The Cost of Reducing Municipal Solid Waste

Karen Palmer, Hilary Sigman,
and Margaret Walls

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Resources for the Future
1616 P Street, NW
Washington, DC 20036
Telephone 202-328-5000
Fax 202-939-3460

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Abstract

This paper explores public policies for reduction of municipal solid waste. We parameterize a simple model of waste disposal using supply and demand elasticities from the economics literature and 1990 prices and quantities of recyclable and recycled materials. Using this model, we calculate the waste reduction in response to three public policies: (i) deposit/refunds, (ii) advance disposal fees, and (iii) recycling subsidies. The results illustrate the effects of the three policies on source reduction and recycling of five recyclable materials that comprise 56 percent of municipal solid waste: aluminum, glass, paper, plastic, and steel. The calculated responses provide information about the cost of reducing municipal solid waste through various policies. This analysis suggests that a 7.5 percent reduction in disposal of the solid wastes in the model might have been optimal in 1990 from a benefit-cost perspective.

Key Words: solid waste, deposit/refund, recycling subsidy, secondary materials

JEL Classifications No(s).: Q2, H2

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I. INTRODUCTION

In recent years, several types of public policies to reduce solid waste disposal and to increase recycling have been considered and implemented. Some policies rely on fees, including deposit/refunds and "advance disposal fees," product taxes intended to discourage consumption of disposable goods. For example, nine states have deposit/refunds on beverage containers, so called "bottle-bills." Congress has considered legislation for a federal beverage container deposit/refund similar to these state policies. Two states, California and Florida, have adopted advance disposal fees. Other public policies use standards rather than fees to accomplish waste reduction. Thirteen states and the District of Columbia have adopted minimum recycled content standards for newsprint. Similarly, some industry and environmental groups advocate recycling rate standards.

Recent research explores these policies from the perspective of economic efficiency. Like most of this literature, the current study assumes that the goal of public policy is to reduce the inefficiency that arises from the failure to charge positive prices for waste disposal. A direct policy response would charge households according to their garbage or place quantity restrictions on their solid waste disposal.[2] However, this direct approach may be undesirable if waste generators can easily substitute illegal disposal for legal alternatives. Several studies

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[1] Karen Palmer, Quality of the Environment Division, Resources for the Future; Hilary Sigman, University of California, Los Angeles; Margaret Walls, Victoria University of Wellington, New Zealand, and Energy and Natural Resources Division, Resources for the Future. This research was funded in part by the U.S. Environmental Protection Agency, Office of Policy, Planning, and Evaluation under a cooperative agreement with Resources for the Future (#R821821-01). Sigman is grateful for financial support for an earlier version of this research from the Marine Policy Center, Woods Hole Oceanographic Institution. The authors appreciate the excellent research assistance of Ken Harrison, Steve Puller, and Richard Akresh and the helpful comments of Dallas Burtraw, Di Jin, Eduardo Ley, Clare Lindsay, Debbie Nestor, Michael Podolsky, Brett Van Akkeren, an anonymous referee, and especially Don Fullerton.

[2] Unit-based pricing of trash collection -- charging residential customers on a per-bag, per-can, or per-pound basis -- is gaining popularity (most residences have traditionally paid a flat rate that is assessed in property taxes or utility bills). In the absence of illegal disposal opportunities, this policy is the most efficient way of reducing solid waste (Jenkins, 1993). However, with illegal disposal, Fullerton and Kinnaman (1995) show that the optimal charge may be zero or even negative.

argue that the deposit/refund is the best option in the presence of illegal disposal (Dinan, 1993; Fullerton and Kinnaman, 1995; Fullerton and Wu, 1996; Menell, 1990; Palmer and Walls, 1995; Sigman, 1995). Under a deposit/refund, the consumer only bears a cost if the product is discarded. Thus, a deposit/refund ensures that the least-cost method of reducing disposal is used, whether it be through source reduction -- reducing disposal through less production and consumption -- or through recycling. By contrast, other policies take advantage of only source reduction or only recycling as opportunities for disposal reduction.

In this paper, we develop a simple partial equilibrium model of waste generation and recycling to evaluate the cost-effectiveness of various policies for reducing solid waste disposal. Our model consists of three equations: a market equilibrium condition for materials that become part of the waste stream, a market equilibrium condition for recycled materials (those reclaimed from the solid waste stream and ultimately reprocessed into useable products), and a mass-balance identity. We calibrate this model with elasticity estimates from previous empirical studies -- price elasticities of demand for materials used to manufacture consumer goods and price elasticities of demand and supply for recycled materials -- and with 1990 price and quantity data for each type of material. The model includes five common recyclable components of the waste stream: paper, glass, plastic, aluminum, and steel. With the calibrated model, we assess the intervention levels necessary for waste disposal reduction through three price-based policies: (i) an advance disposal fee (ADF), (ii) a deposit/refund, and (iii) a recycling subsidy. We conduct a Monte Carlo analysis to explore the sensitivity of our results to uncertainty about the behavioral parameters.

Our results have several policy implications. First, we find a substantial difference in the intervention levels necessary to accomplish reductions in disposal with the various policies. A $45 per ton deposit/refund would reduce all of the wastes in our model by 10 percent. The government could obtain a comparable reduction using an ADF of $85 per ton or a recycling subsidy of $98 per ton. As the paper argues, these results suggest significant costs from using inefficient policies.

Second, our results demonstrate the importance of flexible policies for waste reduction. We compare policies that set common waste reduction targets for specific materials with a least-cost approach that allows greater reductions in some materials than others. The difference in cost between the two approaches is substantial: while the least-cost approach has a marginal cost of $45 per ton of waste reduced for a 10 percent overall reduction, the same reduction would have a marginal cost of $70 per ton if disposal of each material must be reduced by 10 percent. Thus, if different waste types have equal social marginal cost of disposal, setting policy goals for individual materials may cost considerably more than establishing a single disposal price for all materials.

Finally, our calculations suggest that a moderate reduction in municipal solid waste might be desirable from a cost-benefit perspective. The social marginal cost of waste disposal in a new landfill that meets all environmental requirements is approximately $33 per ton (Ackerman et al., 1995). Comparing these avoided disposal costs with the costs from waste reduction in our model suggests that a 7.5% reduction in these wastes would have been desirable in 1990 if it could be implemented in a least-cost manner. However, the wastes in our model account for only 56 percent of all municipal solid waste disposal, so the efficient total reduction in waste remains a subject for future research.

The paper proceeds as follows. In the next section, we describe our model of waste generation and recycling and explain how we incorporate policy interventions in the model. In section III, we show the prices, quantities, and elasticity estimates used to calibrate the model and discuss the sources of these data. In section IV, we report calculations of the intervention levels required for given reductions in disposal. The section also shows the effects of the policies on source reduction and recycling rates by material. Section V evaluates the efficient reduction in solid waste disposal from a cost-benefit perspective by comparing our results with an estimate of the benefits of avoided waste disposal. We summarize our conclusions in section VI.

