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THE ECONOMICS OF RECYCLING HETEROGENEITY

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ABSTRACT

We summarize the economics of recycling municipal solid waste. OECD data suggest that aggregate recycling rates in member countries have plateaued in recent decades. United States recycling rates for some materials remain low, even after decades of learning and participation. Major new policies may be required to increase recycling rates. Aggregate recycling targets are common in the US and OECD but may no longer be effective. We discuss many sources of recycling heterogeneity often ignored. First, recyclable materials are very different from each other. Economies and natural environments differ across space, and technologies change over time. Recycling policies that ignore heterogeneity are likely not set appropriately. If policy costs are low enough to set a unique recycling policy for each material in each locality, then a surgical recycling strategy may better serve society. Specific recycling policies for automobile batteries differ greatly from policies for yard waste, because these two materials are different. A surgical recycling policy could extend this practice, so that rules for aluminum cans differ from plastic jugs or glass bottles. Reaching future recycling targets could be frustrated by ignoring these heterogeneities across materials, locations, and time.

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Developed countries have had nearly fifty years of experience with recycling municipal solid waste. An empirical examination of national recycling averages suggests that a difficult ceiling on aggregate recycling rates may have been reached in developed countries. Overcoming this ceiling may require new aggressive policies to promote recycling. The success of these policies might be hampered by the heterogeneous nature of many aspects related to recycling – making any single one-size-fit-all recycling policy overly blunt and ineffective.

After a summary of the recycling data in the U.S. and more widely across the OECD, this article discusses five sources of heterogeneity and their associated policy challenges. One section's description of relevant market failures helps motivate the subsequent discussion of potential recycling policies. Then the following section points out how heterogeneity makes it difficult to know if more recycling raises social welfare. A more "surgical" approach to recycling policy may be wise, but we also discuss why actual policymakers have resisted surgical policies.

I. Historical Recycling Data

Recycling in the United States

In the 1970's and early 1980's, only portions of the United States organized the recycling of municipal solid waste. A growing public aversion to solid waste disposal facilities led the federal government in 1976 to enact Subtitle D of the Resource Conservation and Recovery Act (RCRA). RCRA laid down new technology-based standards on how solid waste landfills will be constructed and operated and essentially transformed waste disposal in the United States from the reliance on local town dumps to large, highly capitalized, regional landfills.

Public aversion to waste disposal continued, and many state governments responded in the 1980's by passing new laws designed to stimulate the recycling of municipal waste. The most impactful of these state policies required all municipalities to implement a curbside recycling program or to meet a minimum recycling rate. Many states that passed such legislation, roughly 25 in total, were located in the northwest and northeast regions of the country where public aversion to landfills was strong and where disposal costs charged by the new generation of regional landfills was high. Various state governments also stimulated recycling by banning yard waste from landfills (21 states) and banning all recyclable materials from landfills (4 states). The result, even up to now, was a national patchwork of recycling policies with no federal presence.

Within this context, municipal governments in large sections of the country were compelled to offer households free curbside and drop-off recycling programs. These programs sharply reduced recycling costs to households. Early recycling programs typically collected only a few materials such as newspapers, aluminum, and maybe glass. Additional materials such as plastics and textiles were added over time.

Households eventually became accustomed to the new curbside and drop-off recycling practices, at least for some materials. **Table 1** shows the percentage of each material recycled in the United States today – roughly 40 years or two generations after recycling was first initiated. Americans today recycle roughly 0.67 million tons of aluminum beer and soft drink cans per year – about 50 percent of all aluminum cans purchased. The recycling rate for all aluminum containers is 35%, including beverage containers, food containers, foil, and other aluminum packaging. Americans recycle 31% of their glass bottles, 29% of their PET bottles, 29% of their

HDPE containers, and 33% of their tin cans. Households recycle 93% of their cardboard boxes and 68% of their mixed paper (including, *e.g*., office paper, cereal boxes, envelopes).[1]

Perhaps ideally, households would now recycle closer to 100% of each type of material. But after 40 years of household experience with recycling, it's hard still to employ the old argument that households are simply not informed about how to recycle. Future increases in recycling might instead require aggressive new incentives at this point. If so, those incentives would need to originate with new public policy.

Table 1: Recycling Rates for Common Recyclable Materials in the U.S.

Recyclable Item	Recycling Rate
Aluminum Cans	50%
Glass Bottles	31%
PET Plastic Beverage Bottles	29%
Corrugated Boxes	93%
Mixed Paper	68%
HDPE Plastic Jugs	29%
Bi-metal "tin" cans	33%

HDPE is high-density polyethylene. PET is polyethylene terephthalate.

Recycling Across the World

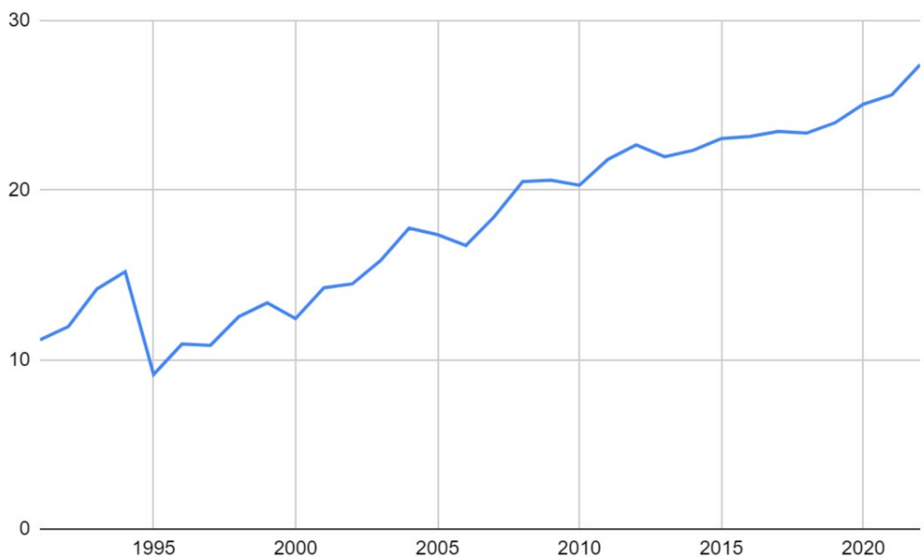
Presented in **Figure 1** is the average recycling rate each year for all developed countries, based on data obtained from the OECD Data Explorer.[2] The data are based on municipal solid waste (MSW), which is comprised mostly of residential and commercial waste but not industrial waste, construction waste, or demolition waste. Thus, the figure does not count recycling carried out by private entities outside of MSW. The automobile, for example, is often considered the most recycled consumer product. The steel industry recycles roughly 14 million tons of steel from old automobiles every year – nearly 100% of the original steel used to produce those autos. Glass and plastic are also extracted from scrapped automobiles. Only what remains of the automobile – the "automotive shredded residue" – is taken to a landfill. But because auto recycling is unrelated to MSW, it is not included in the recycling rates reported in Figure 1.

