Greenhouse gas emissions, waste and recycling policy*

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Abstract

This paper examines least-cost policies for waste reduction, incorporating upstream greenhouse gas externalities associated with the production of consumption goods from various materials. In particular, we decompose the effect of deposit/refund, advance disposal fees, and recycling subsidies on upstream greenhouse gas emissions. We find that the benefits of reducing greenhouse gas emissions are of the same order as or larger than the benefits of reducing solid waste disposal, implying larger optimal total waste reduction than previous studies. Furthermore, the least-cost intervention levels will be material-specific and vary substantially across materials. Finally, despite the reductions in emissions implied by increased recycling rates, direct recycling subsidies are more costly and generate less emissions reductions than a deposit/refund or advance disposal fee.

Keywords: greenhouse gas emissions, waste, recycling, upstream externalities, environmental policy

1 Introduction

The era of contemporary household recycling began in 1987 following the voyage of the garbage barge *Mobro 4000*, whose story sparked a surge of interest in waste reduction within the United States. The infamous barge was loaded with trash, and for two months it roamed the Atlantic Coast as far south as Belize, searching for a port to dock and unload its cargo. By the time it returned to Long Island, still carrying a full load of garbage, the barge had made national news headlines. Nicknamed the "Gar-barge" by the media, it gained the attention of the public who viewed its fate as an indicator of a looming trash crisis due to insufficient landfill space.[1] In addition, images of medical waste and oozing trash

[1] The Mobro was originally intended to undergo an experimental program in North Carolina to convert trash into methane. It was rejected by authorities there and the fate of the *Mobro 4000* had more to do with poor business planning and concerns about mob connections and hazardous waste than with space constraints for waste disposal.

aboard the barge led to rising concerns over hazardous waste and the resulting damage to health and the environment. In response, between 1990 and 2000, recycling programs grew tenfold, increasing the number of households with access to curbside recycling to over half and achieving an overall recovery rate near one-third of total municipal solid waste.

However, consumer interest regarding recycling and waste disposal waned in the years to follow (Katz 2002). Several mechanisms drove this decline in interest in the aftermath of the "trash crisis." First, land was not as scarce as the story of the *Mobro 4000* suggested. Although the number of landfills decreased between 1988 and 1997, the actual landfill capacity grew during this time (Kinnaman and Fullerton 2000). In addition, Benjamin (2010) notes that disposal techniques have improved and leakage has declined, somewhat mitigating concerns of potential contamination from hazardous material. The decline in recycling interest may have also been driven by the recognition that the social costs of disposal are relatively small. Palmer et al. (1997) use a value of $33 per ton as the social cost of waste disposal, justifying a modest policy intervention to achieve a 7.5% reduction in total waste. More recently, Kinnaman (2006) provides a calculation of social costs of waste disposal at $5-$9 per ton, noting that many costs associated with waste disposal are internalized by landfills via tipping and host fees. At this social cost of waste disposal, extremely small total waste reductions would be justified.[2]

A renewed interest in recycling has emerged more than 20 years after the story of the Gar-barge, not in response to the social cost of waste disposal, looming landfill constraints, or contamination concerns, but rather from growing concerns over upstream greenhouse

[2] Kinnaman (2006) notes that most of the benefits from recycling are primarily "warm-glow" utility received by recyclers.

gas emissions associated with production of consumer goods. Specifically, the fact that production of goods from recycled inputs is typically less emissions intensive than production from virgin inputs has led to calls for increased recycling to reduce this upstream externality.[3] However, both Palmer et al. (1997) and Kinnaman (2006) note that upstream externalities such as greenhouse gas emissions would be more efficiently handled via policy interventions at their source, rather than adjusting waste and recycling policies. Walls and Palmer (2001) show that in a world where Pigovian instruments are available, the optimal policy is to set an emissions tax equal to the marginal social damage of the upstream externality coupled with a downstream deposit/refund set equal to the marginal social damage of waste disposal.

However, in the United States it is unlikely that a carbon pricing scheme will be implemented in the near future. When Pigovian taxes are infeasible, Walls and Palmer (2001) show that the setting of alternative instruments could benefit from using life cycle assessment (LCA) information regarding upstream greenhouse gas emissions associated with a product's life.[4] In such a case, the downstream waste and recycling policy should be adjusted to account for the upstream emissions externality, reflecting the fact that waste and recycling policy will alter the quantity and mix of recycled and virgin inputs used in production. Following this logic, we re-evaluate market-based waste reduction and recycling policies in the presence of unpriced greenhouse gas emissions.

Several questions are addressed in this analysis. First, what are the channels of adjust-

[3] For example, aluminum production from recycled inputs uses only 5% of the energy required for the production of virgin material.

[4] Life cycle assessments are becoming a common technique to compare materials by evaluating the environmental performance of goods by accounting for all upstream energy and material input required in its production process, as well as quantifying the impact of disposal, degree of recyclability, and a measure of the functionality of the good.

ment that lead to emission savings and how do emissions savings vary across waste and recycling policies? How does this variation in emission savings affect the relative cost of alternative waste and recycling policies and what are the implications for policy design? Next, how do the social benefits from reduced emissions compare to the social benefits of reduced waste disposal? Lastly, given substantial heterogeneity in the emissions released during the production of varying types of consumer goods, how much would material-specific policy interventions reduce the cost of waste and recycling policies?

This paper extends the simulation analysis presented in Palmer et al. (1997) (hereafter PSW) which determines the least-cost policies for reducing municipal solid waste when a unit pricing, "pay-as-you-throw" system is not feasible.[5] Their numerical results confirm prior theoretical and empirical studies that conclude that a deposit-refund is the most efficient policy instrument for reducing waste (Dinan 1993; Fullerton and Kinnaman 1995; Sigman 1995; Palmer and Walls 1997).[6] The intuitive explanation for this result is that deposit/refunds exploit both channels of waste reduction by decreasing consumption and promoting recycling of goods. While these studies on downstream waste disposal ignore upstream considerations, Walls and Palmer (2001) provide a theoretical framework in which they incorporate upstream

[5] Most households experience a flat fee for waste disposal, although it has been shown that a unit-based charge on waste, either by weight or volume, is the most efficient policy to reduce total residential waste (Jenkins 1993). When households are charged the appropriate nonzero marginal cost per unit waste, the social cost of waste is properly internalized (Kinnaman 2006). In response to the fee, households can voluntarily elect to reduce consumption or increase recycling to avoid additional charges. However, Fullerton and Kinnaman (1996) show that this only holds in the absence of illegal disposal. Ino (2011) revisits the illegal disposal issue from the perspective of firm-level decisions, and derives the second-best deposit/refund policy in the presence of monitoring costs.

[6] Several additional empirical studies examine the effectiveness and consequences of various waste and recycling policies "in the field" (Fullerton and Kinnaman 1996; Jenkins et al. 2003; Yang and Innes 2007; Beatty et al. 2007; Ashenmiller 2009). See Kinnaman (2006) for a summary of recycling benefits and empirical findings of unit-based pricing.

externalities associated with the production of consumer goods. Building off this theoretical framework, our study "closes the loop" in the numerical partial equilibrium model provided by PSW to incorporate greenhouse gas emissions from the production of consumer goods using virgin and recycled material inputs. The simulation analysis calculates the costs of policies to achieve a given reduction in waste, net the benefits of emissions reduction for three price-based policy tools: deposit/refund, advance disposal fees (ADF), and direct recycling subsidies. Calibrated to the PSW model, we directly compare results to identify potential benefits and policy implications as a consequence of accounting for greenhouse gas emissions from upstream production activities in setting waste and recycling policies.

