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Wasting Time? Recycling Incentives in Urban Taiwan and Japan

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Abstract What makes recycling work? We study the factors driving household waste disposal and recycling in 18 cities in Taiwan and Japan in order to understand the impact of alternative waste management incentives. We show that this depends on the effect of distinct policies on the relative costs of the main alternative disposal methods: recycling, disposal to landfill and illegal dumping. The willingness both to recycle and to dispose to landfill depends on the relative costs of the waste collection regime, and these are dominated by the time cost of alternative disposal methods. The higher the frequency of waste collection, the less recycling and the more disposal to landfill there will be. This is because frequent collection reduces the marginal time-cost of disposal to landfill. Curbside collection of recyclable material, and the frequency of that collection, has a similar effect on the recycling rate. Although direct incentives, such as unit pricing are important in the waste disposal decision, recycling depends primarily on management of the time-costs it involves.

Keywords Solid waste · Recycling · Illegal dumping · Waste disposal costs

List of symbols

Legal disposal related

d The volume of (legal) disposals to landfill/incineration
α The labor factor for legal disposals
p_d Unit price for non-recyclable collections

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Recycling related

- r The volume of recycled waste
- β The labor factor for recycling
- p_{rj} Price of recyclable collection for the material j
- f_m The nominal fine for non-compliance with mandatory recycling
- μ_r The probability of prosecution for non-compliance

Illegal dumping related

- b The volume of illegal dumping
- γ The labor factor for illegal dumping
- f_b The nominal fine for illegal dumping
- π_b The probability of prosecution for illegal dumping

Proxies and policy dummies of above variables used in empirical study

- F The collection frequency of legal disposal
- N Recycling enforcement
- D Population density
- I Inspection and enforcement for illegal dumping
- C How many material can be recycled on curbside
- R Total recyclables
- T Enforcement for unit pricing
- D Non-recyclable waste
- W Total waste
- R Recycling ratio
- Y Purchasing-power based income
- T Transparent bags for non-recyclable collection (waste disposal)
- M Mandatory recycling
- K_j Curbside recycling provided for jth material
- p_r The maximum of p_{rj}
- U On-time collection—demanding residents to hand garbage bags to haulers
- f_z A policy of imposing fines for illegal dumping
- Z A dummy for adoption of unit pricing

1 Introduction

Municipal solid waste (MSW) has a special place amongst the polluting by-products of economic growth. For decades it has been at the head of the small list of waste materials that monotonically increase with per capita incomes in every country for which there are reliable data (Shafik 1994; Cole et al. 1997). As domestic consumption increases, so does solid waste. As a result, many cities have introduced measures, including both regulations and market-based incentives, to increase the rate at which solid waste is recycled. These measures are designed to have three main effects: to increase recycling, to reduce disposal to legal landfills or incinerators, and to reduce the use of illegal dumpsites. While mandatory recycling supported by penalties for non-compliance are the instrument of choice in some locations, most municipalities rely on a mix of regulation and charges that collectively change the relative private cost of the different waste disposal options open to households. These generally include the unit pricing of waste. Waste collection fees based on the volume or weight of waste products simultaneously discourage disposals to landfills and encourage waste recycling. Since unit pricing may be expected to encourage illegal dumping, however, it is seldom sufficient. It is therefore usually used as part of a mix of instruments. In this paper we consider what it is that makes a mix of instruments effective in achieving municipalities' recycling goals, using evidence on MSW regimes and material-specific recycling practices in 18 cities in Taiwan and Japan.

There are relatively few empirical studies of municipal recycling, and only Reschovsky and Stone (1994); Sterner and Bartelings (1999); Jenkins et al. (2003) and Kipperberg (2007) use household material-specific recycling data to analyze the effect of recycling programs. Most existing studies treat all recyclable materials as uniform, and use aggregate data. While the studies that do make use of material-specific data disagree on whether unit pricing significantly increases recycling, they all find that when combined with measures that reduce the cost of recycling to the household such measures can have a positive effect. The studies that rely on aggregated data are more consistent in their findings. During the 1990s, one set of studies suggested that the incentive effects of unit pricing could be quite substantial. Miranda et al. (1994), for example, found that, for 7 out of 9 communities, waste to landfills and incinerators declined by between 20 and 50% following the adoption of unit pricing of waste, while all communities saw recycling increase by between 30 and 100%. Seguino et al. (1995) found that per capita waste in communities with unit pricing (pay-by-the-bag) was 56% lower than that in communities without it. Miranda and Aldy (1998) again showed that most communities reported at least a 20% reduction in waste and a 30% increase in recycling after unit pricing was adopted.