[1] Data in Table 1 were obtained from a variety of web pages. Aluminum: https://www.epa.gov/facts-and-figures-about-materials-waste-and-recycling/aluminum-material-specific-data, glass: https://www.epa.gov/facts-and-figures-about-materials-waste-and-recycling/glass-material-specific-data, plastic: https://napcor.com/news/2022-pet-recycling-report/#:~:text=FIGURE%201%3A%20PET%20Bottle%20Collection%20Rates%2C%202001%20%E2%80%93%202022&text=The%20PET%20recycling%20rate%20in,slightly%20from%2038.4%25%20in%202021, tin: https://www.internationaltin.org/recycling/#:~:text=For%20tin%20products%2C%20the%20average,RIR%20of%20tin%20was%2033.1%25, paper: https://www.afandpa.org/news/2023/us-paper-and-cardboard-recycling-rates-continue-hold-strong-2022

[2] https://dataexplorer.oecd.org/vis?tm=recycling%20rates&pg=0&snb=3&vw=tb&df[ds]=dsDisseminateFinalDMZ&df[id]=DSD_MUNW%40DF_MUNW&df[ag]=OECD.ENV.EPI&df[vs]=1.0&dq=.A.RECYCLING%2BMUNICIPAL.PT_WST_TR&pd=1980%2C&to[TIME_PERIOD]=false

According to these data, this average has steadily increased from roughly 10% in the early 1990s to over 25% in 2022. On the surface, this historical steady growth in the recycling rate appears sustainable. A future circular economy featuring high recycling rates seem promising.

FIGURE 1: Aggregate MSW Recycling Percentage in OECD Countries by Year

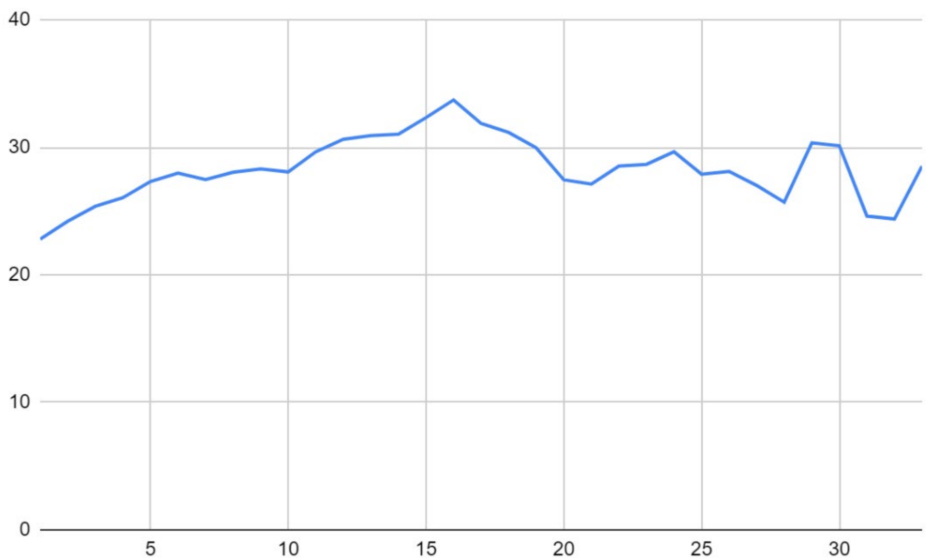


But, are these data deceiving? The gradual increase in the global average recycling rate can be attributed to two broad factors. The first is that the number of countries engaging in any recycling has gradually increased over time. When a country first initiates recycling and increases their recycling rate from 0% to something like 10%, then the average recycling rate across all countries increases for that year and subsequent years as that country continues to recycle. Second, already mature recycling programs can experience increased recycling rates over time if new technologies allow for an increase in the number of distinct materials accepted for recycling or if household recycling behavior improves.

As the number of countries with zero recycling diminishes, then the second of these two factors must be the driving force for global recycling averages to maintain these historical rates of increase into future decades. So, we ask, which of these two factors is most responsible for the historical growth in recycling? The increase in the recycling rate among mature recycling programs can be estimated by taking a closer look at the OECD data. Somewhat arbitrarily, we define a mature recycling program as one that has achieved a 20% recycling rate. This 20% recycling rate was attained very early in countries such as Japan (1990), Switzerland (1990), Austria (1992), Germany (1993), Finland (1994), Sweden (1995), and the United States (1995). A few decades later, they were joined by countries such as France (2010), Czechia (2012), Hungary (2012), Poland (2014), Croatia (2017), Slovakia (2017), and Belarus (2019). Today, very few OECD countries still recycle zero.