II. A MODEL OF WASTE GENERATION AND RECYCLING

This section describes our partial equilibrium model of disposal of a recyclable good. A simple mass balance equation allows us to use market data to estimate the effects of solid waste policies. The amount of waste disposed, W, equals total consumption of the good, Q, less the amount that is recycled, R:

$$W = Q - R \tag{1}$$

Thus, policy intervention may decrease waste disposal through both increased recycling and "source reduction," which we define to mean reductions in Q. Only one form of disposal, W, is available; unlike Fullerton and Kinnaman (1995), we do not allow the possibility of illegal disposal.[3] Further, disposal is free before policy intervention (a typical situation in most communities in the U.S.). Throughout our analysis, we assume that the goal of public policy is to remedy the disparity between this zero private price and the positive social marginal cost of disposal.

In this model, policy intervention influences the equilibria in two markets: the market for Q, and the market for the recycled material R. We begin by outlining a simple framework for these two markets and then illustrate the imposition of the three types of solid waste policies. Finally, we discuss the specific functional forms used to calculate the effects of intervention.

A. Market for the Recyclable Good

Demand for a recyclable good differs from demand for other goods because different consumers may face different effective prices. If the good is not ultimately recycled, its effective price is the market purchase price p_q. However, if the good is ultimately recycled, the effective price equals p_q less the scrap value of the good.[4] The scrap value of the good is the price of the recycled material, p_r. Demand for the good occurs at a combination of these two effective prices. In general terms,

$$Q = D(p_q, p_q - p_r), \tag{2}$$

In our empirical model we focus on markets for intermediate materials such as aluminum or glass that firms use to produce consumer goods such as beverage containers. Thus the demand for what we call "final materials" is a derived demand by firms that use these materials to produce consumer products. Although it is consumers who recycle rather than

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[3] Although it would be desirable to distinguish between legal and illegal disposal, we do not have the empirical elasticities that would be necessary to incorporate this distinction into our quantitative model.

[4] In fact, the effective price of the good to recyclers might equal p_q-p_r plus some marginal effort cost associated with recycling the good. However, it is difficult to know what recycling effort cost characterizes the marginal units of the good purchased. For simplicity, we assume this marginal effort cost equals zero. Kinnaman and Fullerton (1995) model recycling with heterogeneous recycling costs across households.

producers, the derived demand of producers reflects the prices experienced by their consumers.[5] Most of the prices and elasticity estimates required for the model are available only at the intermediate good level.

. These recyclable intermediate materials, hereafter referred to as "final materials," are assumed to be available to firms with perfectly elastic supply. Because p_q is the constant producer price of the good, equation (2) shows the equilibrium in this market.

B. Market for the Recycled Material

The supply of recycled material is written as the recycling rate, r -- the fraction of Q that is recycled -- multiplied by Q. The recycling rate, r, increases with the price of the recycled good, p_r, i.e., collectors of used goods will increase the quantity they supply to the market when the price they receive increases. This supply curve is

$$R^s = r(p_r)\left[D(p_q, p_q - p_r)\right] \tag{3}$$

Demand for recycled material by secondary material processing firms varies with its price:

$$R^d = R^d(p_r) \tag{4}$$

The equilibrium in the recycling market prior to any policy intervention then is:

$$R^d(p_r) = r(p_r)\left[D(p_q, p_q - p_r)\right] \tag{5}$$

Combining the mass balance expression (1) with the two market equilibrium equations (2) and (5), we can write the amount of waste disposed as:

$$W = (1 - r(p_r))\left[D(p_q, p_q - p_r)\right] \tag{6}$$

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[5] Producers face the two different effective prices for these goods because their consumers do. With fixed consumer demand, the derived demand would be perfectly inelastic (because one unit of the intermediate good must be used to produce one unit of the consumption good). Thus, price enters the derived demand only because it affects consumers' demand. As the text argues, consumers may demand at two effective prices, so these two effective prices should enter the derived demand function.

The market clearing-conditions for *Q* in equation (2) and *R* in equation (5), together with the expression for waste disposal in equation (6), provide the basis for the analysis of policy intervention.

This model requires several assumptions about the structure of the markets. First, both the markets for the final material and the recycled material are perfectly competitive. This assumption is probably realistic for the U.S. market in the undifferentiated materials considered in this study. Second, we assume consumers discard or recycle goods made with the final materials within a short time of purchase. This assumption implies that any source reduction that occurs has an immediate effect on disposal and recycling. With alternative assumptions, there would be vintages of goods and a lagged effect of source reduction on disposal. The assumption seems appropriate for the principally nondurable goods studied empirically in this paper.

Third, the demand curve for recycled material does not depend on the quantity of the final material product for which it is an input. As a consequence, a policy such as the ADF that reduces *Q* does not affect the demand for recycled materials. If this feedback does occur, our model overestimates demand for recycled materials. As a result, the deposit/refund and ADF necessary to generate a given reduction in disposal would likely be higher than we calculate, whereas the recycling subsidy would likely be lower.

Finally, this model is partial equilibrium in nature. The model may fail to consider many potential interactions among the markets. For example, we assume that the demand for each final material and each recycled material depends only upon its own effective price and constant parameters. Thus, we study the effects of the policies on each market independently. In practice, some of the materials studied probably substitute for one another; for example, policies that increase the price of plastic may increase demand for glass or aluminum. Ideally, one would model such substitution responses explicitly. Unfortunately, we do not have sufficient information on cross-price elasticities to implement a model with these effects. If cross-price elasticities were positive rather than zero as assumed, the deposit/refund and the ADF necessary to generate a given reduction in disposal would be higher than we estimate.

C. Policy Intervention

We calculate the effects of three types of economic incentive policies to promote solid waste reduction: deposit/refunds, advance disposal fees, and recycling subsidies. Our goal is to calculate the levels of these price instruments that are necessary to achieve particular reductions in solid waste. We also show the predicted effects of the policies on the equilibrium quantities in the final materials markets and the recycling markets.

Since our empirical model focuses on the behavior of firms, we assume that the government imposes these policies at the producer level rather than the household level. Thus, for example, beverage can producers pay the deposit portion of a deposit/refund (or the ADF) when they purchase aluminum sheet. The refund portion of the deposit/refund (or the recycling subsidy) is granted to collectors of used beverage cans who subsequently sell them for reprocessing. Although the statutory incidence of the policies is on producers, the policies will affect household's incentives for consumption, recycling and disposal. Because producers must use one unit of the final material (e.g., a pound of aluminum sheet, for example) to produce one unit of the consumption good (e.g., a pound of aluminum beverage cans), producers will pass changes in intermediate materials prices through to consumers. Imposing the policies at the production rather than the retail level should greatly reduce administrative and transaction costs of the policies because the number of affected producers and products is small compared to the myriad final consumer products.

