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THE ECONOMICS OF RESIDENTIAL SOLID WASTE MANAGEMENT

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ABSTRACT

This paper provides a broad overview of recent trends in solid waste and recycling, related public policy issues, and the economics literature devoted to these topics. Public attention to solid waste and recycling has increased dramatically over the past decade both in the United States and in Europe. In response, economists have developed models to help policy makers choose the efficient mix of policy levers to regulate solid waste and recycling activities. Economists have also employed different kinds of data to estimate the factors that contribute to the generation of residential solid waste and recycling and to estimate the effectiveness of many of the policy options employed.

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The Economics of Residential Solid Waste Management

1. Introduction

The market for residential solid waste management and disposal has experienced dramatic changes over the past 20 years. In the early to mid 1970's, most towns used local garbage dumps. Even though recycling was well known and utilized by the commercial and industrial sectors of the economy, residential recycling was limited to spontaneous collection drives by charitable organizations for old newspapers and aluminum cans. Today, 46% of Americans have access to municipal curbside recycling programs, many other Americans have local access to drop-off recycling facilities, and garbage is often transported tens, hundreds, or even thousands of miles for disposal in a large regional landfill. Recycling has also become more popular in Europe and in other parts of the world.

These market shifts have attracted the attention of economists who have devoted significant attention to understanding the causes and impacts of these events. Economists have also participated in discussions aimed at shaping efficient solid waste policy strategies. This survey article summarizes the economic literature devoted to household solid waste collection and disposal. The next section provides a brief historical introduction to these markets. Section 3 surveys the theoretical literature devoted to suggesting the best way to regulate garbage collection and disposal. Section 4 follows with a summary of solid waste policies in place, and it surveys the empirical studies devoted to those policies. Since household disposal choices determine garbage and recycling totals, Section 5 develops a model of household behavior that generates hypotheses that are subsequently tested by the empirical economics literature.

2. Recent Trends in Residential Solid Waste

The editors of *Biocycle Magazine* (Glenn, 1998) began an annual survey of the 50 states in 1989. Included in these surveys were state estimates of the quantity of solid waste landfilled, incinerated, and recycled in that state. Figure 1 summarizes the total use of these three methods of waste removal over the past decade. Although the percentage of household solid waste incinerated remained near 10% over the last decade, the percentage disposed in a landfill decreased from roughly 85% in 1989 to just over 60% in 1997. This decrease was associated primarily with the simultaneous increase in recycling. As illustrated in Figure 1, the United States recycled nearly 30% of waste in 1997, up from just 10% in 1989.

How were the states able to increase the recycling rate so dramatically over this time period? The *Biocycle* surveys also show that the number of curbside collection programs in operation nationwide increased monotonically from just 1,000 programs in 1989 to nearly 9,000 programs in 1997. Local governments administer all of these programs either by collecting the material directly or by contracting with a single private firm. Growth in the number of programs has steadied of late.

Economists have debated the extent to which the growth in curbside recycling can be attributed to economic factors such as increases in disposal costs or non-economic factors. Although this debate is explored more thoroughly below, we now introduce two important economic variables at play. Figure 2 presents average tipping fees in several states, and Figure 3 presents average prices of recycled materials in the United States over the past 10 years. Tipping fee data were obtained from *Biocycle*'s annual survey of the 50 states (Glenn, 1998). Rather than presenting the average for each state, Figure 2 illustrates the past 10 years' nominal tipping fee for one state from each region of the country. Two lessons can be drawn from this figure. First, the overall trend for tipping fees is weakly positive. But accounting for increases in the general price level, the real tipping fee may not have changed much over the past decade. Therefore, attributing the national rise in curbside recycling to increases in the tipping fee is difficult to support with such casual use of data. However, tipping fees in the northeastern region (New Jersey) are greater than in other regions of the country. And, indeed, curbside recycling programs have become popular in the northeast. Perhaps, then, tipping fees have played an indirect role in encouraging recycling.

The second variable of interest to economists is the price paid for recycled materials. The Bureau of Labor Statistics' data on the prices of corrugated cardboard, old newspaper waste, and scrap aluminum appear in Figure 3. Two lessons can also be learned from Figure 3. First, when accounting for increases in the general price level, the prices of recycled materials have remained rather constant over the past decade (Ackerman, 1997). Second, prices of recycled materials are highly variable over time. For old newspaper, six spikes have appeared over the past 30 years (not all are illustrated in Figure 3). The most recent spike was in 1995 when the price for old newspaper (and many other materials) hit all-time highs. This latest spike has been attributed to new recycled-content laws passed by several state governments (Ackerman, 1997). But overall, these trends do not appear to support the argument that economic forces are responsible for the growth in curbside recycling. This debate is conducted more systematically in economic papers reviewed below.

The dramatic increase in the number of curbside recycling programs in operation in the United States could instead be a function of non-economic influences such as changes in voter tastes for the environment or purely political concerns. Misinformation may have contributed to the public's perception of a shortage of landfill space. This perception may have emerged in 1987, when the barge "Mobro", loaded with Long Island garbage, was unable to unload its cargo after repeated attempts (see Bailey, 1995 for a discussion of the incident). A wave of state and local legislation encouraging or mandating recycling was passed soon after this incident.

Is the United States running out of landfill space? Available landfill capacity is difficult to quantify, but the *number* of landfills in operation can be ascertained and reported quite easily. Figure 4 illustrates the number of landfills (in thousands) operating each year in the United States over the past decade. This number has been steadily decreasing by about 500 landfills each year. Voters could have confused these data with a national shortage in landfill space (Bailey, 1995). While the *number* of landfills has been steadily decreasing over the past 10 years, the estimated *capacity* of remaining landfills has been steadily rising. Based on state-reported estimates (also illustrated in Figure 4), the remaining capacity of landfill space has doubled from roughly 10 years of remaining capacity in 1988 to 20 years in 1997.

The reason for these dual trends has been the replacement of small local town dumps with large regional sanitary landfills. This trend is due mostly to Subtitle D of the Resource Conservation and Recovery Act (RCRA) of 1976. This law was designed to reduce the negative externalities associated with garbage disposal. This law imposed technology-based standards on the construction, operation, and closure of solid waste landfills. Landfills are now required to install thick plastic linings along the base, collect and treat leachate, monitor groundwater, and cover garbage within hours of disposal. Because the fixed costs of constructing and operating a landfill have increased, cost-minimizing landfill sizes increased and fewer landfills have been built. The trend towards large regional landfills may also have been brought on by heightened public awareness over the siting of a landfill in their "back yard". Expanding an existing landfill could be politically more feasible than constructing a new one.

A final general development over the past decade has been the slight increase and subsequent decline of incineration as a method of garbage disposal. Figure 5 illustrates the number of incinerators in operation in the United States over the past decade. Incineration, once considered a dual solution to the solid waste and energy crises, reached a peak in 1991 when 170 incinerators operated nationally. Since then, the number of incinerators in operation has gradually decreased. This decline has been attributed to a number of factors, but most notably the quantity of garbage available to incinerators became lower than expected. If fixed costs are high, then average costs can be reduced with an increase in garbage throughput. But incinerators could not lower tipping fees to levels necessary to encourage more garbage without incurring financial losses. Therefore, many local governments passed laws requiring all local garbage be brought to the incinerator, effectively giving the incinerator monopsony power over local garbage. But the Supreme Court struck down these laws, exposing the incineration industry to competition from cheaper landfills. The Supreme Court dealt a second blow to the incineration industry when it ruled that incinerator ash is toxic and must be disposed in an expensive toxic waste landfill. The increased use of recycling in the early 1990's further reduced the quantity of garbage available to incinerators, adding to their financial dilemmas. Finally, policymakers were not eager to rescue the industry once the public began to oppose the resulting air pollution emitted by incinerators.

Where land is scarce, however, incineration has become a more viable option. The northeastern portion of the United States incinerates 40% of its waste. Incineration is also popular in Japan and several European countries where population densities are large and land values are high. Table 1 indicates the percentage of waste that is landfilled in several European countries in the middle 1980's (the remaining portion is incinerated). Greece, Ireland, and the U.K. rely almost exclusively on landfills. But Switzerland, Sweden, and Denmark rely on incineration to manage the bulk of their garbage. Facing less competition from land-intensive landfills, incinerators in these countries as well as in the Northeast region of the United States can capture the economies of scale necessary to keep the average costs of incineration down (Halstead and Park, 1996). But even though many countries rely heavily on incineration, Brisson (1997) finds the private and full external costs of incineration exceed those associated with landfill disposal in most European countries.

3. The Optimal Policy in the Theoretical Literature

This section reviews the economic literature devoted to designing solid waste management policies to achieve the efficient quantity of garbage and recycling. A skeletal model is developed here to frame discussion of optimal policy design. Notation developed for this model will be used throughout this review.

Assume that N identical households each maximizes utility that is defined over consumption (c). Consumption produces waste that must either be disposed as garbage for collection at the curb (g) or recycled (r). We use $c = c(g, r)$ to represent the various combinations of g and r that are consistent with any particular level of consumption. Given prices paid for consumption (p_c), and garbage collection (p_g), and received for recycled materials (p_r), the household with income (y) will make disposal decisions to maximize utility (u),

$$u = u(c) = u[c(g, r)]$$

subject to the budget constraint,

$$y = p_c c(g, r) + p_g g - p_r r.$$

Producers in the model choose virgin (v) and recycled (r) inputs to produce c according to the production function $c = f(v, r)$. Given input prices p_r and p_v (for recycled and virgin materials, respectively), the producer chooses inputs to maximize profit,

$$\pi = p_c f(v, r) - p_v v - p_r r.$$

Firms in this model would employ virgin and recycled materials so that the ratio of input prices equals the ratio of marginal products. Households would choose between garbage and recycling in a similar manner. In fact, it is easy to show that since agents in this simple model internalize all of the costs and benefits of their choices, resources are allocated efficiently and the optimal quantities of garbage and recycling are produced. But the total amount of solid waste disposed ($G = Ng$) could emit foul odor, pollute groundwater, create an eyesore, or contribute to climate change.[1] Household utility could be impacted by these effects, so assume now that $u = u(c, G)$, where $u_G < 0$. Under this assumption, households fail to internalize the full social costs of their disposal decisions. Too much garbage and too little recycling is produced by a decentralized economy.

In order to internalize disposal costs, economists have suggested several tax and subsidy schemes. This section will review the economic literature devoted to designing the tax/subsidy policy that can achieve the efficient allocation of resources in the presence of external costs from garbage disposal. Households could be taxed on each unit of garbage disposed (at rate t_g) or subsidized for their recycling effort (at rate s_{hr}). Households could

[1] An estimated 6% of the world's emissions of methane (a greenhouse gas) are released from landfills (Beede and Bloom, 1995).

also be required to pay an advanced disposal fee at the time of purchase (t_c). Under these policy schemes, households maximizes utility,

$$u = u[c(g, r), G]$$

subject to the amended budget constraint,

$$y = (p_c+t_c)c(g, r) + (p_g+t_g)g - (p_r+s_{hr})r.$$

The producer's use of virgin material could be taxed (t_v), or use of recycled materials could be subsidized (s_{fr}), resulting in the profit function,

$$\pi = p_c f(v, r) - (p_v+t_v)v - (p_r-s_{fr})r.$$

Economic research reviewed below has found that various combinations of these policies (t_c, t_g, s_{hr}, s_{fr}, t_v) can encourage a decentralized economy to achieve an efficient allocation of resources. Command and control policies such as mandatory household recycling ordinances and minimum recycled-content standards on producers can also achieve efficient outcomes in theory. But economists rarely support such forms of policy because the information required to achieve efficient outcomes is not likely to be available to policymakers. The literature devoted to the study of command and control policies is not rich.

The most direct approach to internalizing the external costs of garbage disposal is to tax each bag of garbage presented by the household (t_g). Most households have traditionally either paid for garbage removal with a flat monthly or quarterly fee, or through local property or income taxes. Households that contribute large quantities of garbage therefore pay the same as a household that contributes smaller quantities, so the cost per bag (p_g+t_g) is zero, even though the social marginal cost of that extra bag is greater than zero. The implementation of a tax (also called a user fee) on each bag of garbage can require households to internalize the social marginal collection and disposal costs.

