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Decision Support

Recycling common materials: Effectiveness, optimal decisions, and coordination mechanisms

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ABSTRACT

A growing amount of municipal solid waste (MSW) is generated worldwide, and common materials (paper, plastic, metals, and glass), which account for more than half of MSW, exhibit low recovery rates. In this paper, we aim to investigate some key questions about recycling across three dimensions: greenhouse gas emissions, operational costs, and aggregate costs (social costs of emissions plus operational costs.) First, we build supply chain models for cradle-to-grave and cradle-to-cradle supply chains to derive an analytical condition for recycling effectiveness, and use US emissions and cost data to empirically validate that recycling is effective in reducing emissions for all the abovementioned materials. Furthermore, our analysis shows that recycling is effective for all materials, with the exception of glass, with respect to both operational and aggregate costs. Second, we study optimal recycling decisions in terms of collection and yield rates in a socially optimal case, as well as in scenarios in which recycling decisions are made by a local government, a product manufacturer, and an independent recycling firm. Unlike some of existing findings, we show that there are instances in which a product manufacturer or an independent firm might be the best choice for organizing recycling operations. Finally, we discuss and analyze incentives that a social planner should offer to recyclers to bring their efforts closer to the socially optimal choice. We obtain a novel result, which shows that a deposit/refund scheme implemented by a social planner with a refund to local governments might lead to a socially optimal collection rate.

1. Introduction

The management of post-consumer products poses serious challenges, particularly when considering the large amount of municipal solid waste (MSW) generated by human activities. The MSW includes everyday items that we use and then dispose of, such as product packaging, bottles, or newspapers (see more at https://archive.epa.gov/epawaste/nonhaz/municipal/web/html/). On a global scale, the annual MSW generation in 2012 was estimated at 1.3 billion tons (short ton unless otherwise specified; 1 short ton = 2000 pounds ≈ 0.907 metric ton), which is expected to increase to 2.2 billion tons by 2025 (Hoornweg & Bhada-Tata, 2012). The US Environmental Protection Agency (EPA) report on MSW—EPA (2016)—shows that the amount generated by Americans increased from 88.1 million tons in 1960 to 251.8 million tons in 2012, out of which EPA estimates that 132.5 million tons were disposed into landfills. The most represented MSW materials in the US are paper (27%), plastics (12.8%), metals (9.1%), and glass (4.5%) (hereafter "*common materials*")—all highly recyclable materials that account for more than half of MSW. Recycling effectiveness and recycling decisions for these common materials are the focus of our paper.

While most people intuitively agree on the environmental benefits of recycling, there are currently opposing views about overall desirability of recycling. One stream of opinions focuses on the environmental impact and argues that the waste reduction is the ultimate goal, and wants to push recycling rate to 100%, which aligns with zero waste philosophy.[1] The others, however, are more concerned with the economic impact of recycling and propose that more waste should be sent to landfill or incinerators. For instance, although Tierney (2015) opines that recycling is costly (compared to landfill) and ineffectual, he acknowledges the potential benefits of greenhouse gas (GHG) emissions reduction. Nash (2016) reports that a decrease in the cost of primary materials makes secondary materials no longer cost competitive, resulting in 4 billion containers disposed in landfills over two years in California. We consider the entire product life cycle to determine the effective-

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[1] http://zwia.org/standards/zw-definition/, http://www1.nyc.gov/assets/dsny/zerowaste/residents.shtml.

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ness of recycling. Specifically, we study three aspects of recycling: GHG emissions, operational costs, and aggregate costs, which include the social costs (the estimated economic societal damages due to increased GHG emissions) and the operational costs.