Deposit/refund

Suppose the deposit, d, equals the refund.[6] A deposit/refund program changes the equations in both the final material and the recycled material markets. First, the deposit acts like a tax on the final material, increasing its price to demanders by the full amount of the deposit. If the fixed supply price of the final material is p_q, the demand price in the presence of the deposit/refund becomes p_q+d. Demanders who recycle get the refund back. Thus they

[6] In our simple model, a deposit equal to the refund is optimal. However, Fullerton and Kinnaman (1995) show that the optimal deposit/refund may have different values for the deposit and refund under more general assumptions.

experience no net increase in the effective price they pay for the final material. The equilibrium in the final material market with a deposit/refund is:

$$\begin{aligned} Q &= D(p_q + d, (p_q + d) - (p_r + d)) \\ &= D(p_q + d, p_q - p_r) \end{aligned} \tag{7}$$

Second, the deposit/refund increases the benefit from recycling the good, changing the equilibrium in the recycled material market. If p_r represents the demand price of the recycled material (the price paid by material reprocessors to suppliers of recycled materials), the full price received by suppliers should equal p_r+d. Thus, in the presence of the deposit/refund, the market clearing condition in the recycling market (equation 5) becomes:

$$R^d(p_r) = r(p_r + d)\left[D(p_q + d, p_q - p_r)\right] \tag{8}$$

With these two new market-clearing conditions, the expression for waste generation is:

$$W = (1 - r(p_r + d))\left[D(p_q + d, p_q - p_r)\right] \tag{9}$$

Advance Disposal Fee

An advance disposal fee is a charge on all consumption of the final material.[7] Like the deposit part of the deposit/refund, it increases the price of the final material to all demanders, including recyclers and non-recyclers. If f is the level of the fee, the equilibrium in the final material market becomes:

$$Q = D(p_q + f, p_q + f - p_r) \tag{10}$$

The ADF indirectly affects the market for the recycled material because it reduces the amount of the good available to be recycled. The equilibrium becomes:

$$R^d(p_r) = r(p_r)\left[D(p_q + f, p_q + f - p_r)\right] \tag{11}$$

[7] Existing ADFs are more complicated than this simple policy. In Florida, the ADF (a penny per container) is automatically repealed lifted if the aggregate recycling rate for the material exceeds 50 percent. In California, the ADF is partially refunded to an individual who returns a container for recycling.

The expression for the amount of waste disposed with an advance disposal fee is:

$$W = (1 - r(p_r))\left[D(p_q + f, p_q + f - p_r)\right] \tag{12}$$

<u>Recycling Subsidy</u>

A recycling subsidy is like the refund part of the deposit/refund. In the recycled material market, the subsidy drives a wedge between the demand price and the price received by suppliers of used goods. If the demand price equals p_r and the level of the subsidy is s, the price received by suppliers is p_r+s. There is also a change in the final materials market. The subsidy decreases the effective price paid by those demanders of the final material who recycle. Equilibrium in the final material market becomes:

$$Q = D(p_q, p_q - (p_r + s)) \tag{13}$$

In the recycled good market, the equilibrium with the subsidy becomes:

$$R^d(p_r) = r(p_r + s)\left[D(p_q, p_q - p_r - s)\right] \tag{14}$$

Thus, the equation for waste disposal in the presence of a subsidy is

$$W = (1 - r(p_r + s))\left[D(p_q, p_q - p_r - s)\right] \tag{15}$$

D. Implementing the Model

Translating these equations into marginal cost estimates requires specific demand and supply functions. We assume that constant elasticities characterize these functions to permit direct application of elasticity estimates from earlier studies to our model. The demand for Q presents special problems because our model requires market demand to respond to two prices. A simple form characterizes market demand as a weighted average of demand at the two prices:

$$D(p_q, p_q - p_r) = \mathrm{b}_1 (p_q)^{\mathrm{e}_q^d} + \mathrm{b}_2 (p_q - p_r)^{\mathrm{e}_q^d} \tag{16}$$

where ε_q^d is an own price elasticity. We choose the weights β_1, and β_2 to reflect the share of Q that is recycled in the pre-intervention equilibrium.[8]

For the recycled good, we also use constant elasticity functional forms:

$$r(p_r) = \alpha(p_r)^{\varepsilon_r^s}$$
$$R^d(p_r) = \gamma(p_r)^{\varepsilon_r^d}$$

where ε_r^d is the own price elasticity of demand for the recycled material by secondary material processors; and ε_r^s is the own price elasticity of supply for the recycled material by material collectors.

We use estimates of the elasticities ε_q^d, ε_r^d, and ε_r^s from the economics literature, as discussed in section III. The parameters α, β_1, β_2, and γ are calculated based on these elasticities and baseline prices and quantities. With this calibrated model, we determine the deposit/refund, the advance disposal fee, and the recycling subsidy -- *d*, *f*, and *s*, respectively, in equations (9), (12), and (15) above -- that will generate a specific percentage reduction in overall solid waste disposal, *W*. In addition to predicting the effects of the policy, we also analyze the relative "cost-effectiveness" of different policy approaches for given percentage reductions. Finally, we perform a "cost-benefit" type of analysis, comparing the marginal costs of disposal reductions with their marginal benefit in avoided marginal social costs of disposal. In both the cost-effectiveness and cost-benefit analysis, we assume that all the materials give rise to the same marginal external cost per ton of waste disposal

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[8] Because the weights are fixed based on the initial recycling rate, this functional form does not allow a change in the share of consumption at each effective price with changes in the recycling rate. In previous versions of this paper, we experimented with different functional forms for demand without marked effects on the quantitative results of the model. The fixed weights do not preclude households that do not recycle initially from becoming recyclers in response to a recycling subsidy or a deposit-refund.

III. BASELINE DATA AND ELASTICITY ESTIMATES

In the model we analyze final materials markets and recycled materials markets for five aggregate categories of materials including paper and paperboard, glass, aluminum, steel and plastics. Table 1 lists the 1990 quantities of consumption, recycling, and disposal in each category and several subcategories of each material.[9]

Together, these five categories of materials accounted for 56% of the total municipal solid waste stream in 1990 (Franklin Associates, 1992). We chose to study the effects of solid waste policies on these particular components of the waste stream because these materials have active recycling markets. Several of these materials, including newsprint and aluminum and glass beverage containers, are the subject of existing or proposed solid waste policies. In addition, we were able to obtain necessary demand and supply elasticities and prices necessary to parameterize the model for these materials.

This table indicates the relative importance of different components of the waste stream and current recycling activity. Paper and paperboard is the largest aggregate component of the municipal solid waste stream followed by yard waste and "other," a category that includes a variety of materials such as non-ferrous metals (excluding aluminum), rubber and leather, textiles and wood. Among those components of the waste stream included in our model, the second largest aggregate category is plastics. The table also shows that aluminum beverage cans have by far the highest baseline recycling rate of all materials, at 63%, followed by newsprint with a recycling rate of 42.5% in 1990. Plastics exhibit the lowest recycling rate of the five aggregate materials categories in the model.[10]

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[9] An appendix containing a complete definition of each of these categories of materials and detailed information about the data sources used to compile the information in Table 1 is available from the authors upon request.