The more systematic evidence on the price elasticity of demand for non-recyclable[1] waste collection services in cities using unit pricing is less positive. Most estimates of the price elasticity lie between −0.12 and −0.39 (Wertz 1976; Jenkins 1993; Strathman et al. 1995; Podolsky and Spiegel 1998; Houtven and Morris 1999; Kinnaman and Fullerton 2000; Dijkgraaf and Gradus 2004). Once again, the biggest difference observed in empirical studies is between US and European cases. The US, applies can/volume and tag systems. Studies of household demand for such services in the 1990s showed the elasticity to be either insignificant (Hong et al. 1993), or very low (0.058 in Fullerton and Kinnaman (1996) 0.013 in Hong and Adams 1999). Although the elasticity under bag and weight systems was higher, (0.26 in Houtven and Morris 1999) it was still lower than elsewhere. Moreover, in some cases the introduction of pricing systems increased waste flows. In Marietta, Georgia, for example, it was found that 'in the can system' generally increased total waste generation, while reductions achieved through 'in the bag system' were insignificant (Nestor and Podolsky 1998).

Studies elsewhere in the world indicate a price elasticity of demand for non-recyclable waste collection that is higher than in the US. A study of 20 Korean cities, for example, yielded an estimate of 0.154 (Hong 1999), while two studies of the effects of a range of waste pricing schemes in the Netherlands yielded estimates of 0.71 (Dijkgraaf and Gradus 2004) and 1.39 (Linderhof et al. (2001)). The evidence on the cross price elasticity of recycling follows a very similar pattern. Estimates range from 0.073 to 0.457, and are systematically

[1] Based on the nature of waste, MSW can be classified as non-recyclable, recyclable, hazardous, and bulky waste. Non-recyclable waste is referred to as unsorted waste, mixed waste, or rubbish and garbage in colloquial language. Recyclable waste (recycling in this paper), consists of the recyclable materials discarded by households via recycling collection systems rather than recycled or reused waste.

lower in the US than elsewhere (Kinnaman and Fullerton 2000; Dijkgraaf and Gradus 2004; Hong et al. 1993; Fullerton and Kinnaman 1996; Hong 1999; Hong and Adams 1999).

While the difference between the US experience and experience elsewhere is not our concern, it is instructive. Aside from cultural factors, one explanation for the systematic difference observed in cities in the US and elsewhere is the effect of the other elements in the mix of measures applied to the problem. In Taiwan, for example, Shaw and Tsai (2002) found that, in Taiwan, both unit pricing and mandatory recycling can significantly increase household recycling time, but only unit pricing can decrease garbage disposal. In fact the evidence on the effect of mandatory schemes is mixed. Studies of the problem in the United States (Reschovsky and Stone 1994; Jenkins et al. 2003) found that mandatory recycling had no significant impact on recycling. A more recent study of mandatory recycling in Norway, however, found that it significantly increased recycling of specific wastes—metal and food (Kipperberg 2007). Similarly, Yang and Innes (2007) found that a combination of unit pricing, a ban on plastic bags, and mandatory recycling in Taiwan both increased recycling and decreased disposal.

In this paper, we seek to understand the impact of the array of measures used to encourage recycling in Taiwan and Japan. Using a household behavior model fitted with data from 18 cities in Taiwan and Japan, we identify the factors driving the household choices between the possible disposal options, including illegal disposal. The study is unusual in including at least one category of illegal disposal[2] (we are aware of just one empirical study of this problem, Miranda and Bynum 2002). We find that the effectiveness of policies based on a combination of mandatory recycling and the unit pricing of waste collection depends on a set of factors that collectively alter the private relative cost of disposal options more profoundly than the waste collection charge and the penalty for non-compliance with mandatory recycling can do alone. More particularly, we find that the dominant factor in household waste disposal decisions is the time cost of alternative options. Wasted time is the main cost of waste disposal.

2 A Household Waste Disposal Model

Most empirical studies of waste disposal policies are based on household behavior models. The first model of household behavior in relation to waste disposal (Smith 1972) included the choice between disposal to landfill and recycling. This remained the standard choice until Fullerton and Kinnaman (1995) added illegal dumping as a third option. Kuo (2005) later added storage/composting. The basic model supposes that in a city of n households, at time t, the representative household (i) consumes a bundle of goods, x_t, a portion of which, ax_t, become waste products in the process of consumption. The disposal options open to the household are disposal to landfill, d_t, recycling, r_t, or illegal disposal, b_t. Illegal dumping generates a public, non-depletable externality, $B_t = \sum_{i=1}^{n} b_t^i$. Waste not discarded in these three ways may be stored, s_t, with decomposition rate, ϕ. Hence stored waste evolves according to:

$$\frac{ds}{dt} = \alpha x_t - d_t - b_t - r_t - \phi s_t \tag{1}$$

[2] This includes littering, burning, dumping in commercial dumpsters and disposal in other jurisdictions. It also includes disposing of mandatory recyclables to landfill. We consider this last category of illegal dumping.

