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The Socially Optimal Recycling Rate: Evidence from Japan☆

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Abstract

This paper estimates the average social cost of municipal waste management as a function of the recycling rate. Social costs include all municipal costs and revenues, costs to recycling households to prepare materials estimated with an original method, external disposal costs, and external recycling benefits. Results suggest average social costs are minimized with recycling rates well below observed and mandated levels in Japan. Cost-minimizing municipalities are estimated to recycle less than the optimal rate. These results are robust to changes in the components of social costs, indicating that Japan and perhaps other developed countries may be setting inefficiently high recycling goals.

Keywords

Solid Waste; Recycling; Environmental Policy

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1. Introduction

Recycling municipal solid waste has become increasingly common over the past 25 years. This trend can be attributable largely to government initiatives. Many individual states within the United States, for example, either mandate curbside recycling or set recycling targets. In the European Union, the Packaging Directive of 1994 (amended in 2004 and 2005) has made recycling a national priority in many member countries. The Law for the Promotion of Sorted Collection and Recycling of Containers and Packaging (1997) has had a similar impact in Japan. Such policy measures have directly or indirectly resulted in recycling rates of 34.1% in the United States (US EPA, 2010), 19.44% in Japan (Table 2), and 34% in the EU27 (EEA, 2013).

Are these recycling rates socially desirable? Is more recycling always preferred to less recycling? Or might some countries have gone too far in terms of promoting municipal recycling? Using data from Japan and external cost and benefit estimates available in the literature, this paper first calculates the social cost of managing municipal waste and then estimates the average social cost as a function of the recycling rate. Social total costs are defined as (1) the total municipal budgetary costs to manage both waste and recycling collection and disposal systems less any revenue earned from the sale of the collected recyclable materials, plus (2) the total resource costs incurred by households to prepare material for recycling collection – estimated in this paper using novel methods, plus (3) the total external costs associated with landfilled waste disposal or incineration, minus (4) the external benefits associated with manufacturing final goods comprised of recycled materials rather than virgin materials less the external costs associated with collecting, transporting, and processing the recycled material. Available data coupled with results from existing literatures allow each of these four components of social costs to be estimated and used to calculate average social costs, which are then estimated as a function of the recycling rate in order to estimate the recycling rate that minimizes average social costs in Japan.

Results suggest that after considering all of the economic and the environmental costs and benefits associated with managing waste and recyclable materials, the estimated optimal recycling rate in Japan is 10%. This estimate is robust to changes in aggregated assumptions regarding the magnitude of the three estimated components of social costs listed above. Cost-minimizing municipalities that externalize costs at landfills and manufacturing centers are estimated to recycle only 7%. Although some of the private and/or external costs that influence these estimated recycling rates may be constant across Japan or even the world such as the production of climate changing gasses or other regional pollutants, other private and external costs may vary substantially across municipalities such as the distance to recycling or disposal facilities. Thus, optimal recycling rates are very likely to vary across municipalities. Yet, the results below suggest that observed and required recycling rates seem to exceed the estimated optimal rate by a substantial margins and may therefore call to question aggressive public attempts to increase recycling rates.

The next section of this paper provides a context for understanding the socially optimal recycling by examining current policy and the economics literature. Section 3 then describes separately the four components of the social cost of waste management and the data and processes used to estimate each component. The socially optimal recycling rate under varying assumptions is estimated in Section 4. We vary assumptions regarding the components of external costs in Section 5 to consider the robustness of the main results. The average social cost of recycling specific materials is considered in Section 6. Section 7 discusses similarities and differences between Japan and other developed countries to consider whether results obtained using data from Japan can be generalized. Section 8 discusses other policy implications of the results and concludes the paper.

2. Recycling Policy and the Economics Literature

What percentage of solid waste should be recycled by municipal governments? Is it none? Is it all? Or is some share in the middle optimal? The answer is unlikely zero because private firms have traditionally found recycling profitable for centuries. If, for example, the market price of scrap aluminum is sufficiently high to cover the resource costs necessary to disassemble old tractor trailers to recover the aluminum siding, then the market will certainly do so. The question must therefore be restated. How much of the otherwise unwanted waste material worth less than the cost to collect and process should society recycle? If the disposal of recyclable materials did no harm to the environment or generate other external benefits or costs and if markets are sufficiently competitive, then the answer is zero - free market internalizes all social benefits and costs of recycling and will find the optimal quantity (Baumol, 1977). But waste disposal facilities have been estimated to generate external costs. Landfills may threaten local groundwater quality, neighborhood property values decline, and climate changing gasses escape from both landfills and incinerators (Davies and Doble, 2004). Although recycling facilities also generate their own external costs, using recyclable materials in industrial production rather than their virgin counterparts is often found to reduce energy demand and the emissions of air and water pollutants (Cleary, 2009). The transportation of both waste and recyclable materials generates road congestion, air pollution, and increases the likelihood of vehicle accidents. Given these factors, rather than rely upon the market recycling rate we must wonder what recycling rate is socially optimal once all economic and environmental costs and benefits are considered.[2]

Economists have devoted very little attention to answering this question. Not because the question is not a good one, but perhaps because, given an assumed nature of the external costs of waste disposal and uncertainties in recycling costs, there may be a better one. For the case of solid waste, the better question may not be what rate to recycle but what price to charge for waste disposal.[3] Economic theory suggests that focusing on pollution prices rather than pollution abatement standards such as required rates of recycling is appropriate wherever the external marginal cost of any given pollutant is constant across quantities of pollutants and marginal abatement costs are unknown to policy makers (Field and Field, 2009). Toxic or hazardous waste associated with the disposal of computers, televisions, and cell phones (such as the lead oxide, cadmium, and mercury imbedded in these products) may generate *increasing* external marginal costs - a threshold can be reached where incremental increases in hazardous waste disposal can mix with existing hazardous waste to form new problems for human health and the natural environment (Kahhat, 2009). But solid waste is comparatively

[2] Only the external costs of energy use are included in the external benefits of recycling category. Changes in private energy costs will be reflected in market prices for recycled materials and are therefore included with municipal costs.
[3] Weitzman (1974) provides a nice background into the question of setting prices versus setting quantities.

benign – adding plastic, food waste, and paper to existing piles of these materials increases environmental costs proportionally, but no new medical or ecological threshold is crossed that results in the emergence of some new problematic issue.

If external marginal disposal costs are assumed constant, then economic theory suggests the policymaker acts efficiently by establishing a tax on waste set equal to the constant external marginal cost of waste disposal with no knowledge of the private recycling costs to municipalities or households. Individual households, firms, and municipalities that know their own recycling costs (even as policymakers do not) will recycle until their own rising marginal cost is equal to the after tax price. By choosing the optimal tax, recycling quantities can go where they may, and any resulting recycling rate will be economically efficient. Perhaps for this reason the economics literature has devoted attention to not only estimating the optimal waste tax, but especially to understanding where along the solid waste stream to assess the tax. One option is to implement a per-bag fee for the collection of solid waste at the curb (Porter, 2002). Concerns have risen over the likelihood of illegal dumping and the high administrative costs of assessing the curbside tax (Kinnaman, 2006). With sufficient competition in the waste industry, the tax could instead be implemented at the landfill, thus leaving to the municipality the decision over how best to manage local solid waste.[4] A third option is to subsidize household recycling or implement a deposit-refund program (Palmer and Walls, 1997). Advanced disposal fees have also been considered (Shinkuma, 2007).

Public policy at the municipal level has evolved within this framework by establishing waste taxes. Some municipalities tax waste at the curb by requiring households to purchase special stickers, tags, or bags prior to collection. Municipalities may also offer free access to curbside or drop-off municipal recycling services – essentially subsidizing the household recycling process. The recycling rate that emerges from these taxes and subsidies may of course vary.

[4] Monopolistic competition probably best describes the structure of the waste disposal industry. From a waste generators perspective, the services of waste disposal landfills and incinerators are only differentiated by their distance from the waste generator. Although historically waste disposal services were vertically integrated with waste collection and supplied primarily by the public sector, today in the United States the private waste disposal sector disposes of 78% of all waste – up from 65% in 1992. See Waste Business Journal (2012) for additional details on the status of the private waste disposal industry.

But perhaps recognizing that optimal tax policy may vary across municipalities due to differences in local cost structures and household waste disposal behavior, policy at the federal and state level often involves setting specific recycling-rate targets rather than specific taxes. In the United States, thirty-seven state governments established goals for the recycling rate. California has set a recycling rate goal of 75% by the year 2020. Texas has set a goal of 40%, which can include source reduction. New Jersey and Oregon penalize local governments that do not achieve their recycling goal, while Florida uses grant money as an incentive for municipalities to reach a recycling target. Only Arkansas and Virginia have met their state-imposed recycling goals, while Alabama, Florida, Missouri, Pennsylvania and South Carolina are within 5% of achieving their targeted recycling rate (Simmons et al., 2006). Although several states have also levied landfill disposal taxes, these taxes are frequently well below the estimated per-ton external marginal cost of waste as estimated in the economics literature.

The European Union has followed a similar regulatory approach by managing waste through recycling-rate targets rather than establishing optimal prices (Price, 2001). The Packaging Directive, as amended in 2005, requires all member countries of the European Union to achieve a recycling rate of 80% by 2009. The United Kingdom serves as an interesting case study for the comparison between setting recycling-rate targets and the optimal tax policy approaches. Britain first attempted to set efficient landfill prices via the use of a landfill tax, but soon had to raise the tax to levels well above the estimated external marginal cost of waste disposal when it became apparent that the existing prices would not reduce waste and increase recycling by levels necessary to satisfy the European law. Arbitrary recycling-rate targets set by the European Union appear to have trumped the use of optimal taxation.

The Japanese government also set recycling targets in 1997. Each municipality in Japan was expected to reduce waste generation by 5%, increase the recycling rate from 10% to 24%, and reduce final disposal by 50% by 2010. Each municipality in Japan is required to submit a plan to the Ministry of the Environment outlining plans for obtaining these three goals.[5]

[5] For the details, please see http://www.env.go.jp/guide/seisaku/h16/pdf/1-6.pdf.

Thus, the repeated theme across much of the developed world is the national regulatory reliance on recycling-rate targets rather than setting optimal prices as is argued by economic theory and often employed at the municipal level. If national and state governments are motivated by recycling-rate targets, then the relevant question is what recycling rate is socially optimal. The known economics literature is silent on this question.