We find several important results with strong policy implications. First, analytical decompositions of the effect of deposit/refund, advance disposal fees, and recycling subsidies on greenhouse gas emissions highlight two channels through which these policies affect emissions: *source reduction* and *increased recycling*. We note that the *source reduction* will generally have a larger impact on emissions as reduced consumption eliminates emissions entirely, while *increased recycling* only reduces emissions corresponding to the difference between virgin and recycled production emissions.[7] From the perspective of emissions reductions, the advance disposal fee provides the largest benefits at $40 per ton of waste reduction, as it operates solely via the *source reduction* channel. Thus, while the advance disposal fee is more costly than the deposit/refund from the perspective of waste reduction, for waste reductions less than 4% the ADF is the least-cost policy due to the benefits of avoided emissions. Beyond

[7] According to the EPA's hierarchy of integrated solid waste management methods, *source reduction* takes priority over *increased recycling*. However, PSW argue that an optimal combination of both methods provides the greatest efficiency in achieving waste disposal reduction. PSW do note that the EPA's hierarchy may be more appropriate when considering upstream production externalities, such as in this study.

that level of waste reduction, the deposit/refund is the least-cost policy. Finally, despite the fact that production of consumer goods from recycled material is less emissions-intensive, the direct recycling subsidy provides the least emission savings and has the highest cost of achieving a given waste reduction. This result may assist policymakers to avoid mistakenly adopting recycling subsidies over less costly policy instruments.

We find that at $25 per MTCO2E price of carbon, the social benefits from reduced greenhouse gas emissions are of the same order of magnitude as the marginal social damage of waste disposal in PSW, and substantially larger than the marginal social damage of waste disposal in Kinnaman (2006).[8] While a 7.5% reduction in waste is justified by the $33 per ton social cost of waste disposal from Palmer et al. (1997), a 14% reduction in waste is justified when the benefits of emissions reductions are incorporated. At the social cost of waste disposal of $5-$9 per ton reported by Kinnaman (2006) a negligible policy intervention is justified; however, when the benefits of emissions reductions are accounted for, an 8% reduction in waste would be justified. Furthermore, for a 10% reduction in waste, PSW find that the least cost policy is a deposit/refund of $45 per ton, while incorporating the benefits of emissions reductions lowers the net cost to $19 per ton for the same reduction in waste.

Finally, we find that a material-specific intervention level reduces the cost of achieving a given reduction in waste by 10%. The intuitive explanation behind this is that while the benefits of waste reduction are homogenous across material types, greenhouse gas emissions

[8] We provide discussion and sensitivity analysis of alternative assumptions regarding the marginal social damage of greenhouse gas emissions. Also, it should be noted that our analysis abstracts from other externalities resulting from the production of consumer goods, such as forestry and land use issues associated with paper production, other local emissions associated with energy production, or mining externalities generated by the production of aluminum and steel.

vary substantially by material type and input source. Thus, the deposit/refund levied per ton of aluminum is more than an order of magnitude larger than the deposit/refund per ton of glass and paper, reflecting the emissions benefits of reducing aluminum production from virgin inputs.

2 Model

The model adopted in this paper builds on the mass-balance model developed by PSW, using their 1990 data on baseline prices, consumption, and recycling by material, as well as own-price elasticities for demand and supply in the consumption and recycling markets.[9] We begin with a description of the analytical model used in PSW, which is then augmented to incorporate greenhouse gas emissions from virgin and recycled production.

2.1 Model of waste and recycling

The basic equilibrium mass-balance model is described by the following system of equations:

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

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

$$R^d(p_r) = r(p_r)D(p_q, p_q - p_r), \tag{3}$$

where W is disposed waste, Q is total consumption, and R is the amount recycled. The mass-balance equation (1) requires that all consumption is either disposed of as waste, or

[9] Using the identical 1990 dataset allows for easy comparison with previous results, ensuring that differences in results are driven solely by inclusion of greenhouse gas externalities, and not differences in parameter assumptions. See Tables I and II in Palmer et al. (1997).

is recycled. Consistent with PSW, supply of the final product Q is assumed to be perfectly elastic, while demand D for the final product Q varies with the price of the final product p_q and, if recycled, with the price net of scrap value $p_q - p_r$ as in Equation (2).[10] The supply of recycled material varies with the scrap price (p_r) and is equal to the recycling rate $r(p_r)$ times total consumption. Finally, demand for recycled material is assumed to also vary with p_r, such that $R^d = R^d(p_r)$. Equation (3) states that the market for recycled scrap clears at endogenous scrap price p_r. The following intuitive assumptions hold: $\frac{\partial D}{\partial p_q} < 0$, $\frac{\partial D}{\partial (p_q - p_r)} < 0$, $\frac{dr}{dp_r} > 0$, and $\frac{dR^d}{dp_r} < 0$.

Several assumptions of the original study should be noted. It is assumed that markets for the final material and recycled material are perfectly competitive. It is also assumed that there are no lags between when the material is purchased and when it is disposed of or recycled. Another important assumption is that the quantity of the consumption good Q does not affect the demand for recycled materials $R^d(p_r)$. The final important assumption is that demand for the consumption good only depends on own price, effectively setting the cross-price elasticity across materials equal to zero. Realistically, increases in the price of glass would likely lead towards substitution towards aluminum. While relaxing these assumptions would alter the quantitative results in PSW and this study, they are unlikely to alter the qualitative comparisons between the two studies.

[10] There may also be a marginal effort cost or marginal psychic benefit associated with recycling the good, but as in PSW, we abstract from such considerations. Nonetheless, the psychic benefits, "warm-glow," associated with recycling may be an important feature of recycling, as argued in Kinnaman (2006).

2.2 Incorporating greenhouse gas externalities

To this basic set-up, we add the CO_2-equivalent emissions associated with the production of the consumption good Q. Walls and Palmer (2001) develop a framework incorporating upstream production into the waste and recycling model in PSW, which we will draw upon. We adopt a similar mass-balance expression $V + R = Q$ which states that production output must come from either virgin inputs V or recycled inputs R.[11] In other words, in a static model, if 30% of consumption is recycled, it implies that 30% of production comes from recycled material, with the remainder produced from virgin materials.[12] Note that this also implies that $W = V$, such that waste disposed of by consumers is equivalent to the amount of virgin inputs required in the production process. Distinguishing between virgin and recycled inputs is extremely important when considering upstream externalities, as CO_2-equivalent emissions from production vary depending on whether virgin or recycled materials are used. Emissions are thus given by:

$$E = \delta_v V + \delta_r R \tag{4}$$

where δ_v transforms production of goods from virgin materials into emissions and δ_r transforms recycled production into emissions.

[Table 1 about here]

[11] The theoretical model in Walls and Palmer (2001) also allows for manufacturing by-products (non-consumer solid waste), a consideration abstracted from in this study.

[12] For certain materials, this strict 1:1 substitutability between virgin and recycled inputs may not precisely hold, and more than one ton of recycled input may be needed to replace one ton of virgin input. For example, slightly more recycled aluminum is required to replace an equivalent amount of virgin aluminum, as some material is lost when melting down the recycled scrap. This would diminish the benefits of recycling relative to the results presented below.

In addition to the fact that emissions vary depending on the use of virgin versus recycled inputs, greenhouse gas emissions also vary considerably by material, as shown in Table 1. The values in Table 1 were obtained from an EPA life-cycle assessment of emissions associated with production. Note that these values also include emissions associated with transportation. A detailed discussion regarding the methods and assumptions used to calculate these values is included in the appendix.[13] Production from virgin inputs for aluminum and steel produce significant amounts of emissions in comparison to emissions from production with recycled inputs.[14] On the other hand, for some paper types, emissions are larger when re-using the recycled material as opposed to utilizing virgin inputs. Virgin production of plastics emits a modest level of greenhouse gases, but recycled production emits an order of magnitude less. Glass produces very little in the way of emissions, regardless of input type. In the following section, we analytically decompose and interpret the effect of changes in policy on emissions.

3 Policy intervention

Following PSW, we study three price-based instruments: deposit/refund, advance disposal fees, and recycling subsidies. As in PSW, we assume these policies are implemented at the producer level, and that the incentives provided by these policies are passed on to consumers.

[13] See "Solid Waste Management and Greenhouse Gases: A Life-Cycle Assessment of Emissions and Sinks" (2009). The EPA study reports metric tons of carbon-equivalent emissions (MTCE) per ton of material produced. To determine CO_2-equivalent metric tons emitted (MTCO2E), the numbers were scaled by $\frac{44}{12}$ for molecular weight.