Households derive utility from consumption and disutility from waste. If the discount rate is δ, the discounted utility stream of the i household at time 0 is given by:

$$U_0^i = \int_{t=0}^{T} u_t^i \left(x_t, s_t, B_t\right) e^{-\delta t} dt \tag{2}$$

with the marginal utility of consumption being positive and diminishing, and the marginal utility of waste being negative and diminishing : i.e.: $u_x > 0$, $u_{xx} < 0$, $u_s < 0$, $u_{ss} < 0$, $u_B < 0$, and $u_{BB} < 0$. The cost of disposal alternatives comprises both the direct time commitment involved, and any other direct or indirect costs associated with each option. Suppressing superscript i the time cost for the three options is L_d, L_r and L_b out of an effective time endowment of L. The direct costs of the first two options are p_d and p_r. If the nominal fine for illegal dumping is f_b and the chance of being prosecuted is $\pi(E, b_t)$, a function of enforcement and the amount dumped, and if the wage rate is w, the household budget constraint takes the form:

$$w\left(L - L_d\left(d_t\right) - L_b\left(b_t\right) - L_r\left(r_t\right)\right) = p_x x_t + p_d d_t + \pi\left(E, b_t\right) f_b \tag{3}$$

Maximization of (2) subject to (3) by choice of the level of consumption and the waste disposal options requires satisfaction of the following first order necessary conditions on those options:

$$\left(w\left(L_{d_{dt}}\right) + p_d\right)\lambda_1 + \lambda_2 = 0 \tag{4}$$

$$\left(w\left(L_{r_{rt}}\right) + p_r\right)\lambda_1 + \lambda_2 = 0 \tag{5}$$

$$-u_B^i + \left(w\left(L_{b_{bt}}\right) + \pi_{bt} f_b\right)\lambda_1 + \lambda_2 = 0 \tag{6}$$

where the co-state variable, λ_1, is the marginal utility of income, and λ_2 is the marginal disutility of waste. The arbitrage condition between waste disposal options,

$$w\left(L_{d_{dt}}\right) + p_d = w\left(L_{b_{bt}}\right) + \pi_{bt} f_b - \left(\frac{u_B^i}{\lambda_1}\right) = w\left(L_{r_{rt}}\right) = \frac{u_s}{\phi} \tag{7}$$

requires that the marginal cost of each disposal option be equal.

The marginal cost of disposal options may be influenced by a number of factors other than the direct price of each option. The waste and recycling collection regimes can, for example, significantly affect the time associated with each disposal method. From the household behavior model, this should impact the effectiveness of the price incentives used as part of the policy. In Taipei, for example, introduction of a unit pricing policy (bag system) decreased waste disposal by 32.79% and increased recycling by 98.87% in two years. While this implies a highly elastic response to a price signal, it should also be noted that charging was introduced alongside a series of collection reforms that significantly affected the net benefits to alternative disposal methods for Taipei residents. These included both high collection frequency (5–6 times per week) and a 'no garbage in the street' policy.[3] Since Taipei residents—as in other cities in the region—mostly live in small, high rise apartments, the costs of waste storage are also high.

To capture the impact of these factors we estimate a reduced form of the household behavior model (1)–(3) that includes a number of properties of the collection system in addition. The time and direct costs of the three disposal options have already been defined. In this

[3] The policy requires that residents hand garbage bags to haulers rather than just leave them at collection points, called on-time collection in this paper.

case we are interested in systems involving mandatory recycling, so the penalty for disposing of recyclable materials as non-recyclable is also part of the cost of disposal to landfill. We expect that the chance of being prosecuted, μ, decreases with recycling, $1/r$, and the penalty for insufficient recycling, f_m. The marginal expected penalty is thus $\mu_r f_m$. Since household behavior has been shown to vary systematically with income, education and other socio-economic variables, we also include a vector of such variables, S. Suppressing time subscripts on the independent variables for clarity, the reduced form of the household decision model for each option is:

$$r_t = r\left(L_r, p_r, \mu_r f_m, S\right) \tag{8}$$

$$d_t = d\left(L_d, p_d, S\right) \tag{9}$$

$$b_t = b\left(L_b, \pi_b f_b, S\right) \tag{10}$$

We approximate the time cost of the different disposal options by a measure of the time elapsed between waste generation and disposal. This is consistent with studies that have used a travel cost approach to estimating the value of waste collection (e.g. Salkie et al. 2001), although in our case the time elapsed includes both storage in house and the time cost of the chosen option for disposing of stored waste. The approach implies that aside from the collection fee, p_d, the cost of disposal can be approximated by the time costs involved. Because most cities provide curbside waste collection, the major part of the cost of disposal is the time that the waste has to be held until disposal. A lower frequency of waste collection a higher time cost—in terms of domestic waste storage as well as waste removal. A reasonable proxy for the time cost of legal disposal is thus the frequency of collection, F:

$$L_d = L_d\left(F\right) \quad \text{and} \quad L_{d_F} \leq 0 \tag{11}$$

An additional cost of disposal to landfill is any penalty for non-compliance with a mandatory recycling policy. This is equal to the marginal expected fine for non-compliance, already defined as $\mu_r f_m$. We take this to be a function of recycling enforcement policies, N. For example, a policy of using transparent bags for waste disposal lowers the costs of inspection and thus increases the likelihood of detection. It follows that:

$$\mu_r = \mu_r\left(N\right) \quad \text{and} \quad \mu_{rN} \geq 0 \tag{12}$$

The labor cost of recycling, L_r, can also be separated into a distance cost and a time cost, and as with disposal to landfill, this can be approximated by the frequency of curbside collection, denoted K_j for the jth recyclable material. So we have:

$$L_{rj} = L_r\left(K_j\right) \quad \text{and} \quad L_{r_{Kj}} \leq 0 \tag{13}$$