Figure 2 illustrates the recycling rate averaged across only mature recycling programs. For each country with a mature program, the data are aligned to the year that it first obtained a 20% recycling rate (labelling the beginning of every country's subsequent years as year one). The figure then follows the maturation of these recycling programs over time as if all such countries achieved the 20% maturation threshold at the same time. These adjusted data suggest the average recycling rate across mature recycling programs increases to about 30% roughly 16 years after first obtaining the 20% recycling rate. But beyond this point, the average recycling rate increases no further. Instead, a 30% ceiling appears to have been sustained for decades after these recycling programs first reached maturity. If mature recycling programs have not increased recycling rates, then the increase in recycling rates observed in Figure 1 may be more attributable to the first factor: new countries initiating recycling programs. If so, then the increasing overall recycling rate observed in Figure 1 is not sustainable. A day will come when all countries take the steps necessary to obtain a 20% recycling rate. On that day, global average recycling rates will tend towards a steady state, perhaps about 30%. Under current technologies and policies, the world may not be on a course towards a future circular economy featuring high recycling rates.

Figure 2: MSW Recycling Percentage for Mature OECD Programs, Plotted Against Number of Years Since Reaching Maturity (For any country, year 1 is the first year after achieving a 20% recycling rate.)



Only a few papers in the economics literature have attempted to explain this apparent stagnancy in average recycling rates across countries with mature recycling programs. Yamamoto and Kinnaman (2022) and Kinnaman and Yamamoto (2023) show that the rapid rise in solid waste incineration in many developed countries may have played a role. Average incineration rates across OECD countries increased from 14% in 1996 to almost 28% in 2018 – the same year that mature recycling programs were apparently hitting their recycling rate ceiling. More than half of solid waste is now incinerated by Denmark, Finland, Japan, Luxembourg, Norway, Sweden, and Switzerland. What are these countries burning? Paper, plastics, and textiles are highly combustible and certainly help incinerators operate efficiently. Incineration is found to replace recycling in Japan (Yamamoto and Kinnaman, 2022), but the evidence is inconclusive for all OECD countries (Kinnaman and Yamamoto, 2023). But maybe incinerating these materials is not a problem. For various reasons, the additional incineration may have benefits similar to those of recycling. First, the incinerator's heat is dispersed into district heating systems and can supply electricity to the grid. Also, any metals making their way into the incinerator are harvested from the ashes (using magnets). Remaining ashes are formed into pavement slabs. Air emissions are treated and monitored.

Weak household participation has traditionally been considered a bottleneck to large and sustainable supplies of recycled materials. Households had little experience in how to separate, clean, and set out recyclable materials in the 1980s, and single stream recycling was thought to be a way to help reduce the costs of these efforts. Phoenix, Arizona, first initiated a single stream program in 1989. Households were given a large bin that could accept a large number of intermingled recyclable materials. The bins were collected by a single truck with one compartment, and the material was taken to a Materials Resource Facility (MRF) to be separated, cleaned, crushed, and baled for sale to a buyer. The idea was quickly adopted in other communities, and by 2014, 80% of communities in the U.S. had single stream recycling. Materials were increasingly exported to other countries, including China and other developing nations (Sigman and Strow, 2024). Would single stream recycling allow already-mature recycling programs to sustain increases in recycling rates?

A problem is that individual efforts by well-intentioned households to increase their own recycling led to "wish-cycling", a phenomenon where households lacking perfect information on recycling guidelines add to the large bin materials that they hope is recyclable but is not.[3] When too many households follow this behavior, then an entire container of materials can be rejected by the company purchasing the material – for reasons of contamination. The material is then diverted to a disposal facility and, thus, does not count towards recycling rates (thus contributing to the apparent 30% ceiling). Indeed, recently, 25% of containers of recycled materials were rejected and sent to landfills. This problem was amplified when China implemented its "National Sword" policy in 2019 that imposed new rules on acceptable levels of contamination – rules that would be costly to meet.

Regardless of whether research achieves a full understanding of how the apparent ceiling relates to recycling rates, the policy community seems to understand it and has taken new steps to increase the recycling rate. In the United States, the EPA in 2020 announced a new recycling goal of 50% by 2030. The Circular Economy Action Plan, passed in 2020 by the European Union, has set a 60% minimum recycling rate for all member countries by 2030. It also plans to double the use of recyclable materials in the economy. In 2022, Japan passed its Act on the Promotion of Resource Circulation for Plastics.

How easily will these economies reach these higher goals after 30 years of stagnating at or around 30%? And, is a 50% recycling rate economically efficient? The remainder of this paper suggests that several sources of heterogeneity within waste and recycling systems might frustrate plans for high recycling rates.

II. Sources of Heterogeneity

After the early partial equilibrium model of recycling in Bohm (1981) and our first-best general equilibrium models of garbage and recycling policy in Fullerton and Kinnaman (1995), virtually all waste disposal models in economics research have continued to assume not only that garbage is a single commodity (*i.e.*, "*G*") but also that that recycling is a single homogeneous

[3] https://www.roadrunnerwm.com/blog/why-cities-are-ending-single-stream-recycling

commodity ("R"). In discussing all of this research since 1995, a major point of this article is to emphasize that recycling is very heterogeneous, in at least five dimensions that matter for any private or social decision about what is best recycled and what is best placed in the landfill.

Clearly, a household or business recycling bin can include plastic PET bottles, HDPE jugs, other kinds of plastic bottles, glass bottles, aluminum cans, other metal cans, paper, cardboard, and sometimes plastic bags, straws, clamshell containers, and even Styrofoam (items indexed $i = 1, \dots, I$). This heterogeneity means that some of these materials can currently be recycled cost-effectively using current technology, and other materials in the list cannot.

A second kind if recycling heterogeneity is across locations (indexed $l = 1, \dots, L$). Large cities have good access to recycled commodity markets, but towns in rural and remote locations can have extremely high costs of recycling because of long distances to a MRF (an abbreviation that applies equivalently to a "materials recovery facility" or a "municipal recycling facility"). Locations across the country also differ by demography or politics, either of which can severely increase the costs of getting people to recycle.