We focus on simple consumer products made of common materials (e.g., bottles or cans) prevalent in MSW. Three rates are generally used to measure recycling: *collection rate* (the fraction of products collected for recycling, also called *recycling rate*, see NAPCOR and APR (2015) for PET recycling), *yield rate* (the fraction recycled into secondary materials, see Venditti, 2014 for paper recycling), and *recovery rate* (product of collection rate and yield rate, also called *utilization rate*). We find that such products exhibit collection rates between 30 and 70%, and yield rates between 60 and 100% (see more details presented in Table B7 in the Online Appendix B.) However, these numbers still do not provide a good indication of whether recycling is beneficial, which motivates us to first derive analytical conditions under which recycling is effective with respect to emissions, operational costs, and aggregate costs. We show the importance of the yield rate in determining the effectiveness of recycling (see, e.g., Pauck, Venditti, Pocock, & Andrew, 2014 for paper recycling.) Using EPA estimates for emissions data, the social cost factor of emissions, and collecting and calculating all relevant operational costs, we demonstrate that in our setting, recycling of common materials is both environmentally beneficial and cost effective in the US, with the exception of glass, for which we achieve an environmental benefit but at a loss.[2]

Following the abovementioned analysis, we ask a natural and fundamental question: Who should be in charge of recycling decisions? Our paper considers the collection and yield rates as recycling decisions, with the assumption that the decision maker may outsource actual recycling operations to a third party. Our benchmark case assumes that a *social planner* makes recycling decisions, and is interested in the entire life cycle of products, which includes primary resources extraction, manufacturing, transportation, consumption, landfill and recycling.[3] We then consider three centralized recycling scenarios in which recycling decisions are determined by one entity: a local government (hereafter, *the government*), a product manufacturer (hereafter, *the firm*), and an independent recycling firm (hereafter, *the recycler*.)

1. When recycling decisions are made by the *the government*, its responsibility includes recycling and landfill and the related costs. Ideally, the local government and the social planner should have the same objective, but in reality the former is more concerned with its direct responsibility Walls, Macauley, and Anderson (2003). We discuss this in detail in Section 5.2.
2. When recycling decisions are made by *a firm* (e.g., Stonyfield recycles plastic containers in collaboration with Whole Foods Market, https://www.preserveproducts.com/recycle/programs/gimme-5-program-171), the firm's primary interests include the cost of primary and secondary materials as well as the cost of product manufacturing. We discuss this in detail in Section 5.3.
3. When *a recycler* makes recycling decisions, its primary concern is the price competition between the secondary materials that the recycler generates and the primary materials. Declines in primary materials' costs (e.g., PET) may put the recycler's business in jeopardy Johnson (2016). We discuss this in detail in Section 5.4.

We characterize and compare the optimal recycling decisions under the abovementioned scenarios, in which one entity determines both the collection and yield rates. We then consider decentralized recycling decisions: namely, what if decisions on collection and yield rates are made by two different entities? For such decentralized recycling, we consider cases in which either the social planner or the government determines the collection rate, because the collection of post-consumer products is usually organized at the municipal level, while the recycling entity chooses the yield rate. We derive novel results and provide simple incentive mechanisms that could be offered to recycling entities to bring their efforts closer to socially optimal choices.

The remainder of the paper is organized as follows. In Section 2, we review the relevant literature. In Section 3, we describe our supply chain models and derive analytical conditions for recycling effectiveness. In Section 4, we use US emissions and cost data to investigate the implications for the common materials. In Section 5, we formulate centralized recycling decision problems and compare optimal decisions of different entities. We explore decentralized recycling and incentives to induce socially optimal recycling in Section 6. In Section 7, we present a case study of a proposed legislation on PET recycling, as well as some extensions in Section 8. We conclude our paper with managerial insights in Section 9. In the online supplemental materials, we present proofs in Appendix A and a detailed derivation of emissions and costs in Appendix B.

2. Literature review

Our research is related to six streams of literature: (1) recycling post-consumer products, (2) multi-objective models in sustainability, (3) environmental analysis of recycling, (4) cost analysis of recycling/remanufacturing, (5) social cost of emissions, and (6) waste management. In what follows, we review the relevant literature in each stream, and highlight our contributions.