Using a panel of twelve cities with direct pricing, Jenkins (1993) estimates that pricing garbage according to its social marginal cost would reduce the quantity of garbage produced by households and therefore improve social welfare by as much as $650 million per year, roughly $3 per person per year. Fullerton and Kinnaman (1996) use household data and also estimate the potential benefits of marginal cost pricing to be in the neighborhood of $3 per person per year. Podolsky and Spiegel (1998) study a cross-section of towns in New Jersey and estimate the economic benefits of charging per unit of garbage to be as great as $12.80 per person per year.

One particular advantage of taxing garbage directly (employing a user fee) is that other tax instruments discussed above are unnecessary for achieving the efficient allocation of resources (Fullerton and Kinnaman, 1995, and Palmer and Walls, 1994). Households may recycle, compost, or engage in source-reduction according to the private costs they face. As long as households face the full social cost of their disposal decisions, they will make those decisions efficiently. Any increase in recycling can reduce the price of recycled materials, making these materials more attractive to manufacturers without a direct tax on virgin materials or subsidy to recycling. In fact, Dinan (1993) finds that a tax on virgin materials (t_v) in combination with a user fee would not be efficient, since the same material is effectively taxed twice. Another advantage of taxing garbage directly is that the only information needed by the local policy maker is the full social cost of each bag of garbage. Repetto et al. (1992) estimate this cost to be $1.43-$1.83 per bag, depending on local private and social disposal costs.[2] Finally, Fullerton and Wu (1998) show that pricing garbage according to its social marginal costs can also encourage firms to produce the optimal amount of packaging per unit and to engage in the optimal amount of green design.[3]

Perhaps in response to these arguments, an estimated 4000 communities in the United States have started to price garbage directly (Miranda and Bynum, 1999). These programs levy a fee on each bag of garbage collected from each household. Garbage collectors can exclude non-payers by utilizing some method of identifying who has paid, such as requiring households to purchase specially marked bags, tags, or stickers.

Several arguments against the use of direct marginal cost pricing of garbage have also appeared in the economics literature. First, taxing garbage may be problematic if illicit or illegal dumping on the part of households is encouraged.[4] Second, the administrative costs of implementing the program may exceed the social benefits estimated above. Fullerton and Kinnaman (1996) estimate that the administrative costs of printing, distributing, and accounting for garbage stickers in Charlottesville, Virginia could exceed the $3 per person per year benefits mentioned above. Third, a uniform tax on all types of garbage may be inefficient if materials within the waste stream produce different social costs (Dinan, 1993). If, for example, the social cost of disposing flashlight batteries is greater than that of old newspapers, then the disposal tax on flashlight batteries should exceed that on old newspapers. But such a precise tax scheme is costly to administer.

To respond to these problems, Dobbs (1991) and Fullerton and Kinnaman (1995) develop models that suggest that if households have the option to litter or dump their garbage, and if the external costs of littered garbage exceeds that of legally-disposed garbage, then the optimal tax on legal garbage disposal (t_g) could be negative. That is, legal garbage disposal should be subsidized. In fact, if the administrative costs of levying a tax

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[2] This estimate is comprised of private and external collection and disposal costs (including a depletion allowance). The external costs are based somewhat on the work of Stone and Ashford (1991) and the Tellus Institute (1991).

[3] Kennedy and Laplante (1994) also develop a model that suggests garbage should be priced at its social marginal cost. But, if governments must balance the disposal *portion* of their budget (and lump sum taxes are not available), then the optimal policy may change. In particular, if the social marginal cost of waste disposal is greater than the household's marginal cost of dumping, then the user fee should be set just equal to the household's private marginal cost of dumping, and the subsidy for recycling should be lowered.

[4] Fullerton and Kinnaman (1996) estimate that 28% of the reduction in garbage resulting from pricing garbage at the curb may have been dumped. Jenkins (1993), Blume (1991), and Miranda and Aldy (1998) also find evidence of increased dumping. A number of other studies find minimal changes in dumping, including Podolsky and Spiegel (1998), Strathman et al. (1995), Miranda et al. (1994), Miranda and Bauer (1996), and Nestor and Podolsky (1998).

on each bag of garbage are significant, then the optimal policy may involve subsidizing garbage at its full price (set $p_g+t_g=0$). Policymakers can instead implement other policies defined below to achieve efficient disposal choices.

If taxing or even pricing garbage directly is problematic, economists have studied whether the implementation of a tax on virgin materials (t_v) can achieve the efficient allocation of resources in a world where garbage disposal produces external costs. Such a tax could increase producer's demand for recycled inputs, drive up the price paid for recycled materials, and thus increase the economic benefits to households that deliver recyclable materials to secondary markets. Miedema (1983) finds that a tax on virgin materials (t_v) set equal to the social marginal cost of disposing any resulting waste material produces welfare gains greater than would result from a subsidy on producer's use of recycled materials (s_{fr}), a direct tax on household solid waste (t_g), or an advanced disposal fee (t_c). The main advantage of virgin materials tax is that it both discourages the economy's use of virgin materials (resulting in less subsequent solid waste) and encourages the development of the market for recycled materials.

Others studies have questioned the use of a tax on virgin materials. Dinan (1993) finds that although a tax on virgin materials encourages the use of recycled materials in industries where the recycled input is a substitute for the taxed virgin input, other industries that do not use the taxed virgin input will not increase demand for recycled materials. For example, farmers could use old newspapers for animal bedding, but a tax on paper manufacturer's use of virgin wood pulp will not encourage this form of recycling. Dinan (1993) also suggests that a domestic tax on virgin materials does not encourage exporters to purchase recycled materials. Significant portions of recyclable paper are currently exported.

Palmer and Walls (1994) develop a model that suggests that although a tax on virgin materials can encourage the efficient mix of inputs, it can discourage production and consumption in the overall economy. The result is an inefficiently low quantity of garbage. Therefore, the virgin materials tax is only efficient when combined with a subsidy on the sales of final goods. Only for the special case where the marginal product of recycled materials is exactly one (1) can a tax on virgin materials lead to the efficient input mix and output level. Finally, both Fullerton and Kinnaman (1995) and Walls and Palmer (1997) find that as long as other policy options are available (namely a deposit/refund system discussed below), then a tax on virgin materials is only necessary to correct for any external costs associated with cutting or extracting the virgin material. The tax is not needed to correct for the external costs associated with garbage disposal.

Palmer and Walls (1994) find that a recycling subsidy (s_{hr} or s_{fr}) by itself can indeed provide the efficient input mix (between virgin and recycled inputs), but it leads to excess production, consumption and waste. Therefore, the subsidy to recycling must be combined with a tax on consumption (t_c). But the implementation of an advanced disposal fee (t_c) by itself only encourages source reduction, not recycling. Only the combination of an advanced disposal fee and a subsidy to recycling encourages both source reduction at the time of production and recycling at the time of disposal (Palmer et al., 1997). This policy is essentially a deposit/refund system.[5]

[5] Palmer et al. (1997) find that a 10% reduction in solid waste can be achieved with a

Several economic studies have favored the use of deposit-refund systems to correct for the external costs associated with garbage disposal, including Dinan (1993), Dobbs (1991), Fullerton and Kinnaman (1995), Palmer and Walls (1994), Palmer et al. (1997), Fullerton and Wu (1998), and Atri and Schellberg (1995). To achieve the efficient allocation, the deposit is set equal to the social marginal cost of disposing the resulting material, and the optimal refund is set equal to the difference between the marginal external cost of garbage and the marginal external cost of recycling. If the external costs of recycling are zero, then the refund matches the deposit. The deposit could be levied either on the production or the sale of goods. As long as transaction costs are low, the refund can be given either to the households that recycle the materials or to the producers that use the recycled materials in production. If the refund is given to the households, then the supply increase will drive down the price of recycled materials to firms. If the refund is given to firms, firms will increase demand for recycled materials and drive up the price received by households (Atri and Schellberg, 1995). In addition, Fullerton and Wu (1998) find that the refund given under a deposit/refund system will encourage firms optimally to engineer products that are easier to recycle. Households will demand such products in order to recycle and receive the refund. This result is important since directly encouraging the recyclability of product design can be administratively difficult.

Economists have also discussed some implementation issues related to a deposit/refund system. Palmer and Walls (1994) argue that a deposit/refund system would be easier to implement than a tax on virgin materials with a subsidy to consumption (an alternative policy combination that could also achieve the efficient outcome). Firms could organize a strong defense against the implementation of a tax on virgin materials. Households may lack this political organization. Furthermore, the subsidy to recycling may earn the support of households with strong tastes for the environment. Also, less information is necessary to implement the deposit/refund system efficiently. The policy maker only needs to know the marginal social cost of waste disposal. The optimal deposit and refund need only be set equal to this value. The application of a virgin materials tax on the other hand requires information on each firm's technical rate of substitution between recycled and virgin inputs. This type of information is normally not available to the policy maker (Palmer and Walls, 1994). If the administrative costs associated with operating the deposit/refund programs are high, then Dinan (1993) suggests that policymakers could single out products that comprise a large segment of the waste stream (newspaper) or that involve very high social marginal disposal costs (batteries). Palmer and Walls (1999) argue that a tax on produced intermediate goods combined with a subsidy paid to collectors of recycling would preserve the efficiency effects of a deposit-refund system but would be less costly to administer.

$45/ton deposit/refund system, an $85/ton advanced disposal fee by itself or a $98/ton recycling subsidy by itself. The latter amounts are larger because these policies must "work harder" to achieve the reduction in garbage since they do not encourage both source reduction at time of production and recycling at the time of disposal. For example, Starreveld and Van Ierland (1994) estimate that using only a disposal fee of $.30 per kilogram (roughly $272 per ton) of plastic will result in the recycling of 25% of disposed plastic in the Netherlands.

One "command and control" policy to receive the attention of environmental economists is a recycled content standard; a law requiring firms to employ a minimum portion of recycled materials in their products. Several states have passed such a law. Palmer and Walls (1997) point out the problems associated with a recycled content standard. First, recycled content standards can only achieve efficiency if carefully implemented with other policies. If recyclable materials are highly productive at the margin, but are not used because of their high price, then a recycled-content standard could increase production and therefore solid waste. A tax on consumption is also necessary. If recycled materials are unproductive on the margin, standards will decrease output (and solid waste) and will therefore require a subsidy to consumption to achieve efficiency. Their model also requires a tax on labor (the other input to production). Finally, the efficient implementation of a recycled-content standard requires information not ordinarily available to policy makers.

This section provided an overview of the economic literature on the best policy approaches to respond to the external costs of traditional garbage disposal. Although a direct tax on garbage disposal (t_g) and a tax on virgin materials (t_v) have been supported by some, the combination of an advanced disposal fee (t_c) and a subsidy to recycling (s_{fr} or s_{hr}) is supported by the majority of studies. The next section provides a survey of the current set of policies implemented by local, state, and the federal governments in the United States and across the world, and it discusses empirical lessons from the vast array of policies currently in place.

4. Solid Waste Policies - A Summary of Empirical Studies

This section provides a broad review of the various solid waste management policies implemented in the United States and abroad. The reader will quickly see that actual approaches used by policymakers often differ from the theoretical policy prescriptions detailed in the last section. The results of empirical economic papers related to each policy are discussed where available.

A. Policy Directives in the United States

1. Federal Government

The most influential disposal regulation passed by the Federal Government of the United States was the Resource Conservation and Recovery Act (RCRA) of 1976. Subtitle D of RCRA imposed technology-based standards on the construction, operation, and closure of solid waste landfills. Prior to RCRA, most every town in the United States had a local dump. These dumps were often formed near the edge of town, perhaps on a flood plain near a river.

Today's regulated landfills are constructed with a base of several inches of various grades of plastic lining to prevent leachate from seeping. Underground plumbing systems capture and treat leachate, and local groundwater supplies are continuously monitored. In terms of operation, garbage must be covered with soil within hours of disposal to reduce foul odor, discourage pests, and reduce the risk of health hazards. Many landfills capture and burn methane to produce electricity. Access roads must be watered several times each day to prevent dust from heavy truck traffic from rising. These regulations have decreased substantially the external costs associated with garbage disposal, but have also increased average disposal costs from an estimated $9 per ton to $20 per ton (Beede and Bloom, 1995).

Even with the recent advances in the technology of landfill construction and operation, local environmental activist groups still often oppose the creation or expansion of landfills in their region. Landfills depress property values. Housing values have been estimated to rise by 6.2% for each mile (up to two miles) away from a landfill (Nelson et al., 1992, as cited in Beede and Bloom, 1995). Roberts et al. (1991) interviewed 150 households in Tennessee and estimated households were willing to pay $227 per year to avoid having a landfill nearby. Reported amounts increase with income, education, and dependency on well water for water consumption.