[14] The substantial emissions associated with aluminum production from virgin inputs stems from the energy usage and emissions of the electrolytic smelting process used to extract pure aluminum metal from aluminum oxide.

The literature has identified two channels that can reduce total waste disposal: *source reduction* and *increased recycling*. Source reduction operates through reductions in consumption, or in terms of the model, by reducing $D(p_q, p_q - p_r)$. The recycling channel operates through increasing the recycling rate, $r(p_r)$. In the context of emissions, similar channels are exploited by the deposit/refund, advance disposal fee, and recycling subsidy policies. By reducing total consumption through source reduction, emissions will fall, and by increasing the recycling rate, more output is produced from recycled inputs, which is less emissions intensive for most materials.

The changes in emissions due to policy changes are analytically determined below, decomposing the effect of each policy into *source reduction* and *increased recycling* terms. It should be noted that the analysis below captures the *direct* effects of the policy instruments on emissions. However, because the scrap price p_r is endogenously determined by Equation (3), this price will also depend on the level and type of policy intervention, generating indirect effects on emissions. While we abstract from this consideration in the derivations below, these scrap price feedbacks are captured in the simulation analysis.[15]

3.1 Deposit/refund

Consider a deposit/refund of d per ton. Consumers who fail to recycle experience a price of $p_q + d$, while those who recycle the product receive the refund back, offsetting the deposit.

[15] Kaffine (2010) analytically explores these indirect scrap price feedbacks in more detail. To summarize, scrap price increases under an advance disposal fee, generating an indirect *increased recycling* effect, while reducing the magnitude of the *source reduction* effect. Under a recycling subsidy, the scrap price falls, reducing the magnitude of both the *increased recycling* effect and negative *source reduction* effect. The scrap price under a deposit/refund can either fall or increase, depending on the magnitudes of the *increased recycling* and *source reduction* effects.

Thus, demand for the final product can be written as: $D(p_q+d, p_q-p_r)$. At the same time, the refund encourages recycling, such that the recycling rate is given by: $r(p_q+d)$. Emissions under this instrument are thus given by:

$$E(d) = \delta_v(1 - r(p_r + d))D(p_q + d, p_q - p_r) + \delta_r r(p_r + d)D(p_q + d, p_q - p_r) \tag{5}$$

or more compactly:

$$E(d) = (\delta_v + (\delta_r - \delta_v)r(p_r + d))D(p_q + d, p_q - p_r). \tag{6}$$

The change in emissions due to a change in the deposit d is given by:

$$\frac{dE}{dd} = (\delta_r - \delta_v)D(p_q + d, p_q - p_r)\frac{dr}{d(p_r + d)} + (\delta_v + (\delta_r - \delta_v)r(p_r + d))\frac{\partial D}{\partial(p_q + d)}. \tag{7}$$

The first term captures the emissions reduction from *increased recycling*. For most materials, $\delta_r < \delta_v$; therefore, as the refund increases the recycling rate, production is shifted from virgin inputs to less emissions-intensive recycled inputs, reducing emissions per unit of consumption.[16] Clearly, the larger the disparity between δ_r and δ_v, the greater the avoided emissions from the *increased recycling* effect. The second term captures the decrease in emissions from *source reduction*, as the deposit discourages consumption, but only for consumers who do not recycle. For those consumers that do recycle, the deposit/refund will not affect the price of the consumption good, and thus no *source reduction* occurs. Interestingly, the higher the recycling rate $r(p_r + d)$, the smaller the emissions reductions from the *source*

[16] This holds for all but writing and printing paper and paperboard.

reduction effect will be, as *source reduction* decreases consumption of goods produced from both high-emission virgin and low-emission recycled inputs.

Comparing these two effects, the magnitude of the *increased recycling* effect is likely smaller than the *source reduction* effect. Reduction of a ton of consumption via *source reduction* generates emission savings equal to the weighted average emissions associated with a ton of production. By contrast, recycling an additional ton generates emissions savings equal to the difference between virgin and recycling production emissions $(\delta_r - \delta_v)$. Thus, unless the recycling rate $r(p_r + d)$ is significantly more elastic with respect to the deposit/refund than consumption $D(p_q+d, p_q-p_r)$, *source reduction* will be a more important source of emissions reductions.[17] This stands in contrast to the case of waste reduction, where recycling an additional ton of material or simply not consuming an additional ton of material has an identical impact on waste disposal.

3.2 Advance disposal fee

Next, consider an advance disposal fee of f per ton. Consumers face this fee regardless if the material is recycled, such that prices for non-recycling consumers are given by $p_q + f$ and $p_q + f - p_r$ if recycled. Demand for the final good is thus $D(p_q + f, p_q + f - p_r)$ while the recycling rate $r(p_r)$ remains unchanged. Emissions under this instrument are thus given by:

$$E(f) = (\delta_v + (\delta_r - \delta_v)r(p_r))D(p_q + f, p_q + f - p_r) \tag{8}$$

[17] Several exceptions exist (aluminum cans, lumber, fiberboard, and carpet), where emissions savings from *increased recycling* are larger than those achieved with *source reduction* at the current input mix. This occurs because the difference between the emissions intensities of recycled and virgin inputs $(\delta_r - \delta_v)$ is large and the recycling rate $r(p_r + d)$ is high.

The change in emissions due to a change in the fee f is given by:

$$\frac{dE}{df} = (\delta_v + (\delta_r - \delta_v)r(p_r))(\frac{\partial D}{\partial(p_q + f)} + \frac{\partial D}{\partial(p_q + f - p_r)}) \tag{9}$$

In this case, because the recycling rate is unchanged, there is no induced shift from virgin to recycled inputs. However, the *source reduction* channel now has a second term, relative to the deposit/refund. Because consumers feel the effect of the ADF whether or not they recycle, the impact of source reduction on emissions will be more pronounced with an ADF than with a deposit/refund. However, it is unclear if the total reduction in emissions under the ADF is greater or less than under the deposit/refund. The ADF will lead to more reductions in emissions if the extra reduction in emissions from *source reduction* outweighs the *increased recycling* under a deposit/refund.

3.3 Recycling subsidy

Finally, consider a recycling subsidy of s per ton. Consumers who recycle receive a price of $p_r + s$, and the effective price of consumption for recycling households is thus $p_q - p_r - s$. Demand for the final good is then given by $D(p_q, p_q - p_r - s)$, and the recycling rate is $r(p_r + s)$. Emissions under this instrument are given by:

$$E(s) = (\delta_v + (\delta_r - \delta_v)r(p_r + s))D(p_q, p_q - p_r - s) \tag{10}$$

The change in emissions due to a change in the subsidy s is given by:

$$\frac{dE}{ds} = (\delta_r - \delta_v)D(p_q, p_q - p_r - s)\frac{dr}{d(p_r + s)} - (\delta_v + (\delta_r - \delta_v)r(p_r + s))\frac{\partial D}{\partial(p_q - p_r - s)} \tag{11}$$

The first term captures the emissions reduction from *increased recycling*, provided that $\delta_r < \delta_v$. On the other hand, the second term of the expression is positive, and actually increases emissions. This occurs because recycling subsidies effectively lower the price of goods for consumers who recycle, and through a negative *source reduction* effect, lead to an increase in consumption and emissions. This raises the theoretical possibility that recycling subsidies may even increase emissions.

To summarize, we can interpret the above results as follows: deposit/refund reduces both emissions per unit of consumption and the total level of consumption, advance disposal fee reduces total consumption but does not alter emissions per unit of consumption, and the recycling subsidy reduces emissions per unit of consumption while increasing total consumption. In the next section, we simulate the effects of these policy instruments on waste, recycling and emissions. This simulation analysis will allow us to explore the relative effects of these instruments for which the analytical decompositions are ambiguous. For example, does the *increased recycling* under a deposit/refund outweigh the additional *source reduction* under an ADF? Do the increases in emissions from the negative *source reduction* under a recycling subsidy outweigh the emission reductions from *increased recycling*? The simulations will also provide estimates of the marginal cost of waste and recycling policies net of emission reduction benefits, as well as allow us to explore optimal instruments and instrument levels across materials.