[10] At the time this study began, 1990 data were the most recent data available. More recent information suggests that recycling rates for some materials have increased substantially. Paper and paperboard had an overall 34 percent recycling rate in 1993, for example (compared with 27.8 percent in 1990), and plastics had a 3.5 percent rate (compared with 1 percent in 1990). PET soda bottles and HDPE milk jug recycling rates stood at 41 and 24 percent in 1993, respectively -- much higher than in 1990 (Franklin Associates, 1994).

Table 1. Generation, Recovery, and Disposal of Municipal Solid Waste in 1990

Material	Quantity Consumed (million tons)	Quantity Recovered (million tons)	Recovery Rate (%)	Quantity of Waste (million tons)	Final Material Price (1990$/ton)	Post-Consumer Scrap Price (1990$/ton)
Included Categories of Materials						
Paper and Paperboard	**73.320**	**20.396**	**27.8**	**52.924**		
newsprint	12.938	5.497	42.5	7.441	479	18
writing and printing	23.881	3.421	14.3	20.460	923	142
paperboard containers	32.605	11.478	35.2	21.127	455	34
other paper	3.896	0	0	3.896	1078	NA
Glass	**13.182**	**2.625**	**19.9**	**9.557**		
beverage containers	11.905	2.625	22.0	9.280	450	18
durables	1.277	0	0	1.277	367	NA
Aluminum	**2.660**	**1.013**	**38.1**	**1.647**		
beverage cans	1.576	0.990	63.2	0.586	2770	1015
other containers/packaging	0.335	0.023	6.9	0.312	2492	1063
durables and misc. nondurable	0.758	0	0	0.758	3830	NA
Steel	**12.302**	**1.890**	**15.4**	**10.412**		
cans	2.689	0.630	23.4	2.059	460	67
other containers/packaging	0.201	0.010	5.0	0.191	460	98
durables	9.412	1.250	13.3	8.162	467	98
Plastics	**16.244**	**0.162**	**1.0**	**16.082**		
PET soft drink bottles	0.435	0.137	31.5	0.298	1548	148
HDPE liquid food containers	0.364	0.025	6.9	0.339	764	138
other plastic nondurables	10.505	0	0	10.505	853	NA
durables	4.940	0	0	4.940	881	NA
Excluded Categories of Materials						
Food Waste	**13.200**	**0**	**0**	**13.200**		
Yard Waste	**35.000**	**4.200**	**12.0**	**30.800**		
Other	**29.824**	**2.414**	**8.1**	**27.410**		
Total - All Categories	**195.732**	**32.700**	**16.7**	**163.032**		
Total - Included Categories	**117.708**	**26.086**	**22.2**	**91.622**		

Sources: Franklin Associates (1992); U.S. Bureau of the Census' ***Current Industrial Reports*; *Recycling Times*; *Resource Recycling*;** and other sources available

The table also indicates the relative costs per ton of different recyclable materials. The materials with the highest cost per ton are the various categories of aluminum, followed by PET for use in soft drink bottles. Glass costs the least of all the materials. In a similar fashion, scrap prices are highest for aluminum and lowest for glass.[11]

Table 2 presents the values of the demand and supply elasticities used to parameterize the model. Materials that are not recycled are assigned an "NA" for the secondary market elasticities. The elasticities in Table 2 are based on estimates from earlier studies wherever possible. Econometric studies provided most of the elasticity estimates, although engineering analyses also contributed a few estimates. Appendix A lists the sources of the estimates. We draw heavily on a study of demand for disaggregated categories of paper by Bingham et al. (1993) and on studies of paper and metals markets by Anderson and Spiegelman (1976), as well as several other studies. In addition, we conducted our own analysis to estimate demand for plastics, based on a limited time-series of prices and quantities; details of this estimation are in Appendix B.

On the basis of the previous literature, Table 2 shows that final demands for all categories of paper products and steel are price inelastic. PET for soft drink bottles is the most elastically demanded product, followed by all categories of aluminum. Source reduction policies are more effective for materials that have higher demand elasticities. All categories of secondary products have price inelastic demands, with the most inelastic demand found in the plastics and paper categories. Secondary supply is most responsive to price for the steel and aluminum categories and least responsive for all categories of paper that are recyclable. The sectors that have greater secondary supply elasticities are particularly responsive to recycling subsidies.

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[11] More recent data would show some differences, particularly in paper and paperboard markets. Old corrugated container prices, for example, went as high as $200 per ton in 1994 (Rabasca, 1994). Many of these prices are quite volatile.

Table 2. Elasticities of Demand and Supply by Material

	Primary Demand Elasticity	Secondary Demand Elasticity	Secondary Supply Elasticity
Paper and Paperboard			
newsprint	-.301	-0.1216	0.200
writing and printing	-.949	-0.1600	0.200
paperboard containers	-.463	-0.1600	0.200
other paper	-.594	NA	NA
Glass			
beverage containers	-1.0	-0.5	0.5
durables	-1.0	NA	NA
Aluminum			
beverage cans	-1.4	-.805	1.1
other containers/packaging	-1.4	-.805	1.1
durables and misc. nondurable	-1.4	NA	NA
Steel			
cans	-0.63	-0.63	1.4
other containers/packaging	-0.63	-0.63	1.4
durables	-0.63	-0.63	1.4
Plastics			
PET soft drink bottles	-2.05	-0.1	0.5
HDPE liquid food containers	-1.20	-0.1	0.5
other plastic nondurables	-1.00	NA	NA
durables	-1.00	NA	NA

Sources: See Appendix A.

IV. MODEL RESULTS

This section presents our calculations of the effects of solid waste reduction policies. The prices and elasticities presented in the previous section are used to calibrate the model from section II. In this section, we present the results for several policy experiments. First, we solve for the deposit/refund, the ADF, and the recycling subsidy that yield given percentage reductions in overall waste disposal. We illustrate the effects of these policies on source reduction and recycling. Second, we explore policies that set specific targets for components of the waste stream. We also solve for the material-specific deposit/refunds that yield uniform percentage reductions in each material in the waste stream. We compare the costs of these policies with policies that reduce materials in the least-cost manner. Finally, we use a Monte Carlo approach to analyze the sensitivity of our calculations to uncertainty about the elasticity estimates.

A. Required Intervention Levels

Figure 1 shows the policy intervention levels necessary to reduce waste through a deposit/refund, an ADF, and a recycling subsidy for various percentage reductions in waste up to 25 percent. For example, the model predicts that a deposit/refund of $45 per ton, an ADF of $85 per ton, or a recycling subsidy of $98 per ton will achieve a 10 percent reduction in overall waste disposal.[12]

Table 3 illustrates the reasons for these differences in the necessary intervention levels. The table shows the amount of waste reduction achieved through recycling and source reduction by material for each of the three policies for a 10 percent overall waste reduction. As the table demonstrates, these policies differ substantially in their reliance on source reduction and recycling. The first block of rows in Table 3 shows that the deposit/refund gives rise to both source reduction and recycling. On the other hand, under the recycling subsidy and ADF one waste reduction approach dominates. The recycling subsidy, shown in the middle rows of the table, encourages recycling but also encourages consumption (because it lowers the effective price of the final material for those users who recycle). The ADF, shown in the last block of rows, discourages consumption; however, in the process, it decreases the amount of material available to be recycled and thus reduces recycling. Because the ADF fails to take advantage of opportunities for recycling and the recycling subsidy fails to take advantage of opportunities for source reduction, these policies have to "work harder" to achieve a given waste reduction.