A second Federal Government initiative that has influenced the market for the collection and disposal of household solid waste is the subsidy of virgin material extraction in the United States. First, income earned by the timber industry has been taxed at the capital gains rate instead of the corporate income tax rate. Second, the depletion of minerals extracted can be deducted from earned income as a form of depreciation. Third, mineral exploration has traditionally been encouraged on public lands. Fourth, freight rates charged for recycled materials have often been higher than for their virgin counterparts. These various forms of favorable tax treatments may have, on the margin, encouraged firms to utilize virgin inputs over recycled inputs, perhaps resulting in the current underdevelopment of the market for recycled materials.

Through a variety of papers, economists have learned a great deal about the market for recyclable materials. For example, Nestor (1992) reports that firms that could purchase recyclable materials are often capital intensive. Most of the existing capital stock is suitable for the use of virgin material in production. Re-tooling these industries to accept recycled inputs could be expensive. She also estimates the paper industry's price elasticity of demand for old newspapers. The short-run price elasticity of demand is estimated at only –0.0475. This elasticity increases to -0.0732 (1 year), -0.1009 (3 years), -0.1128 (5 years), and to –0.1216 in the "long run". These estimates are inelastic because the newsprint industry in many countries is equipped for the use of virgin fiber. The short-run marginal cost to the firm of using substitute inputs is high. The implication of an inelastic demand is that policies aimed at increasing the supply of old newspapers could indeed reduce their price but will not effectively increase the quantity of newspapers recycled. Furthermore, the elimination of existing tax subsidies on virgin inputs in the United States, Nestor (1992) reports, will also have little impact on the quantity of old newspapers recycled. The more effective approach would involve subsidizing the firm's purchases of capital equipment that would allow for the substitute use of both virgin and recycled inputs.

Anderson and Spiegelman (1977) also find the price elasticity of demand for scrap steel and old newspaper to be inelastic (-0.64 and –0.08, respectively). The elimination of tax advantages for virgin inputs is estimated to increase newspaper recycling by only

0.04% and scrap steel recycling by only 0.37%.[6] Anderson and Spiegelman (1977) also forecast that a subsidy to the suppliers of scrap iron (a 15% depletion deduction) would decrease the price of scrap steel by 7.2% and increase its quantity demanded by 2.9%. A similar subsidy to wastepaper suppliers (of 18%) would decrease the price of old newspapers by 8.6% but increase the quantity recycled by only 0.57%. A $10 per ton subsidy to the purchasers of old newspaper is forecasted to increase the quantity of newspaper recycled by only 2.0%. The common theme found throughout these empirical studies is the relative unresponsiveness of quantity demanded for recycled inputs to its price. Policies designed to increase the supply of recycled materials may have little impact on the quantity of recycled materials used in production.

One explanation given for the resistance on the part of many firms to make capital improvements to allow for the use of recycled materials has been the uncertainty over obtaining a steady supply of recycled materials. Prior to the widespread use of municipal recycling programs, the market's supply of recycled materials was highly variable. To determine whether tax or subsidy policies could stimulate the supply of recycled materials, several economists have estimated the effect of price on the quantity supplied. Most of these studies found the supply of recycled materials also to be inelastic. For example, Bingham et al. (1983) estimate the price elasticity of supply of glass (0.165), steel (0.372), and aluminum (0.730). Miedema (1976 - cited in Edwards and Pearce, 1978) also finds the price elasticity of supply of wastepaper to be inelastic (0.09). Ir Vander Kuil (1976 - cited in Edwards and Pearce, 1978) finds evidence that increases in the price of recycled materials simply shifts the source of the supplied materials from municipalities to volunteer scout groups.[7] But now that municipal governments supply the industry with a steady and predictable stream of recycled materials, firms may find the environment more conducive to invest in capital equipment suitable for recycled inputs.

2. State Governments

RCRA also assigned to the states the responsibility of regulating the market for household solid waste collection and recycling. The logic behind this action was based on the inherent differences in industry practices and environmental conditions across the states (Callan and Thomas, 1997). Delegating disposal authority to the states has resulted in a wide variety of policy approaches. Table 2 provides a glance at the policies enacted by each of the 50 states and the District of Columbia to increase recycling. The most common state action is to set a goal for recycling as a percentage of the solid waste stream. These goals range from 20% in Maryland to 70% in Rhode Island. The laws are ceremonious,

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[6] If, in the long run, virgin and recycled inputs are perfect substitutes, then the elimination of tax advantages for virgin input would still only increase newspaper recycling by 1.68% and scrap steel by 3.4%.

[7] Many of these empirical studies also uncover a negative relationship between a previous period's prices and current supply quantities. This relationship is explained by the use of stockpiling. If prices of recycled materials were low in a previous period, then firms may build up their inventories rather than sell at the low price. An increased inventory then increases supply in the current period. The assumption that suppliers stockpile materials to wait for higher prices has not been tested by the economics literature.

for the most part, since they rarely state the consequences of falling short. In fact, the strategy employed by many states facing a failure to achieve the goal is to delay the deadline. Kinnaman and Fullerton (1997) find no significant impact of these goals on recycling quantities.

States have also passed laws that set recycling guidelines for municipalities within the state. The strongest law requires all municipalities to implement curbside recycling programs *and* to pass local ordinances making household participation in the recycling program mandatory. Seven states, including Pennsylvania and New Jersey, have passed such laws. Seven other states have passed similar laws requiring municipalities to offer recycling programs to households, but do not require the implementation of mandatory ordinances. Finally, eight states have set recycling goals for each town or county to satisfy, but allow each town or county to decide how to go about achieving the goal.

In exchange for these various mandates, 34 states provide grants to localities to help finance the costs of recycling expenses. For example, in Pennsylvania, each municipality receives a state grant that is based on the total quantity of materials recycled. Although economists have not devoted attention to estimating the incidence of these various forms of state recycling mandates, anecdotal evidence indicates the laws are costly but have had a dramatic impact on the number of municipal recycling programs operating within these states.

An approach taken by 23 states to regulate household solid waste is to prevent yard waste from being disposed in landfills. Large composting facilities are usually established to accommodate yard waste more cheaply than disposal in landfills. Several other states have passed laws preventing materials such as automobile tires, batteries, motor oil and old appliances from entering landfills (not presented in Table 2). In one highly publicized example, the state of Maine banned the disposal of aseptic packaging (drink boxes) in landfills. The ban was repealed after a Tellus Institute study found them to be environmentally friendly relative to other drink containers (Ackerman, 1997).

The oldest policy implemented at the state level is deposit-refund systems for empty beverage containers. The state of Oregon was the first to pass this form of legislation in 1983. Eight other states have followed suit, though no state has implemented a new deposit-refund system since the early 1980's.

States quickly learned that their policies aimed at stimulating the supply of recyclable materials produced a glut of recycled materials (see a review of economic research on this topic above). To help balance the market, states began to implement policies designed to stimulate the demand for recycled materials. Twenty-nine states provide tax credits to encourage the production of new recycling plants, fifteen states provide low-interest loans for the same, and 29 states require government offices and in some cases private firms to purchase a minimum of their inputs from recycled products. As mentioned above, Palmer and Walls (1997) find recycled-content standards to be a difficult policy to implement and administer.

A final area of state intervention involves the use of restrictions on shipments of solid waste imported from other states. The transition from local dumps to regional landfills also brought an increase in the amount of solid waste transported across state and national boundaries. Today, an estimated 8% of all waste generated in the United States is disposed in another state. A few states, especially Pennsylvania, Virginia, and those in the

Midwest, have recently attempted to restrict the quantity of solid waste imported. Repeated attempts by these states to restrict the importation of garbage were struck down by the Supreme Court, which ruled that import restrictions violate the free flow of interstate commerce.[8] More recently, several governors have petitioned Congress to pass Federal legislation imposing import restrictions on interstate garbage shipments. Congress has yet to pass such legislation.

The top importer of solid waste in the United States is Pennsylvania, followed by Ohio, Virginia, Illinois and Indiana. In 1996, Pennsylvania received its waste from New York (3,300,000 tons), New Jersey (3,100,000 tons), Maryland (819,000 tons), Delaware (261,000 tons) and Connecticut (141,000 tons). Overall garbage imports to Pennsylvania have increased from 3.8 million tons in 1993 to 7.9 million tons in 1996. Similar growth rates have emerged in other importing states.

One reason state governments are frustrated with imported garbage is that their states have devoted significant public resources to reducing the quantity of solid waste generated within the state. As discussed above, resources have been devoted to implementing curbside recycling programs, banning certain materials from being disposed in landfills, providing tax advantages and/or subsidized loans to commercial recycling activities, and distributing grants to help run local recycling services. State officials may wonder what the state has gained by these efforts if the saved landfill space is filled by imports from other states. For example, in 1996 the state of Pennsylvania recycled 1.9 million tons of solid waste, but imported 7.9 million tons.

Traditional economic theory suggests free trading of garbage is efficient since those states with a comparative advantage in garbage disposal can specialize in garbage disposal. Any policy that interferes with the free flow of garbage would therefore be socially costly. Ley et al. (1997) estimate the loss in total surplus resulting from various restrictions on the flow of garbage considered by Congress. First, a $1 per ton surcharge on imported garbage would result in a 4% decrease in the quantity of garbage traded and a loss of total surplus of only $0.02 per person. The implementation of caps on the quantity of garbage traded across state lines (caps consistent with a Senate bill passed in 1995 that would require a reduction in garbage imports to 65% of their 1993 levels after a prolonged introductory phase) results in a surplus loss of $10 per person. Finally, a law that restricted all trading of garbage would result in a $18 per person loss in surplus. This study assumes that all external costs associated with garbage disposal are internalized through the tipping fee.

Other economic arguments can be made that flow controls improve welfare. Copeland (1991) provides two arguments in favor of restrictions of garbage imports. First, governments in some states (or countries) may not adequately regulate the industry to ensure that the external costs of garbage disposal are internalized. Total welfare can improve if exports from a highly-regulated country are prevented from entering a weakly-regulated country. Since landfill regulations across the United States are uniform, this

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[8] As an exception, U.S. courts have often applied the Market Participant doctrine that allows local governments to restrict out-of-state garbage from government-owned disposal sites. See Podolsky and Spiegel (1999) for a thorough review of the case law related to interstate garbage shipments.

rationale is probably more appropriate to inter-country shipments of solid waste. Second, even if regulations are uniform across trading partners, Copeland argues that restricting garbage trade can still improve welfare if evading the regulations is easier in one area than another. Also, Macauley at al. (1993) explain that allowing landfills the option to practice third-degree price discrimination (for example, charging a greater fee on imported garbage relative to local garbage) can be welfare improving if these landfills operate in imperfectly competitive markets. A landfill that can lower prices to local customers (with relatively elastic demand curves for garbage disposal) without having to lower prices to importers (with more inelastic demand curves) can make the local landfill and local residents better off without making the rest of the world worse off.[9]

Interestingly, while many state governments have attempted to restrict out-of-state garbage, other local governments have attempted to prevent local garbage from being exported from the area. As discussed above, such restrictions on garbage flow were designed to help support local incinerators that levy tipping fees that often exceed those of neighboring landfills. The Supreme Court recently struck down the use of such export restrictions. Tawil (1999) estimated that this event did not impact the profit levels of the participating incinerators or waste-hauling firms. Perhaps entry into the waste management industry is easy, eroding any profits that could have followed the Supreme Court's ruling. Finally, Podolsky and Spiegel (1999) argue that the existence of economies of scale in garbage disposal practices could in some cases merit restrictions on garbage exports. The local reduction in average disposal costs attributed to the increase in garbage brought on by the export restriction could exceed the increase in average disposal costs experienced by a distant site.

Public and academic attention devoted to the issue of flow controls may increase when the Fresh Kills Landfill on Staten Island closes in 2002. New York City currently disposes 13,000 tons per day (4.7 million tons per year) in the Fresh Kills Landfill, the largest landfill in the country. Given the recent 38% cut in New York City's recycling budget, all signs indicate that New York City's garbage will be exported to other states.