A third kind of heterogeneity is the available recycling method (indexed $m = 1, \dots, M$). Some small or older facilities must rely on labor-intensive technology to collect bins, to sort materials, and to clean, crush, and bale each material. Other larger or newer facilities have capital-intensive technology that allows the truck to dump materials onto a conveyor belt that runs under video cameras trained by machine learning to identify each type of material and to use a puff of air or a robotic arm to move each item to the appropriate bin. These newer methods can achieve much lower cost per additional amount of recycling, but they require a high initial outlay. Because this high fixed cost is only worthwhile for larger facilities, production exhibits economies of scale.

Table 2: National Average Prices per Ton (if cleaned, crushed, and baled)

Commodity	Dollars per Ton September 2020	 September 2021	Ratio 2021/2020
Corrugated cardboard	$60	$171	2.85
Mixed paper	$18	$96	5.33
HDPE	$1,100	$2,169	1.97
PET	$130	$511	3.93
Polypropylene	$105	$663	6.31
Aluminum cans	$915	$1,550	1.69
Steel cans	$78	$250	3.21

Source: SWANA (2021). HDPE is high-density polyethylene. PET is polyethylene terephthalate.

A fourth kind of heterogeneity is the wide variety of recycled commodity prices, not just across these items ($i = 1, \dots, I$), and for each item across locations ($l = 1, \dots, L$), but also across time (indexed $t = 1, \dots, T$). For a simple example, **Table 2** shows changes in prices of some materials just from 2020 to 2021. It shows that the lowest price increase was 69% but that most of these prices tripled from one year to the next. Any such price increase can be followed the next year by a dramatic price crash. A major part of this problem is that any one year's price can be quite low, near zero, so the next percentage increase can be quite large. Not shown in the table is the price of recycled glass, which is so low that it varies above and below zero. All of this price volatility means that MRFs have much difficulty planning operations for future years.

A fifth kind of heterogeneity for any waste item that can be recycled is that it may have very different external damages from recycling (R), or from disposal as garbage (G), or especially from littering or dumping (D). For those three methods of disposal, indexed by $j \in \{R, G, D\}$, damages for any particular waste item will differ in their toxicity, their persistence in the environment, and their threat to wildlife. We will use μ to indicate marginal external damages (MED). These damages can differ not only across those disposal options for each item but also across recycling method, across locations, and over time.

Facing all five sources of heterogeneity, a local government must choose public policy that decides whether any particular item should be collected for recycling or placed in a landfill. After accounting for the sale of recycled materials, suppose the net private marginal cost is c_{Rilmt} per additional recycling of item i in a particular location, method, and time. The decision to recycle it is socially optimal if the sum of private cost plus external cost of recycling is less than from placing it into garbage:

$$c_{Rilmt} + \mu_{Rilmt} < c_{Gilmt} + \mu_{Gilmt}$$

The heterogeneity in each variable – reflected in the fact that each has five subscripts – leads inevitably to some logical conclusions:

1. No single policy is best. While some towns may choose to recycle most items, others will optimally recycle only few items. Some towns will optimally recycle not at all (for example, a small town with inefficient recycling methods, in a remote location with high transportation costs, low access to commodity markets, and low costs of landfill).
2. The town's policy decision whether to collect any particular item in the recycling bin must also vary across locations with different characteristics, as different methods are available for processing recycled materials, and across time with variation in recycled commodity prices.
3. Consumer households will inevitably be confused about what should be recycled and what cannot be recycled. Information is always imperfect. They experience changes in these rules over time, and they talk to friends in other locations that have a different list of what can be recycled. Thus, the MRF cannot reasonably expect households to sort correctly what can or cannot be recycled. This problem significantly raises costs, because of "wish-cycling" mentioned above.

Our review of some economics literature below tries to account for these many ways in which recycling is heterogeneous. We first discuss whether and how these types of heterogeneity affect results from existing analytical models about policy intended to influence disposal choices and thereby to raise social welfare. We also discuss whether and how heterogeneity affects empirical results in existing literature.

III. Heterogeneity Complicates Market Failures and Recycling Policy

For waste collection and disposal services, two kinds of market failures prevent a fully private market from operating efficiently to maximize social welfare. The first is a "public good." Despite significant benefits from provision of such a good or service, all residents get the benefits whether they pay or not, so a private firm cannot make customers pay (and thus will not provide it). An example is a lighthouse, since boats can use the light whether they pay or not; boats can "free ride" and still get the benefits. For waste collection, if a private firm tried to charge a price high enough to cover all costs, too many people could get a similar free ride and dispose of waste for free by litter and dumping. For millennia, cities have recognized the need to pay for waste collection, to avoid unhealthy waste piling up in the streets. Still today, most cities use tax revenue to pay for waste collection, or they require households to pay monthly fees for private waste collection. Only in the past few decades have cities realized that some of this waste disposal can become more sustainable by recycling and that it can be sold to offset some costs.

Second, waste collection can create environmental damages. Private markets work efficiently if the buyer and seller together face all of the costs and benefits. But if they ignore an external cost on somebody else, then the total of all costs can exceed the total benefits – a failure of private markets. For MSW, as just described, most cities provide or require waste collection, so their costs are not exactly private costs, but they are direct costs or on-budget dollar costs. Environmental damages may or may not be part of the city's calculation of total costs and total benefits. They could be called indirect costs, or non-dollar costs. But many of the environmental damages from a city's waste disposal are still external costs on people outside the jurisdiction.

The first market failure, a public good, pertains to all waste disposal everywhere. This paper is about recycling heterogeneity, so we focus here on the indirect or external costs that differ across materials and are critical to any decision about whether a particular material is best put into landfill or recycled. Each waste material can generate external costs if sent to a landfill, a different external cost if sent for recycling, and another external cost if it needs to be replaced by virgin material extraction.