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[12] This 10% reduction is equivalent to a 5.6 percent reduction in total municipal solid waste because the wastes in the model comprise only 56 percent of the municipal solid waste stream (see Table 1).

Figure 1 is available from the authors
at Resources for the Future.

Table 3. Waste Reduction by Policy and Material for a 10% Total Reduction in Disposal (millions of tons)

Policy		Material				
		Paper	Glass	Aluminum	Steel	Plastic
Deposit Refund	Source Reduction	2.363	1.051	0.046	0.666	0.813
	Increased Recycling	2.240	1.469	0.014	0.496	0.003
	Total Waste Reduction	4.604	2.520	0.060	1.163	0.816
Recycling Subsidy	Source Reduction	-1.560	-0.620	-0.028	-0.158	-0.005
	Increased Recycling	6.000	3.828	0.055	1.639	0.013
	Total Waste Reduction	4.440	3.208	0.027	1.481	0.007
Advance Disposal Fee (ADF)	Source Reduction	5.101	2.137	0.108	1.262	1.473
	Increased Recycling	-0.616	-0.218	-0.019	-0.063	-0.003
	Total Waste Reduction	4.485	1.918	0.089	1.199	1.471

Source: Authors' calculations.
Note: Negative source reduction values indicate increased use of the material.

Source reduction and recycling by material

The extent of source reduction and recycling opportunities varies across materials. For example, disposal of plastics and aluminum is achieved much more cheaply through reduced consumption than increased recycling because PET and aluminum have large demand elasticities and because a large portion of plastics can not be recycled. In addition, aluminum also has a scrap price that is one to two orders of magnitude greater per ton than the scrap prices of other materials; thus, a given deposit-refund represents a smaller percentage change in its scrap price. As a consequence of these combined effects, the deposit/refund reduces consumption of these materials, but barely increases recycling. The recycling subsidy causes very little reduction in plastic and aluminum waste because it must rely entirely on recycling.

By contrast, wastepaper, which has relatively low final material and scrap prices, may be reduced inexpensively both by source reduction and recycling, as the response to the deposit/refund in Table 3 shows.

These results highlight the inefficiency of EPA's "hierarchy" of integrated solid waste management (EPA, 1989) as a method of reducing waste disposal.[13] In this hierarchy, source reduction takes priority over recycling, which takes priority over landfilling and incineration. Our model suggests that there is an optimal mix of source reduction and recycling that cost-effectively reduces waste disposal. Relying solely on source reduction for paper, for example, forgoes low-cost recycling opportunities.

Table 3 also shows large differences in the amount of the reduction in waste disposal attributable to different materials under each of the three policies. Paper always accounts for the largest fraction of the overall reduction in absolute terms. Paper comprises a large portion of the waste stream: it is 67.5 percent of the waste materials in our model and 32 percent of all discarded materials (see Table 1). In addition, the initial price of paper per ton indicates a low cost of source reduction (low marginal benefits of paper to consumers) and hence inexpensive waste reduction.

Of all the materials in our model, glass exhibits the largest percentage waste reduction due primarily to the low prices of final glass and scrap glass. When overall waste is reduced by 10 percent with a deposit/refund, glass disposal falls by 24 percent, paper by 9 percent, steel by 11 percent, and aluminum and plastics by only 4 and 5 percent, respectively. These findings foreshadow the results in the next section: the cost of achieving a uniform percentage reduction in each category of waste is much greater than the costs of reducing materials in a least-cost fashion.

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[13] This hierarchy might be more appropriate if externalities associated with production must be addressed simultaneously with those associated with disposal. However, as argued in footnote 2222 below, it is more desirable to address upstream externalities directly at their source.

Recycling rates

Figure 2 shows the aggregate recycling rate for materials in our model under each of the three policies for percentage waste reductions up to 25 percent. The recycling subsidy leads to the greatest increase in recycling rates: a 10 percent waste reduction achieved with the subsidy yields an average recycling rate of 31 percent, up from 22 percent in the absence of any policy. By comparison, the deposit/refund and the ADF achieve 27 percent and 23 percent aggregate recycling rates respectively.

Recycling rates increase the most for glass beverage containers and steel cans. With a 10 percent waste reduction achieved with a subsidy of $98 per ton, the recycling rate for glass beverage containers increases from 22 to 52 percent and the rate for steel cans increases from 23 to 51 percent.[14] The increase in beverage container recycling reflects a low scrap price, whereas the increase for steel cans is attributable largely to the high value of the steel recycling supply elasticity shown in Table 2.

Implications for costs of waste disposal reduction

The intervention levels provide information about the marginal cost of disposal reduction. If the initial price for waste disposal is zero, the deposit/refund per ton of waste necessary to accomplish a given reduction is also the marginal cost of this waste reduction. The deposit/refund is a charge on waste that is disposed; waste generators will adjust their behavior so that this charge equals the marginal cost of waste disposal reduction. Thus, with a 10 percent reduction in waste disposal, the marginal cost of another ton is $45 according to Figure 1.

The levels of the ADF and recycling subsidy also provide information about the marginal private cost of disposal reductions; however, the relationship is less direct than the deposit/refund. In equilibrium, agents will adjust their consumption so that they set the ADF equal to the marginal cost of source reduction. Similarly, in equilibrium, the recycling subsidy equals the marginal cost of recycling. As the results above show, an increase in the ADF

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[14] Higher recycling rates may have been achieved since 1990. By 1993, the recycling rate for steel cans had risen to 46 percent (Franklin Associates, 1994).

Figure 2 is available from the authors
at Resources for the Future.

generates more source reduction than disposal reduction; likewise, an increase in the recycling subsidy generates a greater increase in recycling than decrease in disposal. As a consequence, both *underestimate* the true marginal cost of reducing disposal. The differences that we estimate between the ADF or recycling subsidy and the deposit/refund therefore provide a lower bound on the marginal cost differences between these programs.

The comparison between the level of the deposit/refund and the levels of other policies suggests that the cost differences may be large. An ADF must be $85 per ton to accomplish the same 10 percent reduction in waste, while a recycling subsidy must be $98 per ton. Thus, the results suggest that alternative policies are at least twice as costly as the deposit/refund. The deposit/refund is the least expensive because it encourages both source reduction and recycling. The ADF and recycling subsidy focus on source reduction and increased recycling, respectively, and thus miss some low cost opportunities for waste reduction involving a mix of the two activities.

