
ln density		0.272	0.947**
		(0.742)	(0.465)
ln density^2		−0.0332	−0.0804*
		(0.0690)	(0.0434)
ln income		−0.0841	−0.0289
		(0.111)	(0.0590)
Household size		0.0293	0.0896
		(0.126)	(0.0558)
%<20 years		−0.00900*	−0.00773**
		(0.00534)	(0.00311)
%>60 years		0.000311	0.00164
		(0.00536)	(0.00284)
Municipalities fixed effects	Yes	Yes	Yes
Time fixed effects	Yes	Yes	Yes
Number of observations	1007	1007	1000
Number of clusters	132	132	132
Adjusted R^2	0.118	0.115	0.107
Hansen test (*p*-value)			0.476
First stage *F*-statistic			23.814
Number of instruments			63

Clustered standard errors in parentheses. *p < 0.10 **p < 0.05 ***p < 0.01.

price variable is treated as exogenous, in line with previous research (Kinnaman and Fullerton, 2000).

5.1. Price elasticities when there is no collection of organic waste

We first focus on the municipalities that do not collect organic waste at the curbside.

As we can see in Table 6 column (3), the own-price elasticity of RUW is −1.2 and is statistically significant at the 1 % level. This result implies that the demand for residual waste is elastic in municipalities where no curbside collection of organic waste is organised, and households react notably to an increase in the price of waste by decreasing the quantities presented for collection. A 1 % increase in the price of residual waste reduces residual waste production by 1.2 %.

The most plausible explanation is that households exert more effort to decrease their total bill by reverting to zero-waste practices or home composting, especially with the spread of home composting containers available at moderate prices. Home composting might be more complicated for households living in smaller areas, which is confirmed by the positive coefficient for the variable density.

The estimate on the quadratic term is negative, even though it is marginally significant, it deserves some attention. It implies that the quantities increase at high densities, and this can be explained in two ways. First, it may be due to illegal dumping (Kinnaman and Fullerton, 2000). Second, it may be the effect of a low generation of waste because, in big cities, households spend more time outside their apartments, in cafés and restaurants for example (Callan and Thomas, 2006), which decreases the residential quantities collected. Finally, we find a negative coefficient for the proportion of the population below 20 years as this age group eventually spend most of their time in schools, universities, and other recreational activities and less at home.

For the municipalities that do not collect organic waste, we cannot test the impact of the UBP systems as there is no variability and the effect of the UBP is absorbed into the municipality's fixed effect.

5.2. Price elasticities when there is a collection of organic waste

Next, we estimate the price elasticities in municipalities that organise a collection of organic waste.

5.2.1. Elasticities of residual waste

We find that the price elasticity of residual waste is about −0.3 and is significant at the 5 % level (Table 7, column 4). A price increase of 1 % reduces the residual waste production by 0.3 %. This is much lower than the elasticity computed for RUW. Once the organic fraction is collected separately at the curb, households have less room to reduce their waste, and therefore, their reaction to the price is lower. Once all sorting options have been proposed to households, they are left with an incompressible residual fraction that responds hardly to the price variation. In such a case, a reduction of waste at the source, i.e., waste avoidance is needed to have a significant decrease in the quantities collected.

The coefficient on the price of organics measuring the cross-price elasticity is positive as expected, but it is neither economically nor statistically significant. The estimations also show that the pricing system matters. For a given price level, municipalities using containers with a price per kilo and a price per levy produce on average 31 %[13] less residual waste than the ones using the bag. These results are compatible with the economic rationale since households would try to put as much waste as the bag can handle without being penalized. This is not the case when they have to pay for the actual weight of the waste they generate, where each additional material put for disposal will increase their cost. This finding is extremely important since more municipalities are adopting a weight-based pricing system. It shows that the weight-based system is an effective tool and provides an incentive for households to alter their purchasing and consumption behaviour over time.

Looking at the socio-economic variables, we observe that the income

[13] The exact percentage change was calculated as [Exp (ζj) − 1] × 100, where the dependent variable is expressed in logarithm.

Table 7
Own-price and cross-price elasticities for residual solid waste (RSW).