We begin with the direct costs and the indirect/external costs of garbage disposal. Direct costs include the hauler's collection costs, the landfill's tipping fee, and the scarcity rents from limited landfill capacity (which may or may not be covered within the tipping fee). External costs are comprised of noise and litter from collection trucks, transportation emissions, and landfill emissions of leachate and methane. The private market or the city itself might be expected to make efficient decisions to minimize direct costs, but further public policy might be needed to account for indirect or external costs – if the goal is to raise social and economic welfare. Such policies might include a subsidy to recycling, a tax on garbage, or a particular command-and-control regulatory mandate or a cap-and-trade system.

Pigou (1932) points out that the first-best optimal (FBO) allocation of resources can be achieved by a tax on any polluting activity, at a rate equal to the marginal environmental damage (MED). With the heterogeneity described above, this Pigovian tax policy would require a tax on garbage at a rate equal to MED. For perfect efficiency, this rate would have to depend on the item for disposal, its location, and year (since MED can change over time). Thus, using the notation above, the tax is $\tau_{Gilmt} = \mu_{Gilmt}$. But any garbage tax could shift disposal not just toward recycling but toward illegal dumping as well. Perfect efficiency would require a Pigovian tax on every externality.

Fullerton and Kinnaman (1995) use a simple general equilibrium model to solve for FBO policies when households can choose among three methods of disposal: garbage, recycling, and dumping. They start off by ignoring heterogeneity, which keeps the model simple and yet still conveys important intuition, but they note that recycling may have its own MED from collection, litter, noise, and emissions. In that case, it's easy to show that the FBO policy can tax each type of disposal at a rate equal to its MED: $\tau_G = \mu_G$, $\tau_R = \mu_R$ and $\tau_D = \mu_D$. Then the next problem is that dumping cannot easily be taxed at all! It is evasive behavior, hidden from regulators, and not a market transaction with an invoice to help enforce a tax. Assuming logically that damages from dumping are the worst ($\mu_R < \mu_G < \mu_D$), and that τ_D must be zero, then a tax on garbage could induce dumping with higher damages, and thus reduce social welfare.

The more interesting result in their paper is that the exact same FBO allocation of resources can equivalently be attained with no tax on dumping, but only with taxes on market transactions like a commodity purchase with an invoice, or on collection of garbage and recycling by a commercial hauler. The easiest way to describe their solution is shown in the first column of **Table 3**. It is essentially a deposit-refund system (DRS), where the consumer upon purchase pays a "bond" equal to the high MED of dumping the item (μ_D), in case they dump it, but receives back that bond if the item is placed into garbage (minus the garbage external cost) or if the item is placed into the recycling bin (minus the recycling external cost).

Table 3: Considering Heterogeneity within a Deposit-Refund System

	Fullerton and Kinnaman (1995)		
	No Heterogeneity	Item-specific DRS	Full Heterogeneity
Tax on Purchase	μ_D	μ_{Di}	μ_{Dilmt}
Subsidy to Garbage	$\mu_G - \mu_D$	$\mu_{Gi} - \mu_{Di}$	$\mu_{Gilmt} - \mu_{Dilmt}$
Subsidy to Recycling	$\mu_R - \mu_D$	$\mu_{Ri} - \mu_{Di}$	$\mu_{Rilmt} - \mu_{Dilmt}$

While most of their paper focusses on the simple case in the first column, the last section of their paper points out that the same solution could apply item by item (in the second column). The point here, then, is that the same solution could apply with perfect efficiency in the case with full heterogeneity in the last column, but only if each tax and refund is specific to the item, the time of disposal, the location it would be dumped (for the deposit, μ_{Dilmt}), and the method by which it would be recycled (for μ_{Rilmt}). This perfect efficiency is very demanding, for at least two reasons. First, the economy includes many thousands of different items for purchase, so the list of purchase taxes would be long. Second, the location it would be dumped is unknown, and may differ across consumers. These reasons generally drive policymakers toward approximate solutions, putting a whole set of items into a categorical bin for each sales tax rate, and making a guess about the size of the dumping externality. Without precision, an approximate solution could still improve upon a free market that ignores external costs.

Subsequent papers generally also ignore most heterogeneity. Looking at a partial equilibrium model of a particular industry output, Palmer and Walls (1997) consider how to avoid dumping by use of other policies such as a recycled content standard, but they conclude the DRS is superior. Palmer *et al.* (1997) use a similar model to compare the DRS to an advance disposal fee, and a recycling subsidy, and they calculate effects for each policy using 1990 prices and quantities of aluminum, glass, paper, plastic, and steel (finding the DRS is least costly). Walls and Palmer (2001) add consideration of product-specific life-cycle assessment (LCA).

Fullerton and Wu (1998) use a simple general equilibrium model to add consideration of producer choices about both packaging and product design. They find that the FBO outcome can be achieved using any of three policies: (a) Pigovian taxes on each and every type of household disposal, because then consumers would demand packaging reductions and products designed for recycling; (b) a DRS; or (c) an extended producer responsibility (EPR) policy called a "take-back" rule. If firms are required to take back their packaging and the product itself after its useful life, and if those firms have to pay the full social cost of disposal, then they will optimally choose eco-friendly packaging and design for recycling. Also see Eichner and Pethig (2001) and Forlina and Scholz (2020).

These papers often study individual products or materials, but they miss part of the problem by ignoring other heterogeneity and detail – such as recycling costs that depend on technology and transportation costs that depend on location. For one example, EPR policy has indeed been adopted in many European countries and some U.S. states, but the actual implementation of these policies fully realizes that it's far too expensive for each producer to drive around to collect only their own packaging and their own used products from their own customers who may live far apart from each other. Instead, those producers each pay a share of the costs of a single collection agency (called a "producer responsibility organization", PRO) that can collect all companies' packaging and used products from every house on the street.