3. Local Governments

Markets for household solid waste collection and disposal were once decentralized. As cities began to develop, dumps often formed near the outskirts of each town, and households were typically responsible for transporting their own waste to this dump. To ensure that all garbage was removed from neighborhoods, and to help capture economies of density, many communities designating a single collector for household solid waste.

In the United States, this intervention has typically taken one of two forms. First, direct government provision meant that municipalities would purchase trucks, hire drivers, and define collection routes. The costs of this local service was typically financed out of general tax revenue or the issuing of monthly or quarterly bills to each household. Second, the local government could regulate a single private collector. The town could contract with a single firm to collect all garbage or it could award a franchise permission to collect

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[9] This conclusion is an application of more general findings related to the efficiency of Ramsey pricing.

garbage to a single private garbage collector. The main difference between these two latter forms is that under a franchise agreement the private collector bills the households rather than the town.

Town governments could also pass local ordinances requiring households to hire their own company. Although such competitive garbage systems still operate today, the single collector model is the norm. Dubin and Navarro (1988) estimate that 43% of communities in the United States rely upon contract or franchise agreements, 26% of municipalities operate municipal collection programs, and 30% rely on the competitive market.[10]

Economies of density suggest that a single collector could reduce the overall collection costs. Dubin and Navarro (1988) find that an increase in the population density by 100 persons per square mile decreases the average cost per ton of collected materials by $1.62. Kemper and Quigly (1976) estimate that competitive markets are 25% to 36% more expensive than a single collector, and that contract or franchise agreements reduce costs over municipal collections by another 13 to 30% (depending on the level of service). Stevens (1978) estimates that the contract or franchise agreements are 26% to 48% cheaper than a competitive private market and 27 to 37% cheaper than municipal provision (for cities over 50,000 population). Savas (1977) finds that municipal collection is 14% more costly than that by a single private firm. Bohm et al (1999) estimates that municipally-run curbside recycling programs are on average $82,000 more costly per year than private recycling programs. Finally, McDavid (1985) finds in Canada that public collection is 41% more costly than private collection. This difference is identified (by McDavid) to arise from the fact that workers in private firms receive productivity bonuses and private collectors are more likely to use larger trucks with smaller crew sizes.

Why don't all communities employ the most efficient contract or franchise method? Dubin and Navarro (1988) find that the community's choice of method depends upon the power of rent-seeking interest groups (such as labor unions) and the ideological preferences of the community. Conservative towns are more likely to rely on the free market than liberal towns, but liberal towns are more likely to use municipal collection rather than contract or franchise agreements.

Beyond the mere collection of household garbage, local governments have also attempted to influence the decisions of households to reduce the quantity of garbage collected and disposed. Drop-off and curbside recycling programs, unit-based pricing programs, and mandatory recycling ordinances have been passed. Although precise year-to-year data are unavailable, recent estimates indicate that over 9000 curbside recycling programs and 4000 unit-based pricing programs are currently in operation in the United States. Economic studies of the impact of these policies are summarized in Section 5 below.

At first, towns began to offer drop-off recycling services. Towns would usually purchase (or rent) a few large trailers, and would leave those trailers on municipal property,

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[10] In Canada, McDavid (1985) estimates that only 20.6% of cities with populations in excess of 10,000 use municipal collection, though another 37.3% rely partly on the municipality to collect household garbage at the curb and partly on private firms to compete for collection from commercial establishments and apartment buildings.

usually a parking lot or near the entrance of a park or other municipal property. Residents would voluntarily transport certain materials (usually newspaper, aluminum cans, and perhaps glass). Jakus et al. (1996) estimate that rural households devote an average of 90 seconds to recycle one unit of glass and one unit of old newspaper. Given the opportunity cost of household time, households paid $1.29 to recycle one pound of each material. Based on quantities recycled, Jakus et al. (1996) estimate that these households value local access to drop-off facilities at $5.78 per month.

As municipal governments gained expertise in the area of marketing recycled products, they began to implement curbside recycling programs. Curbside recycling programs decrease the household's time and effort devoted to recycling. Households are expected to respond by recycling more, while municipal governments collect more, save disposal costs, and earn greater revenues from the sale of materials. The external costs of garbage collection and disposal could also decrease. Powell et al. (1996) find that the costs associated with vehicle emissions, traffic accidents, and road congestion are much less for curbside programs (4.99 British pounds sterling per ton recycled) than for drop-off programs (22.95 British pounds sterling per ton recycled). Direct estimates of the impact of the implementation of curbside recycling programs on household disposal choices are presented in Section 5 below, where the disposal choices of the household are carefully modeled.

Are economic or non-economic forces responsible for the recent increase in the number of municipal curbside recycling programs? The answer is probably both. Tawil (1995) and Kinnaman and Fullerton (1997) estimate the probability of implementing a curbside recycling program. Tawil (1995) employs a cross-sectional database of 80 towns in Massachusetts to estimate that every $1000 that can be saved by curbside recycling increases the probability of adoption by 11%. But Tawil (1995) also finds that a 1% increase in the percentage of households belonging to an environmental interest group increases the probability of adoption by 4%. Kinnaman and Fullerton (1997) also uncover economic reasons for implementing a recycling program. The likelihood increases by .78% with a $1 increase in the tipping fee (from the average tipping fee of $26) and by .39% with a 100-person increase per square mile (from the average density of 2,600) since average collection costs could decrease with the population density (Bohm et al, 1999). However, non-economic variables also partly explain the move towards recycling. A one-percent increase in the percentage of the population with a college degree (from the average of 23.6%) increases the likelihood that a town implements curbside recycling by 0.77%.

Several economic studies have estimated directly the benefits and costs of curbside recycling programs. Most suggest that the costs of operating a curbside program exceed the benefits resulting from the subsequent decrease in garbage disposal costs and sale of collected materials. Franklin Associates (1994) use national cost averages to estimate that recycling costs the municipality $9.52 to $16.53 per ton more than the cost of landfill disposal. Other studies suggest recycling is much more costly. The Solid Waste Association of North America (SWANA, 1995) estimates it costs an extra $74 per ton to recycle in a sample of 6 communities. Kinnaman (1998) estimates that a recycling program costs an extra $55.45 per ton recycled. This estimate includes costs to firms that are required by local ordinance to recycle. Carroll (1997) uses cross-section data from Wisconsin to estimate that recycling costs over $140 per ton, roughly $100 more than the cost of disposing the material. Only Hanley and Slark (1994) estimate recycling to be economically beneficial for the recycling of newspaper in Scotland. Palmer et al. (1997) estimate the benefits of recycling exceed the costs if the recycling rate is less than 7.5% of total waste. Recycling beyond this threshold is costly.

Kinnaman (1998) and Jakus et al. (1996) estimate the political/environmental benefits of curbside recycling through use of contingent valuation surveys. In a survey of 100 households, Kinnaman (1998) finds that households are on average willing to pay about $86 per year to keep curbside recycling of newspaper, glass, and aluminum. Jakus et al. (1996) estimate that households are willing to pay $69.36 per year for curbside collection of newspaper and glass. In addition, Tiller et al. (1997) estimate that suburban households that classify themselves as recyclers are willing to pay $11.74 per month for drop-off recycling facilities. If such preferences influence the decisions of local officials, then some of the trend towards greater recycling may in fact be attributable to political or environmental forces.

Other studies have estimated the costs of curbside recycling programs. Judge and Becker (1993) estimate that such costs increase with the addition of weekly collection (as opposed to monthly) of commingled (rather than separated) material collected from the porch of households (rather than the curb). Carroll (1997) uses self-reported cost figures from 1,103 programs in the state of Wisconsin to estimate that the costs of curbside recycling programs increase with the population, the tons recycled, and the number of materials collected. Interestingly, Carroll does not find a relationship between population density and collection costs. Bohm et al. (1999) estimate the costs of recycling with data based on a national survey of 1,021 municipal recycling programs in the United States. They find that the average costs of recycling decrease with the quantity collected, indicating economies of scale in collection. The total costs of recycling are estimated to increase with the cost of labor, the cost of capital, and if the municipality collects the material rather than a private company. Butterfield and Kubursi (1993) also find that recycling is costly. Laws that require or encourage recycling in Canada are found to decrease employment levels in several industries.

Huhtala (1997) and Brisson (1997) break down the private and external costs of recycling by type of material. Huhtala develops a dynamic model of waste accumulation with recycling as a backstop technology. The model is simulated using 1993 data from the Helsinki region. Results show that the social benefits of recycling paper, cardboard, and metal exceed the social costs. Glass and plastic do not pass the benefit/cost criterion. Brisson (1997) finds that the recycling of aluminum produces the greatest social benefits, followed by glass, ferrous metals, paper board, and rigid plastic.

As described above, several states in America have implemented recycling goals. England has also set a 50% recycling goal and the Netherlands set a goal for plastics of 42%. Palmer et al. (1997) and Huhtala (1997) estimate the optimal recycling rate. Using the lowest cost policy to encourage recycling (a deposit-refund of $45 per ton), Palmer et al. (1997) find that only 7% of solid waste should be recycled in the United States (where the social marginal cost of garbage disposal is estimated to be $33 per ton). Huhtala (1997) find the optimal recycling rate to be between 31% and 52% in Finland (where the *private* marginal cost of garbage disposal is estimated at $101/ton). In addition, Huhtala (1997) adds a contingent valuation estimate of the non-market benefits of recycling to the analysis. Such benefits include the value of less air pollution from solid waste incinerators plus an estimate of the "environmental friendliness of recycling".

To ensure participation in the curbside recycling program, some local governments have passed a local ordinance making it illegal to include recyclable waste with regular garbage. As mentioned above, several states have passed laws requiring all towns to implement such mandatory ordinances. Kinnaman and Fullerton (1997) find mandatory recycling ordinances have little significant impact on recycling or garbage quantities. A plausible reason for this non-result is that municipalities do not adequately enforce their mandatory ordinances. Garbage collectors rarely inspect household garbage carefully. Any found violators usually just receive a written warning (Kinnaman, 1998). Duggal et al. (1991) find that communities that enforce mandatory recycling laws with fines experience no more recycling than towns without such enforcement.

Four thousand local governments have also implemented unit-based pricing programs. Most empirical papers devoted to user fees for garbage collection estimate the impact of the programs on household garbage and recycling behavior. These studies are discussed in Section 5 below. In addition to estimating the incidence of the programs, a few studies have estimated the likelihood such programs are implemented, the change in illegal dumping, and the benefits and costs of implementing a price-per-bag. Miranda and Aldy (1998) provide an in-depth analysis of the experiences of nine communities in the United States that implemented a price-per-bag.

Kinnaman and Fullerton (1997) and Callan and Thomas (1999) estimate the likelihood that a community will implement a unit-based pricing program. Kinnaman and Fullerton (1997) use data representing a national cross-section of 909 communities with and without unit-based pricing programs. They find that the likelihood increases with the local tipping fee, with the use of municipal (rather than private) resources to collect garbage, and with the education level of the community. Callan and Thomas (1999) find that the likelihood increases with household income, housing value, the age of the population, and whether the regional landfill is due to close within the next two years. They use data representing 317 communities in Massachusetts.

Available data rarely allow for direct comparisons between illegal dumping quantities before and after the implementation of unit pricing. Many economists have requested town officials to provide their opinion over whether they believe illegal dumping has increased. Many local officials have stated that it has, though many more have stated otherwise. Reschovsky and Stone (1994) and Fullerton and Kinnaman (1996) asked individual households whether they observed any change. In the former study, 51% of respondents reported an increase in dumping. The most popular method was household use of commercial dumpsters. For the 20% who admitted to burning trash, the authors were unable to confirm whether these burners did so in response to the program. Roughly 40% of the respondents to the Fullerton and Kinnaman (1996) survey indicated that illegal dumping had increased in response to the unit-pricing program. Many of these lived in the more densely populated urban areas of the city. Fullerton and Kinnaman (1996) also use survey responses with direct household garbage observations to estimate that 28% of the reduction of garbage observed at the curb was redirected to illicit forms of disposal. See Footnote 4 for a list of other papers that study the dumping issue.