4 Simulation results

In the section below, we report the results of our simulation results, with an emphasis on how they compare with the results in Palmer et al. (1997).[18] We begin by examining the marginal benefit of emissions reduction per ton of waste reduced under a deposit/refund, advance disposal fee, and recycling subsidy. Next, we compare the net marginal cost of achieving a given reduction in total waste across policies. We then compare uniform intervention levels for each instrument (imposing the same deposit/refund, advance disposal fee or recycling subsidy across all materials), with the least-cost mix of material-specific instrument levels.

4.1 Benefits of greenhouse gas reductions from waste and recycling policies

For a given percentage reduction in waste, we calculate the necessary deposit/refund, advance disposal fee, and recycling subsidy. Based on the calculated instrument level, the emissions associated with the waste and recycling portfolio are determined, and the marginal benefits of reduced emissions (at \$25 per MTCO2E) per ton of waste reduced are plotted in Figure 1.[19] The vertical axis represents the marginal benefit per ton from GHG emission reductions

[18] PSW find that, for a 10% reduction in total waste, the marginal cost of the deposit/refund is \$45 per ton, the marginal cost of the ADF is \$90 per ton, and the marginal cost of the recycling subsidy is \$98 per ton. Based on an assumed social cost of \$33 dollars per ton of waste disposal, the authors conclude that a 7.5% reduction in total waste under a deposit/refund would be justified. These numbers will provide a useful benchmark for the discussion below.

[19] The US Interagency Working Group On Social Cost Of Carbon calculated several estimates of the social costs of carbon. \$25 per MTCO2E in 1990 dollars is roughly equivalent to the Working Groups's \$35 per short ton of CO_2 in 2007 dollars under an assumed 2.5% discount rate. See "Social Cost of Carbon for Regulatory Impact Analysis Under Executive

while the horizontal axis represents the percentage reduction in total waste.

[Figure 1 about here]

Several important points emerge from Figure 1. First, the advance disposal fee generates more emissions reductions (and thus benefits) than the deposit/refund, which suggests that the *increased recycling* effect from the deposit/refund (Equation 7) does not offset the larger *source reduction* effect from the advance disposal fee (Equation 9). The benefit of emissions avoided under the ADF is roughly $38 per ton of waste reduced, while the emission benefits under the deposit/refund varies from $25-$30 per ton. Second, the recycling subsidy generates substantially less reductions in emissions relative to the other policies (roughly $10 per ton of waste reduced). The analytical decomposition in the previous section provides intuition as to why this occurs - despite the decrease in emissions due to increased recycling rates, the subsidy increases consumption by lowering the cost of consumer goods, increasing emissions and offsetting the reduction from increased recycling. Note, however, that because the benefit curve under the recycling subsidy is still positive, the *increased recycling* effect dominates the negative *source reduction* effect. Finally, the total benefits from emissions reductions caused by waste and recycling policies can be substantial. Under an advance disposal fee, a 10% reduction in total waste would generate $342 million dollars in benefits through a reduction in greenhouse gas emissions of 13.7 million tons.

Order 12866" at http://www.epa.gov/oms/climate/regulations/scc-tsd.pdf and Greenstone et al. (2011) for further details on methodologies and assumptions. In section 5, we explore the sensitivity of our main results to alternative specifications of the social cost of GHG emissions.

4.2 Least-cost policy with greenhouse gas externalities

We now examine the net marginal cost of the three policy instruments, incorporating the benefits of greenhouse gas reductions. Again, for a given percentage reduction in waste, we calculate the necessary deposit/refund, advance disposal fee, and recycling subsidy. Based on the level of the instrument, the emissions associated with the waste and recycling portfolio are determined, and the benefits of reduced emissions (at $25 per MTCO2E) are subtracted from the marginal cost of the policy. Figure 2 displays the net marginal cost per ton of achieving a given percentage reduction in total waste, where the vertical axis represents the marginal cost of the policy net of greenhouse gas benefits (scaled to $/ton of waste), and the horizontal axis represents the percentage reduction in total waste disposal.

[Figure 2 about here]

Several results are to be highlighted. At $25 dollars per MTCO2E as the social cost of carbon and $33 dollars as the marginal social damage of a ton of waste (PSW's estimate of the social costs of waste disposal), a 14% reduction in total solid waste would be justified (under deposit/refund).[20] This represents a nearly 100% increase in total waste reduction relative to 7.5% in PSW. A social cost of waste disposal of $5-9 dollars as in Kinnaman (2006) would justify an 8% reduction in total waste, compared to a less than 1% reduction based on the the social cost of waste disposal alone. In the extreme case where waste disposal generates no social costs, a reduction in total waste of roughly 6% is justified solely by emissions savings. Finally, we note that the net marginal cost of a deposit/refund to achieve a 10% reduction in solid waste is substantially smaller than in PSW, at $19 dollars per ton

[20] Because the benefits of emissions reductions per ton of waste reduced have been netted from the vertical axis, the level of intervention is given where the net marginal cost per ton of waste equals $33 dollars.

versus $45 dollars per ton. This suggests that the emissions avoided via waste and recycling policies are an important factor that should be considered when selecting policy intervention levels.

Surprisingly, the advanced disposal fee is the least-cost policy for waste reductions of less than 4%. This occurs because the ADF provides the most benefit per ton in terms of reduced greenhouse gas emissions per Figure 1, while the primary cost for all policies is relatively low for small reductions in waste. It should be noted that the marginal benefit per ton in terms of reduced greenhouse gases is relatively flat, while the marginal primary cost per ton increases with respect to the reduction in total waste (per Figure 1 in PSW). The deposit/refund has the lowest marginal primary cost per ton, and as a result, the deposit/refund is the least-cost policy for larger waste reductions above 4%. In other words, the larger marginal benefits per ton (from reduced greenhouse gases) from the advance disposal fee initially makes it more attractive than the deposit/refund, but the increasing magnitude of the primary costs eventually leads the deposit/refund to be the least-cost policy.

Finally, the net marginal cost of the recycling subsidy is only slightly reduced by the inclusion of benefits from reduced emissions. The net marginal cost for a 10% reduction in total waste is $87 dollars per ton, only slightly less than the $98 dollars per ton in PSW. By contrast, both the deposit/refund and the advance disposal fee see large reductions in net marginal cost, from $45 per ton down to $19 per ton for the deposit/refund, and from $90 dollars per ton down to $45 dollars per ton for the advance disposal fee.

4.3 Material-specific instruments

In the above analysis, the instrument levels were assumed to be uniform across all material types. In Palmer et al. (1997), this is the optimal policy because materials are undifferentiated in terms of the social costs of disposal. Here, we set material-specific instrument levels to minimize the marginal cost of waste reduction net of emissions benefits; for example, allowing the advance disposal fee for glass to be different than the advance disposal fee for aluminum.[21] The intuition is that, due to heterogeneity in emissions by material, we should adjust the instrument level by material accordingly - increasing instrument levels for materials that provide larger emissions reductions and correspondingly reducing instrument levels for materials with less emissions reductions.

For a 10% reduction in total waste, the use of material-specific instrument levels provides a modest reduction in net marginal cost. For the deposit/refund, net marginal cost with material-specific levels is \$16.9 per ton versus \$18.7 per ton. A material-specific ADF has a net marginal cost of \$41.0 per ton compared to \$45.2 per ton under a uniform ADF. Under a material-specific recycling subsidy, net marginal cost per ton is \$80.7 compared to \$87.3 under a uniform recycling subsidy. Thus the cost-savings from material-specific instruments levels is roughly a 10% reduction in the net marginal cost of achieving a given waste reduction.