The PRO takes advantage of substantial economies of scale in collection, and then it charges the various responsible producers according to their share of the weight of the total amount collected. This "collective" EPR is very cost-efficient, and it provides firms with some incentives to reduce the weight of their packaging or product, but it does not provide the incentive of the "individual" EPR (take-back rule) to design products that are more easily recycled. In other words, it is not a targeted policy to encourage recycling *per se*, though it has become a major source of financing for recycling efforts. It is effectively an advance disposal fee. The MRFs no longer have to rely as much on local property or sales taxes, paid by households, because the collective-EPR is like a tax on firms instead.[4]

Other recycling policies have been suggested, of course, and we cannot review all of them in this short article. But the best organizing principle for discussion of recycling policies is first to ask "what is the problem?". The answer helps identify what is the best policy to fix it.

So far, we have discussed external costs from disposal of garbage, from recycling, and from dumping. Another critical externality that relates to recycling policy is from virgin materials extraction. As discussed next, these external costs from mining can be quite large. If virgin materials and recycled materials are good substitutes in production of a new product, then a subsidy to recycling makes the recycled input cheaper and thus could lead to reductions in virgin material extraction, also reducing those high environmental damages. But this logic does not justify a uniform quantity goal or subsidy for all recycling. Because of heterogeneity, it suggests a surgical recycling quantity target or subsidy for the individual recycled materials that can displace virgin materials with the highest external costs of extraction.

For one example, the Eagle Mine in Colorado for a hundred years extracted virgin materials such as gold, silver, and zinc. It was designated a Superfund site in 1986 by the EPA, which cited

[4] Lifset *et al* (2023) discuss ideas for "restoring the incentives for eco-design" within EPR systems by using "eco-modulation" standards that are not incentives for design, but mandates instead.

large amounts of arsenic, cadmium, copper, lead, and zinc in the soil, large fish kills in the Eagle River, and threatened drinking water downstream. Damigos (2006) cites estimates from hedonic models that residences within six miles of the mine experienced reductions in property value ranging around $25,000 (in 1985 USD). In developing countries with less regulation, the external costs can be even larger. Furoida and Susilowati (2021) study Indonesia's mining of minerals in Brown Canyon and associated damages (mining noise, decreased supply and quality of groundwater, a lot of dust, and other air pollution). For a sample of 50 households, they collect a combination of quantitative data and qualitative data from interviews about costs of replacing those resources and of illnesses. We convert their monthly estimates in IDR to annual costs in USD to find that the average family's loss is about $143 USD/year in a community where the average annual income is about $205 USD.

Using many such studies of external damages from virgin materials extraction, LCA models estimate total environmental costs associated with final products sold to consumers. Kinnaman (2014) then uses those LCA studies to summarize both damages per ton of virgin materials extraction and damages per ton of garbage at the end of a product's life. He finds that the MED per ton of garbage is small, about $10-15 per ton, but that the MED of extracting a ton of some virgin materials can be twenty times that amount.

A recycling subsidy is a second-best optimal (SBO) policy. It cannot reduce externalities from garbage as effectively as a FBO tax on garbage, and it cannot reduce mining externalities as effectively as a FBO tax or regulation of emissions from mining. But the garbage externality is small, and policymakers *can* impose a tax or other direct policy to reduce landfill damages. In contrast, external damages from extraction are large, and U.S. policymakers *cannot* impose taxes or other direct policies to control large damages from mining in developing countries. For these reasons, discussion of recycling subsidies might be better justified not by pointing to garbage externalities but instead pointing to the larger external damages from virgin material extraction.

IV. Heterogeneity Makes It Hard to Know if More Recycling Raises Social Welfare

Some materials are easy to recycle and costly for landfill disposal. Other materials are different in several ways. First, some materials are difficult to recycle because of their toxicity, size, or other attributes. Toxic items are not easily handled by households and can be easy to discard. Small items like plastic straws or food wrappers may be considered inconsequential and are therefore too easy to throw into a garbage bin (especially when a recycling bin is not available). Second, some materials have virtually no market price, like glass, so the private benefits associated with recycling are small. Third, locations differ with respect to the market price or access of small rural towns to recycling markets. Fourth, not recycling these materials generates different environmental damages. Fifth, even with recycling, these materials generate different environmental damages. For example, primitive methods used to recycle circuit boards in developing countries threaten human health. And the hot water used by households to clean various materials for recycling can generate external costs. Sixth, behavior and habits differ across households, so only huge marketing efforts have a chance to get them to recycle anything. Because of these diversities, recycling additional types of materials can have high social costs that exceed low benefits. Society would simply not be wise to try to recycle a very large percentage of all items everywhere.

Consider the case of recycling PET bottles into shirt fabrics, a common practice throughout the global economy. It might seem like a successful example of a circular economy supported by recycling. When washed, however, fabrics made of recycled PET plastic often leach micro-plastics into water runoff. One study estimated that 73% of microfiber found in the Arctic Ocean originated from PET plastic.[5] Burying plastic in a landfill can sequester its carbon content and may do more to protect the natural environment than recycling it. But this argument against recycling may pertain only to PET bottles and only using current technologies. Recycling other plastics or PET bottles using future technologies may still raise social welfare.

Because of these multiple heterogeneities, a policy that sets any overall recycling rate target such as 50% is too blunt, obtuse, and perhaps inefficient. Kinnaman *et al*. (2014) consider the private and external costs and benefits associated with recycling, and they find that the optimal aggregate recycling rate is currently less than 30%. Surgical recycling targets might be more effective. If the favored policy approach involves choosing quantity targets rather than prices, then a surgical recycling policy would set different recycling goals for different circumstances, based on all sources of heterogeneity discussed above. A target recycling rate of 100% for automobile batteries can efficiently coexist with a zero target for recycling used plastic peanut butter jars, given the differences in external costs and benefits of each such item.