Many important research questions exist on the topic of *recycling post-consumer products*. For products such as electronics and appliances, the literature on Extended Producer Responsibility generally focuses on take-back legislation (e.g., Massarutto, 2014; Zhou, Zheng, & Huang, 2016), as well as the remanufacturing of end-of-use products—e.g., disposable cameras or photocopiers (Geyer, Van Wassenhove, & Atasu, 2007). Unlike these papers, our study focuses on simple products made of common materials that are prevalent in MSW. For common materials recycling, many studies have examined problems such as dual stream vs. single stream collection and separation of recyclables (Fitzgerald, Krones, & Themelis, 2012), curbside recycling (Aadland & Caplan, 2006), curbside vs. non-curbside recycling (Abbott, Nandeibam, & O'Shea, 2017), collection planning (Teixeira, Antunes, & Sousac, 2004), vehicle routing for collection (De Bruecker, Belien, De Boeck, De Jaeger, & Demeulemeester, 2018), multi-period collection (Elbek & Wohlk, 2016), sorting recyclable materials (Toso & Alem, 2014), flexibility of recycling (Coratoa & Montinari, 2014), and technological, design, marketing innovations aspects of recycling (D'Amato, Mazzanti, Montini, & Nicolli, 2013; Zoboli et al., 2014). In contrast to the abovementioned papers, we focus on two aspects of recycling decisions: collection of post-consumer products and recycling such products into secondary materials.

The first part of our work belongs to a stream of sustainability research that uses *multi-objective models*. Niakan, Baboli, Botta-Genoulaz, Tavakkoli-Moghaddam, and Camapgne (2013) focus on inventory and transportation costs and carbon emissions. Accorsi, Manzini, Pini, and Penazzi (2015) build a mixed-integer linear programming model to determine the optimal geographic location of the network with an application to the furniture industry in Italy. Govindan, Paam, and Abtahi (2016) develop a fuzzy multi-objective

[2] While our empirical results are restricted to the US, our analytical models can be applied to different regions and yield potentially different results, depending on the local emission and cost structures. For more information in EU setting, see, e.g., European Environment Agency 2017 report, EU Circular Economy Action Plan.

[3] The federal government can play this role (if we consider the scope of the US) or the California state government (if we restrict our attention to the State of California).

optimization model with an application to a medical syringe and needle producer in Iran. By comparison, our paper focuses on an environmental and economic analysis (considering both societal and operational costs) of various common materials' recycling. We analyze conditions that lead to a reduction in long-term average emissions and costs, and demonstrate the importance of yield rate. We collect US data for emissions and costs to confirm our abovementioned intuition on the environmental benefits of common materials' recycling and show, through our analysis, that recycling is also cost-effective for all materials except glass.

The research on recycling of common materials has often focused on *environmental impact* (without considering operational cost) of a single recycling cycle. For example, Craighill and Powell (1996) provide a life cycle assessment and economic evaluation of the environmental impact of a few common materials, and compare a single waste disposal cycle with a single recycling cycle. Unlike their paper, our model considers multiple recycling cycles, incorporates operational costs of the underlying systems, and provides conditions under which recycling is preferred cost-wise to not recycling. Several studies have considered the impact of recycling specific materials of interests. Schmidt, Ostermayer, and Bevers (2000) analyze the use of PET and glass bottles for packaging of carbonated soft drinks in Germany and consider the broader environmental impacts, such as water and wood consumption, in addition to emissions. However, their work does not include the underlying costs and realized collection rates. Perugini, Mastellone, and Arena (2005) quantify environmental performances of recycling of plastic containers in Italy and compare them with landfilling, incineration, and feedstock recycling. Their model focuses on a single post-consumer use cycle, does not consider operational costs, and finds that recycling is always environmentally preferable. Kuczenski and Geyer (2011) provide a comprehensive analysis of the resource requirements and environmental impact of PET bottles in California in 2009. They conclude that material recovery makes a small contribution to the environmental impact, as the majority of such impacts come from pre-consumer stages, and potential improvement could come from improved utilization of secondary materials. Although this conclusion is in line with our results, this study focused on a single cycle and did not consider operational costs. Our empirical analysis provides minimum yield rates under which recycling is beneficial as well as estimates of primary materials costs under which recycling ceases to be profitable.