Table 2 reports the reductions in waste and emissions under the material-specific intervention levels. The material-specific deposit/refund varies substantially by material, from a high of \$423.78 dollars per ton for aluminum to a low of \$25.65 per ton for glass. The large

[21] Operationally, intervention levels were adjusted for each material until the specified reduction in total waste was met at minimum total cost.

deposit/refund on aluminum reflects the fact that substantial emissions savings are achieved through reducing the use of virgin inputs in production. Despite the large deposit/refund, the smallest share of total waste reduction comes from aluminum, with paper's $36.45 per ton deposit/refund generating over one-third of the total waste reduction. Though aluminum provides a small share in total waste reduction, aluminum has the highest percentage reduction in waste (31%) of all materials under a material-specific deposit/refund. This stands in stark contrast to the uniform deposit/refund, which results in a minimal reduction in aluminum waste of 3.6%. Comparing with the uniform deposit/refund, the percentage reduction in steel waste rises from 11.2% to 19.8%, while glass falls from 26.4% to 16.4%. Finally, despite aluminum's small contribution to total waste reduction, it contributes the most in terms of emissions reductions (6.484 million MTCO2E), followed closely by steel (4.810 million MTCO2E). Also of note, emissions reductions are substantially larger (50% more) under the variable deposit/refund compared to the uniform deposit/refund.

[Table 2 about here.]

Similar trends across materials occur for the advance disposal fee, though it should be noted that total emissions reductions under the ADF are larger under both the uniform and material-specific ADF relative to deposit/refund. By comparison, we see a slightly different pattern of waste reduction by material under a recycling subsidy. The percentage reduction in glass waste is substantially higher and the reduction in aluminum waste substantially lower. Furthermore, the reduction in plastics under a recycling subsidy is effectively zero. Recycling subsidies generate no reductions in plastic waste (the increased recycling effect is offset by the negative source reduction effect), and as such, the marginal cost of waste reduction is essentially infinite. In general, material-specific intervention levels tend to favor larger percentage waste reductions in aluminum and steel relative to the uniform policy.

To put this in perspective, under a 10% reduction in waste via a uniform deposit/refund, the deposit/refund is equal to 0.12 cents per aluminum can, 0.22 cents per plastic bottle, 5.6 cents per newspaper, 0.5 cents per steel can, and 1.7 cents per glass bottle.[22] When the deposit/refund is set variably across materials to minimize the net marginal cost of achieving a 10% reduction in waste, the deposit/refund is equal to 1.1 cents per aluminum can, 0.35 cents per plastic bottle, 4.5 cents per newspaper, 0.83 cents per steel can, and 1 cent per glass bottle. Relative to the uniform deposit/refunds, the material-specific deposit/refunds are substantially larger per aluminum and steel can, reflecting the emissions savings from targeting those materials.

5 Further analysis

In this section, we explore two additional questions. In the previous section, we find that there are modest cost-savings associated with allowing material-specific intervention levels by policy. We now look at potential cost-savings from allowing both variable policies and variable intervention levels by material (for example, a $25 dollar per ton advance disposal fee for glass and a $40 dollar per ton deposit/refund for paper). We also explore the sensitivity of our primary results to alternative assumptions regarding the social costs of greenhouse gas emissions.

[22] The deposit/refund has been inflation-adjusted to give an idea of the level of the deposit/refund in 2011$.

5.1 Variable policies by material

We first consider the least-cost combination of instruments and instrument levels by material. Table 3 reports the least-cost policy by instrument and the corresponding instrument level for 2.5%, 5%, 10% and 20% reductions in total waste. For larger reductions in total waste, the deposit/refund is selected for all materials. However, for smaller reductions in waste, the advance disposal fee is preferred for paper and glass.[23] Consistent with the previous sections, the recycling subsidy is never the least-cost instrument for any material at any level of total waste reduction.

[Table 3 about here]

The final two rows for Table 3 compare the net marginal cost per ton when instruments are allowed to vary by material (Variable Policy) with the net marginal cost per ton when only a single instrument can be used (Uniform Policy). Allowing variable policy instruments by material does produce minor cost-savings, but only at the smallest levels of total waste reduction. For larger reductions in total waste, the deposit/refund is always the least-cost instrument. From the perspective of policymakers, the administrative costs of managing separate policy instruments for different materials may outweigh the small potential cost-savings offered by the flexibility of material-specific instruments.

5.2 Sensitivity to alternative social costs of GHG emissions

Next, we consider the importance of the assumed $25 per MTCO2E cost of GHG emissions. To explore the sensitivity of our results to emissions assumption, we compare, for a 10%

[23] At 2.5% reduction in total waste, no intervention is necessary in the glass market.

reduction in total waste, the net marginal costs of the deposit/refund, advance disposal fee, and recycling subsidy across varying values for the social costs of GHG emissions. The net marginal costs are plotted on the vertical axis of Figure 3, with the social costs of CO_2 emissions on the horizontal axis.

[Figure 3 about here]

From Figure 3, we see that the deposit/refund is the least-cost policy for a 10% waste reduction within a wide-range of the $25 dollar per MTCO2E assumption. However, it should be noted that the net marginal cost of the advance disposal fee decreases the fastest of all the policies, and at $80 dollars per MTCO2E, it emerges as the least-cost policy. This is intuitive, as the advance disposal fee produces the greatest reductions in CO_2 equivalent emissions, per Figure 1. As Figure 1 also foreshadows, the net marginal costs of the recycling subsidy fall the slowest as the social costs of emissions rise. Strikingly, even at $100 dollars per MTCO2E, the recycling subsidy with emissions savings included is more costly than the deposit/refund with *none* of the benefits of emissions reductions included.

6 Conclusions

In recent years, a renewed interest in recycling and waste policies has surfaced, as a consequence of rising concerns over greenhouse gas emissions from the production of various consumer goods. In the absence of a carbon pricing policy, waste and recycling policies could be adjusted to address upstream greenhouse gas emissions externalities. We compare three waste reduction and recycling policy alternatives and weigh the costs of each policy net the benefits of greenhouse gas emissions reductions. Accounting for the emissions reductions from waste and recycling policies substantially increases the justified level of total waste reduction. We find that the social benefits of emissions reductions from waste and recycling policy are of the same order or significantly larger than the social benefits of avoided waste disposal. Furthermore, when emissions reductions are incorporated, the net marginal cost of achieving a given waste reduction is substantially reduced, particularly for the deposit/refund and advance disposal fee.

We also provide a decomposition of the three policies to highlight the two channels of emissions reductions that are exploited by the policies. The *source reduction* channel generates greater emissions savings for most materials, by reducing all emissions associated with production, while the *increased recycling* channel only generates emissions savings equal to the emissions difference between production from virgin and recycled inputs. The ADF provides the greatest benefits in emissions reductions. Moreover, the ADF is the least-cost policy for small reductions in total waste disposal, with the deposit/refund emerging as the least-cost policy when total waste reduction increases beyond 4%. Despite the fact that fewer emissions are produced from recycled inputs relative to virgin inputs, a direct recycling subsidy is the most costly policy of the three instruments examined in this study and produces the least emissions reductions.[24] This is driven in part by the negative *source*

[24] A limitation of the current study is that it only considers market instruments. A beneficial extension of this study would be to consider non-market policies such as curbside recycling and extended producer responsibility. Conceptually, curbside recycling functions similar to a direct recycling subsidy, with the key difference that it does not encourage additional consumption. Thus, emissions reductions would occur solely through the *increased recycling* channel. Extended producer responsibility (EPR) would be challenging to evaluate in the framework presented above, given the heterogeneity in EPR implementation (Fleckinger and Glachant 2010). On the one hand, EPR may function like an advance disposal fee by simply raising the cost of consumption goods, generating fewer emissions via *source reduction.* On the other hand, if EPR encourages firms to "Design for Environment" (Calcott and Walls 2000) by improving recyclability of products, then EPR may function

reduction channel, effectively increasing consumption by lowering the final price of consumer goods for consumers who recycle.

This study also finds that a modest cost-savings of 10% is achieved when policy instrument levels are allowed to vary by material, finding a substantial variation in the instrument level across materials. For a 10% reduction in total solid waste, the optimal deposit/refund assigns a high of $423.78 per ton of aluminum to a low of $25.65 per ton of glass, highlighting the large disparity in emissions savings between the materials. We also consider an optimal policy design that allows instrument levels and instrument type to vary across material, but find that potential cost-savings are minor and may be outweighed by increased implementation costs. Finally, a sensitivity analysis shows that rising social costs of greenhouse gas emissions enhance the attractiveness of the ADF, as the emissions savings benefits from the *source reduction* channel increase.