The cost of illegal dumping is the expected value of the fine, already defined. We assume that the probability of being prosecuted and the volume of illegally dumped waste are a function of population density, P, and enforcement effort, E. The probability of being prosecuted and the marginal distance cost of illegal dumping are both lower in low population density areas. So we have:

$$\pi_b = \pi_b\left(P, E\right), \pi_{b_P} \geq 0 \quad \text{and} \quad \pi_{b_E} \geq 0 \tag{14}$$

$$L_b = L_b\left(P\right) \quad \text{and} \quad L_b' \geq 0 \tag{15}$$

Hence the reduced-form model of each recyclable material depends on the legal disposal variables, p_d and F, recycling policy variables, p_{rj}, f_m, K_j, and N, the illegal dumping variables f_b, E and P, together with a vector of socioeconomic factors, S.

$$r_{jt} = r\left(p_d, F; p_{rj}, f_m, K_j, N; f_b, E, P; S\right) \tag{16}$$

Aggregate recycling is simply the sum of all recyclable materials (*C* kinds of materials):

$$r_t = \sum_{j=1}^{c} r_{jt} \tag{17}$$

and the reduced-form aggregate recycling model is:

$$r_t = r\left(p_d, F; \mathbf{p}_r, f_m, K, N; f_b, E, P; S\right) \tag{18}$$

where $\mathbf{p}_r$ is the vector recyclable prices. The household's disposal of non-recyclable wastes similarly depends on the marginal cost of each disposal option, and this in turn depends on the effect of collection methods on both the time and direct costs of collection. In a number of the cities investigated in this study, a system of 'on time collection'—under which residents are required to hand waste directly to the collectors—allows haulers to check paid bag usage. Denoting 'on time collection' by *U,* we can write the reduced-form for the non-recyclable disposal model as:

$$d_t = d(p_d, F, U; \mathbf{p}_r, f_m, K, N, C; f_b, E, P; S) \tag{19}$$

while the recycling ratio is the ratio of the aggregate recycling and total waste.

$$R_t = \frac{r_t}{r_t + d_t} = R(p_d, F, U; \mathbf{p}_r, f_m, K, N, C; f_b, E, P; S) \tag{20}$$

3 Data and Model Estimation

Two kinds of data were used to estimate these models. The first comprises household level data on disposal and recycling behavior. The second comprises municipal data on the waste management system. The main sources for these data are municipal statistics and individual household surveys. In this paper we focus on city level data, partly because of well-recognized biases in self reported recycling activities (Reschovsky and Stone 1994; Martin et al. 2006).

The data derive from a set of 18 cities in Taiwan and Japan (Table 1). The selected cities use a range of waste-management policies. They also operate a range of frequencies or methods of garbage and recycling collection. Data on household waste disposal and recycling were obtained from online databases of the Environmental Protection Administration, Taiwan and

Table 1 Chosen cities and their population density in 2002

Country	Japan				Taiwan	
City name and density (people/ km^2)	Sapporo	1646.60	Osaka	11225.09	Taipei	9720.39
	Sendai	1270.28	Kobe	2697.95	Hsinchu	3640.86
	Chiba	3257.53	Kitakyushu	2060.86	Tainan	4241.50
	Yokohama	7995.17	Fukuoka	3864.42	Kaohsiung	9831.20
	Nagoya	6696.51	Nagasaki	1244.27	Keelung	2945.19
	Kyoto	2406.30	Naha	7831.70	Taichung	6100.64

Table 2 Descriptive statistics of dependent variables (g/person.day)

Variable	Mean	SD	Minimum	Maximum	Cases
Non-recyclable	839.35	179.94	472.00	1491.00	108
Japan cases	794.85	133.83	571.47	1180.62	72
Taiwan cases	928.36	224.58	472.00	1491.00	36
Total waste	978.90	162.71	634.28	1639.89	108
Japan cases	954.48	142.44	634.28	1295.18	72
Taiwan cases	1027.74	189.99	635.92	1639.89	36
Recyclable paper	81.86	68.74	0	325.92	108
Japan cases	98.45	75.14	0	325.92	72
Taiwan cases	48.68	35.98	0.82	148.83	36
Recyclable metal	22.14	13.64	0.29	57.95	108
Japan cases	24.01	11.64	2.52	52.86	72
Taiwan cases	18.42	16.51	0.29	57.95	36
Recyclable glass	11.34	11.03	0.02	49.32	108
Japan cases	15.85	10.92	0.23	49.32	72
Taiwan cases	2.32	2.00	0.02	6.92	36
Recyclable PET	4.03	3.07	0	11.95	108
Japan cases	3.41	2.44	0	9.84	72
Taiwan cases	5.25	3.79	0.12	11.95	36
Recyclable plastics	5.77	8.95	0	36.35	108
Japan cases	3.94	9.22	0	36.35	72
Taiwan cases	9.44	7.18	0	28.99	36
Total recyclables	139.55	94.43	2.09	419.45	108
Japan cases	159.64	99.01	29.64	419.45	72
Taiwan cases	99.38	69.89	2.09	241.17	36

from the Ministry of Environment, Japan.[4] The data cover the period from 1998 to 2003. Descriptive statistics on dependent variables, including per capita per day non-recyclable and total waste generation, the recyclable paper, metal, glass, PET bottles, plastics and total recyclables, are listed in Table 2.