Several papers in operations literature have studied recycling and/or remanufacturing in supply chains and focused on *costs and profits*, but have not considered the environmental impact. Savaskan, Bhattacharya, and Van Wassenhove (2004) analyze the choice of the party responsible for collection in closed-loop supply chains with remanufacturing. They assume a fixed unit remanufacturing cost and define collection cost as a convex function of the collection rate (independent of the scale of operations.) Their paper concludes that the party closest to the customer (i.e., the retailer) is the best choice for collection from the manufacturer's perspective. Atasu, Toktay, and Van Wassenhove (2013) expand on the model from Savaskan et al. (2004) by adding to the collection cost a component that captures economies/diseconomies of scale, hence, the collection cost could be convex or concave. They conclude that a manufacturer's collection is preferable with diseconomies of scale. Both papers assume a price-setting retailer facing a price-sensitive demand. We consider recycling (instead of remanufacturing) of common materials with a price-independent demand and allow for different parties to be responsible for selecting collection and yield rates; in addition, we allow the unit recycling costs depend on yield rate (i.e., are not constant) and show that the recycler might be the best choice for the recycling entity under some settings. We believe that our assumption of price-independent demand is reasonable in a setting which focuses on post-consumer products, such as plastic and glass bottles, or aluminum and steel cans, that are found in MSW. Most of these products are packaging materials/containers for beverages (e.g., soda, water, wine) and canned foods (e.g., processed fruits, vegetables, fish), thus it is reasonable to assume that a manufacturer (e.g., Nestlé, Pepsi) will determine the quantity they need based on demand for their final product (drink or food), and not on the price of the container. We further validated this assumption with the management of one of the largest PET recycling firms in the US, who confirmed that their customers (mainly manufacturers of consumer beverages) are not sensitive to the price of recycled plastic resins. We want to note that our manuscript is different from the studies that analyze durable goods (e.g., electronics, cars), which are more likely to be price-dependent (for a more detailed discussion and the scope of common materials, see EPA background, 2015).

GHG emissions have an impact on agricultural productivity, human health, property damages from increased flood risks, and the ecosystem due to climate change. Knowlton, Rotkin-Ellman, Geballe, Max, and Solomon (2011) estimate that climate change-related events have caused health costs in excess of USD $14 billion. Despite such high *social costs of emissions*, there is limited work related to its impact on operational decisions. Aflaki and Netessine (2017) consider both system and emissions costs in renewable energy and show that charging a higher price for emissions could have a negative impact by discouraging investment. In our setting, an increase in the social cost of emissions could cause an increase or decrease in recycling efforts depending on the relationship between emissions and cost structures.

The second part of our paper is related to a stream of literature that analyzes *waste management* and its service providers. Our paper considers scenarios in which recycling decisions are made by a local government, a firm, or a recycler, and analyzes the optimal collection and yield rates for each case. Walls (2003) provide an empirical look at waste management and recycling markets for 1000 US communities. Their data shows that government provisions dominate in central cities of metropolitan areas (e.g., in 70% of central city communities, government employees handle waste and recyclable collection). This supports our use of a government recycling model in addition to firm's and recycler's model. This paper concludes that political influence and regulations have little impact on a government's choice of service provider, and that costs of providing service and transaction costs play a significant role in deciding whether to choose a public or private option. This is in line with our use of cost minimization as the government's objective. Walls (2003) studies contracts between governments and waste management service providers and explores incentives for improving waste diversion. The paper discusses contracts for seven US communities that mainly use private contractors and achieve high-waste diversion rates, finding that very diverse options can be used (i.e., incentives for achieving a desired collection rate, ownership of revenues from sale of materials, and so forth.) In most cases, no direct incentives were used for achieving a desired collection rate. Our model suggests that such an incentive would be desirable, which could be indirectly implemented through a deposit/refund scheme. Palmer, Sigman, and Walls (1996) develop a model of waste disposal and calculate the waste reduction in response to deposit/refunds, advance disposal fees, and recycling subsidies. They apply their model to common materials to evaluate the intervention level required to reduce waste by 10%, and conclude that a deposit/refund policy dominates over the other options. Their model assumes a price-dependent demand, and as such, a deposit that acts as a tax, which increases the price and impacts the demand. We assume that demand is constant and not influenced by a deposit and show that under this assumption, the