Instead, if the favored policy approach is to fix prices rather than quantities, then efficient tax rates vary by item, location, and the other sources of heterogeneity. For example, the disposal tax on aluminum cans could be substantially higher than the disposal tax on PET bottles. The SBO disposal tax on PET bottles could even be negative if landfill disposal is socially more desirable than recycling PET plastic into textiles.

Table 4 provides some data that reflect three of the above sources of heterogeneity: across weight of each item, across damage from each item in a landfill, and across damage at different landfill locations. The first source of heterogeneity across items is their weight. Material prices are often quoted in tons, which can be unfamiliar to most readers and do not reflect underlying heterogeneity across items of different weight. Thus, each price in Table 4 has been converted to the price of a single unit such as one aluminum can, one Sunday newspaper, or one of several other individual items. The weight of each of these individual items is also given in the table. Most of these household recyclable items are worth less than ten cents each, a realization first made by Palmer *et al*. (1997). One empty aluminum can, for example, has a market value of 1.5 cents (in states without a DRS). One plastic PET water bottle is worth a fraction of a cent. One uncolored HDPE milk jug is worth a little over a nickel.[6] The most valuable common household item might be a Sunday newspaper, which is worth about 17 cents. But the main point made in examining these prices is just to see how different they are from each other. One aluminum can, for example, has the same market value as ten PET plastic water bottles. It's difficult to imagine a single optimal tax or recycling rate target percentage for these two kinds of containers. Instead, different policies tailored around the sources heterogeneity are not difficult to fathom.

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[5] https://fibershed.org/2022/01/11/what-you-need-to-know-about-microplastics-and-textile/

[6] Other items are not shown in the table. Plastic bottles containing carbonated beverages need to be thicker, so they weigh a little more and are worth a little more. Colored HDPE bottles that hold liquid laundry detergent are worth about one-fifth as much as HDPE milk jugs, because removing the colored dye is an extra expense for the recycler.

Table 4: Market Prices and MED from the Disposal of Common Recyclable Materials

Recyclable Item	Price	External Marginal Damage	
		At \$10/ton	At \$100/ton
One Empty Aluminum Can (14.9 grams)	1.5 cents	0.02 cents	0.16 cents
One Empty Glass Wine Bottle (500 grams)	6.6 cents	0.55 cents	5.5 cents
One Empty 500cc PET Plastic Water Bottle (9 grams)	0.15 cents	0.001 cents	0.01 cents
One Empty 12”×10”×8” Corrugated Box (250 grams)	2.4 cents	0.27 cents	2.7 cents
One Sunday Newspaper (4.2 pounds)	17.22 cents	2.1 cents	21 cents
30 Sheets of Office Paper (4.8 ounces)	3.6 cents	0.15 cents	1.5 cents
One Empty Gallon HDPE Plastic Milk Jug (61 grams)	5.23 cents	0.07 cents	0.7 cents
One average bi-metal “tin” can (2.6 ounces)	1.53 cents	0.08 cents	0.8 cents

Note: Market prices were obtained via quick web searches at the time of this writing. Many prices are found at https://resource-recycling.com/recycling/2023/03/13/scrap-plastic-prices-rise-notably-this-month/. These prices change frequently (as shown in Table 2), so prices in Table 4 are already outdated.

In a second source of heterogeneity, the external marginal damage (MED) across different recyclable materials in a landfill is presented in the second to last column of Table 4. These estimates are based on Kinnaman (2014), who surveyed empirical literature to find that external costs are roughly \$10 per compacted ton of mixed waste. Note that the heterogeneity in the MED as presented in this column is based purely on differences in the weight of each material, which has two key limitations. First, use of a landfill is based on compacted volume of each material, which is only loosely related to uncompacted weight. For example, PET plastic might compact more efficiently than glass. If so, then the MED of glass relative to PET plastic is understated in each column of Table 4. Second, each material presents its own threat to the natural environment when buried in a landfill. Glass and plastic bottles may have low risks to surrounding water tables, but unused household paints and other hazardous materials have high risks.

Unfortunately, this source of heterogeneity is not fully reflected in the MED estimates used within this column of Table 4, although most of these materials in a monitored landfill are likely relatively benign to the natural environment. But the heterogeneity generated simply by the differences in the weight of each material is still observable in this column. For example, the MED of one aluminum can is twenty times that of a plastic water bottle. The MED of one Sunday newspaper is fourteen times that of 30 pages of office paper.

The third source of heterogeneity presented in Table 4 relates to differences in the underlying per-ton estimate of the MED of mixed waste disposal. This MED can vary by location. Imagine a case where one landfill is located above an important aquifer that is a source of fresh drinking water or near an environmentally sensitive area. It might generate a high MED per mixed ton, while another landfill might only generate a low MED per mixed ton. Kinnaman (2014) settles on an overall average estimate of \$10 per ton, based on the known literature, and Palmer *et al.* (1997) use a value of about \$33 per ton. Some studies cited in Kinnaman (2014) estimate the MED at \$100 or more. Thus, the heterogeneity is not only across locations but across estimates for the average location. The final column of Table 4 recalculates the MED of each material based on a location or estimate where the underlying MED is $100 per mixed ton of waste. Unsurprisingly, compared to the previous column, the MED from disposing of each material increases by a factor of ten. The location source of heterogeneity illustrates how optimal recycling policies can differ across two landfills.

One blunt recycling rate target, as recently passed in the US and in the EU, fails to account for these three important sources of heterogeneity, as well as the other sources of heterogeneity described above that are not illustrated in Table 4. These differences could inform optimal recycling policy for each material.

V. Why Have Policymakers Resisted More Surgical Recycling Policies?

Well, to be fair, governments have based recycling policy around heterogeneity in many cases. State governments across the United States vary substantially from each other in terms of recycling laws, resulting in a patchwork of differing approaches. These differences by state are likely related to one or more of the sources of heterogeneity discussed above, such as differences in the private cost of disposal, the value that the local population places on natural environments, and perhaps other underlying natural, social, and political differences. In addition, some materials such as automobile tires and batteries, as well as bulky waste such as appliances and furniture have their own unique recycling laws in many parts of the world.