Of the socio-economic variables average, Y, income turns out to be the dominant influence on disposal choices. Hence we set $Y \simeq S$, and approximate Y by per capita Regional Gross Domestic Product. Japanese cities' per capita GDP is unavailable but prefectural per capita GDP[5] is a reasonable proxy—since the Prefecture is a higher level of local government than the City and most selected cities are capitals of prefectures. To ensure that income data for each city and year in the two countries are consistent, nominal per capita regional GDP was converted to common unit. Specifically, we converted nominal per capita regional GDP to real

[4] a. Environmental Database, Environmental Protection Administration, Executive Yuan, R.O.C. (Taiwan) (http://edb.epa.gov.tw/); b. Survey of Municipal Solid Waste Management of Ministry of Environment, Japan (http://www.env.go.jp/recycle/waste_tech/ippan/).

[5] Prefectural per capita GDP is obtained from the Prefectural Account, Economic and Social Research Institute, Cabinet Office, Japan (http://www.esri.cao.go.jp/jp/sna/toukei.html#kenmin).

Table 3 Descriptive statistics of per capita income and population density

Variable	Mean	SD	Minimum	Maximum	Cases
Y[a]	20946.24	3549.61	14676.60	30608.02	108
Japan cases	20230.76	3330.16	14676.60	27716.36	72
Taiwan cases	22377.20	3586.51	17517.54	30608.02	36
P[b]	4872.92	3130.94	1239.09	11690.07	108
Japan cases	4313.62	3150.25	1239.09	11690.07	72
Taiwan cases	5991.53	2811.83	2862.33	9831.20	36

[a] International dollars/person year; [b] People/km^2

per capita regional GDP (1995 prices) using consumer price indices,[6] and then converted real per capita regional GDP to purchasing-power-parity income.[7] This has a significant effect. While average per capita income (1995 US dollars) in Japanese cities was double that in Taiwanese cities, average purchasing-power-parity income in Taiwan and Japan are much closes. Data on income and population density[8] are summarized in Table 3.

We obtained data on the waste management variables from a survey of cities' solid waste management authorities administered in January 2005. The survey posed a number of questions about waste management policies and procedures in the city during the period 1998 to 2004. It attracted a 100% response rate. Importantly, it added additional explanatory variables to the data on the cost of alternative disposal methods. Two kinds of recycling enforcement policies are applied in this set of cities. First, a number of Taiwanese cities have mandatory recycling policies, M. In fact several of these were initiated during the survey period: Taichung City, Kaohsiung City, Keelung City and Tainan City started mandatory recycling in July 1998, January 2001, July 2001 and July 2003, respectively. Second, several Japanese cities, and Taipei City, Taiwan, require residents to use transparent bags or authorized transparent bag for the collection of non-recyclable wastes, T. This is designed to lower inspection costs of insufficient recycling.

These policies are supported in several cities by unit pricing policies, Z. Between 1998 and 2003, six cities adopted unit pricing through a requirement that residents use an authorized garbage bag for dumping. Taipei City has applied a "Per Bag Garbage Collection Fee" from July 2000. In Japan, five cities have started to use unit pricing: Sendai City from April 1991, Chiba City from January 1995, Fukuoka City from December 1997, Kitakyushu City from July 1998, and Nagoya City from August 2000. The price of a garbage bag, p_d, is 0.047US$ per 10 liters of waste in Japanese cities, and 0.145US$ per 10 liters in Taipei City. This reflects the fact that the price in Taipei City includes full collection and treatment costs. An additional difference between Japanese and Taiwanese cities is that non-recyclable waste collection frequencies are much higher in Taiwan (5–7 times per week) than in Japan (2–3 times per week). The descriptive statistics of these policy variables are listed in Table 4.

[6] The general (price) index of the city area is obtained from the Japanese Statistical Yearbook 2005 (http://www.stat.go.jp/data/nenkan/index.htm), and the price index is obtained from the Consumer Price Index, R.O.C. (Taiwan) Statistical Yearbook 2005 (http://eng.dgbas.gov.tw/).

[7] Would Economic Outlook Database, International Monetary Fund (http://www.imf.org/external/pubs/ft/weo/2005/02/data/index.htm).