The literature has estimated the *municipal* costs of operating recycling programs as a function of recycling quantities, but ignores recycling costs to households and all external costs and benefits associated with recycling, landfills, and incinerators. Using 1992 data from a sample of 57 municipalities in the state of Wisconsin, Carrol (1995) estimate the municipal cost of curbside recycling as functions of several variables including the quantity recycled. The municipality's marginal cost of curbside recycling is estimated to be constant, implying no economies of scale in municipal recycling services. Callan and Thomas (2001) expand upon the work of Carrol by considering a larger data set of 101 municipalities located within the state of Massachusetts and estimate economies of scale in municipal curbside recycling efforts. Bohm et al. (2010) use 1996 data from a random sample of municipalities across all of the United States and estimate a u-shaped marginal cost curve for curbside recycling services. The municipality's marginal cost decreases for quantities up to 4,600 tons per year but rises for quantities above this threshold. The municipal marginal cost of recycling is estimated to exceed the marginal costs of utilizing the landfill. In a related body of literature Criner et al. (1995), Steuteville (1996), and Renkow and Rubin (1998) estimate the costs of municipal composting programs. Criner et al. (1995) estimates composting is worthwhile for landfill disposal fees between $75 and $115 per ton.

This paper extends this literature by focusing on social costs of waste and recycling rather than only municipal costs. The emphasis is on estimating the optimal recycling *rate* (the quantity of material recycled divided by the total quantity of waste material managed by the municipality) rather than the recycling *quantity* for two reasons. First, virtually every known federal and state policy targets municipal rates of recycling rather than recycling quantities. Estimating average social costs as a function of recycling rates may therefore yield results that prove helpful to policy formation.

Second, although recycling quantities vary substantially across the sample and a model estimating average social costs as a function of recycling quantity could yield an optimal recycling quantity, that quantity may prove useless to municipalities of varying sizes and with varying quantities of waste to manage. We believe the more relevant question is; given any exogenously determined level of waste, what percentage should be recycled.

The weakness of focusing upon recycling rates rather than recycling quantities is that, by holding constant the total quantity of municipal waste to manage, the paper does not consider the costs of policies designed to reduce or reuse municipal waste. If municipalities are devoting costly resources to the reduction and reuse of waste materials, then the estimated coefficient on the recycling rate will be biased if source reduction is correlated with the recycling rates. This potential bias disappears if resources devoted to the reduction and reuse of waste (informational ad campaigns, for example) are very low or if such costs are uncorrelated with recycling rates.

3. Deriving the Social Cost of Waste: Data Sources

This section describes the process to derive each of the four broad components that comprise the social costs of managing municipal waste and the data necessary to estimate each component. Data on municipal waste and recycling costs and data employed to estimate household recycling costs originate from Japan. Estimates of the external marginal costs and benefits associated with both waste disposal and recycling are obtained through a survey of the relevant literatures that rely upon data from across the developed world. Definitions of each variable introduced in this section used below in the estimation are provided in Table 1 with statistical summaries appearing in Table 2.

A. Budgetary Costs to the Municipality (Data from Japan)

Data on municipal costs of collecting and managing waste and recyclable materials are rare or non-existent in the United States and many parts of Europe. Folz (1999) mailed surveys to 2,096 municipalities in the United States in 1996, and 1,021 municipalities responded by reporting the cost of their waste and recycling activities. But this data represents only one year, and no data are known to have been gathered across the United States since 1996. Municipal cost data do not exist in the United Kingdom and in many other European countries.

Japan stands out among perhaps every other country in the world in terms of making available for public consumption high quality data on municipal solid waste and recycling programs. Beginning in 1979, the Ministry of Environment in Japan organized a centralized data gathering process whereby each of the 1,700 Japanese municipalities submitted waste management data to its prefecture government (a prefecture government is similar to a state government in the U.S.). Each prefecture compiles and submits the municipal data to the Ministry of the Environment, which then makes the data available for public consumption. This hierarchical data gathering process is used in many areas of Japanese government including employment, agriculture, manufacturing, and education. In recent years the data gathering process has been performed over the internet.[6]

Data are obtained for the 84 largest municipalities in Japan over the six year period spanning 2005 to 2010.[7] The panel nature of this data allow for the use of econometric methods that eliminate potential biases resulting from unobserved municipal variables that may affect recycling rates and costs but do not vary over time.

Municipal waste management costs (*MUNICOST* in Table 1) include the budgetary costs to operate both a municipal solid waste collection and disposal program and a municipal recycling program. Both waste and recycling efforts require labor, trucks, machinery, land, and administrative services. A recycling program could also involve separate curbside collection, drop-off recycling facilities, or both. The disposal of waste materials at landfills or incinerators requires the payment of a tipping fee. Collected recyclable materials need to be stored, processed, and transported to markets that utilize recycled materials. All of these costs are included in municipal cost variable.

Recycled aluminum, metal, paper, glass, and some plastics have economic value as inputs to the production of a variety of manufacturing industries. With sufficient competition among municipalities and private recyclers, the revenue earned from the sale of recyclable materials to the recyclers approximates the economic benefit of providing

[6] For the latest survey result, please see http://www.env.go.jp/recycle/waste_tech/ippan/h22/index.html.

[7] Japan has experienced a period of merging among neighboring small municipal governments making data from small jurisdictions impossible to follow over time. Thus, all municipalities in the sample have populations in excess of 180,000 persons. These municipalities comprise about 40% of Japan's entire population.

recycled materials and offset the municipal government's cost to operate recycling programs. Thus, any economic benefit associated with generating recyclable materials are internalized by municipal governments. But these revenues are not included in the initial municipal cost data. We therefore calculate these revenues by multiplying the quantity of each recyclable material by the market price of each material that year.[8] Annual data on market prices of each recyclable material were obtained via the Ministry of the Environment. Any revenue earned was then subtracted from municipal costs. As seen in Table 2, the municipal cost variable (*MUNICOST*) averages nearly $70 million per year in the sample, averaging $265 per ton to collect, transport, process, and dispose waste and recyclable materials.

Also included in the Japanese data are the tons of waste generated (*WASTE*) in each municipality in the years 2005 through to 2010. Municipal waste varies in the sample between 72,040 and 1,926,718 tons per year. This range is likely explained by differences in the human population (*POP*) across municipalities. Also included in the data are the tons recycled (r), which allows for the calculation of the recycling rate. The recycling rate (*RATE*) varies between 3.90% and 48.33% and averages 19.44%. The wage variable (*WAGE*) is measured as the total payroll paid to the municipal waste management work force divided by the number of waste management employees in the municipality, and averages about $80,350.

B. Recycling Costs to Households (Data from Japan)

Municipal recycling programs require participating households to devote household resources such as time, effort, and storage space to generate recyclable materials. That many municipalities in Japan levy a curbside fee on each bag of waste collected allows for the estimation of these resource costs. Assume that some level of autonomous recycling can be generated by households at no cost ($\bar{r}$). This autonomous recycling could be the consequence of individual norms, social norms, altruism, or direct utility associated with the recycling process (Kinnaman, 2006; Halvorsen, 2008; Koford et al., 2012; and Abbot et al., 2013). Separating, storing, and possibly transporting

[8] Each municipality in Japan faces two options for selling its collected recyclable material. It can participate in live auctions for the materials or contract directly with individual parties.

recyclable materials in excess of this autonomous level requires the household to devote its resources (*k*) according to the production function $r - \bar{r} = f^{-1}(k)$. This function can be inverted to solve for $k = f(r - \bar{r})$, which assume for econometric specification takes on the functional form,

$$k = f(r - \bar{r}) = \frac{1}{2}\delta(r - \bar{r})^2, for\ r \geq \bar{r}$$

where δ is a positive constant.

Households allocate all waste in excess of autonomous recycling $(w - \bar{r})$ between curbside disposal (*g*) and curbside or drop-off recycling (r). Let p_g and p_k denote the household per-bag cost of curbside waste collection and the per-unit cost of the household resource, respectively. Assume the household opportunity cost of time (p_k) does not vary across municipalities within any given year. Also assume the household marginal cost to prepare waste material is constant within the range of waste affected by changes in recycling quantities attributable to the user fee. Thus, only recycling is resource costly on the margin.[9] Assume households choose *g* and *r* to minimize the total cost of removing all waste,[10]

$$\begin{array}{ll} \text{MIN} & p_g g + p_k k \\ \text{subject to} & w - \bar{r} = g + r \\ & k = \frac{1}{2}\delta(r - \bar{r})^2. \end{array}$$

The cost-minimizing quantity of recycling is given by $r^* = \bar{r} + \frac{1}{\delta p_k} p_g$.

Estimates of $\bar{r}$ and $\frac{1}{\delta p_k}$ can be obtained by regressing the per-capita quantity of recycling (*r/POP* = *r**) on a constant and the curbside price of waste collection (p_g – where for ease of interpretation we convert price per bag to price per ton in the regression

[9] If the household marginal cost of waste preparation is positive and constant, then the coefficient used to estimate autonomous recycling is biased in the upward direction.
[10] If households enjoy warm-glow or altruistic benefits from recycling, then recycling up to the autonomous level would result in household benefits. Including these benefits will not affect the optimal recycling rate because they are fixed when adding recycling quantities above the autonomous level. If households continue to gain utility when choosing how much additional recycling to contribute in response to the user fee, then the household recycling marginal cost curve estimated here is interpreted as the net-costs to households –costs minus recycling benefits.

below by assuming an average bag weighs 15 pounds). Data useful to this estimation are once again obtained from Japan. Roughly 20 percent of the municipalities in the data set have implemented a unit-based pricing program for waste collection. The per-bag price varies between \$0.18 and \$0.80 per bag, and the average price charged is \$0.45 per bag. Recall that the fixed-effects model allows for the control of all unobserved variables within each municipality that remain constant across time. These variables could include public tastes for the environment and specific attributes of the recycling program such as frequency of collection. The fixed-effects regression results are summarized in Table 3. Based upon these estimates, the estimated autonomous recycling level ($\hat{\bar{r}}$) is 0.076 tons per person per year, which is equivalent to about 7.40 pounds per week for a household comprised of 2.5 individuals – thus most current household recycling is conducted at no costs to households.[11] The coefficient on p_g suggests a curbside user fee of \$0.80 per bag is estimated to cause a 2.5-person household to increase recycling by 0.74 pounds per week, about a 10% increase over autonomous recycling levels. This extra recycling must be costly to households because, in the absence of the user fee, households choose not to recycle this material. Note that the fixed-effects coefficient on price is statistically significant at the 5% level. If this estimated coefficient on $\frac{1}{\delta p_k}$ is not statistically significant, then the null hypothesis that the coefficient is zero is not rejected, and therefore the marginal cost to households to recycle above the autonomous level of recycling may be infinite.