We note several results of particular policy relevance. First, though we consider only one externality of upstream production (greenhouse gas emissions reduction), the benefits of addressing this single upstream externality are roughly as large or larger than the benefits associated with avoided waste disposal. Incorporating additional externalities such as land-use and forestry-related externalities from paper production or extraction externalities of aluminum and iron ore production would further highlight the importance of considering upstream production when deciding amongst waste and recycling policies. In fact, it may be the case that in the absence of carbon pricing, the most important aspect of waste and recycling policy is the impact on these upstream production externalities, with avoided disposal of waste in landfills a secondary concern. Finally, while the reduction in emissions from the

similar to a deposit/refund as emissions per unit of consumption are also reduced.

use of recycled inputs may make direct recycling subsides intuitively appealing, our results should give policymakers pause. While recycling rates will increase under recycling subsidies, the rebound effect of increased consumption can significantly erode emissions reductions, generating smaller reductions in emissions at a higher cost than either deposit/refunds or advance disposal fees.

References

Ashenmiller, B. (2009). Cash Recycling, Waste Disposal Costs, and the Incomes of the Working Poor: Evidence from California. *Land Economics 85*(3), 539.

Beatty, T., P. Berck, and J. Shimshack (2007). Curbside recycling in the presence of alternatives. *Economic Inquiry 45*(4), 739.

Benjamin, D. K. (2010). *Recycling Myths Revisited*. PERC Policy Series, No. 47.

Calcott, P. and M. Walls (2000). Can downstream waste disposal policies encourage upstream "design for environment". *American Economic Review 90*(2), 233–237.

Dinan, M. T. (1993). Economic efficiency effects of alternative policies for reducing waste disposal. *Journal of Environmental Economics and Management 25*(3), 242–256.

Fleckinger, P. and M. Glachant (2010). The organization of extended producer responsibility in waste policy with product differentiation. *Journal of Environmental Economics and Management 59*(1), 57–66.

Fullerton, D. and T. Kinnaman (1995). Garbage, recycling, and illicit burning or dumping. *Journal of Environmental Economics and Management 29*(1), 78–91.

Fullerton, D. and T. Kinnaman (1996). Household responses to pricing garbage by the bag. *American Economic Review 86*(4), 971–984.

Greenstone, M., E. Kopits, and A. Wolverton (2011). Estimating the social cost of carbon for use in U.S. federal rulemakings: A summary and interprtation. *NBER Working Paper 16913*.

Ino, H. (2011). Optimal environmental policy for waste disposal and recycling when firms are not compliant. *Journal of Environmental Economics and Management In press*.

Jenkins, R. (1993). *The Economics of Solid Waste Reduction: The Impact of User Fees*. Edward Elgar Press, Brookfield, VT.

Jenkins, R., S. Martinez, K. Palmer, and M. Podolsky (2003). The determinants of household recycling: a material-specific analysis of recycling program features and unit pricing. *Journal of Environmental Economics and Management 45*(2), 294–318.

Kaffine, D. T. (2010). The importance of scrap prices in waste and recycling policy. *Working paper*.

Katz, J. (2002). What a waste. *Regional Review 12*(1), 22–30.

Kinnaman, T. and D. Fullerton (2000). 3. The economics of residential solid waste management. *The International Yearbook of Environmental and Resource Economics 2000/2001: A Survey of Current Issues*, 100.

Kinnaman, T. C. (2006). Examining the justification for residential recycling. *Journal of Economic Perspectives 20*(4), 219–232.

Palmer, K., H. Sigman, and M. Walls (1997). The cost of reducing municipal solid waste. *Journal of Environmental Economics and Management 33*(2), 128–150.

Palmer, K. and M. Walls (1997). Optimal policies for solid waste disposal Taxes, subsidies, and standards. *Journal of Public Economics 65*(2), 193–205.

Sigman, H. (1995). A comparison of public policies for lead recycling. *The RAND journal of Economics 26*(3), 452–478.

Walls, M. and K. Palmer (2001). Upstream Pollution, Downstream Waste Disposal, and the Design of Comprehensive Environmental Policies. *Journal of Environmental Economics and Management 41*(1), 94–108.

Yang, H.-L. and R. Innes (2007). Economic incentives and residential waste management in taiwan: An empirical investigation. *Environmental and Resoure Economics 37*, 489–519.

A Greenhouse gas emissions data sources and assumptions

The data used in the current study was obtained from"Solid Waste Management and Greenhouse Gases: A Life-Cycle Assessment of Emissions and Sinks" an analysis conducted by the EPA in 2002.[25] The EPA estimated the greenhouse gas emissions resulting from fossil fuel combustion during the acquisition, manufacturing and transportation of raw materials into manufactured goods. All energy data originated from the EPA's Office of Research and Development (ORD) and from Franklin Associates Ltd. (FAL). The report was created to highlight the greenhouse gas emissions implications of solid waste policy decisions. The degree to which greenhouse gas emissions from various waste materials differ determines the impact of alternative waste and recyclign policies across material type.

There were four greenhouse gas emissions estimates made for each material, representing the process energy and transportation energy for materials made from either 100% virgin or 100% recycled inputs. Exceptions were made for steel cans and corrugated box which assume 20% and 14.7% minimum recycled contents respectively.

A.1 Emissions from process energy

Energy required for acquisition and manufacturing of raw materials is denoted as process energy, with emissions primarily consisting of CO_2. Specifically, the CO_2 emissions from

[25] We include the following information as an appendix, primarily as information for reviewers, due to the fact that emissions by material and input type are crucial values for this study - the authors are happy to have this information posted as an online supplemental to conserve journal space.

combustion of fuels for mining activities and from generation of electricity for manufacturing activities were calculated in the process energy emissions. Process energy emissions also include those generated from activities involved in oil exploration and extraction, coal mining, natural gas production, and other precombustion activities requiring energy. In addition to CO_2 considerations, total process emissions from raw material acquisition and manufacturing also include CH_4 emissions associated with producing, processing, and transporting of coal, oil, and natural gas.

A.2 Emissions from transportation energy

Transportation data was obtained using the US Bureau of Transportation Statistics Commodity Flow Survey which reported the average distance commodities are shipped as well as the breakdown of transportation modes used within the country. Transportation emissions were added to both virgin and recycled materials. In the case of emissions reduction from transportation, source reduction of materials would have a more significant impact than increased recycling, reflecting the similar energy requirements for transportation of both virgin and recycled materials.

Greenhouse gas emissions from transportation activities were calculated as the CO_2 emissions from the combustion of fuels used to transport raw materials to the final manufacturing facilities. For recycled materials, transportation from curbside to the materials recovery facility (MRF), from MRF to broker, and from broker to final manufacturing facility was computed. Additionally, the transportation emissions values for recycled materials include the energy used to process inputs at the MRF. For both material types, emissions from transportation of the final good from the manufacturer to the retailer was included and the transportation to consumers was excluded. Emissions values of CH_4 or N_2O, however, were not included in the transportation calculations as these emissions are assumed to be significantly smaller in magnitude than the CO_2 values produced by transportation activities. The last assumption results in a slight understatement of the total greenhouse gas emissions and of the share of emissions from transportation activities.

A.3 Energy requirement assumptions

For the energy requirements for each material, both the ORD and FAL data sets were created using published engineering and production data in addition to industry experts and trade organizations. All known energy consumption in the production processes, whether used abroad or in the United States, was incorporated in the data. For instance, the energy required for mining and processing bauxite overseas for the production of aluminum was included.

The energy requirements in the analysis assume that materials undergo closed loop recycling, where primary materials are recycled into the same material. In reality, some materials are not remanufactured into the same material type and thus would require an open loop. Such open loop life cycles are commonly observed for mixed paper, cardboard, carpet, and computers. For plastics, recycled materials were given a single energy profile because recycling processes for HDPE, LDPE, and PET are similar. For steel can production, the EPA made estimates for greenhouse gas emissions from virgin production using a basic oxygen process and from recycled production using an electric arc furnace process.