[8] The data source for population is the same as that in Footnote 3. City territory data for Taiwanese cities are obtained from the Environmental Database, Environmental Protection Administration, Executive Yuan, R.O.C. (Taiwan), and for Japanese cities are obtained from the Japanese Local Governments website (http://uub.jp/).

Table 4 Descriptive statistics of independent variables

Variable	Mean	SD	Minimum	Maximum	Cases
F^{a}	3.57	1.93	2	7	108
Japan cases	2.25	0.44	2	3	72
Taiwan cases	6.22	0.48	5	7	36
C^{b}	2.69	1.74	0	7	108
Japan cases	3.17	1.70	2	7	72
Taiwan cases	1.75	1.42	0	5	36
T^{c}	0.57	0.50	0	1	108
Japan cases	0.81	0.40	0	1	72
Taiwan cases	0.11	0.32	0	1	36
M^{c}	0.12	0.33	0	1	108
Japan cases	0	0	0	0	72
Taiwan cases	0.36	0.49	0	1	36
K^{c}_{paper}	0.35	0.48	0	1	108
Japan cases	0.14	0.35	0	1	72
Taiwan cases	0.78	0.42	0	1	36
K_{metal} and K^{c}_{glass}	0.93	0.26	0	1	108
Japan cases	1.00	0	1	1	72
Taiwan cases	0.78	0.42	0	1	36
K^{c}_{pet}	0.86	0.35	0	1	108
Japan cases	0.90	0.30	0	1	72
Taiwan cases	0.78	0.42	0	1	36
$K^{c}_{plastics}$	0.40	0.49	0	1	108
Japan cases	0.21	0.41	0	1	72
Taiwan cases	0.78	0.42	0	1	36
U^{c}	0.27	0.45	0	1	108
Japan cases	0	0	0	0	72
Taiwan cases	0.81	0.40	0	1	36
f^{d}_{z}	0.65	0.47	0	1	108
Japan cases	0.50	0.50	0	1	72
Taiwan cases	0.94	0.23	0	1	36
p^{e}_{d}	0.024	0.030	0.005	0.152	108
Japan cases	0.024	0.023	0.005	0.086	72
Taiwan cases	0.023	0.040	0.010	0.152	36
p^{e}_{r}	0.016	0.057	0	0.294	108
Japan cases	0.024	0.068	0	0.294	72
Taiwan cases	0	0	0	0	36

[a] Times/week; [b] categories of recyclable waste; [c] policy adopted = 1, policy not adopted = 0; [d] the authority tries to fine illegal dumping = 1, does not try to fine = 0; [e] USD/10 liters

Of the two curbside recycling variables used, K_j measures the frequency of curbside recycling for the jth recyclable waste material. We define C to be the number of categories of recyclable waste material that can be recycled on curbside. Curbside recycling is free in most cases except in Sendai City, Nagoya City, Kyoto City, Fukuoka City and Nagasaki City. These cities charge for cans, jars, or PET bottles for curbside recycling by demanding that residents use only authorized bags for collection. The price of these recyclable materials for curbside recycling, p_{rj}, is the same as the price of non-recyclable collection in those cities. p_r in Table 4 is the maximum price of p_{rj}.

One policy that ensures that residents pay waste collection fees is on-time collection. This policy requires that residents hand garbage bags to haulers rather than leaving them at collection points. Taiwanese cities started to adopt on-time collection during the survey period, and all of them adopted it after 2003. Although both Japan and Taiwan have nominal penalties for illegal dumping and insufficient recycling in national waste management laws or local government laws/rules, the actual penalty depends on cities and cases. The data for nominal fines (f_m and f_b) could not therefore be retrieved through the questionnaire, which instead asked whether the waste management authority implemented fines for illegal dumping, f_z.

Taking these features of the waste collection procedures into account led to the following reduced form models for recyclable, non-recyclable wastes and recycling ratio:

$$r_{jt} = r\left(p_d, F, p_{rj}, f_z, T, M, K_j, P, Y\right) \tag{21}$$

$$r_t = r(p_d, F, \mathbf{p}_r, f_z, T, C, M, K, P, Y) \tag{22}$$

$$d_t = d(p_d, F, U, Z, \mathbf{p}_r, f_Z, T, M, K, P, Y) \tag{23}$$

$$R_t = R(p_d, F, \mathbf{p}_r, f_z, T, C, M, K, P, Y) \tag{24}$$

In our estimation of these models we tested four functional forms: linear with linear population density, linear with logged population density, log-linear and log-log. By using the method suggested in Wooldridge (2003), the adjusted R^2 for the log-log was compared with models without the logged dependant variable. This selected the linear and log-linear over the log-log models. The results are reported below.

4 Results

Residential recycling, waste disposal and waste generation models (21), (22), (23) and (24) were estimated for eighteen cities over a six year period, using a panel data analysis implemented with Limdep 8. A summary of the results is presented in Tables 5 and 6. The coefficient of p_{rj} is unavailable in the recyclable paper model because no city charges for paper recycling. Since the correlation between p_{rj} for plastics and K_j for plastics is high, K_j in recyclable plastics was also dropped. All models reject the hypothesis that the coefficients are jointly zero at the 99% confidence level. A Breusch and Pagan (1980) test confirmed that the panel data models performed better than non-panel models, while a Hausman Test (Mundlak 1978) favored a fixed-effects over a random-effects model. The effects of stable national and/or city characteristics are reflected in the city-dummies in the fixed effects models.