The recycling cost to all municipal households is given by

$$HHRECCOST = p_k k * POP = p_k \frac{1}{2}\delta(r - \bar{r})^2 * POP$$

By substitution for $\bar{r}$ and $\frac{1}{\delta p_k}$ with the estimates above, we get

[11] The small value of R^2 in this regression suggests that the curbside price plays only a minute role in describing the overall variation in recycling quantities across the sample. Fortunately, the objective of this regression was not to describe variations in recycling quantities but to obtain an unbiased estimate of the effect of garbage price on recycling quantities. This estimate is biased if unobserved variables are correlated with both price and recycling quantity. Recall that the fixed-effects estimator eliminates potential bias from any such variables that are constant across time such as attributes of the recycling program or public tastes for the environment.

$$HHRECCOST = 6{,}963(r - .076)^2 * POP$$

The total cost to households in each municipality is therefore estimated by inserting the observed values of *r* and *POP* for each municipality. Municipalities that recycle less than the estimated level of autonomous recycling are assigned a value of zero for *HHRECCOST*. Summary statistics on *HHRECCOST* are provided in Table 2. Household costs vary across the sample between zero and $61,489,610 per year. The average value of $2,310,502 per year equates to about $4.25 per person per year. Apparently recycling is not a resource intensive activity for many households. At current recycling levels these costs to households amount to only 3.31% of the municipal costs (*MUNICOST*). But due to the assumed increasing marginal household recycling costs, these total costs may become more substantial if recycling rates rise above current levels. To our knowledge, the resource costs to households to recycle have not been estimated in the existing economics literature. Therefore these results cannot be compared to other estimates.

C. *The External Costs of Waste Disposal (Data from Various Countries)*

The components of the external costs of waste disposal include nuisance to neighboring property owners, air pollution and congestion from garbage and recycling trucks, and the emission of climate gasses. In addition to these components, landfill disposal threatens adjacent groundwater supplies, and incineration threatens local air quality and generates ashes that require disposal.

The component of these external costs that has received the most scholarly attention is the disamenity value of landfills. Research utilizing hedonic land pricing models has estimated the reduction in property values near landfills. Department for Environment Food & Rural Affairs (DEFRA, 2004) reviews this extensive literature, most based upon data sets for local housing prices in the United States and Europe, and, using plausible assumptions regarding the quantity of waste disposed, the size of landfills, and the discount rate, estimates a $4.39 per ton nuisance effect to neighboring properties. Davies and Doble (2004) estimate the external marginal cost of waste disposal attributable to climate change emissions and waste transportation externalities are $4.96 per ton of waste disposed. This estimate varies according to whether the landfill is local or remote and whether the landfill captures methane for electricity production. Combining these two sources, as is done by Kinnaman (2006), results in total external costs of about $9 per ton. Nahnan (2011) examine data gathered in South Africa to estimate the external costs associated with climate gas emissions and local disamenities are $16 per ton – but external costs associated with transportation are not considered. This estimate decreases if regional landfills replace local landfills or if the landfill captures methane to generate electricity. Porter (2002) summarizes work by Miranda and Hale (1997) who estimate external costs of waste in the $3 to $15 per ton range in Germany, Sweden, the United Kingdom, and the United States. Dijkgraf and Vollebergh (2004) examine shadow prices associated with landfills complying with environmental standards (assumed to have been implemented efficiently) to estimate external costs at about $30 per ton in the Netherlands. External costs associated with transportation are not included. An OECD working group assumes external marginal costs of between a lower bound of about $12.50 and an upper bound of about $60 (OECD, 2006), but does not explain how these estimates were obtained.

Based on these prior estimates, this study assumes a baseline external marginal cost of $15 per ton for landfill disposal.[12] This value is then doubled to $30 and then halved to $7.50 to learn how sensitive the main result is to changes in assumed external costs of waste. These constant external marginal costs are multiplied by the total quantity of waste landfilled in each municipality to estimate the external total costs of waste disposal in each municipality.

Previous research estimating the external costs of incineration is less prominent than that for landfills. Once again, Porter (2002) summarizes the work of Miranda and Hale (1997) who estimate the external marginal cost of incineration between $5 and $14 per ton in Germany, $7 to $15 per ton in Sweden, $24 to $33 per ton in the United Kingdom, and between $11 and $20 per ton in the United States. Dijkgraaf and Vollebergh (2004) once again examine shadow prices (costs) of complying with (assumed) efficient regulations to estimate external costs at about $24 per ton in the

[12] The arithmetic average of the five point estimates ($3, $9, $15, $16, and $30) is $14.60.

Netherlands. All of these estimates consider air pollution, the disposal of ashes, and the generation of climate gasses but ignore transportation costs. OECD (2006) assumes these costs fall within the range of about $43 and $83 per ton but once again does not explain the details.

Based on these prior estimates, this study assumes external marginal costs of $30 per ton for incineration, and then considers for sensitivity analysis values of $15 and $60 per ton.[13] These amounts are multiplied by the quantity of waste incinerated in each municipality to estimate external total costs of incineration. Estimated external total costs of waste disposal (*ECWASTE*) are calculated by adding the external costs associated with both landfill disposal and incinerations and are summarized in Table 2. Using the baseline assumptions, these external costs average $6,168,214 per year that, in relative terms, amount to about 8.82% of municipal costs.

D. *The External Benefits of Recycling (Data from Various Countries)*

Although municipal recycling programs generate external costs through the use of collection trucks, the processing of various materials, the transportation of recyclables to potentially distant markets, and the manufacturing of final goods with recycled materials, recycling also generate external benefits as it reduces the extraction of virgin materials and associated external costs of producing final goods with those virgin materials. Life-cycle analyses often conducted by engineers, geographers, and occasionally economists consider each of these various factors to estimate the overall environmental consequences of municipal recycling programs.[14] Life-cycle models categorize the environmental impacts into those that affect climate change, human health, and acidification and eutrophication of the natural environment. The various pollutants that contribute to each environmental impact are combined into a single functional unit. For example, the various pollutants contributing to climate change can be expressed in terms of 'CO_2 equivalents' allowing the climate impact from various pollutants to be expressed solely in

[13] The arithmetic average of these estimates is just over $25 but includes estimates as high as $83. We use $30 as a base to help capture this range of estimates.
[14] Life-cycle analyses focus entirely on these primary sources of pollutants. Secondary sources of pollutants associated with the production of capital goods required for recycling, such as collection trucks and processing facilities, are often ignored because, when spread over the lifetime of the capital unit, these secondary emissions are very small in comparison to primary pollutants.

terms of CO2. Acidification is often reported in units of sulfur dioxide (SO2) equivalents, nitrification in units of nitrogen oxides (NOx) equivalents, and human health in units of fine particulate matter ($PM_{2.5}$) or in Disability Adjusted Life-Years (DALY) equivalents. Each recyclable material will have its own life-cycle signature as manufacturing processes vary. The overall emissions are then found by weighing each recyclable material according to its portion of the total recycling stream.

Cleary (2009) provides a nice gateway into this literature by surveying twenty peer reviewed and published life-cycle analyses associated with municipal waste management. The vast majority of these papers focus upon the recycling program of a single municipality or region, and all but three of these studies take place in Europe and Asia (with two in the United States and one in Brazil). In most cases, emissions, effluents, and energy use from various sub-processes are imputed using existing emission data bases. For example, the life-cycle researcher will estimate the distance driven to a specific recycling center, and then appeal to existing emissions database to learn the emissions associated with each mile driven. These emission databases are almost always constructed in locations far removed from the recycling program under analysis – only occasionally will life-cycle researchers measure actual emissions. Although most existing studies reviewed by Cleary (2009) compare the life-cycle impacts of landfilling versus incineration, a few also examine municipal recycling systems. Results vary widely across these studies due to differences in the list of pollutants considered, the range of recycling activities considered, and the emissions data sets employed.

Morris (2005) estimates the lifecycle impacts of a municipal recycling program in San Louis Obispo County, California. The collection, processing, and transportation of recycled materials and the subsequent manufacture of products with the recycled materials is estimated to generate emissions of CO2-equivalents, SO2-equivalents, NOx-equivalents, Toluene-equivalents (a human toxin), and reduces DALY's. But these added emissions are offset via reductions in the mining of virgin materials and subsequent manufacturing with those virgin materials. Per ton of material recycled, the overall emission reductions are estimates at 4,500 pounds of CO2, 10.94 pounds of SO2, and 0.25 pounds of NOx. This paper is the most comprehensive in the literature in terms of estimating a broad category of environmental impacts.

A report published by the Japanese Ministry of the Environment (JMOE, 2005) estimates that the overall effect of recycling one ton of recyclable material reduces CO2-equivalents by 1,595 pounds, reduces SO2-equivalents by 5.58 pounds, and reduces NOx-equivalents by 5.19 pounds. Craighill and Powell (1996) estimate the life-cycle emissions associated with recycling in Milton Keynes in the United Kingdom. Each ton recycled is estimated to reduce CO2 by about 1,430 pounds, SO2 by 148.5 pounds and NOx by 46.91 pounds. Erickson et al. (2005) relies on municipal data in Sweden to examine two recycling systems – one that recycles 70% of household HDPE plastic and nothing else and another that recycles 70% of household cardboard and nothing else. The recycling of one ton of plastic is estimated to reduce overall CO2-equivalents by 161 pounds, SO2-equivalents by 0.12 pounds, and NOx-equivalents by 6.34 pounds. Results for cardboard are similar. Other life-cycle studies of municipal recycling programs limit the analysis to a single pollutant such as CO2.[15]

Comparing the three complete sets of results above reveals the extent of variation across the results of life-cycle analyses. A portion of this variation could be attributable to real differences across recycling programs with respect to transportation differences, materials collected, and assumptions made in relation to avoided manufacturing costs. But differences might also be attributable to heterogeneous research methods, and Clearly (2009) suggests a greater degree of uniformity in life-cycle practices and assumptions than has been practiced by the existing life-cycle literature devoted to municipal solid waste.