However, there may be differences in costs among the three policies that are absent from our model. First, the deposit/refund may give rise to greater administrative costs than other policies. By comparison with an advance disposal fee, a deposit/refund requires a potentially costly mechanism for refunding deposits. Traditional bottle bills, such as Massachusetts', require retailers to pay refunds to consumers, sort containers by brand name, and store containers until bottlers collect them. Ackerman et al. (1995) find that this deposit/refund system costs 2.3 cents per container, which corresponds to approximately $320 per ton for a typical steel can or one-gallon HDPE milk jug or over $1300 per ton for an aluminum beverage can.[15] Other designs for deposit/refunds may result in lower average administrative costs. In particular, Ackerman et al. find that California's bottle bill has administrative costs of only 0.2 cents per container, or $28 per ton for steel cans and HDPE

[15] These estimates are based on a typical weight of 0.143 pounds for HDPE in a milk jug, 0.144 pounds for a steel can, and 0.033 pounds for an aluminum beverage can (EPA, 1994; Apotheker, 1995).

milk jugs and $120 per ton for aluminum cans.[16] Moreover, a deposit/refund levied upstream on producers rather than on final consumers may save administrative costs.

Because Ackerman et al. estimate average costs rather than marginal costs, a direct comparison with our calculations is not possible. However, the figures do suggest that administrative costs may be on the same order as the cost savings from using a deposit/refund. Thus, a definitive policy prescription will require further research on administrative costs and strategies to reduce them.

Second, the wastes that are absent from the model could also alter the policy rankings. Wastes that are not currently recycled, such as wood, textiles, and rubber and leather, may not experience increased recycling in response to small policy interventions. However, even small policy incentives may give rise to source reduction of these wastes. Thus, although the government may impose any of these policies on the entire universe of wastes, policies that rely on source reduction such as the ADF and deposit/refund may reduce a wider group of wastes and therefore have a lower cost per ton of avoided disposal.

A third cost consideration missing from the model is the shadow cost of public funds. Again, this consideration discourages reliance on a recycling subsidy relative to other policy options. The government must raise the funds to subsidize recycling by additional taxation somewhere in the economy. If these taxes are distortionary, there could be an additional cost associated with the recycling subsidy. By contrast, the ADF and the deposit/refund raise revenues that the government could use to reduce existing distortionary taxes or to fund public goods; this opportunity may lower the net cost of the program. These additional cost considerations are important to an overall assessment of these policies.

Comparison with earlier research

In our model, the deposit/refund is equivalent to a disposal charge. Therefore, we may compare our calculations of the response of waste disposal to deposit/refunds to the response

[16] In California, retailers are not required to redeem containers; administrative costs are borne by the state, which pays handling fees to recycling centers that collect and process recyclables. Containers are not sorted by brand name or returned intact to bottlers.

that earlier studies have observed empirically to disposal charges. Two recent studies estimate the responsiveness of waste reduction to disposal charges. Jenkins (1993) compiled data on waste disposal in several municipalities with disposal charges. She estimates that an $0.80 per 32-gallon container charge would decrease waste by 9.5% without curb-side recycling and 16% with curb-side recycling.[17] Fullerton and Kinnaman (forthcoming) examine the response of the weight and volume of discards in Charlottesville, Virginia before and after the town imposed a charge of $0.80 per 32-gallon container. In response to this charge, households reduced the weight of their garbage by 14 percent (in the presence of curb-side recycling). However, recent work by Kinnaman and Fullerton (1996) suggests that garbage is much less sensitive to disposal charges than found by this earlier research.[18]

By comparison with these results, our calculated elasticity of waste reduction is similar in magnitude but slightly higher: a $98 per ton charge (which would be equivalent to an $0.80 per 32-gallon container charge given the pre-charge density of garbage observed by Fullerton and Kinnaman) would reduce disposal by slightly less than 20 percent. Our more elastic response may be due to the selection of wastes in our model (specifically, wastes with low cost recycling options). Nonetheless, the comparison appears to confirm the view that waste reduction is moderately sensitive to price incentives.

B. Uniform Percentage Reductions

Some states have adopted material-specific waste reduction goals that vary across materials (Macauley and Walls, 1995). To illustrate the potential costs of this approach, we use the model to determine the *material-specific* deposit/refunds that will generate uniform percentage reductions in disposal of each material.

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[17] These values are based on the description of Jenkins' empirical analysis presented in Repetto et al. (1992).

[18] Fullerton and Kinnaman (1996) use a large cross-section of cities and correct for selection in the municipalities that charge for disposal and offer curbside recycling, perhaps explaining the difference between their conclusion and Jenkins'. Other recent studies that estimate solid waste demand elasticities include Podolsky (1995) and Stratham et al. (1995).

Figure 3 shows the marginal costs of a deposit/refund achieving various percentage reductions in overall waste when each material in the waste stream must be reduced by the same percentage; the marginal costs under the least cost approach (from Figure 1) are shown for comparison purposes. The uniform reduction is substantially more costly than the least-cost reduction. A 10 percent across the board reduction in all included components of waste yields a marginal cost of $70 per ton, compared to $45 per ton for the least-cost approach.

The material-specific deposit/refunds necessary to yield the 10 percent reductions in every waste type vary from only $16 per ton for glass beverage containers to almost $300 per ton for aluminum durables. Those materials that we assume cannot be recycled, such as aluminum durables, "other" paper, and glass durables, require a relatively high deposit/refund to generate a 10 percent waste reduction. Other materials, such as glass beverage containers and newspapers, that have relatively high elasticities of demand and initially low scrap prices need a relatively low deposit/refund to reduce waste. These differences in the characteristics of materials in the waste stream make the uniform approach more costly than an approach that allows materials to be reduced in the least-cost fashion.

C. Sensitivity Analysis

The calculations presented above use estimates of elasticities of demand and supply from existing studies of the markets for the different materials. These values are either the point estimate from a single study or an approximate mid-point of a range of estimates given by several different studies. In both cases, the parameter is only one estimate from a range of possible values implied by the study or studies.

To explore the sensitivity of our results to the choice of elasticities, we use a Monte Carlo method. A value for each elasticity was selected independently from a distribution for the parameter. Distributions of elasticity values were generated in two ways.[19] When the elasticity is based on a single econometric estimate, the parameter is assumed to be distributed

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[19] The table in Appendix A shows distributional assumptions by material.

Figure 3 is available from the authors
at Resources for the Future.

normally with standard deviation equal to the standard error of the point estimate.[20] When the estimate is a mid-value from several studies, a lognormal distribution is assumed. We choose parameters for the lognormal to best fit a distribution with three points: a median equal to the middle range value from the literature (the value that is used for our central case calculations) and 5th and 95th percentiles equal to the lowest and highest point estimates in the literature.[21] Using these distributional assumptions, 1000 sets of elasticities were drawn and used to calculate the probability distribution for the intervention level.