All significant coefficients of the models in Tables 5 and 6 have the expected signs. The higher the price of non-recyclable collection (the unit price of collection) the higher the recycling ratio, the total quantity of waste recycled, and the quantity of metal, glass and PET bottles recycled. The price elasticity of demand for non-recyclable waste collection services, was derived as follows: $\varepsilon = \frac{d'-d}{d} / \frac{p_d\prime - p_d}{p_d} \cong \beta_{p_d}\left(\frac{\bar{p}_d}{\bar{d}}\right)$. This yielded an estimate of -2424.15 without any interaction term (the non-recyclable model in Table 6). Since $\bar{p}_d$ is 0.024 and $\overline{d}$ is 839.35, the price elasticity of demand for non-recyclable waste collection services at the average price was found to be 0.069. We note that this is a lower elasticity than has been

Table 5 Summary of recycling estimated models (FEM)

	Recycling ratio	Total recyclables	Recyclable paper	Recyclable plastics
p_d^{a}	0.028 (0.010)***	20.49 (10.88)*	10.59 (9.11)	0.70 (1.20)
F^{a}	−0.63 (0.17)***	−574.56 (182.31)***	−246.20 (146.82)*	−11.18 (19.29)
P^{a}	2.02 (0.26)***	1791.07 (277.42)***	1095.60 (231.34)***	111.42 (30.02)***
C	0.019 (0.0074)**	18.44 (8.01)**	na	na
Y^{a}	0.16 (0.15)	193.56 (167.54)	−12.39 (133.04)	−10.43 (16.96)
T	0.028 (0.025)	41.21 (26.79)	62.53 (21.44)***	1.35 (2.87)
M	0.063 (0.023)***	44.40 (25.50)*	3.54 (20.66)	1.60 (2.59)
K_j	−0.0047 (0.013)	11.44 (34.15)	44.80 (21.38)**	13.38 (1.74)***
f_z	−0.046 (0.044)	−47.71 (48.03)	−54.11 (40.14)	−4.71 (5.13)
p_{rj}	−0.11 (0.11)	−91.01 (123.16)	na.	na.
Adj. R^2	0.83	0.81	0.73	0.72
F-sta.	20.39	17.38	12.54	12.17
	Recyclable metal	Recyclable glass	Recyclable PET	
p_d	171.68 (50.71)***	60.64 (21.63)***	40.81 (11.09)***	
F	−16.55 (4.93)***	−0.82 (2.10)	0.44 (1.08)	
P^{a}	114.53 (46.98)**	47.21 (20.04)**	58.05 (10.44)***	
Y	0.0047 (0.0013)***	−0.00029 (0.00056)	0.00053 (0.00027)*	
T	−5.06 (3.82)	−1.32 (1.63)	1.34 (0.85)	
M	12.46 (4.37)***	0.86 (1.86)	3.37 (0.89)***	
K_j	6.22 (5.29)	2.01 (2.26)	1.85 (0.83)**	
f_z	−0.16 (8.42)	0.39 (3.59)	0.57 (1.79)	
p_{rj}	−25.55 (21.33)	−11.29 (9.10)	0.44 (4.67)	
Adj. R^2	0.72	0.92	0.74	
F-sta.	11.64	49.85	12.44	

Standard error in parentheses
*, ** and *** Indicate significance at the 0.1, 0.05 and 0.01 levels, respectively
[a] Indicates that the independent variable is in log form

reported in many studies of cities outside the USA. p_d also decreased total waste generation (source reduction), as shown in the total waste model in Table 6 but not in the model with an interaction term (total waste-1 model). That means the source reduction may only happened in the Taipei cases which adopted both unit pricing and on-time collection.

More important than the impact of the direct costs of disposal options, however, is the impact of indirect—especially time—costs. The effect of unit pricing in many cities is reinforced by a series of ancillary measures. Specific recycling policies include both the frequency of collection, targeted collection and transparent containers. The higher frequency of non-recyclable collection decreased the recycling ratio, total recyclables, recyclable paper and metal, largely by reducing the marginal time cost of this disposal option. Mandatory recycling policies also increased recycling and decreased non-recyclable and total waste generation, but the effect varied between recyclable waste materials. Targeted collections of different recyclable materials had a marked effect on recyclable paper, PET bottles and plastics. The

Table 6 Summary of waste estimated models (FEM)