With the exception of Craighill and Powell (1996), all life-cycle analyses reviewed above, often conducted by civil engineers and geologists, are not monetized and are therefore not directly useful to the estimation of the social costs of managing waste. The monetization process involves dollar amounts placed on each unit of CO2, SO2, and NOx. These dollar amounts are estimated by an entirely different literature. For CO2,

[15] Finnveden et al. (2005) examines data in Sweden to estimate that CO2 emissions drop by about 2,867 pounds for each ton of newsprint recycled, 4,851 pounds for each ton of PET recycled, and 1,102 pounds for food waste. The entire recycling system reduces CO2 emissions by about 1,000 pounds per ton recycled. Acuff and Kaffine (2013) compares optimal waste policy when carbon dioxide emissions are considered with those same policies that do not consider reductions in carbon dioxide as a benefit of recycling.

the United States government established external cost per-ton estimates of $5, $21, and $35, for discount rates of 5%, 3%, and 2.5% respectively, for use in the cost-benefit analyses of all federal regulations (Interagency Working Group on the Social Cost of Carbon, 2010). We select the middle estimate ($21) for this study.

To gain estimates for the external costs of SO2 and NOx emissions, we rely upon three studies. First, Muller and Mendelsohn (2009) estimates the external marginal cost of various air pollutants including SO2 and NOx. Damages arise from impaired visibility, depreciation of man-made materials, illness, recreation, and timber and agriculture yields (although damages to human health are estimated to comprise 95% of all damages). Using $2 million as the value of a statistical life (after a review of the hedonic wage literature), the estimated marginal damages are $1,310 and $260 for SO2 and NOx, respectively. These amounts are estimated to vary across municipalities owing to differences in atmospheric conditions, the composition of the air pollutants, and the height of emitting smoke stacks in urban areas.

Second, Fann et al. (2009) assumes the value of a statistical life is $6.2 million. Perhaps owing to this assumed difference, the external marginal damages associated with pollution emissions are estimated at levels roughly one hundred times higher than in Muller and Mendelsohn (2009). For example, the external marginal cost of one ton of SO2 is estimated at $87,000 and one ton of NOx at $14,000. Fann et al. (2009) also estimates wide differences across municipalities owing to variation in populations, stack heights, stack temperatures, the velocity of emissions, and the chemical processes that govern the formation of air pollution.

Third, Holland and Watkiss (2002) develop external marginal cost estimates for the European Commission. Although the assumed value of a statistical life is only one million euros, the estimated external marginal costs are also much higher than reported by Muller and Mendelsohn (2009). For a city of 1,000,000 persons, external marginal cost estimates are $60,750 per ton of SO2 and $5,670 for NOx.

Applying these three external marginal cost estimates of SO2 and NOx to each of the three complete life-cycle analyses reviewed above leads to nine separate estimates of the external marginal benefit of recycling. These nine estimates are summarized in Table 4. Note that for CO2 we use the estimation result published by the United States

Federal government (Interagency Working Group on the Social Cost of Carbon, 2010). Recall that Craighill and Powell (1996) monetizes their own results and estimate a ton of recyclable material generates $516 in external benefits, which is also included in Table 4 as a tenth estimate.

With no justifiable reason for favoring one source over any of the others, this study will attempt to capture the range of possible external recycling benefits by first considering the mean value of these ten estimates (about $200). We then increase the value to $400, the approximate average of the five largest values in Table 4. We then consider a value of $25, which represents the approximate mean of the five lowest values. Multiplying the baseline estimate of $200 by the quantity recycled in each municipality in each year provides one estimate of the total external benefit of recycling, also summarized in Table 2 (*EBREC*). External benefits of recycling average $9,092,434 per year across municipalities in the sample. On average these benefits amount to about 13% of municipal costs. The external recycling benefits calculated by assuming $25 per ton estimate comprise only 1.6% of municipal costs.

Table 5 provides the means of each of these four components of social costs under each assumption made to test for sensitivity analysis. The external benefits of recycling exceed the external costs of waste disposal in two of the three scenarios. Making this observation particularly interesting is that external waste costs are cited by nearly the entire economics literature as the primary source of the market failure in solid waste markets. The external benefits of recycling are rarely mentioned even though their magnitude could be larger than that of external waste costs. Perhaps external disposal costs at landfills and incinerators are intuitively linked to municipal solid waste whereas air and water pollution surrounding manufacturing regions are linked to overall production in the economy – making them easy to overlook when studying optimal local solid waste policy.

4. Estimating Average Social Costs as a Function of the Recycling Rate

Based on the data described in the last section, the social cost of waste management (*SC*) is calculated by first summing the municipal costs of waste management, the costs to recycling households, and the external total costs of waste disposal, and then subtracting the external benefits of recycling. The average social cost (*ASC = SC/WASTE*) is summarized in Table 2 using the \$200/ton assumption for the external recycling benefit and averages \$260.29 per ton. A flexible functional form econometric model is utilized to estimate how the average social costs of waste management are affected by changes in the recycling rate (*REC_RATE*).

$$\ln(ASC)_{it} = \beta_1 + \beta_2 \ln(REC\_RATE_{it}) + \beta_3 ln^2(REC\_RATE_{it}) \\ +\beta_4 NUMB_{it} + \beta_5 \ln(WAGE_{it}) + \beta_6 \ln(FUEL_{it}) + \alpha_i + \lambda_t + u_{it}$$

where α_i is an unobserved time-invariant municipal effect such as the prevailing tastes for the environment, population density, attributes of local recycling programs, and geological constraints on waste management disposal practices and λ_t is an unobserved time-effect that is held constant across municipalities such as the interest rate and other macroeconomic conditions, and u_{it}, the error term, includes any measurement error in the dependent variables as well as any unobserved variables that vary across time and across municipalities such as household income and demographic characteristics. We assume u_{it} has mean zero and constant variance, σ^2. The estimation controls for changes in the number of materials recycled (*NUMB*), the wage rate paid to waste collectors (*WAGE*), and the cost of fuel (*FUEL*)[16]. The econometric equation above is estimated via the fixed-effects model using the within transformation.

Results are reported in Table 6 for each assumed level of external benefits of recycling. We will focus first upon the results when external benefits of recycling are assumed equal to \$200 per ton (the middle column of Table 6). The estimated coefficients on the two recycling rate variables ($\hat{\beta}_2 and\ \hat{\beta}_3$) are both statistically significant. In combination, these two estimates suggest a u-shaped relationship between the recycling rate and the average social costs of managing waste, where other variables are held constant. Figure 1 illustrates this relationship, where all other variables in the model are held constant at their sample mean levels. Increases in the recycling rate

[16] Capital costs in Japan, as captured by the interest rate, did not change over the years in the sample, Nakada and Adachi (2006) state that "Looking back over previous research, we find that the composition of regions with high and low loan interest has been fixed over the last 20 years, at least." Thus the municipal-specific fixed effects constants control for these changes in capital costs.

initially decrease the average social cost of waste management from an estimated $455 per ton with no recycling to a minimum of $233 per ton that corresponds with a socially optimal recycling rate of 10%. The social savings, $223 per ton, amounts to an average of $98 per person per year. A family of four individuals would save $392 in social costs by increasing the recycling rate from just above zero to 10%. The distribution of these gains in social costs may vary sharply among recycling households, municipal governments, and individuals that value the quality of the natural environment as recycling rates change. Any recycling above 10% is estimated to slightly increase the average social cost of managing waste. The average social cost of recycling 48%, the highest recycling rate observed in the sample, is estimated at $324/ton, about 39% more than the average social costs of recycling 10%.

To understand what might be guiding the independent recycling decisions of municipal governments, it may prove helpful to first know what recycling rate minimizes the costs that municipal governments internalize. These costs include the budgetary costs of managing both forms of waste and household recycling costs internalized via local political pressures from citizens. External costs accruing around remote landfills or external benefits accruing in manufacturing centers are not likely internalized by municipal governments and might therefore be ignored. The natural log of the per-ton value of these costs (*MUNICOST + HHRECCOST*) was regressed on the same set of independent variables above. Fixed-effects regression results are reported in Table 7. The estimated coefficients on the two recycling rate variables ($\hat{\beta}_2 and\ \hat{\beta}_3$) are both statistically significant at the 1% level. In combination, these two estimates generate a u-shaped relationship between the recycling rate and the average municipal costs of managing municipal waste, where other variables are held constant at their sample mean levels (also illustrated in Figure 1). The recycling rate that minimizes internalized costs of managing waste, and therefore that rate municipalities might choose if guided only by internalized costs, is estimated to be 7%. This recycling rate may be obtainable by municipal governments simply providing drop-off facilities for households to bring various materials. Left to their own decisions, cost-minimizing municipal governments are estimated to recycle less than the optimal recycling rate (10%) and the mandated recycling rate in Japan (20%) suggesting that national policies may be required to increase municipal recycling rates.

5. Altering the Magnitude of the Three Components of External Costs

Household recycling costs, external costs of waste disposal, and external benefits of recycling were all derived using results from the existing literature or, in the case of household recycling costs, assumptions about the functional form of the household recycling production function. To determine how robust the main results are to changes in these three measures, the model was estimated again after varying the magnitudes of these three components of social costs. Household recycling costs and external waste costs were separately doubled and then halved. External recycling benefits were lowered to \$25 per ton and increased to \$400 per ton to characterize the literature summarized in Table 4. Each of these six redefined measures of average social cost was regressed on the same set of variables described above. Estimated coefficients from these regressions were then used to generate new predicted relationships between average social costs and the recycling rate. Results from these additional estimations are illustrated in Figures 2, 3, and 4, where once again other variables in the model are held constant at their mean levels. The estimated optimal recycling rate for each definition of social costs is summarized in Table 8.

a. Average Social Costs with Varying External Recycling Benefits

Recall from the information in Table 4 that the external benefits of recycling, as based on estimates of the existing literature, could be as low as \$2.43 per ton and as high as \$525.83 per ton. The results above assumed recycling benefits of \$200 per ton. To characterize the existing literature, the model is re-estimated assuming the external marginal benefit of recycling a ton of material is \$25 and then \$400.[17] Results are reported in the first and third columns of Table 9 and the predicted average social costs curves with respect to the recycling rate are illustrated in Figure 2.