To estimate the emissions from electricity, the EPA used a national average mix of fuels which may vary regionally. It is important to note that many consumer products are increasingly originating from international sources where the fuel mixes and energy requirements may differ significantly. Similarly, different manufacturers within the US will have different processes with varied energy requirements and efficiency profiles.

A.4 Carbon coefficient assumptions

EPA used the carbon coefficients provided by the Depart of Energy's Energy Information Administration for all energy sources with exception of electricity. The carbon coefficients used included the CO_2 emissions generated from fuel combustion. To elaborate on these values, the EPA added the average CH_4 emitted from the production, processing, and transportation of fossil fuels and the CO_2 emitted from oil production and flaring of natural gas. The resulting values were used as the average greenhouse gas emissions related to production of coal, oil, and natural gas. The coefficient for electricity was calculated as a weighted average of the national mix of fuel usage. For the carbon coefficient of electricity, the EPA used data calculated by the FAL that already incorporated the efficiency of converting each fuel to electricity and the corresponding loss in transmission and distribution of electricity.

A.5 Limitations to emissions data

The EPA believes that these average emissions values provide a good approximation of the actual greenhouse gas emissions; however, it must be noted that these are averages and not marginal emissions. These values do not represent the marginal emissions rate per incremental ton of material produced. It may be the case for some materials that the marginal emissions may differ significantly from the averages used here. Moreover, the relationship between recycling and greenhouse gas emissions may be non-linear, such that reducing the production of a material from virgin inputs may not result in a proportional decrease in the greenhouse gas emissions. This suggests that as recycling rates increase, emissions reductions may occur at a declining rate. Greenhouse gas emissions from production and transportation of materials may shift in a nonlinear fashion if capacity constraints, scale economies, or other factors exist. In the case of electricity a long-term decline in the demand for electricity may reduce demand for specific fuels and not an overall reduction in all fuels in proportion to their share within the average fuel mix. Nonetheless, the average carbon coefficients are used primarily because the marginal rates would be difficult to estimate.

Additionally, national averages for electricity generations are general representations of emissions. A more specific regional analysis would be required in order to more accurately capture the effect of the local fuel mix on overall emissions from electricity.

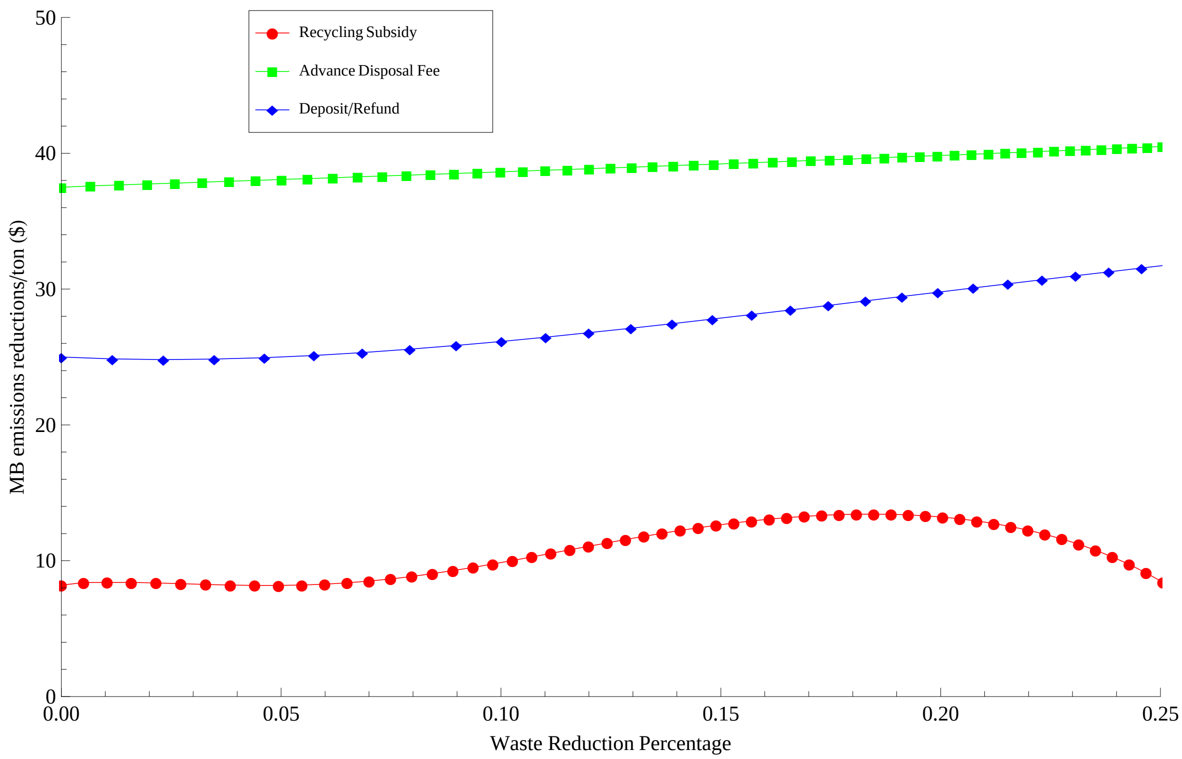

Figure 1: Marginal benefits per ton associated with greenhouse gas reductions under the deposit/refund, advance disposal fee, and recycling subsidy necessary to achieve various percentage waste reductions.

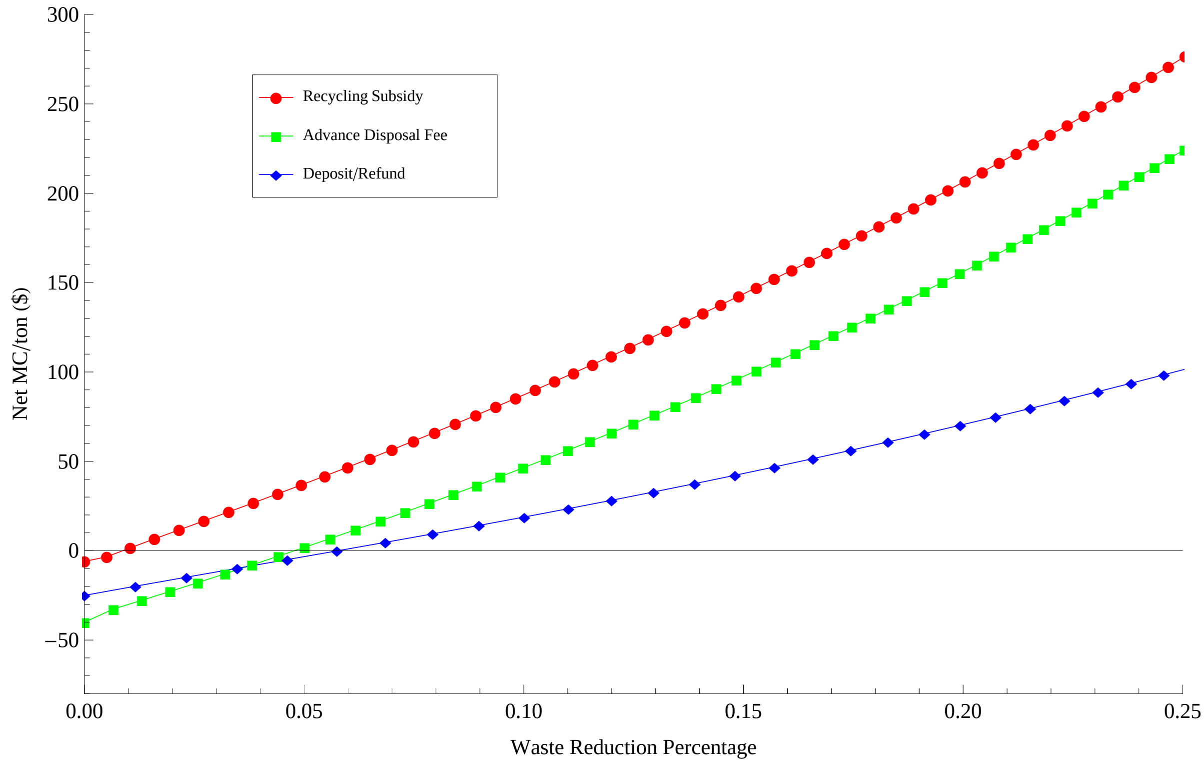

Figure 2: Net marginal cost per ton of the deposit/refund, advance disposal fee, and recycling subsidy necessary to achieve various percentage reductions in total waste disposal.