	Non-recyclable	Non-recyclable-1	Total waste	Total waste-1
p_d	−2424.15 (573.72)***	−1675.41 (718.45)**	−1980.19 (604.70)***	−692.15 (734.19)
F	118.29 (56.20)**	112.42 (55.66)**	54.07 (59.23)	43.97 (56.88)
ln(P)	−3078.23 (522.67)***	−3162.48 (519.08)***	−1183.00 (550.89)**	−1327.94 (530.45)**
Y	0.0048 (0.015)	0.0092 (0.015)	0.012 (0.016)	0.020 (0.015)
T	−82.61 (43.16)*	−69.99 (43.31)	2.75 (45.49)	24.47 (44.26)
M	−264.96 (49.39)***	−260.37 (48.90)***	−223.58 (52.05)***	−215.69 (49.97)***
K	106.24 (95.15)	103.38 (94.08)	150.31 (100.29)	145.39 (96.14)
U	−48.35 (106.79)	−50.53 (105.58)	−51.89 (112.55)	−55.64 (107.89)
f_z	11.41 (98.71)	6.70 (97.63)	−45.35 (104.04)	−53.44 (99.76)
U*Z	na	−186.84 (110.05)*	na	−321.42 (112.46)***
Adj. R^2	0.79	0.80	0.72	0.74
F-statistics	16.95	16.80	11.65	12.51

Standard error in parentheses
*, ** and *** Indicate significance at the 0.1, 0.05 and 0.01 levels, respectively

policy of using transparent garbage bags only increases paper recycling, possibly because paper is most easily visible.

On-time collection itself had no significant effect on waste disposal. To evaluate the effect of enforcement, however, an interaction term (on-time collection with the adoption of unit pricing) was added. The significant negative interaction term and p_d shows that the effect of the unit pricing policy has been strengthened by on-time collection (see the non-recyclable-1 model reported in Table 6). We conjecture that this is because on-time collection can decrease the cost of inspection.

Recycling of all kinds was found to increase with population density. Population density was also found to decrease non-recyclable and total waste generation. There are three potential reasons for these finding. One is that due to the higher marginal cost of space, households in high-density areas may consume less physical products than households of equivalent size in low-density areas—reflecting differences in income, and hence generate less waste. A second reason is that the non-depletable external costs of illegal waste disposal are greater the higher the population density of a city. It follows that, other things being equal, high-density cities would be expected to commit more effort into promoting recycling and waste reduction than low density-cities. A third reason is that the household's marginal cost of illegal disposals may be higher in high-density areas, not just because of the non-depletable externality, but also because the probability of detection may be higher.

Finally, whether or not the authority states that it is willing to impose fines for illegal dumping turned out to be insignificant in all models. We note that since this paper focuses only on household's curbside waste and recycling practices, it does not evaluate the effectiveness of policies that require that residents bring recyclables to specific recycling centers under a deposit-refund system or Extended Producer Responsibility.

5 Discussion

Municipal solid waste is still one of a small class of pollutants that increases monotonically with income. It also increases with population. Indeed, a combination of economic and population growth is historically the main driver behind the growth of both consumption and the generation of waste products. At the same time, economic and population growth also increases the cost of disposals. On the one hand, the storage or illegal disposal of waste materials imposes greater human health costs—by the effect it has on the abundance of disease vectors, the quality of surface water flows, and the like—the more dense the human population. Economic and population growth both strengthen the incentive to address the waste burden in cities.

We used data for 18 high-density cities in Taiwan and Japan to evaluate the effects of waste management policies that typically include an array of measures, including unit pricing, mandatory recycling, the effect of punishing on non-compliance, transparency of waste containers and the frequency of waste and recycling collections. Our results show that unit pricing and mandatory recycling can both significantly decrease waste disposal and increase recycling, but that the effectiveness of both measures depends on a series of ancillary mechanisms, each of which has more or less direct effects on the remaining costs of disposal options. The higher frequency of non-recyclable collection, for example, reduces the time costs of this form of disposal. A transparent garbage bag policy increases paper recycling and decreases disposal to landfills by increasing the likelihood that illegal disposal of this form of waste will be detected. This is incidentally that unit pricing is reinforced by on-time collection, primarily designed to make the street clean. This policy becomes providing a chance for haulers to check the usage of paid bag after unit pricing was adopted.

Increased illegal disposal is potentially one of the costs of unit pricing. We found that illegal dumping did not increase after unit pricing was adopted, whereas the recycling ratio increased significantly. Whether or not waste management authorities declare their willingness to impose fines for illegal dumping turned out to have no significant effect on either disposal to landfill or recycling. Since data on illegal dumping cannot be retrieved from official statistics, the cross price elasticity of demand for this disposal option cannot be estimated.

Ultimately, the critical factor in determining household recycling strategies is the time cost involved in alternative disposal options. Direct charges for disposal—the price of 'legal' garbage bags, or the expected penalty for illegal disposal—are the prime determinant of household decisions on disposal to landfill. However, direct charges do not determine all recycling decisions. These are dominated by the time cost of disposal options. People dislike 'wasting' time in recycling. Moreover, the wealthier a community is, the larger the time cost looms. It follows that the most effective measures reduce the marginal cost of favored disposal options or raise the marginal cost of unfavored options, either directly or through the disposal and recycling collection protocols that affect the time cost of those options.

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