	(1)	(2)	(3)	(4)
Residual solid waste				
$p_{residual}$ endogenous	no	no	no	yes
ln $p_{residual}$	-0.590^{**}	-0.0963	-0.0937	-0.290^{**}
	(0.277)	(0.131)	(0.134)	(0.147)
ln $p_{organic}$	0.0633	0.0276	0.0249	0.0664
	(0.116)	(0.0859)	(0.0883)	(0.0893)
Container $_{weight\&volume}$		-0.378^{***}	-0.376^{***}	-0.378^{***}
		(0.104)	(0.105)	(0.0624)
$\text{Container}_{volume}$		-0.0538	-0.0527	-0.0331
		(0.0428)	(0.0433)	(0.0236)
ln density			-0.400	-0.373
			(0.813)	(0.562)
ln density^2			0.0462	0.0344
			(0.0968)	(0.0668)
ln income			0.240*	0.334^{***}
			(0.139)	(0.0947)
Household size			-0.0312	-0.0287
			(0.169)	(0.103)
%<20 years			-0.0111	-0.00428
			(0.00855)	(0.00521)
%>60 years			0.00476	0.00750^{**}
			(0.00462)	(0.00363)
change_{org}	0.0479^{**}	0.0464^{**}	0.0467^{**}	0.0473^{***}
	(0.0218)	(0.0215)	(0.0218)	(0.00956)
Municipalities fixed effects	Yes	Yes	Yes	Yes
Time fixed effects	Yes	Yes	Yes	Yes
Number of observations	1214	1214	1214	1209
Number of clusters	155	155	155	155
Adjusted R^2	0.167	0.263	0.264	0.256
Hansen test (*p*-value)				0.325
First stage *F*-statistic				13.789
Number of instruments				54

Clustered standard errors in parentheses. *$p < 0.10$ **$p < 0.05$ ***$p < 0.01$.

Table 8
Own-price and cross-price elasticities for organic waste (ORG).

	(1)	(2)	(3)	(4)
Organic				
$p_{organic}$ endogenous	no	no	no	yes
ln $p_{organic}$	-0.711*	-0.735*	-0.725*	-0.0561
	(0.416)	(0.436)	(0.426)	(0.134)
ln $p_{residual}$	1.578^{**}	1.579^{**}	1.561^{**}	1.396^{***}
	(0.793)	(0.793)	(0.783)	(0.229)
Container organic$_{weight\&volume}$		0.0150	0.0408	0.0285^{**}
		(0.0252)	(0.0252)	(0.0137)
Container organic$_{volume}$		-0.00833	0.00746	0.0188
		(0.0278)	(0.0293)	(0.0144)
ln density			-5.536^{**}	-5.822^{***}
			(2.567)	(1.443)
ln density^2			0.716^{**}	0.729^{***}
			(0.314)	(0.179)
ln income			-0.102	-0.553^{**}
			(0.345)	(0.228)
Household size			-0.468	-0.591^{**}
			(0.465)	(0.242)
%<20 years			0.0203	0.000598
			(0.0206)	(0.0122)
%>60 years			0.00642	0.00456
			(0.0140)	(0.00795)
change_{org}	-0.521^{***}	-0.521^{***}	-0.513^{***}	-0.475^{***}
	(0.0649)	(0.0649)	(0.0644)	(0.0319)
Municipalities fixed effects	Yes	Yes	Yes	Yes
Time fixed effects	Yes	Yes	Yes	Yes
Number of observations	1214	1214	1214	1208
Number of clusters	155	155	155	155
adj. R^2	0.388	0.388	0.389	0.378
Hansen test (p-value)				0.277
First stage F-statistic				15.877
Number of instruments				63

Clustered standard errors in parentheses. *$p < 0.10$ **$p < 0.05$ ***$p < 0.01$.

elasticity is positive, and its magnitude is comparable to previous work (Gellynck and Verhelst, 2008). Consumption increases with income, and richer households produce more residual waste. The coefficients for the density variables are not significant, confirming that the collection of organics is a substitute to home composting. The proportion of elderly above 60 years of age has a positive estimate, possibly because they spend more time at home and generate more waste compared to their younger counterparts who spend most of their time at work. It can also be that the elderly recycles less but, as we will see, the results of the regression on organic waste do not support this claim.

5.2.2. Elasticities of organic waste

The estimates of Equation (1) for the organic waste are displayed in Table 8. The own-price elasticity is negative but does not appear to be significant when we treat the price as endogenous (column 4). The cross-price elasticity with respect to RSW carries the expected sign and is highly significant.