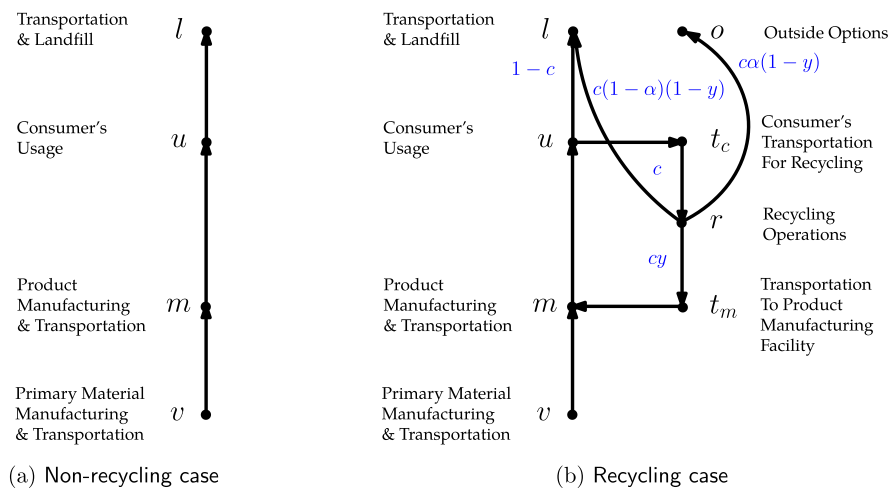


Fig. 1. A simplified product life cycle.

government might reduce the collection rate if it implements a deposit/refund fee with a firm or recycler, but that a social planner can implement a deposit/refund scheme to increase the collection rate chosen by the government.

3. The model

Suppose that a simple product made of a single material is either recycled into the same product (e.g., bottle) or diverted to a different product (e.g., non-bottle.) The life-cycle of a product starts with virgin (primary) input; by recycling a post-consumer product instead of sending it to a landfill, some additional quantity of secondary material is generated. We now introduce some notations.

Definition 1. Let $n \in IN_0$ be the number of recycling cycles (non-negative integer). We will use $\varphi(n)$ to denote a general impact function of the number of recycling cycles. Depending on the context, it can denote the amount of GHG emissions ($\varphi(\cdot) = e(\cdot)$), the operational cost ($\varphi(\cdot) = s(\cdot)$), or the aggregate cost—social cost of emissions plus operational cost ($\varphi(\cdot) = \sigma(\cdot)$).

- $u(n)$ denotes the *total quantity* of the product obtained by sending one product made from primary materials through n recycling cycles.
- $\varphi(n)$ denotes the *total value of* φ resulting from sending one product made from primary materials through n recycling cycles.
- $\varphi_{eff}(n)$ denotes the *effective value of* φ from making one product, that combines manufacturing from primary and secondary materials. Formally, $\varphi_{eff}(n) := \frac{\varphi(n)}{u(n)}$.

Fig. 1a depicts a cradle-to-grave life cycle in which a product is not recycled and goes directly to a landfill. We analyze the emissions and costs that occur through the life cycle of a product manufactured from primary materials and the notations for φ from each process for a single product unit are defined in Table 1. Subscripts denote different processes in the life cycle: v denotes processes related to manipulation of primary materials, m denotes processes related to manufacturing and transportation of the product, u denotes processes related to the end consumer usage, and l denotes processes related to the landfill. We note here that for common materials under consideration we have $\varphi_u = 0$ (no cost or emissions.)

As $n = 0$ without recycling, we have $u(0) = 1$, $\varphi(0) = \varphi_v + \varphi_m + \varphi_u + \varphi_l$, $\varphi_{eff}(0) = \frac{\varphi(0)}{u(0)} = \varphi(0)$.

Table 1
Notation for φ (per unit of product).

Supply chain member	Node i	φ at node i
Primary materials manufacturing and transportation	v	φ_v
Product manufacturing and transportation	m	φ_m
Consumer's usage	u	φ_u
Transportation to landfill and landfill	l	φ_l
Consumer's transportation for recycling	t_c	φ_{t_c}
Recycling operations	r	φ_r
Transportation to product manufacturing facility	t_m	φ_{t_m}
Outside option	o	φ_o

Product recycling ($n > 0$)

Suppose that a product can be recycled, as depicted in Fig. 1b. We use $c \in [0, 1]$ to denote the collection rate; thus, $1 - c$ portion is going to the landfill, corresponding to path $u \to l$. When $c = 0$, this case collapses to the cradle-to-grave model shown in Fig. 1a. As an example, NAPCOR and APR (2015) shows a 31.2% collection rate for PET recycling in the US in 2013.