When external benefits are \$25 per ton (first column of Table 9), the estimated optimal recycling rate is 8% - essentially the private cost-minimizing rate chosen by

[17] Results based on any assumed external recycling benefit of less than \$25 are very similar.

municipalities and probably accomplished with drop-off recycling programs. Any recycling above 8% is estimated to sharply increase the social total cost of managing waste. The average social cost of recycling 48% of waste, the highest recycling rate observed in the sample, is 66% higher than the average social cost of recycling the optimal 8% of waste. The estimated optimal recycling rate increases modestly to 17% when the external marginal benefit of recycling is increased to $400 per ton. Although changing the magnitude of the external benefits of recycling has a greater effect on the optimal recycling rate than changing the other components of social costs below, the optimal recycling rate is still below both the observed and required recycling rates in Japan.

b. *Average Social Costs with Varying Household Recycling Costs*

As illustrated in Figure 3, varying the household recycling costs up or down by a factor of 2 does not appreciably change the estimated relationship between the recycling rate and average social costs at low levels of recycling (recall that the threshold quantity of recycling is free to households). But for recycling rates above 25% the assumptions over the costs to households begins to matter. If costs to households are doubled, then the average social cost of waste management rises rather sharply with increases in the recycling rate. But if costs to households are half of our original estimates, then the average social costs remain relatively flat across the spectrum of possible recycling rates. The optimal recycling rate remains close to 10% for all three cases.

c. *Average Social Costs with Varying External Waste Costs*

Figure 4 illustrates how changes in the magnitude of external costs associated with waste disposed at landfills or incinerators affects the estimated relationship between average social costs and the recycling rate. Recall that a review of the literature suggests that these costs were assumed at $15 per ton for landfilling and $30 for incineration. But the literature also suggests a range of values. This range is accommodated by first doubling and then halving these two values to yield two new definitions of the social costs of waste management. Once again, results suggest that such changes make only modest changes to the estimated relationship between average social costs and the recycling rate. As illustrated in Figure 4, the overall shape of the relationship seems rather robust to changes in external disposal costs. The optimal recycling rate remains within 10% of the base result. Thus, variations in the external costs associated with waste disposal and incineration do not appear important to shaping optimal recycling decisions. These results are rather interesting because historically the external costs of waste disposal appeared to have been the driving force for implementing municipal recycling programs.

Finally, assume a perfect storm of low recycling costs to households, high external waste costs, and high external recycling benefits. Regressing the natural log of the resulting average social costs on the recycling rate yields a slightly decreasing average social cost curve throughout all recycling rates (Figure 5), but the coefficient on the natural log of the recycling rate and its squared term are both statistically insignificant (t-statistics of -0.18 and -0.38, respectively). Thus, the average social cost curve is essentially flat, indicating all recycling rates within the range of the sample (0 to 48%) may be equally costly.

6. Average Social Costs and the Recycling of Specific Materials

The data also allow for the estimation of how the recycling of each specific material affects the average social costs of waste management. Each municipality in Japan reports the quantity recycled of each of six categories of materials (metal, paper, glass, PET plastic, other plastics, and other materials). To remain consistent with econometric specification above, we divided the quantity of each recycled material by the total amount of municipal waste to create a series of recycling rate variables for each material. These variables define the percentage of all waste that is recycled as each specific material. Municipalities in Japan, for example, recycle as much as 2% of overall waste in the form of PET plastic, whereas metal recycling can comprise as much as 8% of all waste in the sample. The model defined above was altered by first eliminating the two broad “recycling rate” variables and then adding variables (log and log squared) for the portion of waste material recycled as paper, metal, glass, PET plastic, other plastics, and other materials. Because natural logs are used, municipalities with zero levels of recycling any of the materials were dropped from the data.

Regression results appear in Table 10. The recycling of just metal, just glass and just plastic is estimated to have no statistical effect on average social costs. The recycling of just paper is estimated to decrease average social costs – both recycling rate variables are significant at the 10% level. The recycling of PET plastic is estimated to increase average social costs with statistical significance at the 1% level. The recycling of other materials is also estimated to increase average social costs based on the significant squared term.

Holding constant the quantity of other recycled materials at their mean levels, Figure 6 illustrates how the average social costs change with the recycling rate of the three specific materials with significant estimated coefficients (paper, PET, and other materials). The unit of measurement along the horizontal axis is the ratio of the quantity of each material recycled and the total waste. The length of each best-fit line reflects the range of observed recycling rates in the sample. No municipality in Japan, for example, recycles more than 2% of its overall waste in the form of PET plastic, whereas paper recycling can comprise as much as 20% of all waste in the sample. The estimated best fit lines for the three materials with insignificant estimated coefficients (metal, glass, and plastic) may be flat. Increasing the rate of recycling these three materials may have no measurable impact on social costs. Results suggest municipalities interested in decreasing average social costs of waste management should focus policy on increasing the specific recycling of paper but reducing the recycling of PET plastic and other materials.

7. Generalizing Results to Other Countries

Recall that the data used to estimate the optimal recycling rate were obtained from Japan, chosen over other countries because of the availability of panel municipal cost data. Although optimal recycling rates can be expected to vary across countries with differences in tastes, technologies, and the availability of economic resources, of interest is whether the estimated optimal recycling rate estimated for Japan might be relevant to understanding optimal recycling policy in other developed countries. Towards this end, this section provides basic comparisons of Japanese waste management practices with those of other developed countries.

How typical is solid waste management in Japan in relation to other developed countries? A comparison of a few common statistics of a few large cities provides some indication. In terms of waste generation, residents of Tokyo generate 1.1 kilograms of waste per day. This amount is similar to London (1.2 kg/day) but less than New York (2.0 kg/day). Collection costs may be linked to population densities. The population density in Tokyo, at 4,750 persons per square kilometer, again resembles London (5,100) more than New York (2,050) or Rome (2,950). Although the average wage rate for a waste collector in the Japanese sample is about $80,000, the minimum wage rate is readily observed in many developed countries and may serve as a proxy for the wages paid to some low-skilled waste collectors and processors. The minimum wage in Japan is $10.11 per hour, which is similar to London's "living wage" ($11.21) and to a lesser extent New York ($7.25), but is much higher than the minimum wage rate in Rome ($2.90). Finally, the price of petrol in Tokyo ($7.19/gallon) is similar to that in London ($8.17) and Rome ($9.19), which are all much higher than the price in New York ($3.11). Thus, in these respects Tokyo is similar to other large cities in developed countries, especially London. If the costs incurred to collect, process, and transport waste and recyclable materials are based on these factors, then municipal costs in Tokyo and these other cities may be similar. One difference between Tokyo and these three other cities is that Tokyo relies almost exclusively on incineration for waste disposal. The use of landfill disposal is more common in New York, London, and Rome. Private costs to operate incinerators tend to exceed slightly the private costs of operating landfills, although tipping fees paid by municipal governments are roughly equal.[18]

Matsuto and Ham (1990) provides another snapshot of the differences and similarities between Tokyo and the United States. This study compared the content of waste generated by a sample of households in Madison, Wisconsin in the United States and the municipality of Sapporo in Japan. Although these two municipalities may not be representative of their respective national populations, these comparisons provide some insight into waste management practices across the two countries. For example, the average individual in Madison generated 1,016.4 grams of waste per day and the average

[18] See http://www.wastebusinessjournal.com/wbjpriceindex.htm (accessed 4/20/13) for a detailed comparison of tipping fees for both landfills and incinerators in the United States.

individual in Sapporo generated 866 grams – about a 17% difference. The average individual in Madison recycled 22% of waste, compared to 21% in Sapporo. The average individual in the Madison generated slightly more paper, metal, and slightly less glass, textiles, and food waste when compared to the average individual in Sapporo. The quantity of plastics and bulky waste are estimated to be about equal across the two municipalities. Although disposal patterns are not identical, these data may suggest the constitution of the recycling stream may be similar in Japan and the United States.

Of course the optimal recycling rate can be directly estimated in any developed country with available municipal-level data. Municipal costs should include all costs to collect, process, transport, and dispose all waste and recyclable materials (less revenue gained from the sale of such material). Costs to recycling households can be estimated if some municipalities in the sample have implemented unit-based pricing programs for waste collection as was done in Section 3b above.[19] Then the external cost and benefit measures estimated above can be applied to the quantities in landfilled, incinerated, and recycled to estimate average social costs for each municipality. Problems may arise if municipalities use municipal-owned disposal facilities and therefore do not pay market tipping fees for disposal. In these cases data on the costs of operating landfills, including the depreciating value of the land, would need to be obtained.

8. Conclusions and Policy Implications

Although the previous literature has focused upon only municipal costs to recycle, this paper estimated the average social cost of waste management in Japan as a function of the recycling rate. The social costs of waste management include all municipal costs and revenues, costs to recycling households, external disposal costs and external benefits of recycling. Results suggest the recycling rate that minimizes these average social costs is 10%. This result is rather robust to individual changes to household recycling costs,

[19] Developing countries that rely upon open dumping disposal practices or incomplete incineration would likely have very different external costs and benefits of recycling than estimated by the life-cycle literature.

external disposal costs, and external benefits of recycling. The collective reduction to household recycling costs with increases in the external costs of both waste and recycling yields a statistically flat average social cost curve suggesting all recycling rates are equally costly.

Based upon the main result of this paper, it appears that the 20% recycling rate in Japan is higher than the socially optimal rate. If results of this paper can be extended to other parts of the developed world, then current recycling rates in the United States (35%) and the EU27 (34%) may also be too high. That recent increases in the recycling rate in many developed countries have slowed relative to historical growth rates might be suggestive of rising average social costs estimated in this paper. For example, after the recycling rate in the United States grew from 10.1% in 1985 to 28.6% in 2000, the past decade has seen the recycling rate grow to only 34.1%. The recycling rate in Japan was roughly the same in 2007 (20.5%) as it was in 2002 (19.9%).

But recent policy has also promoted large increases in recycling. Caroline Spelman, the Secretary of State for Communities and Local Government (a cabinet position within the United Kingdom government) has asked each local council to develop a strategy to achieve zero residential solid waste. Scotland has approved a national goal to become the world's first zero-waste country by 2025. If these goals are to be achieved via additional recycling, then they may not be socially efficient.