We use this Monte Carlo method to find the probability distribution of the deposit/refund necessary to achieve a 10 percent reduction in disposal of the wastes in the model. Figure 4 shows the cumulative distribution function of the deposit/refund for this 10 percent reduction. Our central calculation of the marginal cost of the deposit/refund for a 10 percent reduction is $45 per ton. As indicated in Figure 4, the 90 percent confidence interval lies between $28 per ton and $56 per ton.

These results suggest that our calculations are fairly robust to uncertainty about supply and demand elasticities. One reason for this robustness is the importance of pre-intervention prices in determining costs for small reductions in waste disposal. The price of the final good to consumers indicates the marginal benefits of the good to them and hence the marginal cost of the first unit of source reduction. Similarly, the price of the recycled good indicates the cost that was borne for the marginal unit of recycling prior to intervention and thus the cost of expanding this activity. These prices are observable in the market and not subject to the uncertainty of our supply and demand elasticities. However, the prices do vary substantially over time, making the model's recommendations sensitive to the baseline year.

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[20] We require that all draws be positive for supply elasticities and negative for demand elasticities. When a draw has an inappropriate sign, we assign it a value of zero.

[21] The best fit was chosen to minimize the mean square error at these points.

Figure 4 is available from the authors
at Resources for the Future.

V. EFFICIENT SOLID WASTE REDUCTION FROM A COST-BENEFIT PERSPECTIVE

This section compares our numerical calculations for the marginal cost of disposal reduction through a deposit/refund with estimates of the benefits of disposal reduction. This comparison suggests the extent to which current levels of disposal are excessive. Unfortunately, there are no reliable estimates of these benefits; this section discusses the information that is available to provide a general comparison with our estimates.

Reducing waste generates benefits by avoiding the costs associated with disposing waste in a landfill or incinerator.[22] The avoided costs include the direct monetary costs of the landfill or incineration reflected in the "tipping fee," the price charged to dispose of a ton of waste. In addition, there may be costs that the tipping fee does not internalize: such costs include environmental consequences and nuisance costs imposed on the community. Although tipping fees are observable, it is unclear what share of the social cost of waste disposal these fees capture.

Tipping fees at municipal solid waste landfills in the U.S. in 1992 averaged $33 per ton (Repa, 1993). The variance in tipping fees is large, however, with a range in 1992 from $4 per ton in New Mexico to $74 per ton in New Jersey (Steuteville and Goldstein, 1993). Tipping fees at waste-to-energy facilities (incinerators) are substantially higher, averaging approximately $56 per ton in 1993.

As a measure of the marginal benefits of waste reduction, we would like to know the marginal avoided social disposal cost at new state-of-the-art landfills that meet all RCRA

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[22]As stated at the outset, we consider the goal of these policies to be correction of the inefficiencies generated by a zero charge on disposal (i.e., reduction in the social costs of *disposal* only) rather than environmental costs at other stages in the product's life-cycle. Some analysts argue for a "life-cycle" approach that would consider upstream environmental costs, such as the pollution associated with production and transportation of goods, and factor these costs into a charge on waste disposal (Ackerman, 1993; U.S. Environmental Protection Agency/Office of Solid Waste and Emergency Response, 1995a). However, many analysts suggest that life-cycle assessments are, for the most part superfluous: market prices, in combination with existing environmental regulations, already reflect the resource costs measured in life-cycle assessments (Arnold, 1993; Portney, 1993/94; Menell, 1993). To the extent that some environmental externalities are not internalized, these authors and others (see Fullerton and Kinnaman, 1995; and Macauley and Walls, 1995) argue that policies that deal with environmental problems *at their source* -- for example, by setting taxes or standards on air or water emissions from a manufacturing process -- are likely to be more efficient than solid waste policies which are several steps removed.

Subtitle D requirements. These landfills have been designed to have low environmental costs and hence the tipping fees at these facilities may be close to the social cost of waste disposal.[23] Franklin Associates and BFI, Inc. estimate these costs to be approximately $24 per ton (cited by Ackerman et al., 1995). Ackerman et al. also argue that avoided transfer and transportation costs of $6 per ton should be included in the avoided cost calculation. These estimates yield an average transportation and disposal cost of $30 per ton. This average cost approximately equals the marginal cost because the cost of disposing an additional ton in a new landfill is constant over a wide range (Gallagher, 1994; U.S. EPA, 1995b).

In addition to environmental costs, there are other costs to local communities associated with waste. These costs include truck traffic and noise around the facility, the general unsightliness and occasional odor of a landfill, and the possible *perception* on the part of nearby residents that the facility presents a health risk. However, these nuisance costs are unlikely to increase the marginal cost of waste disposal greatly. Many costs may be incurred from the simple existence of the facility and not by its size; as a result the nuisance costs may not have a large impact on marginal waste disposal costs. Moreover, at many facilities, the so-called "host fees" that private landfills pay to their communities may internalize these costs.[24]

Thus, approximately $30-$33 per ton appears to be a reasonable estimate of the marginal avoided social waste disposal cost, although this estimate is subject to considerable uncertainty and fails to reflect large geographical variations in cost. If the government imposed this cost of waste disposal as a Pigouvian tax (through a deposit/refund), the policy would reduce the wastes included in the model by about 7.5 percent.

A $33 per ton fee is equivalent to the following deposits (and refunds) on individual materials:

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[23] Environmental costs include the leaching of toxic substances into groundwater and soil and the accumulation of methane, as well as odors and other problems. Subtitle D landfills reduce these costs with measures such as excluding hazardous wastes, installing liners to prevent leachate, covering each day's waste with dirt or other material, monitoring groundwater and methane gas, and ensuring that there will be post-closure care of the facility.

0.06 cents per aluminum can (1.4% of can price)
0.24 cents per steel can (7.3% of can price)
2.4 cents per newspaper (7% of newsprint price)
0.24 cents per plastic milk jug (4.4% of milk jug price)

Thus, the deposit/refunds would be much lower than current state bottle bills but would apply to a much larger universe of wastes. As in our model, such deposit/refunds could be levied on manufacturers of consumer products and collectors of recycled materials. This upstream application of the deposit/refund might reduce the transaction costs associated with implementing this policy relative to conventional retail-level deposit/refunds such as bottle bills.

VI. CONCLUSIONS

In this study, we analyze the costs of three economic incentive policies for reducing disposal of municipal solid waste: deposit/refunds, recycling subsidies, and advance disposal fees. We find that a deposit/refund is significantly less costly than either a recycling subsidy or an ADF. However, high administrative costs might alter this conclusion, making an ADF appear more attractive.

Our analysis also suggests that a modest reduction in municipal solid waste would be efficient if it could be accomplished without large administrative and transactions costs. We consider the marginal social benefits of waste reduction to result from avoided disposal and transportation costs. These avoided social costs currently amount to approximately $33 per ton, although the costs vary substantially by region. This marginal benefit implies that a 7.5 percent reduction in the wastes in our model would have been optimal in 1990 if the reduction were accomplished by a deposit/refund. Other wastes not included in the model might be reduced in the optimum as well, so the total percentage reduction in municipal solid waste remains to be determined.