Two types of unit-based programs have been implemented in the United States. Traditional bag or tag programs require households to pay for each additional bag of garbage presented at the curb for collection. The second program type requires households to pre-commit or "subscribe" to the collection of a specific number of containers each week. The household pays for the subscribed number whether these containers are filled with garbage or not. Many communities in California and Oregon have utilized subscription programs since early in the century. One advantage of subscription programs is that their direct billing systems may reduce administrative costs. Yet, economists believe the first type of user fee more truly represents marginal cost pricing. Kinnaman and Fullerton (1997) use city-wide data from over 700 communities to estimate that subscription programs have less of an impact than bag/tag programs on garbage and recycling quantities. Miranda and Aldy (1998) find that subscription programs can be effective if pricing applies to smaller trash containers. Nestor and Podolsky (1998) employ self-reported household data to estimate that subscription programs are about as effective as bag/tag programs at reducing garbage. Neither program is found to encourage source reduction in the presence of a curbside recycling program, since such programs subsidize recycling households' overall disposal practices.

B. Policy Directives in Europe

Many of the approaches taken above in the United States have also been pursued, to a greater or lesser extent, in other countries. For example, the United Kingdom has established a 50% recycling goal to be achieved by 2000. The current recycling rate in the UK is just 5% (Powell et al., 1996). To increase the recycling rate, the UK implemented credits for recycling and has been considering a tax on the disposal of solid waste in landfills. Seven other EC countries (Belgium, Finland, France, Germany, Italy, Luxenbourg, and the Netherlands) have implemented some variation of user fees for garbage collection. The UK rejected the idea of user fees due to the uncertainty of their effects. Also, deposit-refund systems for beverage containers have been implemented in Australia, Canada, France, Germany, and Switzerland. Germany has also implemented deposit-refund programs for detergent and paint containers.

Germany implemented a unique policy in 1991 called the "Law on Waste Management" that is designed to internalize the external costs of packaging choices by industry. This law requires the original product manufacturers to pay to recycle the packaging it produces even after the product is sold to retail firms or directly to consumers. The law also set an original recycling target of 80%. That is, firms would be required to recycle 80% of all packaging they produce. Amendments to the original legislation are expected to ease these targets to 60-70%.

Over 400 retail and packaging firms have combined with the large waste-hauling firms to create the Duales System of Deutschland (DSD). The purpose of this syndicate is to reduce the administrative costs associated with satisfying the minimum recycling standards. Rather than requiring that each bottle be delivered back to its original manufacturer, local waste management firms agree to collect for recycling *all* bottles of member organizations in exchange for payment from the DSD. Participating manufacturers identify their membership in the DSD by affixing a green dot on their packaging. In essence, the program becomes a national recycling effort operated by the DSD rather than by independent municipal governments, as is common in the United States.[11]

The collection, sorting, and marketing costs incurred by the waste management firms are paid by the DSD. The DSD then charges manufacturers according to the quantity and type of packaging used. For example, manufacturers pay the DSD $.82 for each pound of plastic packaging produced, $.27 per pound for aluminum, and only $.04 for each pound of glass. These charges represent the marginal cost to the DSD of collecting and sorting each type of material. The cost of glass is low because consumers traditionally separate and transport glass bottles themselves, these costs are paid by consumers and are therefore not internalized by the DSD or product manufacturers. Fullerton and Wu (1998) find that if the charges to manufacturers are set optimally, then the German Green Dot program can encourage firms to produce the optimal amount and type of packaging. The quantity of packaging consumed by households decreased by 4% following the implementation of the Green Dot program (Rousso and Shah, 1994)

The success of the Green Dot program in achieving the efficient quantities of garbage and recycling rests on two critical issues (Fenton and Hanley, 1995). First, households must be willing to separate materials for recycling. A mandatory deposit on non-refillable beverage containers gives consumers the incentive to return these forms of packaging. But lacking such incentives for other types of packaging, the household cannot be expected to recycle efficiently. Second, private collectors must recycle the materials. But in the absence of other regulations, the private collectors face private rather than social disposal costs. Thus, the collectors of recyclable material may find disposal in other countries cheaper than negotiating with a recycler to take the material.[12] Palmer and Walls (1999) argue that replacing Extended Producer Responsibility programs (like the Green Dot program) with a combined tax on intermediate goods and a subsidy paid to the collectors of recycled materials could alleviate these problems while preserving the more desirable outcomes.

Countries within the European Union have implemented other versions of producer responsibility programs, but few have set recycling goals as lofty as Germany's 60-70% target. Austria, Denmark, France, Italy, the Netherlands, and Sweden have made manufacturers at least partly responsible for the management of their packaging materials. The European Union itself has set a recycling target of between 50% and 75% to be met by the year 2000, and is watching the German experience carefully. The UK has dropped its national eco-labeling program but is cooperating with all other EU policy guidelines.

[11] Michaelis (1995) and Roussa and Shah (1994) provide further background on Germany's green dot program.

[12] Such concerns arose after several packages with green dots were found in French landfills. In response, the European Union recently banned the export of recyclable materials headed for foreign landfills or incinerators. Reliable data are not available to characterize the quantity of residential solid waste that is shipped between European countries. Europe has been exporting solid waste to Africa.

C. Developing Nations

This paper is concerned predominantly with residential solid waste in industrialized countries, but we discuss briefly some events in less developed countries. Solid waste management is a different story in developing countries. First, only 50-70% of the solid waste generated is actually collected (Cointreau-Levine, 1994). Second, the collection that does take place is very labor intensive. Households bring garbage to transfer stations, or collectors (scavengers) agree to carry garbage to a transfer station in exchange for any recyclable material found in the garbage. The World Bank estimates that 7,000 such workers operate in Manila, 8,000 in Jakarta, and 10,000 in Mexico City. In poorer sections of Egypt, India, Indonesia, and the Philippines, individuals using handcarts collect garbage door-to-door (Beede and Bloom, 1995).

The experiences in developing countries have allowed economists to estimate the relationship between per-capita income and garbage generation rates. Beede and Bloom (1995) find that per-day garbage generation rates vary between 0.5 kilograms per-capita in underdeveloped Mozambique to 1.9 kilograms per-capita in developed Australia. These cross-national data are used to estimate that the income elasticity of supply of garbage is 0.34, quite similar to estimates based on data sets gathered entirely within developed countries (described below). On the policy front, Cyprus, Egypt, India, Lebanon, and Syria have implemented deposit-refund systems for glass containers.

5. A Model of Household Behavior with Empirical Implications

The household is at center stage in the market for solid waste collection and disposal because the household chooses among various abatement options, including whether to devote resources to the separation and storage of recyclable materials. Every policy discussed above from a tax on virgin materials to a per-bag user fee on garbage disposal or the German green dot program depend crucially on household behavior to influence disposal quantities.

The model of household disposal decisions developed in Section 3 derived normative propositions about the optimal pricing of garbage, recycling, and virgin material. The model developed in this section can be used to derive empirical propositions for testing and estimation. This model is quite simple, but demonstrates the main forces influencing the disposal decisions of households. Specific functional forms are assigned to the equations above to simplify the interpretation of results. Some of the comparative statics generated from the model are tested in the available economics literature.

Assume the household consumes a single composite commodity good c that generates waste material m, according to

$$m = (1/\alpha)c, \tag{1}$$

where $1/\alpha$ is the portion of consumption that forms waste material. Assume $(1/\alpha) < 1$. Material m can either be presented at the curb for garbage collection (g) or recycled (r):

$$m = g + r. \tag{2}$$

Since these two equations imply that $c = \alpha(g + r)$, they are just a more restrictive version of the expression $c = c(g, r)$ given in Section 3 above.

Household utility is a function of it's own consumption of the composite commodity good,

$$u = u(c), \tag{3}$$

where $u_C > 0$ and $u_{CC} < 0$. The impact of aggregate garbage (G) on household utility is suppressed here for ease of presentation. Households do not notice a change in aggregate garbage attributable to their own disposal when making such choices.

Instead of having fixed income as in Section 3 above, the household here is endowed with $\bar{k}$ units of a resource such as time that can be exchanged in a labor market k^m for a wage p_k. Therefore, $y = p_k k^m$. The household resource can also be used to prepare waste material for recycling (k^r). The resource is fully employed ($k^m + k^r = \bar{k}$).

The amount of recycling generated by the household (r) is a function of the time allocated to recycling (k^r),

$$r = r(k^r), \tag{4}$$

where the marginal product of labor in recycling is positive ($r_k > 0$) and labor devoted to recycling experiences diminishing marginal returns ($r_{kk} < 0$). Equation (4) can be solved for k^r to give the cost of recycling:

$$k^r = k(r), \tag{5}$$

where $k_r > 0$ and $k_{rr} > 0$. For simplification, we specify

$$k(r) = 0.5\delta r^2 \tag{6}$$

where the first derivative $k_r = \delta r$ and the second derivative $k_{rr} = \delta$. Thus δ is the rate at which the marginal cost rises with r. A decrease in the parameter δ implies less household effort is required for recycling.

Household income ($p_k \bar{k} - p_k k^r$) can either be used to purchase the composite commodity good (for a price p_c), or to pay for each bag of garbage (at cost p_g) presented at the curb for collection. Using (6) to substitute for k^r in the above resource constraint, the household's budget constraint is:

$$p_k \bar{k} - p_k(0.5\delta r^2) = p_c c + p_g g. \tag{7}$$

Each household maximizes utility (3) subject to technological constraints (1) and (2) and the budget constraint (7), by choosing the quantity of material to discard (g) and to recycle (r). The Lagrange Function from this maximization problem is

$$L = u[\alpha(g + r)] + \lambda[p_k \bar{k} - \alpha(g + r)p_c - p_g g - p_k(0.5\delta r^2)]. \tag{8}$$

Assuming the existence of interior solutions for g and r, first-order conditions are

(9a) $\alpha u_c/\lambda = [\alpha p_c + p_g]$

(9b) $\alpha u_c/\lambda = [\alpha p_c + p_k\delta r]$

(9c) $p_k\bar{k} - \alpha(g + r)p_c - p_g g - p_k k(r) = 0$

where λ is the marginal utility of income. At the utility-maximizing choices, condition (9a) requires the marginal benefit of acquiring an additional unit of material (measured in dollars) to equal the purchase price of the material plus the price of discarding the material at the curb. Condition (9b) has a similar interpretation, except the marginal cost of acquiring an additional unit of material is comprised of the purchase price plus the resource cost of recycling it ($p_k\delta r = p_k k_r$). Solving conditions (9a) and (9b) provides the relationship $p_g = p_k\delta r$ at the utility-maximizing choices of g and r. The household increases recycling to the point where the marginal cost of recycling another unit of the material ($p_k\delta r^*$) equals the marginal cost of discarding the material (p_g).

Utility-maximizing solutions for the choice variables take the form:

(10a) $g^* = g^*(\alpha, \bar{k}, p_c, p_g, p_k, \delta)$

(10b) $r^* = r^*(\alpha, \bar{k}, p_c, p_g, p_k, \delta)$

Equations (1) and (2) can be used to solve for the utility-maximizing consumption level,

(11) $c^* = \alpha(g^* + r^*)$.

How would the equilibrium values of g^* and r^* be affected by an exogenous change in the values of p_g, δ, p_k, or α? The comparative statics reported below are obtained by first substituting the solutions (10) into the first-order conditions (9), then differentiating with respect to the exogenous variable of interest, and finally solving the system of differential equations for the comparative static terms (as in Silberberg, 1990, page 323).[13]

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[13] One implication of the model presented here is that if the price of garbage is zero, then the household has no incentive to engage in recycling since garbage is free and recycling requires scarce household resources. This result is clearly inconsistent with the available data. In fact, Fullerton and Kinnaman (1996) find that 73.3% of households recycled even in the absence of any legal or economic incentive. Why do these households recycle? Even if households value the quality of the environment (a public good) and their recycling efforts improve the quality of the environment, households cannot be expected to provide this public good at their own cost. Perhaps households simply enjoy recycling or feel a civic duty to participate in the recycling program. Understanding why households have been willing to participate in municipal recycling programs remains an interesting question to economists and policy makers.

A. A Change in the User Fee (p_g)

If the town has implemented a unit pricing program, the representative household in the model is required to pay for each bag of garbage collection (p_g). How will the household respond to an increase in the per-bag fee? Comparative static analysis indicates that the change in recycling attributable to a change in the value of the user fee is[14]

$$\frac{\partial r^*}{\partial p_g} \equiv \frac{1}{p_k \delta} > 0 \qquad (12)$$

which is unambiguously positive. A household will respond to an increase in the user fee by increasing recycling. This increase varies across households with different wage levels (p_k), and would be the greatest for households with the lowest wage. The increase also varies across households with different recycling production functions (value of δ in Equation 6). The change in recycling would be greater for a household that experiences less-rapidly diminishing marginal product of time in recycling (a low value of δ). Proxies for δ could include household size, age composition, and other demographic variables.