But the materials listed in Table 4 are traditionally included by MSW in curbside collection, so they are largely grouped together and subjected to identical recycling policies despite their heterogeneities. Why do we rarely see surgical recycling policies among these materials? A different deposit-refund system could be enacted for each. One reason might be the sizeable administrative costs associated with implementing each perfect policy.

The transactions costs associated with enforcing a set of differing recycling laws across these materials may be substantial, relative to the efficiency gains from regulating each product separately. Consider a household going through the process of cleaning and storing a weekly supply of six aluminum cans, two wine bottles, ten water bottles, two corrugated boxes, three newspapers, two empty HDPE jugs, and four tin cans. According to the figures in Table 4, a household facing a tax based on the unique MED of each material would pay a total tax bill of only 8.53 cents per week if the MED is $10 per ton (or 85.3 cents/week if the MED were $100 per ton). These numbers are low. A household with access to free disposal of additional waste can skip this tax – and thus has no incentive to recycle. Kinnaman (2014) estimates that the deadweight loss from failing to impose an efficient tax is a mere 0.3 cents per household per week, or 15.6 cents per household per year (assuming MED of disposal is $10 per ton). This estimate is miniscule. An efficient and surgical recycling policy would eliminate this dead weight loss and thus generate a benefit of 15.6 cents per household per year. Even without going through the effort to estimate the administrative and enforcement costs, it is not difficult to imagine that these costs would eclipse 15.6 cents per household per year. Even one stamp placed on a letter mailed once a year informing or reminding the household of the recycling law would cost more than the expected benefit of 15.6 cents.

But a surgical recycling policy on used automobile batteries, for example, does likely pass the threshold ratio of benefits to administrative cost. A driver with a dead battery can enter an auto parts store and purchase a new one, where bringing the old battery for recycling earns a reduced price on a new battery (the return of a deposit). The cost of bringing the old battery on a trip they are making anyway might be no more than placing that battery into their garbage. Even though the market value of an old auto battery is about $10, a government refund directed only at auto batteries can be more than $10 within a unique DRS for batteries. With these incentives in place, a DRS directed only at auto batteries is nearly costless to administer. For these reasons, Americans recycle over 90% of automobile batteries. In comparison, households generating the mixed basket of various recyclable materials in the previous paragraph save the MRF only 97 cents from their costly recycling efforts (the sum of sales prices for those items in Table 4). Implementing a surgical policy to recycle the optimal amount of each of those materials may not be worthwhile.

But a more surgical recycling policy can always be on the menu of possible future policies when heterogeneities are substantial. Materials such as used flashlight batteries, consumer electronic waste, household hazardous waste, and other selected materials can deserve their own unique DRS or recycling rate target. Relying on a single broad recycling rate goal for these products misses important opportunities to reduce deadweight loss.

VI. Concluding Remarks

The empirical evidence discussed above highlights recycling rates that appear to stagnate. On average, they hover at about 30% across countries with mature recycling programs. These averages fall well short of the 50% to 60% targets set by governments in the U.S. and European Union for municipal recycling. This paper describes multiple challenges that these government targets are likely to face. First, the five types of heterogeneity related to recycling mean that any “one size fit all” recycling policy is too blunt and likely inefficient. Second, because of transaction costs, the meager net benefits associated with recycling any particular item in the MSW waste stream will likely frustrate the achievement of these lofty targets. Perhaps the policy community can take a breather. Kinnaman *et al.* (2014) estimate the optimal aggregate recycling rate is less than 30%, but they also find that the social cost of setting the recycling rate too low or too high is not terribly large. In other words, small mistakes are not costly.

One additional challenge associated with further promotion of recycling is that the material recycled might not just displace landfill disposal but also displace other more beneficial options such as reducing overall consumption, repair of used items, resale, or remanufacturing. A large subsidy to recycle automobiles, for example, may induce households to obtain the subsidy rather than to resell the auto. Yet, further use of the auto may be socially more beneficial than recycling it. The same argument pertains to old clothing and other household materials. An aggressive recycling subsidy may literally remove the shirt off the back of a consumer who might otherwise reuse it or sell it to another consumer. Measurement can be extremely difficult for the amount of reuse, renovation, or re-manufacture. Government policymakers thirsty for hard numbers to justify their policies may choose measurable recycling over unmeasurable activities that might be more efficient.

The various arguments against recycling just articulated are of course not relevant for all materials in all places but only for some items in some places at certain times. Recycling may never get to 100% of all materials everywhere, and such a goal is probably not wise. But recycling 100% of some materials in some places may be efficient, while recycling none of other materials in other places may also be efficient. Adopting a surgical approach, when and where transactions costs are low, may yield better outcomes than adopting a blanket aggregate approach for everything. Old cell phones and old glass jars might best not be lumped together into a single recycling policy.

And, of course, future changes may alter the efficiency calculus for each material and place. First, recycling technology may improve and reduce the cost of recycling certain materials in several places over time. Second, new markets may emerge for some materials not currently recycled, thus making worthwhile the recycling of those materials. Third, new public policies may emerge that better internalize all costs associated with mining and manufacturing, which would raise the cost of producing new virgin materials and thus increase the demand for some recycled materials. Finally, new methods to encourage households to recycle some materials can emerge. If distributing flyers with recycling information continues to offer little impact, then strategies such as "community-based social marketing" (see, *e.g*., McKenzie-Mohr, 2000) can be emplyed, where community organizers contact individual households directly to explain and encourage the right kind of recycling. Any or all of these potential future changes can alter the optimal amount of recycling for affected materials. Although surgical recycling could remain the focus of public policy, the aggregate recycling rate in developed countries might efficiently and effectively increase above 30% as a byproduct.

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