Monte Carlo analysis that varies the elasticities used in the model reinforces the conclusion that a modest reduction in waste disposal might be desirable. Indeed, there may be greater uncertainty about the benefits of waste reduction than its marginal costs. Thus, our study highlights the need for more research on the social benefits of waste reduction.

Appendix A: Distributional Assumptions and Sources for Elasticity Estimates

Material	**Elasticity**		
	Final Demand	**Secondary Demand**	**Secondary Supply**
Paper and Paperboard			
Newsprint	N(-.301,.279) Bingham et al. (1993)	N(-.122, .003) Nestor (1992)	Mid-value: -.20 Log-Normal with 90% confidence interval: .06 to 1.70 Literature[a]
Writing and Printing	N(-.949, .0645) Bingham et al. (1993)[b]	N(-.160, .012) Anderson and Spiegelman (1976)	(As above)
Paperboard Containers	N(-.463, .0149) Bingham et al. (1993)[b]	(As above)	(As above)
Other Paper	N(-.594, .972) Bingham et al. (1993)[b]	No recycling	No recycling
Glass			
Beverage Containers	Mid-value: -1.0 Log-normal with 90% confidence interval: -.1 to -2.0	Mid-value: -1.0 Log-normal with 90% confidence interval: -.1 to -2.0	Mid-value: .5 No dispersion
Durables	(As above)	No recycling	No recycling
Aluminum			
Beverage Containers	Mid-value: -1.4 Log-normal with 90% confidence interval: -.7 to -2.0 Suslow (1986)[c]	Mid-value: -.805 Log-normal with 90% confidence interval: -.73 to -.88 Suslow (1986)	Mid-value: 1.1 Log-normal with 90% confidence interval: .8 to 4.3 Literature[d]
Other Containers and Packaging	(As above)	(As above)	(As above)
Durables and Misc. Nondurables	(As above)	(As above)	(As above)
Steel			
Cans	Mid-value:-.63 Anderson and Spiegelman (1976) No dispersion	Mid-value: -.63 Anderson and Spiegelman (1976) No dispersion	Mid-value: 1.4 Anderson and Spiegelman (1976) Log-normal with 90% confidence interval: 0.1 to 3.9 ICF-1979
Other Containers and Packaging	(As above)	(As above)	(As above)
Durables	(As above)	(As above)	(As above)
Plastics			
PET Soft Drink Bottles	N(-2.05, 5.29) Authors' estimates	Mid-value: -.1 Log-normal with 90% confidence interval: -0.05 to -0.5	Mid-value: .5 Log-normal with 90% confidence interval: .1 to .8
HDPE Liquid Food Containers	N(-1.20, 1.17) Authors' estimates	(As above)	(As above)

Other Plastic Nondurables	Mid-value: -1.0 Log-normal with 90% confidence interval: -0.5 to -2.0	No recycling	No recycling
Durables	(As above)	No recycling	No recycling

Notes:

When an econometric estimate was the basis for the elasticity, the distribution is assumed to be normal with mean and variance as shown. All normal distributions are truncated at zero, so that supply and demand elasticities have the anticipated sign. When the table presents an interval, the Monte Carlo analysis assumes a lognormal distribution with this range as the 90% confidence interval.

(a) Secondary supply elasticity estimates for paper range from .06 estimated by Edgren and Moreland (1989) to 1.70 estimated by ICF-1979 (cited by Bingham et al, 1983).

(b) Paper demand elasticities (except newsprint) are production-weighted averages of elasticities for more disaggregated categories by Bingham, et al (1993).

(c) The final aluminum demand is the average of primary and secondary demand elasticities estimated by Suslow. The 90% confidence interval spans the two elasticities.

(d) Secondary aluminum supply elasticity ranges from .8 estimated by Bingham et al. (1983) to 4.3 estimated by ICF-1979.

Appendix B: Estimates of the Demand for HDPE and PET Bottles

Estimates of the price elasticities of supply and demand for plastics did not exist in the literature, perhaps partly due to a lack of available data. We assembled data on PET and HDPE prices and quantities for the period 1979-1994. Prices are from Plaspec for both types of plastics, while data on resin sales are from the January issues of *Modern Plastics*. Recycling of plastic containers has occurred only since about 1979. Although we were able to find PET recycling quantities back to that point, we were unable to match it with prices. We were also unable to obtain HDPE recycling information for a sufficient number of years. As a result, we estimated only final goods demand equations.

We estimated instrumental variables regression equations for the demand for HDPE and PET. In both models, the quantity demand depends on its own price and GDP as a measure of income. The equations also include total expenditures on the principal good packaged in that type of plastic (milk for HDPE and soda for PET) and prices of substitutes (paper prices in the HDPE model and aluminum and glass prices in the PET model). All prices and expenditures are in real dollars. In the PET regression, the estimated coefficients on the aluminum and glass price variables often had the wrong sign and were not statistically significant in any estimated equations; therefore, the equations shown below exclude these variables. We use current and lagged oil prices as instruments for the resin prices because oil is a major production input and thus should shift the supply curve.

The estimates for PET are as follows (the absolute value of t-statistics are in parentheses):

$$LPETQ = \underset{(1.04)}{-17.56} - \underset{(0.89)}{2.05}\,LPETPRICE + \underset{(3.08)}{2.41}\,LGDP + \underset{(0.77)}{0.54}\,LSODA \tag{B.1}$$

$R^2 = 0.95$

number of obs. = 15

where *LPETQ* is the natural log of annual PET resin consumption; *LPETPRICE* is the natural log of average annual PET resin price; *LGDP* is the natural log of per capita GDP; and *LSODA* is the natural log of total annual expenditures on nonalcoholic beverages. All coefficients have the expected sign but only the coefficient on *LGDP* is significant at the 5 percent level.

Similarly the estimated equation for HDPE is:

$$LHDPEQ = \underset{(1.40)}{31.63} - \underset{(1.11)}{1.20}\, LHDPEPRICE - \underset{(0.63)}{0.83}\, LGDP - \underset{(1.61)}{1.32}\, LMILK + \underset{(1.24)}{2.22}\, LPBDPRICE \quad (B.2)$$

$R^2 = 0.86$

number of obs. = 15

where *LHDPEQ* is the natural log of annual HDPE resin consumption; *LHDPEPRICE* is the natural log of average annual HDPE resin price; *LMILK* is the natural log of total annual expenditures on milk; and *LPPRPRICE* is the average annual producer price index for paperboard (a substitute). The coefficients on the both prices have the expected signs but none of the coefficients in the equation is significantly different from zero.

The price elasticities of demand, which are the coefficients on *LHDPEPRICE* and *LPETPRICE*, both have the correct sign and seem to be a reasonable order of magnitude. A one percent increase in the price of PET resin leads to a 2 percent reduction in demand for PET. A one percent increase in the price of HDPE leads to a 1.2 percent reduction in demand for HDPE. These estimates are reported in Table 2 and are used to calibrate our solid waste disposal model.

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