One important source of uncertainty in these results is the assumed external marginal costs of various air pollutants necessary to monetize the life-cycle impacts of recycling programs. Recall that Fann et al. (2009) and Holland and Watkiss (2002) estimates the external costs associated with the emissions of SO2 and NOx at levels that are generally 50 to 100 times larger than those estimated by Muller and Mendelsohn (2009). The optimal recycling rate may hang in the balance of this debate. Life-cycle models employed to estimate the impacts of recycling also vary, and optimal recycling rates may be better understood as the life-cycle literature continues to mature.

How might very large recycling rates be socially optimal? Recall that average social costs do not increase appreciably with the recycling rate when costs to recycling households were low (see Figure 3). These costs would fall with the emergence of single stream waste collection systems. Technologies that allow for automated separation of recyclable materials from ordinary waste would spare households the costs of separating recyclable materials and household recycling costs would disappear (although these technologies would likely increase municipal costs). Very large recycling rates may also be optimal if external benefits of recycling are estimated in the range of $1,000 per ton.

Another policy approach would be to abandon recycling-rate targets entirely and instead target waste and recycling *prices*, as is suggested by economic theory and discussed in Section 2 above. Setting state or national waste taxes at roughly $15 (for landfill disposal) and $30 (for incineration) as well as $200 per-ton subsidies for recyclable materials would result in municipal governments internalizing all benefits and costs of their waste management decisions. If instead these taxes and subsidies were applied at the curb, then households paying garbage taxes of $0.15 (for landfill disposal) or $0.30 (for incineration) for a 20-pound bag of waste and receiving a subsidy of $2.00 for each 20-pound bag of recyclable materials would make efficient disposal decisions.

References

Abbott, A., Nandeibam, S., and O'Shea L., 2013, 'Recycling: social norms and warm-glow revisited', Ecological Economics, 90: 11-18.

Acuff, K. and D.T. Kaffine. 2013. 'Greenhouse Gas Emissions, Waste and Recycling Policy.' *Journal of Environmental Economics and Management*, 65, pp. 74-86.

Baumol, William J. 1977. 'On Recycling as a Moot Environmental Issue.' *Journal of Environmental Economics and Management*, 4(1), March, pp. 83-87.

Bohm, Robert A., David H. Folz, Thomas C. Kinnaman and Michael J. Podolsky. 2010. "The Costs of Municipal Curbside Recycling and Waste Collection." *Resources, Conservation, and Recycling*. 54(11): September, pp. 864-871.

Callan, S. and J. Thomas. 2001. "Economies of Scale and Scope: A Cost Analysis of Municipal Solid Waste Services", *Land Economics*, 77(4): 548-560.

Carroll, W. (1995) "The Organization and Efficiency of Residential Recycling Services", *Eastern Economic Journal*, 21(2): 215-225.

Cleary, Julian. 2009. 'Life cycle assessments of municipal solid waste management systems: A comparative analysis of selected peer-reviewed literature.' *Environment International.* 35(8): November, pp. 1256-1266.

Craighill, Amelia L. and Jane C. Powell. 1996. 'Lifecycle Assessment and Economic Evaluation of Recycling: A Case Study.' *Resources, Conservation, and Recycling*. 17: pp. 75-96.

Criner, G., A. Kezis, and J. O'Connor (1995) "Regional Composting of Waste Paper and Food", *BioCycle*, 36(1): 66-67.

Davies, Bob and Michael Doble. 2004. 'The Development and Implementation of a Landfill Tax in the UK', in Addressing the Economics of Waste, Paris, France: OECD, 63-80.

DEFRA. 2004. 'Valuation of the External Costs and Benefits to Health and Environment of Waste Management Options,' Final Report for DEFRA by Enviros Consulting Limited in Association with EFTEC (http://webarchive.nationalarchives.gov.uk/20130123162956/http:/www.defra.gov.uk/environment/waste/research/health/pdf/costbenefit-valuation.pdf).

Dijkgraf, E. and H.R.J. Vollebergh. 2004. '*Burn or bury?: A Social Cost Comparison of Final Waste Disposal Methods*.' Ecological Economics. 50: pp. 233-247.

Eriksson O, Carlsson Reich M, Frostell B, Björklund A, Assefa G, Sundqvist J-O, et al. Municipal solid waste management from a systems perspective. J Clean Prod 2005;13:241–52.

European Environment Agency (EEA). 2013. ‘Managing Municipal Solid Waste – A Review of Achievements in 32 European Countries. http://www.eea.europa.eu/publications/managing-municipal-solid-waste (accessed 4/19/13).

Fann, Neal, Charles M. Fulcher, and Bryan Hubbell. 2009. “The influence of Location, Source, and Emission Type in Estimates of the Human Health Benefits of Reducing a Ton of Air Pollution.” *Atmosphere and Health,* 2(3): 169-176.

Field, Barry C. and Martha K. Field (2009) Environmental Economics, An Introduction, 5th Edition. New York: McGraw-Hill, Irwin:

Finnveden G, Johansson J, Lind P, Moberg A. Life cycle assessment of energy from solid waste — part 1: general methodology and results. J Clean Prod 2005;13:213–29.

Folz, D. H. 1999. “Recycling Policy and Performance: Trends in Participation, Diversion, and Costs”, *Public Works Management and Policy*, 4(2): 131-142.

Halvorsen, B. 2008. ‘Effects of Norms and Opportunity Cost of Time on Household Recycling’. *Land Economics* 84, 501-16.

Holland, Mike and Paul Watkiss. 2002. ‘Benefits Table Database: Estimates of the Marginal External Costs of Air Pollution in Europe – Version E1.02a.’ Created for European Commission DG Environment by Netcen.

Interagency Working Group on the Social Cost of Carbon, U. S. G. (2010). Appendix 15a. Social cost of carbon for regulatory impact analysis under Executive Order 12866. http://go.usa.gov/3fH.

Japanese Ministry of the Environment (JMOE). 2005. ‘LCA Report on Recycling of Containers and Packaging (in Japanese) http://www.env.go.jp/recycle/c_3_report/pdf/h16_lca_chousa_honpen.pdf

Kahhat, R., J. Kim, M. Xu, B. Allenby, E. Williams and P. Zhang (2008) Exploring E-Waste Management Systems in the United States’, *Resources, Conservation and Recycling* 52: 955–964.

Kinnaman, Thomas C. 2006. ‘Examining the Justification for Residential Recycling.’ *The Journal of Economic Perspectives.* 20(4), fall, pp.219-232.

Koford, B.C., Blomquist, G.C., Hardesty, D.M., and Troske K.R., 2012, 'Estimating consumer willingness to supply and willingness to pay for curbside recycling', Land Economics, 88, 745-763.

Matsuto, Toshihiko and Robert K. Ham. 1990. 'Residential Solid Waste Generation and Recycling in the U.S.A. and Japan', *Waste Management & Research.* **8**, pp**.** 229-242.

Miranda, Marie Lynn and Brack Hale. 1997. ' Waste Not, Want Not: The Private and Social Costs of Waste-to-Energy Production.' *Energy Policy.* 25(6), pp. 587-600.

Morris J. 2005. 'Comparative LCAs For Curbside Recycling Versus Either Landfilling or Incineration with Energy Recovery.' *International Journal of Life Cycle Assessment*. 10(4):273–84.

Muller, Nicholas Z. and Robert Mendelsohn. 2009. 'Efficient Pollution Regulation: Getting the Prices Right.' *American Economic Review*. 99(5): pp. 1714-1739.

Nahman, Anton. 2011. 'Pricing Landfill Externalities: Emissions and Disamenity Costs in Cape Town, South Africa.' *Waste Management.* 31, pp. 2045-56.

Nakada, M. and S. Adachi. 2006. 'Why has regional disparity in loan interest not been resolved? – Empirical analysis using panel data from second regional banks and credit associations.' *Financial Review.* 86, pp. 161-193 (in Japanese).

OECD Working Group on Waste Prevention and Re-cycling.2006. 'Impacts of Unit-based Waste Collection Charges.' OECD Papers, 6(8):58 –214

Palmer, Karen and Margaret Walls. 1997. 'Optimal Policies for Solid Waste: An Evaluation of Policies.' *Journal of Public Economics.* 65(2), August, pp. 193-205.

Porter, Richard C (2002) 'The Economics of Waste', Washington DC: Resources for the Future.

Price, Jane L. 2001.'The Landfill Directive and the Challenge Ahead: Demands and Pressures on the UK Householder." *Resources, Conservation, and Recycling* 32(3-4), July, pp. 333-348.

Renkow, M. and A.R. Rubin. 1998. "Does municipal solid waste composting make economic sense?" *Journal of Environmental Management*, 53(4): 339-347.

Shinkuma, T. 2007. "Reconsideration of an advanced disposal fee policy for end-of-life durable goods", *Journal of Environmental Economics and Management* 53: 110-121.

Simmons Phil, Nora Goldstein, Scott M. Kaufman, Nickolas J. Themelis and James Thompson, Jr. 2006. 'The State of Garbage in America' *BioCycle*. 47 (4), April, pp. 26-34.

Steuteville, R. (1996) "How Much Does it Cost to Compost Yard Trimmings?" *BioCycle*, 37(9): 39-43.

Unites States Environmental Protection Agency. 2010. 'Municipal Solid Waste Generation, Recycling, and Disposal in the United States: Facts and Figures for 2010.' http://www.epa.gov/wastes/nonhaz/municipal/pubs/msw_2010_rev_factsheet.pdf (accessed 4/19/2013).

Waste Business Journal. 2012. 'Waste Market Overview and Outlook 2012.' http://www.wastebusinessjournal.com/overview.htm (accessed 4/19/13).

Weitzman, Martin L. 1974. "Prices vs. Quantities." *The Review of Economic Studies*, 41(4), October: 477-491.