An increase in the price per bag of garbage collection also changes the utility-maximizing quantity of garbage discarded[15]:

$$\frac{\partial g^*}{\partial p_g} \equiv -\frac{\partial r^*}{\partial p_g} - \frac{g^*}{\alpha u_c / \lambda^*} < 0 \qquad (13)$$

which is unambiguously negative. Households are predicted to respond to an increase in the value of the user fee by decreasing the quantity of garbage presented at the curb. The first component of the right-hand side might be called the “substitution” effect since it represents the change in garbage directly attributable to the increase in recycling. The second component of this comparative static might be called the “income” effect since it represents the decrease in garbage brought about by the reduction in material generated from less consumption. The increase in the price per bag reduces the amount of income available to purchase other goods, decreasing the quantity of waste material. To see this more formally, note that

$$\frac{\partial m^*}{\partial p_g} \equiv \frac{\partial g^*}{\partial p_g} + \frac{\partial r^*}{\partial p_g} \equiv -\left[\frac{g^*}{\alpha u_c / \lambda^*}\right] < 0 \qquad (14)$$

Relative to the average household, this “income” effect is greatest for households that generate more garbage (high g^*), generate more waste material from consumption (low α), exhibit a low marginal utility of consumption (low u_c, perhaps because of a large c^*), or possess a high marginal utility of income (λ^*). The denominator of (14) is identical to the

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[14] A simple way to see this result is to solve (9a) and (9b) to get $r = p_g/\delta p_k$ and then differentiate that with respect to p_g.

[15] This result requires the use of all equations (9) and (10).

left-hand side of the first-order condition in (9a). A household that experiences a low marginal benefit of generating an additional unit of waste material ($\alpha u_c/\lambda^*$) will react to the user fee by reducing garbage more than other households.

To see why the income effect only reduces g and not r in this simple model, consider Figure 6. Total waste ($g + r$) on the horizontal axis is divided between r^* and g^* at the point where the flat marginal cost of g (equal to p_g) intersects the rising marginal cost of r (equal to $p_k\delta r$). When the income effect reduces consumption c (and thus the sum $g + r$), the right vertical axis shifts to the left, reducing g but leaving r unchanged.

Several economic papers have estimated these comparative static relationships. A brief overview of some of these studies appears in Table 3. One element common to every study mentioned in Table 3 is the use of original data. Data collection techniques include interviews with local solid waste officials, direct phoning of households, and actual measurement of household waste.

Wertz (1976) was the first to derive the impact of a user fee on garbage quantities. By simply comparing the average quantity of garbage collected in San Francisco, a town with a user fee, with the average town in the United States, Wertz calculates a price elasticity of demand equal to –0.15.

Jenkins (1993) expanded the understanding of the impact of user fees on garbage totals by gathering monthly data from 14 towns (10 with unit-pricing) over several years. Jenkins also found inelastic demand for garbage collection services; a 1% increase in the user fee is estimated to lead to a 0.12 percent decrease in the quantity of garbage.

Two studies rely on self-reported garbage quantities from individual households (rather than as reported by municipal governments). Hong et al. (1993) utilize data based on 4,306 surveys. Households indicate whether they recycle and how much they pay for garbage collection. Results indicate that a user fee increases the probability that a household recycles, but does not appreciably affect the quantity of garbage produced at the curb. Reschovsky and Stone (1994) mailed questionnaires to 3040 households and received 1422 replies. Each household reported its recycling behavior and income and demographic information. The price of garbage was estimated to have no significant impact on the probability that a household recycles. When combined with a curbside recycling program, recycling rates increase by 27 to 58%, depending on type of material.

Miranda et al. (1994) gather data from 21 communities with unit-pricing programs and compare the quantity of garbage and recycling over the year preceding the implementation of unit-pricing with the year following it. Results indicate that these towns reduce garbage by between 17% and 74% and increase recycling by 128%. These large estimates cannot be attributed directly to pricing garbage, since in every program curbside recycling programs were implemented during the same year as the unit-pricing program. Callan and Thomas (1997) predict that the implementation of a user fee increases the portion of waste recycled by 6.6 percentage points. This impact increases to 12.1% points when the user fee is accompanied by a curbside recycling program.

Only Fullerton and Kinnaman (1996) use household data that are not based on self-reported surveys. The weight and volume of the garbage and recycling of 75 households were measured by hand over four weeks prior to, and following, the implementation of a price-per-bag program in Charlottesville, VA. A curbside recycling program had already been in operation for over one year. Results indicate that the weight of garbage decreased slightly, but the volume of garbage (number of bags or cans) decreased by more. Indeed, the density of garbage increased from 15 pounds per bag to just over 20 pounds per bag.

Two studies expanded on the work of Jenkins (1993) by increasing the number of communities in the sample. Podolsky and Spiegel (1998) employ a 1992 cross-section of 159 towns clustered in New Jersey, twelve with unit-based pricing programs. They estimate the largest price elasticity of demand in the literature (-0.39). The authors attribute this estimate to the fact that no towns in their sample had implemented subscription programs (as was the case for Wertz and Jenkins) and had mature recycling programs in place. Kinnaman and Fullerton (1997) use a 1991 national cross- section of 959 towns, 114 that implemented user fees (none with subscription programs). The estimated demand elasticities are also higher than Jenkins, but not as large as Podolsky and Spiegel (1998). The Kinnaman and Fullerton estimates account for possible endogeneity of the policy variables. They find that towns with high garbage totals and low recycling totals are more likely to introduce a user fee. Previous estimates may have under-reported this elasticity by assuming that these policy variables are exogenous.

Strathman et al. (1995) employ data obtained by officials near Portland, OR, and they find that a 10% increase in the tipping fee decreases garbage disposed at the landfill by 1.1%.[16] Seguino et al. (1995) find that the implementation of user fee programs in 29 towns in Maine decrease solid waste by 8.73 pounds per person per week (a 56% decrease). Regarding illegal dumping, almost half of the towns reported initial increases in roadside dumping, and over half reported increases in backyard burning (30% say it is a continuing problem). Backyard burning is permitted in the state of Maine.

Only Klein and Robison (1993) estimate the impact of disposal fees on commercial behavior. Firms are estimated to reduce solid waste generation when faced with higher disposal rates.

What can be learned from all of these empirical studies? First, demand for garbage collection services is inelastic. Substitutes are not readily available. Advocates of unit-based pricing suggest demand may become more elastic in the long run as households learn of available substitutes for garbage disposal. The empirical economics literature has yet to address this point.

B. A Change in Ease of Recycling (δ)

Recall that household resources are required to recycle materials. According to the cost function given in (6), the implementation of a curbside recycling can be modeled by a decrease in the value of δ. Many expect the ease of curbside recycling to increase the quantity of recycling chosen by the household. Comparative static results of the model make a similar prediction,[17]

(15) $$\frac{\partial r^*}{\partial \delta} \equiv \frac{-r^*}{\delta} < 0$$

[16] Nestor and Podolsky (1996) published a comment suggesting that the changes in tipping fees may not have been passed on to households - the generators of garbage.

[17] From (9a,b) we get $r = p_g/\delta p_k$, so differentiation yields $\partial r/\partial\delta = -p_g/\delta^2 p_k = -r/\delta$.

Kinnaman and Fullerton (1997) confirm that this effect is positive. The implementation of a curbside recycling program is estimated to increase the annual quantity of recycling by 195 pounds per person (this estimate corrects for policy endogeneity). Reschovsky and Stone (1994) also find that a recycling program, especially when combined with a mandatory ordinance, increases recycling rates. Callan and Thomas (1997) find that a curbside recycling program increases by 4.15% the ratio of material recycled to all materials disposed. This impact increases to 9.67% when the curbside recycling program is accompanied with a unit-based pricing program.

The comparative static result in (15) predicts a greater than average increase in recycling for households that already recycle (a high r^*) and households that are very efficient recyclers (have a low value of δ). Reschovsky and Stone (1994) find that households reporting adequate storage space are much more likely to report that they recycle (using self-reported data). Judge and Becker (1993) study the recycling habits of 1000 households in towns of Minnesota (with different program attributes). They estimate that recycling totals are increased by allowing households to co-mingle recyclable materials, offering weekly collections (rather than biweekly), and not requiring households to put materials on the curb. They also find that special information about the recycling program did not increase recycling when controlling for other factors. Once a curbside recycling program has been implemented, Duggal et al. (1991) estimate that recycling totals increase with the age of the program, the frequency of collection, and the number of items collected.

The model does not provide a refutable hypothesis regarding the change in garbage attributable to the implementation of a curbside recycling program,

(16) $$\frac{\partial g^*}{\partial \delta} = \frac{-\partial r^*}{\partial \delta} - \frac{p_k k(r^*)}{\delta\left(\frac{\alpha u_c}{\lambda}\right)} > or < 0$$

The implementation of a municipal recycling program diverts some material from the garbage pile to the recycling pile (thus the first component of the comparative static is positive), but it frees up additional household resources for consumption, which may result in more material (the second term is negative). In order for the overall effect to be negative, the first component must exceed the second in absolute value. Most policymakers believe the direction of the comparative static in (16) to be positive. That is, the implementation of a curbside recycling program (a decrease in δ) reduces garbage.

The empirical evidence testing that assumption is inconclusive. Only Kinnaman and Fullerton (1997) estimate the impact of curbside recycling on household garbage totals, but they find the impact on garbage is not statistically significant.

C. A Change in the Wage (p_k)

Households may also change their utility-maximizing disposal choices with a change in their wage. As the wage rises, households face a higher opportunity cost of recycling and thus may recycle less. The comparative static result verifies this claim:

(17) $$\frac{\partial r^*}{\partial p_k} \equiv -\frac{r}{p_k} < 0$$

Relative to the average household, this negative effect is greater for households that recycle more (r) or earn low wages (p_k). Thus, poorer households are expected to respond to an increase in wage by decreasing recycling by a greater amount than richer households, *ceteris paribus*.

Hong et al. (1993) test the relationship in (17). They regress the probability of recycling on the wage rate of the female member of the household and find that as the wage rate increases, the probability of recycling decreases. Kinnaman (1994) also finds that recycling decreases with the number of full-time workers in the household.

A change in the wage is also predicted to affect the optimal quantity of garbage:

(18) $$\frac{\partial g^*}{\partial p_k} \equiv -\frac{\partial r^*}{\partial p_k} + \frac{k^m}{\alpha u_c / \lambda^*} > 0$$

which is unambiguously positive. Again, this comparative static can be partitioned into an "income" and "substitution" effect. Part of the increase in garbage is a direct result of the decrease in recycling. The remaining portion arises from the fact that more material is being generated by the household with the higher wage. In Figure 6, the marginal cost of recycling ($p_k \delta r$) would rotate upward with the increase in p_k, so r falls. Garbage is increased both by the decrease in recycling and by the *rightward* shift of the right vertical axis. This can be expressed more formally by:

(19) $$\frac{\partial m^*}{\partial p_k} \equiv \frac{\partial g^*}{\partial p_k} + \frac{\partial r^*}{p_k} \equiv \frac{k^m}{\alpha u_c / \lambda^*} > 0$$

The increase in total waste material is particularly high for households that devote more time to working (high k^m) since these households will enjoy the greatest boost to income for an increase in p_k. *Ceteris paribus*, households that experience a low marginal benefit of consumption ($\alpha u_c/\lambda^*$) will generate more additional material than the average household (following a boost in p_k).

Though Hong et al. (1993) find a positive relationship between garbage and the wage rate, the estimate is statistically insignificant. Podolsky and Spiegel (1998) estimate that an increase in the ratio of employees to household members increases garbage. Kinnaman (1994) also finds that an increase in the portion of the household that are full-time workers increases garbage.

D. A Change in α

The portion of consumption that becomes waste material ($1/\alpha$) is exogenous to the household.[18] This exogenous value of α could change if firms reduce the quantity of

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[18] An extension of the model would allow α to be a choice variable. Households could

material used to package their products.[19] How would households respond to an exogenous change in α? The comparative static results are[20]

$$\frac{\partial r^*}{\partial \alpha} \equiv 0 \tag{20}$$

$$\frac{\partial g^*}{\partial \alpha} \equiv -\frac{p_c m^*}{\alpha u_c / \lambda^*} < 0 \tag{21}$$

An increase in the value of α is interpreted as a decrease in the portion of consumption that becomes waste material. Households respond to this increase by decreasing garbage, but do not change recycling. The change in garbage is especially large for households that discard a high amount of material (m^*), face high prices for goods and services (p_c), or experience a low marginal benefit of acquiring an additional unit of material ($\alpha u_c/\lambda^*$). No empirical evidence has been found to test these predictions.