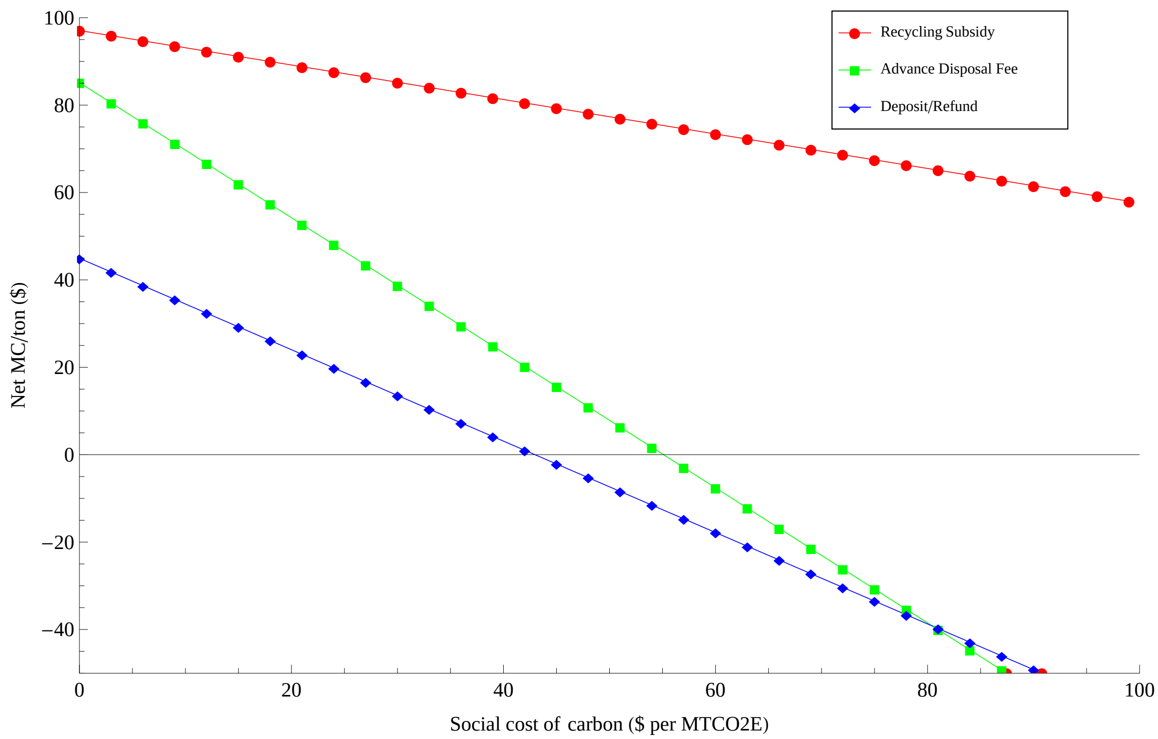

Figure 3: Net marginal cost per ton of the deposit/refund, advance disposal fee, and recycling subsidy for a 10% reduction in total waste for varying values of the social cost of greenhouse gas emissions

Table 1: Combined Process and Transportation Energy Emissions (MTCO2E/ton)

Material	Virgin	Recycled
Paper and paperboard		
Newsprint	2.13	1.25
Writing and printing (office)	0.99	1.36
Paperboard (cardboard)	0.84	0.92
Other	1.98	0.92
Glass		
Beverage containers	0.48	0.33
Durables	0.48	0.33
Aluminum		
Cans	12.94	0.95
Other containers	12.94	0.95
Durables and misc. nondurable	12.94	0.95
Steel		
Cans	2.82	0.99
Other containers	2.82	0.99
Durables	2.82	0.99
Plastics		
PET	2.05	0.18
HDPE	1.76	0.18
Other nondurables	1.76	0.18
Durables	2.16	0.18

Note: Values obtained from "Solid Waste Management and Greenhouse Gases: A Life-Cycle Assessment of Emissions and Sinks, Chapter 2: Raw Materials Acquisition and Manufacturing," EPA. Values for aluminum and steel were not differentiated by type.

Table 2: Uniform and material-specific intervention levels for a 10% reduction in waste

	Uniform	Material-specific				
		Paper	Glass	Aluminum	Steel	Plastic
Deposit/refund ($ per ton)	$44.94	$36.72	$25.93	$424.26	$77.31	$70.46
Waste reduction (million tons)	9.163	3.813	1.543	0.511	2.056	1.240
Percent waste reduction	10%	7.2%	16.1%	31.0%	19.7%	7.7%
Emissions reduction (million MTCO2E)	9.212	2.846	0.437	6.473	4.837	2.340
Percent emissions reduction	5.3%	3.4%	7.4%	28.9%	15.5%	7.7%
Advance disposal fee ($ per ton)	$85.15	$75.39	$59.91	$505.23	$123.26	$95.94
Waste reduction (million tons)	9.163	4.020	1.420	0.446	1.640	1.638
Percent waste reduction	10%	7.6%	14.9%	27.1%	15.8%	10.2%
Emissions reduction (million MTCO2E)	13.944	4.976	0.729	5.875	4.717	3.093
Percent emissions reduction	8.0%	5.9%	12.4%	26.2%	15.1%	10.2%
Recycling Subsidy ($ per ton)	$97.07	$79.13	$85.24	$597.22	$126.98	$0
Waste reduction (million tons)	9.163	3.813	2.927	0.208	2.215	0
Percent waste reduction	10%	7.2%	30.6%	12.6%	21.3%	0%
Emissions reduction (million MTCO2E)	3.120	0.220	0.261	2.241	3.824	0
Percent emissions reduction	1.8%	0.3%	4.64%	10.0%	12.2%	0%

Note: Material-specific instrument levels calculated as the instrument level that achieves a 10% reduction in total waste at a minimum net cost. As a point of comparison, Palmer et al. (1997) find that for the uniform deposit/refund required to achieve a 10% reduction in total waste, paper, glass, aluminum, steel, and plastic waste are reduced by 8.7%, 26.4%, 3.6%, 11.2%, and 5.1% respectively. For the advance disposal fees, material reductions are 8.5%, 20.0%, 5.4%, 11.5%, and 9.1%, and for the recycling subsidy, material reductions are 8.4%, 33.6%, 1.6%, 14.2%, and .04%.

Table 3: Material-specific optimal instrument and intervention level for various waste reduction percentages

	Total Waste Reduction			
	2.5%	5%	10%	20%
Paper and paperboard				
Instrument	ADF	ADF	Dep/Ref	Dep/Ref
Instrument Level	\$6.54	\$26.72	\$36.72	\$89.18
Glass				
Instrument	N/A	ADF	Dep/Ref	Dep/Ref
Instrument Level	\$0	\$10.67	\$25.93	\$78.44
Aluminum				
Instrument	Dep/Ref	Dep/Ref	Dep/Ref	Dep/Ref
Instrument Level	\$379.34	\$400.17	\$424.26	\$477.65
Steel				
Instrument	Dep/Ref	Dep/Ref	Dep/Ref	Dep/Ref
Instrument Level	\$37.42	\$55.89	\$77.31	\$125.53
Plastics				
Instrument	Dep/Ref	Dep/Ref	Dep/Ref	Dep/Ref
Instrument Level	\$26.70	\$46.99	\$70.46	\$122.54
Net Marginal Cost/Variable policy	-\$26.02	-\$5.99	\$17.17	\$68.54
Net Marginal Cost/Uniform policy	-\$23.37	-\$5.30	\$17.17	\$68.54

Note: Level indicates the price of the specific intervention required to achieve the target waste reduction across all materials.