Table 1: Variable Definitions

WASTE	Annual tons of waste plus recycling managed by the municipality
r	Annual tons of municipal recycling
RATE	*r/WASTE* times 100
NUMB	Number of items collected for recycling
WAGE	Annual salary of municipal waste workers ($/year)
FUEL	Cost of fuel ($ per liter)
p_g	Curbside price per bag of waste collected ($ per bag)
POP	Human population residing within the municipality
MUNICOST	Annual cost to manage all waste less recycling revenues ($)
AMUNICOST	*MUNICOST/WASTE*
HHRECCOST	Calculated total costs to households to recycle ($)
ECWASTE	Calculated total external costs of waste disposal ($)
EBREC	Calculated total external benefits of municipal recycling ($)
SC	*MUNICOST + HHRECCOST + ECWASTE – EBREC*
ASC	*SC/WASTE*

Table 2: Summary of Variables

Variable	N	Mean	Standard Dev.	Min	Max
WASTE	497	266,900	268,568	72,040	1,926,718
r	497	45,462	47,023	8,153	362,881
RATE	497	19.44	7.63	3.90	48.53
NUMB	497	11.50	4.29	4	26
WAGE	497	80.35	13.74	32.63	152.83
FUEL	497	1.15	0.13	0.96	1.46
p_g	497	0.10	0.20	0	0.80
POP	497	605,413	548,387	188,562	3,673,036
MUNICOST	497	69,898,480	71,385,220	19,243,040	474,000,000
AMUNICOST	497	264.55	73.97	126.14	531.34
HHRECCOST	497	2,310,502	7,464,753	0	61,489,610
ECWASTE	497	6,168,214	6,484,421	1,482,990	50,619,088
EBREC	497	9,092,434	9,404,641	1,630,600	70,260,000
SC	497	69,300,000	71,700,000	18,400,000	509,000,000
ASC	497	260.29	74.24	122.75	505.27

Table 3: Household Recycling Costs (Dependent Variable: r/POP)

Variable	Coefficient	Standard Error	Significance
Constant	0.076	0.0006	1% level
PRICE	0.0000718	0.0000357	5% level

N = 497; R^2 (within) = 0.010; R^2 (between) = 0.007; R^2 (overall) = 0.007
F test that all u_i=0: F(84, 411) = 48.53; Prob > F = 0.0000

Table 4: External Marginal Benefits of Recycling

Source of Estimated Life-Cycle Emissions (Units of CO_2, SO_2, NO_x per Ton Recycled)	Source of Estimated External Marginal Costs ($ per Unit of SO_2 and NO_x)*	$ per Ton Recycled
Morris (2005)	Muller and Mendelsohn (2009)	$54.77
	Fann et al. (2009)	$525.33
	Holland and Watkiss (2002)	$380.66
Erickson et al. (2005)	Muller and Mendelsohn (2009)	$2.43
	Fann et al. (2009)	$40.92
	Holland and Watkiss (2002)	$16.06
JMOE (2005)	Muller and Mendelsohn (2009)	$21.07
	Fann et al. (2009)	$295.69
	Holland and Watkiss (2002)	$200.86
Craighill and Powell (1996)	Craighill and Powell (1996)	$516

*External marginal cost of CO_2 assumed at $21 per ton (Interagency Working Group on the Social Cost of Carbon, 2010).

Table 5: The Components of Social Cost: Means

	Base Value	High Value	Low Value
MUNICOST	Observed: $69,898,480	n/a	n/a
HHRECCOST	Estimated: $2,310,502	Doubled: $4,621,004	Halved: $1,155,251
ECWASTE (Landfill and Incineration)	$15 and $30/ton: $6,168,214	$30 and $60/ton: $12,336,428	$7.5 and $15/ton: $3,084,107
EBREC	$200/ton: $9,092,434	$400/ton: $18,184,868	$25/ton: $1,136,554

Table 6: The Average Social Costs of Waste Disposal

Dependent Variable = ln(ASC)

Variable	Coefficient	Standard Error	Significance
Constant	4.94	0.493	1% level
ln(REC_RATE)	-0.590	0.223	1% level
[ln(REC_RATE)]^2	0.130	0.041	1% level
NUMB	0.012	0.003	1% level
ln(WAGE)	0.079	0.040	5% level
ln(fuel)	0.067	0.037	10% level

N = 497; R^2 (within) = 0.118; R^2 (between) = 0.138; R^2 (overall) = 0.136
F test that all u_i =0: F(84, 407) = 34.02; Prob > F = 0.0000

Table 7: The Average Municipal Costs of Waste Disposal

Dependent Variable = ln((*MUNICOST+HHRECCOST)/WASTE)*

Variable	Coefficient	Standard Error	Significance
Constant	4.95	0.473	1% level
ln(REC_RATE)	-0.668	0.214	1% level
[ln(REC_RATE)]^2	0.169	0.040	1% level
NUMB	0.012	0.002	1% level
ln(WAGE)	0.076	0.039	5% level
ln(fuel)	0.062	0.035	10% level

N = 497; R^2 (within) = 0.207; R^2 (between) = 0.275; R^2 (overall) = 0.271
F test that all u_i =0: F(84, 407) = 34.58; Prob > F = 0.0000

Table 8: Optimal Recycling Rates

Variable	Base	Doubled	Halved
HHRECOST	10%	9%	12%
ECWASTE	10%	10%	10%
EBREC	10%	17%	8%

Table 9: The Average Social Costs of Waste Disposal (Dependent Variable: ln*ASC*)

	External Marginal Benefit of Recycling		
Variable	\$25/ton	\$200/ton	\$400/ton
Constant	5.095*** (0.440)	4.944*** (0.493)	4.671*** (0.577)
ln(REC_RATE)	-0.608*** (0.199)	-0.590*** (0.223)	-0.516** (0.261)
[ln(REC_RATE)]^2	0.149*** (0.037)	0.130*** (0.042)	0.091* (0.049)
NUMB	0.011*** (0.002)	0.013*** (0.003)	0.015*** (0.003)
ln(WAGE)	0.069* (0.036)	0.079** (0.040)	0.095** (0.047)
ln(fuel)	0.059* (0.033)	0.067* (0.037)	0.080* (0.043)
N, R^2	497, 0.245	497, 0.136	497, 0.033

*, **, and *** denote statistical significance at the 10%, 5%, and 1%, respectively

Table 10: The Social Cost of Recycling in Japan (Dependent Variable: ln*ASC*)

Variable	Coefficient	Standard Error	Significance
LN(paper recycle rate)	-0.051	0.020	5% level
[LN(paper recycle rate)]^2	-0.010	0.005	10% level
LN(metal recycle rate)	-0.002	0.023	-
[LN(metal recycle rate)]^2	-0.023	0.037	-
LN(glass recycle rate)	0.012	0.031	-
[LN(glass recycle rate)]^2	0.027	0.019	-
LN(PET recycle rate)	0.337	0.059	1% level
[LN(PET recycle rate)]^2	0.069	0.025	1% level
LN(plastic recycle rate)	0.013	0.010	-
[LN(plastic recycle rate)]^2	0.000	0.002	-
LN(other recycle rate)	0.010	0.008	-
[LN(other recycle rate)]^2	0.010	0.003	1% level
NUMB	0.009	0.003	1% level
Ln(wage)	0.053	0.040	-
Ln(fuel)	0.002	0.038	-
Constant	5.269	0.397	1% level

N = 419; R^2 (within) = 0.319; R^2 (between) = 0.008; R^2 (overall) = 0.038
F test that all u_i =0: F(78, 325) = 36.56; Prob > F = 0.0000

Figure 1: Municipal and Social Average Costs of Waste Disposal ($ per Ton)

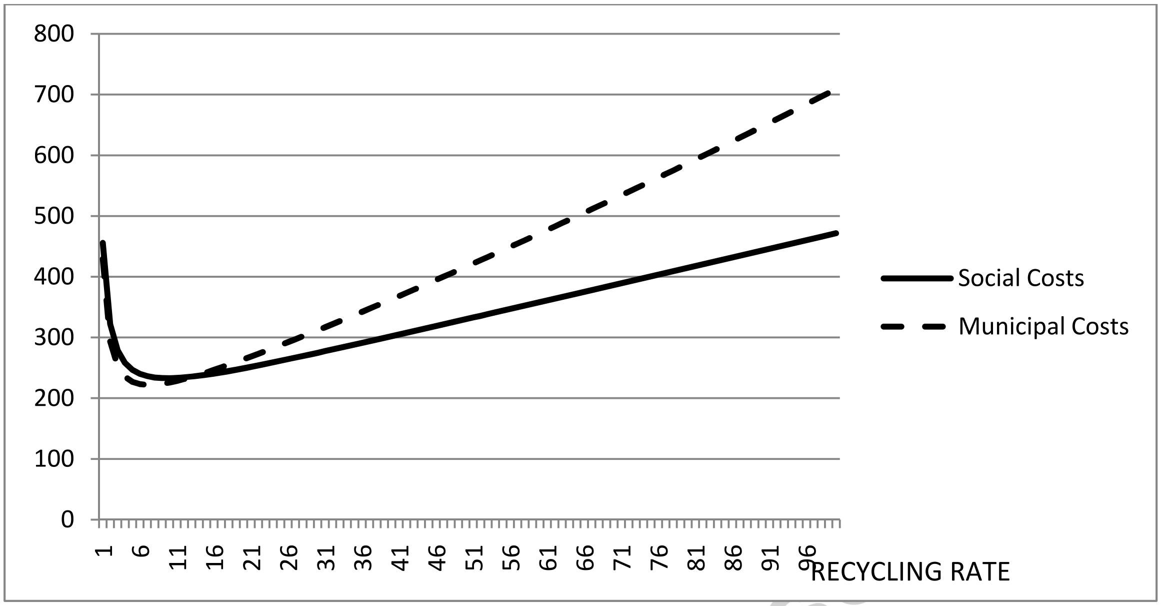

Figure 2: Varying the External Benefits of Recycling ($ per Ton)