E. Other Considerations

Many of the empirical studies mentioned above control for income and demographic variables in the regression when estimating the quantity of garbage and recycling produced by households. The estimated coefficients on these variables could assist local governments to forecast future garbage disposal needs.

A change in the wage rate, as modeled above, has both an income effect and a price effect (on the cost of recycling). The pure income effect of a change in nonlabor income on household garbage has been estimated in several empirical studies. This relationship could be expected to be positive if additional income implies more consumption and garbage. However, if increases in income are spent on dining out and longer vacations, household garbage totals could decrease with income. The empirical literature finds more evidence supporting the former prediction. In fact, Podolsky and Spiegel (1998) find the strongest relationship between garbage quantities and income by estimating the income elasticity of demand for garbage collection to be 0.55. Other studies also find a positive but weaker relationship between income and garbage. Jenkins (1993) estimates an income elasticity of demand equal to 0.41, Wertz (1976) at 0.279 and 0.272 using two sets of data, Kinnaman and Fullerton (1997) at 0.262, Richardson and Havlicek (1978) at 0.242, 0.22 by Reschovsky and Stone (1994), 0.2 by Petrovic and Jaffee (1978), and finally 0.049 by Hong et al. (1993). Strathman et al. (1995) find that garbage disposed at landfill decreases with the average manufacturing income of the city.

The effect of nonlabor income on recycling is not as well understood. (The simple model in Figure 6 would predict no effect.) Callan and Thomas (1997) and Duggal et al.

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choose the mix of consumption goods to include less waste-intensive goods. Additional constraints would have to be imposed on the current model, or households here would simply choose α to be 0.

[19] See Fullerton and Wu (1998) for a further discussion of the packaging decisions of firms.

[20] Again, the first result follows directly from differentiating $r = p_g/\delta p_k$.

(1991) find that income increases household recycling quantities, but Hong et al. (1993) find income does not impact self-reported recycling participation. Jakus et al. (1996) find income increases the recycling of paper but not glass. Saltzman et al. (1993) find that additional income increases the recycling of newspaper but decreases the recycling of glass.

Economists have also estimated the relationship between education and household garbage totals. Educated households could be more aware of recycling opportunities. Educated households may also have greater tastes for the environment. Indeed, Hong et al. (1993), Callan and Thomas (1997), Judge and Becker (1993), Reschovsky and Stone (1994), and Duggal et al. (1991) find education increases recycling. Using household data, Kinnaman (1994) estimates that educated households produce less garbage. Using a cross-section of 959 communities, Kinnaman and Fullerton (1997) find a similar result. Though Judge and Becker (1993) find no impact from publicity efforts to increase the awareness of municipal recycling opportunities, Callan and Thomas (1997) find that an extra dollar spent per household on such efforts increases the recycling rate by 2.55%.

The effects of other demographic variables have also been estimated. Jenkins (1993), Kinnaman (1994), and Podolsky and Spiegel (1998) find that increases in the size of the household decrease the per-capita quantity of garbage disposed. Larger families could share meals in a way that produces less waste than the same number of people eating separately. Hong et al. (1993) find that larger households also are more likely to report participation in recycling. Regarding the age of the household and its impact on garbage totals, Podolsky and Spiegel (1998) find that an increase in median age decreases garbage. Jenkins (1993) finds that an increase in the portion of population between 18 and 49 increases garbage. Jakus et al. (1996) find that older individuals are more likely to recycle glass. Kinnaman (1994) estimates that households with married couples produce less per-capita garbage and recycling. Reschovsky and Stone (1994) find that married households produce more total recycling (not controlling for household size). Finally, Kinnaman (1994) estimates that homeowners produce more garbage and recycling than renters do.

6. Conclusion

The solid waste collection and disposal industry has undergone dramatic changes over the past two decades. First, the structure of landfills has changed from local town dumps to large regional landfills equipped to reduce the negative externalities associated with garbage disposal. Second, Japan, much of Europe, and the northeast regions of the United States have turned to incineration to manage residential solid waste since the 1970's. Financially, incineration has been most successful where land is scarce (and hence the costs of landfills are high). Some still question the environmental benefits of incineration. Third, the portion of solid waste that is recycled has risen sharply over the past decade. This growth has been facilitated by greater government involvement designed to encourage households to separate waste. The growth in the supply of recycled materials has resulted in a short-run glut of materials, and governments have been active in finding markets for these materials. Several states in the U.S. have passed an assortment of policies with this goal in mind. Finally, roughly 4000 local communities in the U.S. have begun to price garbage by the bag. These local programs have helped to pay the rising costs of disposal in some areas, and they provide an incentive for households to recycle more. The extent to which these programs produce positive net benefits is still debated in the economics literature.

As residential solid waste became a more important issue to policy makers, intellectual attention from economists increased. The number of economic papers devoted to residential solid waste and recycling has risen sharply over the past 10 years. The bulk of these papers provide empirical estimates of the effects of government policies on household disposal behavior. Another portion is devoted to prescribing the efficient policy approach. Most models support the use of some form of a "deposit-refund" system. The deposit or advanced disposal fee could be applied at either the point of production or purchase. The refund or subsidy to recycling could be given to households that recycle or to firms that purchase recycled materials. Other economic models support a tax on virgin material or a direct tax on the household's disposal choices.

Even though the economic literature has reached some consensus over the choice of policy directives, very few of these recommendations have been pursued explicitly by the policy-making community. Advanced disposal fees exist only for some products in some countries. Explicit recycling subsidies are also few and far between. Deposit-refund systems have been implemented only for beverage containers and have only been implemented in some countries. Perhaps additional work could design structures for these policies to help minimize the administrative costs. Palmer and Walls (1999) have begun work in this area. On the other hand, many jurisdictions already have *implicit* deposit-refund systems on all goods, to the extent that they impose a general sales tax on all purchases and use some of the money to pay for free curbside recycling collection.

Many economic predictions have been confirmed by empirical work: a higher price per bag of garbage is found to reduce demand for garbage collection, and higher incomes are found to increase waste for disposal. Other behaviors are not yet well understood, however, such as observed amounts of recycling even when households have no incentive to recycle.

Figure 1: Disposal Trends (%)
Source: Biocycle Magazine

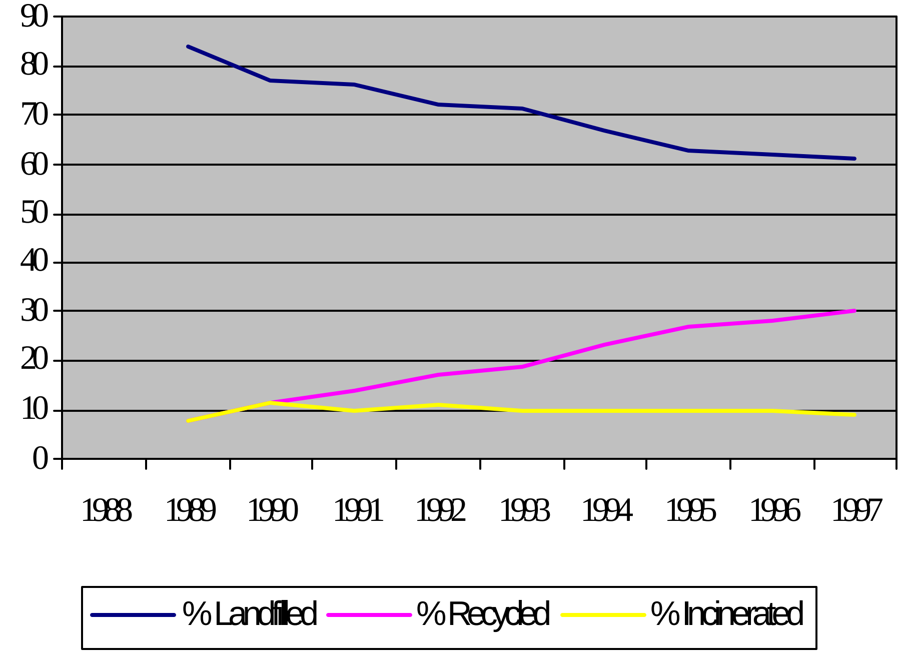


Figure 2: Tipping Fees ($/ton)
Source: Biocycle Magazine

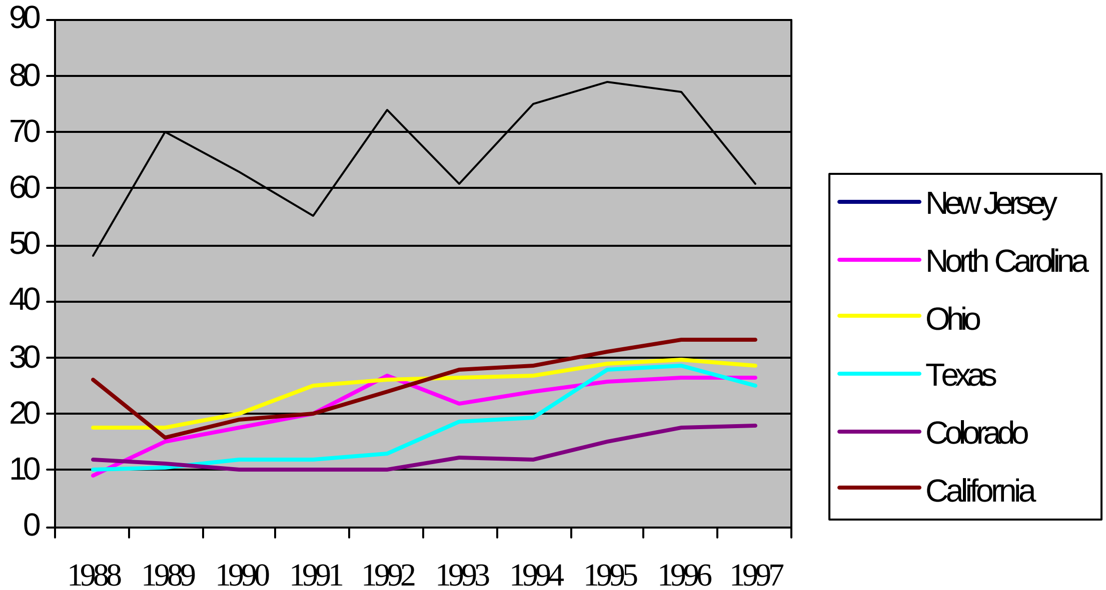


Figure 3: Price Index of Recyclable Materials
Source: Bureau of Labor Statistics (1982=100)