An increase in the price of RSW by 1 % will increase the quantities of organic waste by 1.4 % and will decrease the residual waste by 0.3 % as we have shown in Table 7. This means that sorting improves and households pay more attention to diverting the organic fraction from the residual one when the residual price gets higher. The high cross-price elasticity can be explained by the fact that sorting organic waste remains optional, i.e., households can still use the residual bin for their organic waste. However, as the price of residual waste increases, it increases participation in sorting and therefore reduces the residual fraction. The price of residual waste has a significant impact on the quantities of both organic and residual waste. On the other hand, the price of the organic fraction has a negative but not significant impact on the residual fraction. Therefore, our estimations show that it is the price of the residual fraction that incentivizes households to sort their organic waste and not the price of the organic fraction.

Unlike RSW, the weight-based system does not seem to outperform the bag for the organic fraction, the coefficient is even slightly positive. A plausible justification is that municipalities using weight-based pricing for the organic fraction necessarily also use the same pricing system for RSW. Since the price differential is large between the two streams, this gives more incentives for households to improve sorting of the organic fraction, and so the quantities presented for collection increase. Furthermore, when the bags are used for collection, they are (time) costlier to households since they have to purchase the bags and they have to find a place to store them.

The variable $change_{org}$ is negative for organic and positive for the residual solid waste, meaning that the quantities captured are significantly higher the years after the introduction. This may reflect learning effects or a progressive introduction of organic collection during the year.

Most of the socio-economic variables are significant. What is particularly interesting is that the sign of density and income. They are both negative, meaning that households living in more dense and poorer places produce less organic waste. In small and low-income areas, households have less space to store a separate bag/container for organic waste beside the one for residual waste and given the putrescible nature of organic waste, they are less prone to participate in sorting. In a study of the attitudes towards the selective collection of organic waste, Bernad-Beltrán et al. (2014) found that the most important barrier to participate in the selective collection is the lack of space in the house. Our results confirm that claim.

6. Conclusion

Several reforms have taken place in Wallonia. Since 2010, all municipalities have introduced unit-based pricing for residual waste. The weight-based pricing system expanded to 42 % of the municipalities. Finally, the collection of organic waste, currently done by 62 % of the municipalities, will be compulsory by 2025. In this paper, we examined the effectiveness of those instruments.

We find that the price elasticity depends on the possibility for the households to dispose of their organic waste at the curbside. When this possibility exists, households have a lower sensitivity to prices, with an estimated price elasticity of −0.3. When this possibility does not exist, the estimated elasticity is higher at −1.2. The collection of organic waste substantially reduces the quantity of residual waste, but it also limits the effectiveness of the price instrument for reducing residual waste. This implies that other tools are needed to decrease this less desirable fraction. These can be information campaigns that encourage households to decrease waste generation from the source, to promote the reuse of products and waste, to increase the awareness for better consumption habits and the reduction of food wasting, and to encourage the consumption of zero-waste products.

Another tool that can be used is the pricing unit. By comparing different unit pricing systems, we found that the weight-based system outperforms all others and reduces significantly residual waste. It is therefore an efficient tool for municipalities to achieve their environmental objectives. Hence, high prices for the residual fraction could be efficiently combined with weight-based pricing to reduce the residual fraction and encourage sorting.

Turning to organic waste, our results show that it is the price of the residual fraction that has the highest impact on the production of organic waste. The cross-price elasticity with respect to residual solid waste is highly significant. This shows the importance of the price signal to give incentives to households to improve sorting, and policymakers should design their tariffs such that the price differential makes it worthy for households to spend the necessary time to appropriately separate the different fractions.

While the present results of this paper give an important assessment of the price instruments in place and show their effectiveness and their limitations, there are two points that need further investigation. First, municipalities using weight-based pricing do apply non-linear pricing, this may be another reason that makes it perform better than other systems. To detangle this effect, we need to have more granular data. Second, there are municipalities that started the curbside collection of organic waste earlier than the others. It would be interesting to test whether there is a learning effect through a better understanding of what constitutes organic waste and the financial and environmental implications of sorting associated with the time elapsed since the implementation of curbside collection of organics. These topics are on our agenda for future research.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

This paper uses publicly available data. Links to data source are included in the paper.

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