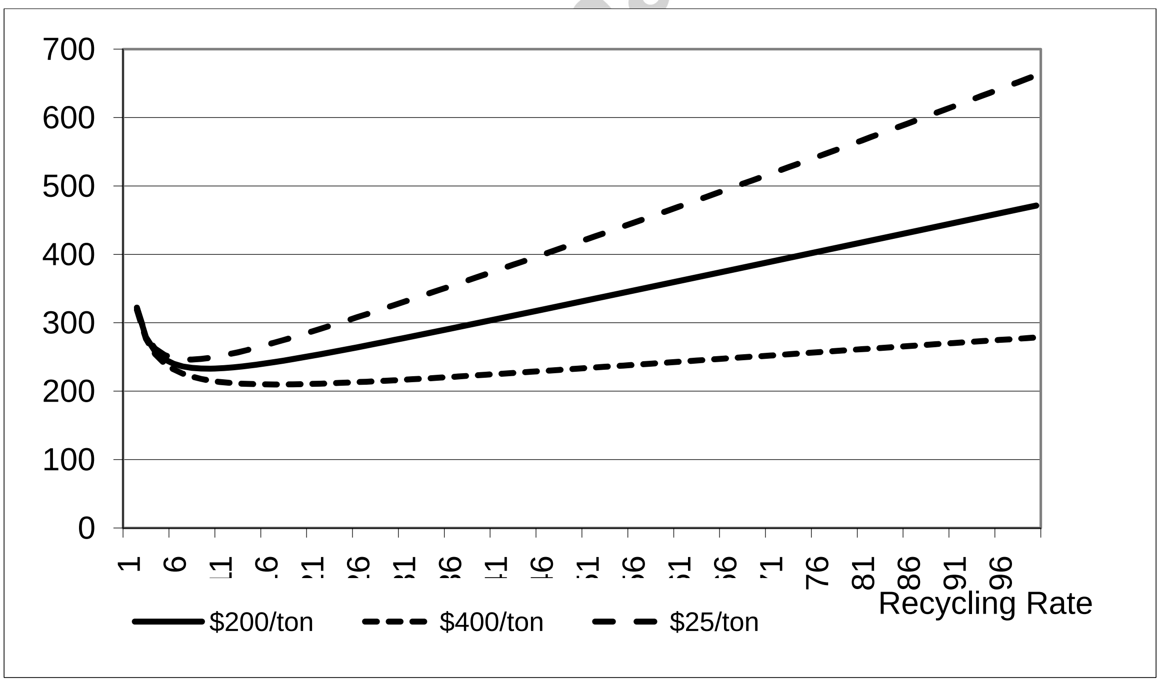

Figure 3: Varying Household Recycling Costs ($ Per Ton)

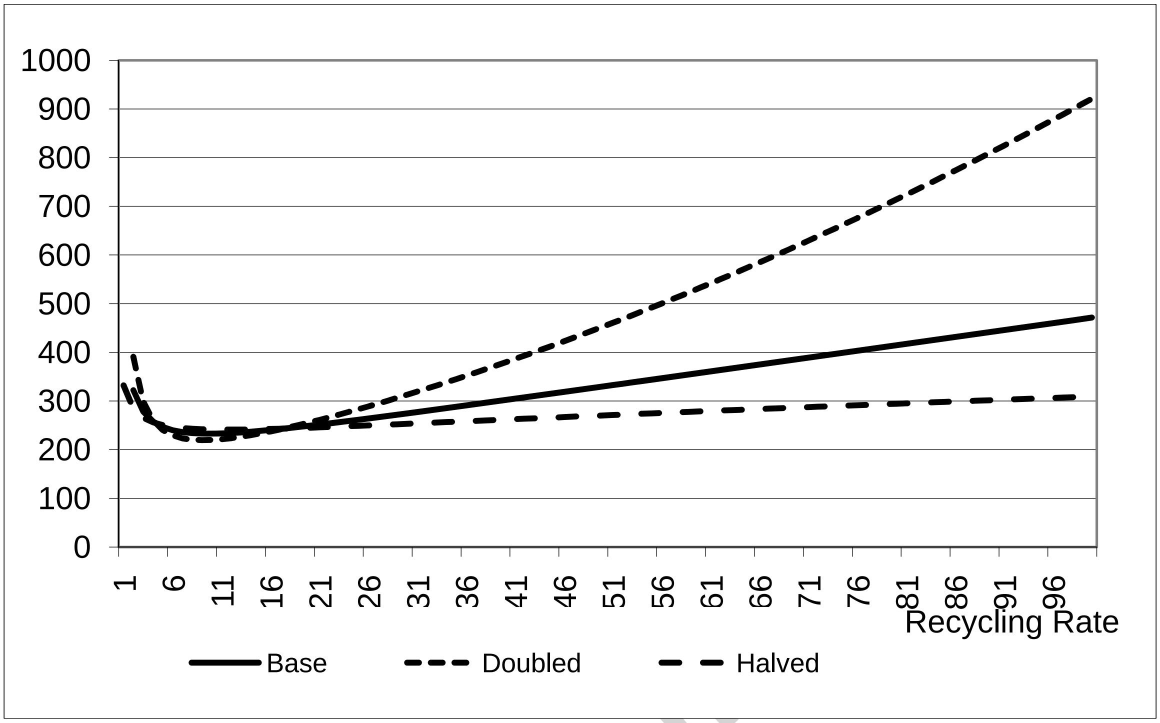

Figure 4: Varying the External Costs of Waste Disposal ($ per Ton)

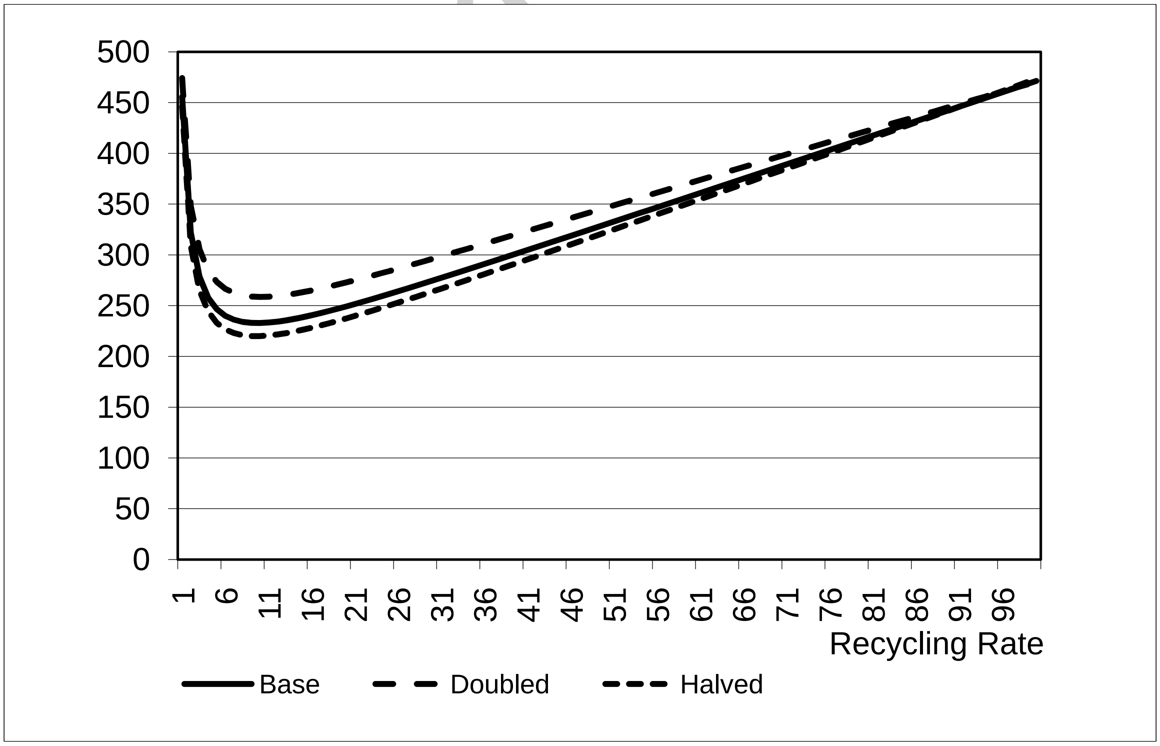

Figure 5: Low Household Recycling Costs and High External Waste and Rec. Costs

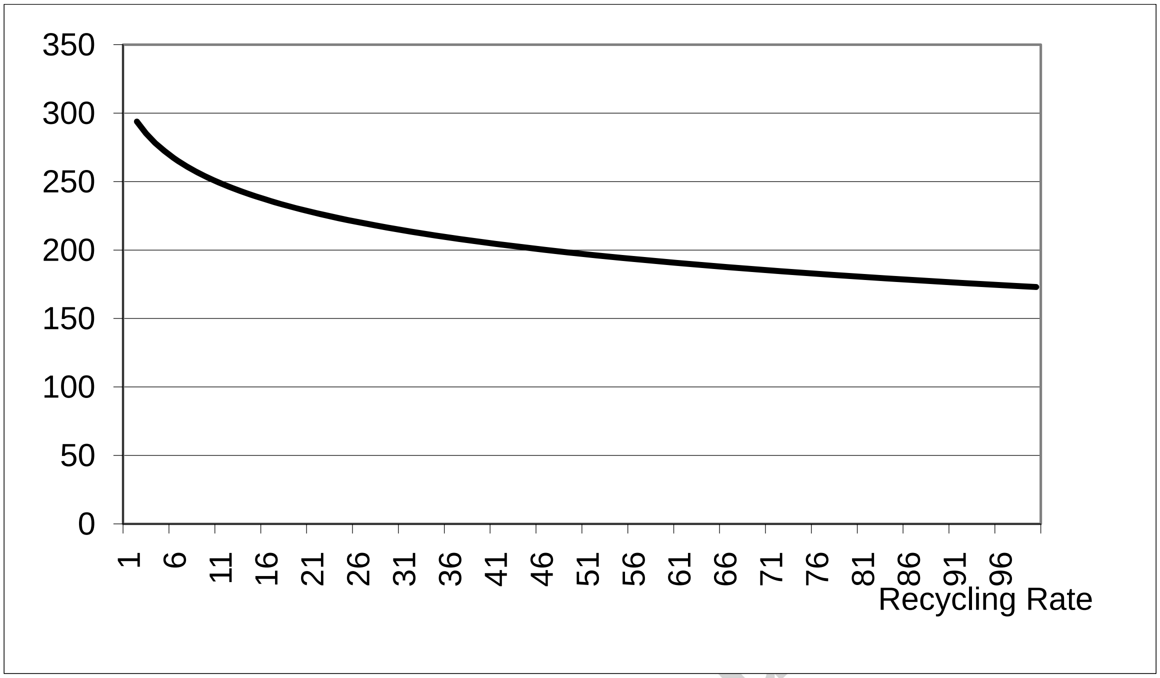

Figure 6: The Recycling of Specific Materials ($ per Ton)