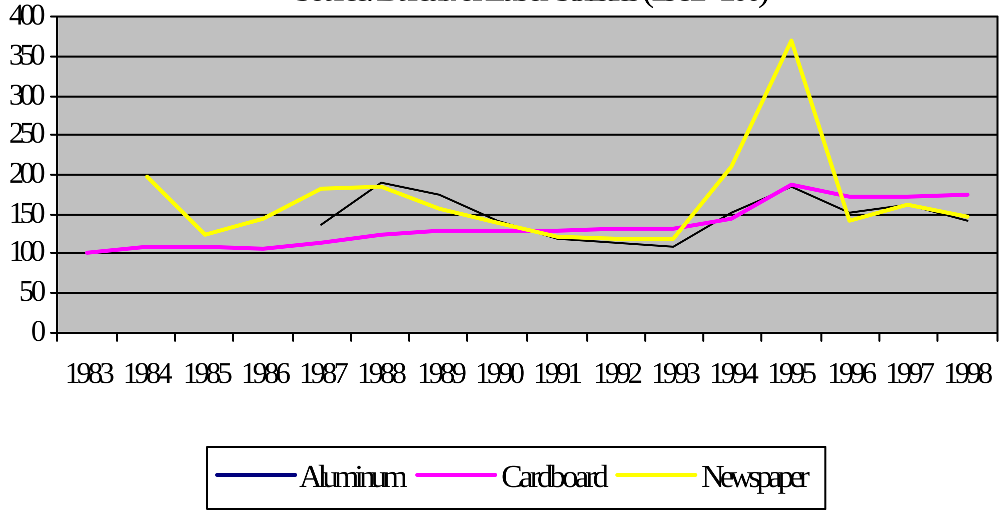
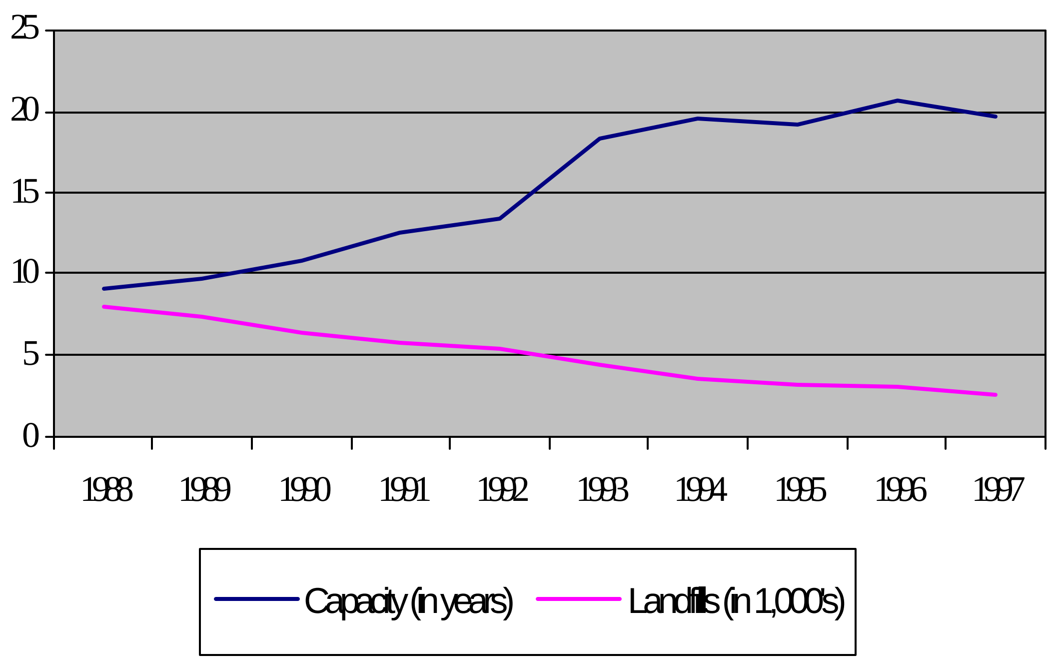


Figure 4: Landfill Crisis?
Source: Biocycle Magazine



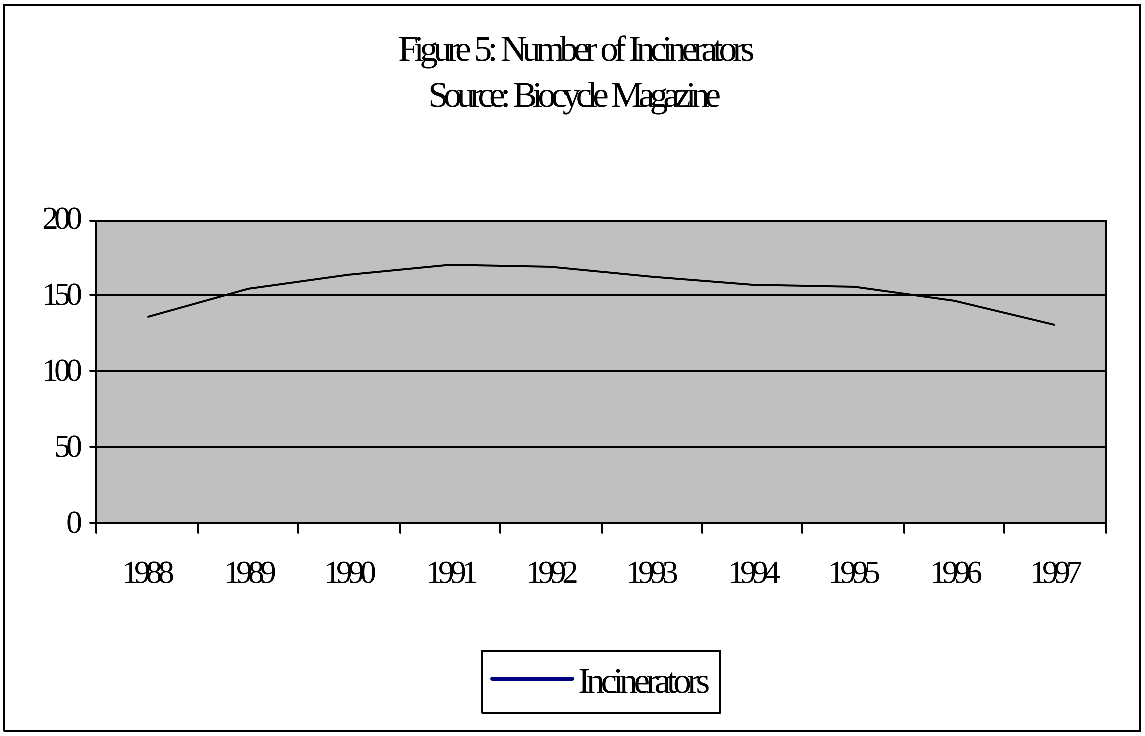
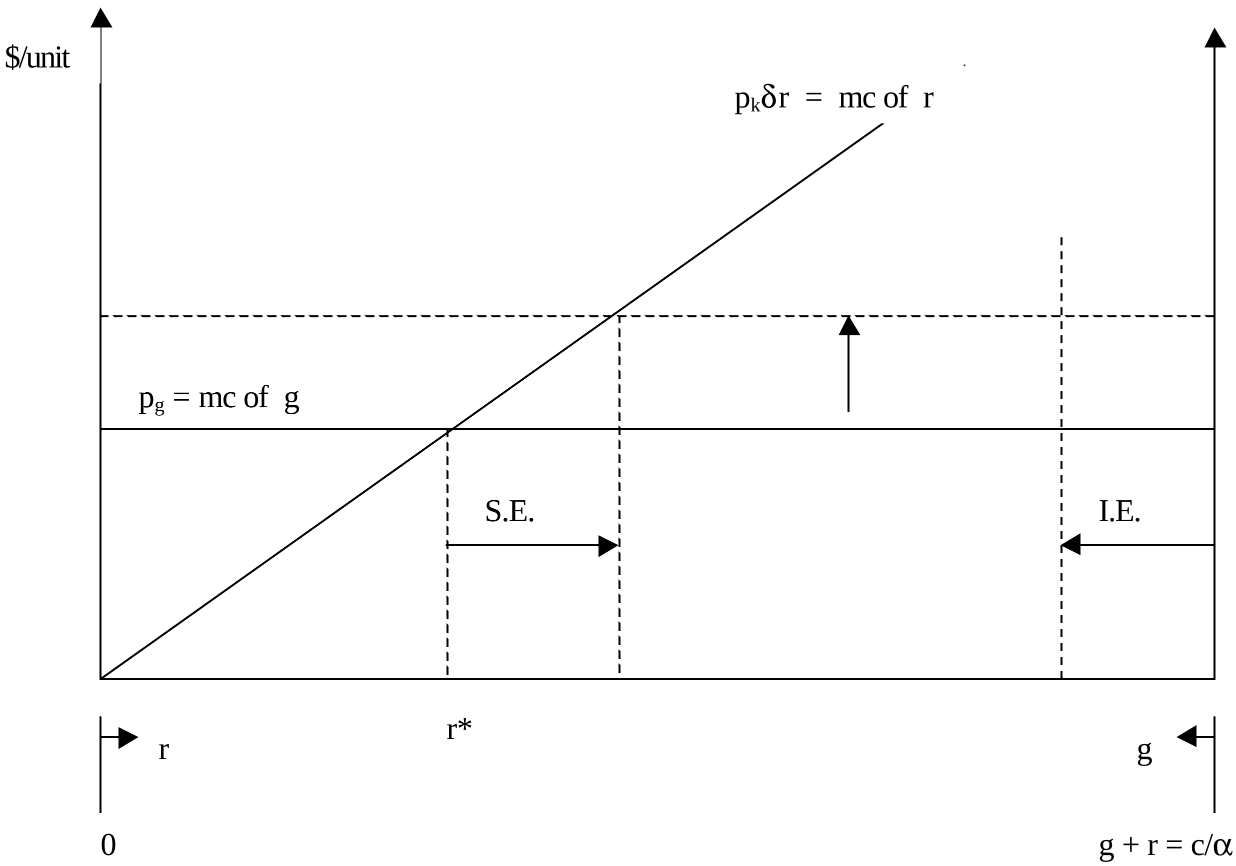


Figure 5: Number of Incinerators
Source: Biocycle Magazine
200
150
100
50
0
1988
1989
1990
1991
1992
1993
1994
1995
1996
1997
Incinerators

Figure 6: The Choice of Garbage (g) and Recycling (r)



An increase in p_g raises the flat marginal cost of garbage disposal (mc of g). It thus moves r* to the right by a substitution effect (S.E.), and it moves the right-hand origin (c/α) to the left by an income effect (I.E.). Both effects reduce garbage g.

TABLE 1: USE OF LANDFILLS FOR WASTE DISPOSAL IN EUROPE

COUNTRY	PERCENT LANDFILLED (NET OF RECYCLING)
Denmark	44
France	54
Greece	100
Ireland	100
Italy	85
Netherlands	56-61
Sweden	35-49
Switzerland	22-25
United Kingdom	90
United States	90
West Germany	66-74

Source: Jenkins (1993), based on data gathered by: US Congress, Office of Technology Assessment (1989).

TABLE 2: U.S. STATE POLICIES DESIGNED TO INCREASE RECYCLING

POLICY	NUMBER OF STATES IMPLEMENTED
Pass a recycling goal	45
Require all municipalities to implement curbside recycling programs and pass a local ordinances making household and commercial recycling mandatory	7
Require all municipalities to implement curbside or drop-off recycling programs but not a mandatory ordinance	7
Require all municipalities and counties to satisfy a minimum recycling quota without designating the method to achieve it	8
Provide grants to municipalities to help finance recycling programs	34
Ban yard waste from being disposed in landfills	23
Implement a deposit/refund system for beverage containers	9
Provide tax credits for new recycling facilities	29
Provide low-interest loans for new recycling facilities	15
Require all state government offices to purchase recycled materials	29

Source: Glenn (1998).

TABLE 3: EMPIRICAL ESTIMATES OF THE EFFECT OF UNIT-PRICING

Study	Data	Model	Change in Garbage	Change in Recycling
Wertz (1976)	Compares subscription program in San Francisco with flat fees imposed by "all urban areas"	Comparison of Means	ε = -0.15	
Jenkins (1993)	Panel of 14 cities (10 with user fees) over 1980-88		ε = -0.12	
Hong et al. (1993)	1990 survey of 4,306 households in and around Portland, Oregon.	Ordered Probit and 2SLS	No significant impact	Unspecified positive relationship
Reschovsky and Stone (1994)	1992 mail survey of 1,422 households in and around Ithaca, NY.	Probit		No significant impact
Miranda et al. (1994)	Panel of 21 cities over 18 months beginning in 1990	Comparison of Means	17%-74% reduction in garbage	Average increase of 128%
Callan and Thomas (1997)	1994 cross-section of 324 towns in MA, 55 with unit-pricing programs	OLS		6.6%-12.1% increase
Fullerton and Kinnaman (1996)	Two-period panel of 75 households in 1992	OLS	ε = -0.076 (weight) ε = -0.226 (volume)	Cross-price elasticity is 0.073
Podolsky and Spiegel (1998)	1992 cross-section of 159 municipalities in NJ, 12 with unit-pricing	OLS	ε = -0.39	
Kinnaman and Fullerton (1997)	1991 cross-section of 959 towns across the U. S., 114 with unit-pricing	OLS 2SLS	ε = -0.19 ε = -0.28	ε = 0.23 ε = 0.22
Strathman et al. (1995)	Seven year (1984-1991) time series in Portland, OR	OLS	ε = -0.11	
Seguino et al. (1995)	1993-1994 cross section of 60 towns in Maine, 29 with unit-pricing	Comparison of Means	56% decrease	

ε = price elasticity of demand, OLS = ordinary least squares, 2SLS = two stage least squares.

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