The collected fraction, c, for recycling has three possible recycling outcomes: closed-loop recycling back to the same product (branch $r \to t_m \to m$) in the amount of cy, where $y \in [0, 1]$ captures the yield rate; open-loop recycling into outside options denoted by $c\alpha(1 - y)$ for some $\alpha \in [0, 1]$ (branch $r \to o$, see Section 8.1 for a PET example in California); and disposal of the remaining portion $c(1 - \alpha)(1 - y)$ (branch $r \to l$). Emissions and costs of the three abovementioned processes are captured in φ_r, which may depend on collection rate c, yield rate y, and the outside option α; therefore, we assume $\varphi_r = \varphi_{r,\alpha}(c, y)$.[4] The recycling decisions on collection and yield rates have implications on the costs (and emissions): for example, if a recycling firm provides more recycling stations or advertises to increase its collection rate, its unit operational cost may go up. Similarly, if a firm wants to increase its yield rate, it may need to invest in new technology for recycling, which may increase its unit cost. Specifically, we make the following assumption:

[4] The reason we use α as a subscript rather than a decision variable is that we assume that the outside option is exogenous to the recycling decision maker. For example, Freytas-Tamura (2018) reports China announced that it would no longer accept plastics and paper products for recycling, which is an outside option to the US and is exogenous to the US recycling decisions.

Table 2
Minimum yield rates with respect to emissions.

Product	Material	e_v	e_r	e_{t_c}	e_{t_m}	e_l	x_e (%)	Actual yield rate, y (%)
PET container	PET resin	2.21	0.62	0.018	0.085	0.04	28.29	69.68
HDPE container	HDPE resin	1.54	0.38	0.018	0.075	0.04	24.44	81.80
Office paper	Paper pulp	0.53	0.53	0.000	0.000	1.75	≥ 0	60.29
Aluminum cans	Aluminum ingot	7.46	0.27	0.018	0.008	0.04	3.35	100.00
Steel cans	Tinplate	2.74	0.76	0.018	0.148	0.04	28.48	98.00
Glass container	Glass container	0.46	0.14	0.018	0.010	0.04	25.12	85.00

Assumption 1. We can decompose $\varphi_r = \varphi_{r,\alpha}(c, y)$, the impact function of converting 1-unit of collected post-consumer material into secondary materials, into three processes as follows:

$\varphi_c(c, y) = \varphi_c(c)$: the impact function of *collecting* c-unit out of every 1-unit of post-consumer material available. We assume this function is only dependent on c, increasing and strictly convex on [0, 1].

$\varphi_p(c, y) = \varphi_p(y)$: the impact function of *processing* 1-unit of collected post-consumer material into y-unit of secondary materials. We assume this function is only dependent on y, increasing and strictly convex on [0, 1].

$\varphi_{d,\alpha}(c, y) = \varphi_{d,\alpha}(y)$: the *disposal cost/diversion benefit* of the remaining $(1-y)$-unit of material (out of 1-unit of collected material). For some $\alpha \in [0, 1]$, we assume that a fraction $(1-\alpha)$ goes to the landfill, while fraction α goes to open-loop recycling: $\varphi_{d,\alpha}(y) = (\varphi_{l'}(1-\alpha) - \varphi_o\alpha)(1-y)$, where $\varphi_{l'} \geq \varphi_l > 0$ is disposal cost, and $\varphi_o \geq 0$ is the benefit generated from selling the post-consumer product as input for outside products.

Based on the above assumption on $\varphi_{r,\,\alpha}(c,\ y)$, it costs $\frac{1}{c}\varphi_c(c)$ to collect $\frac{1}{c} \cdot c = 1$ unit of post-consumer material for the downstream manufacturing process with recycled input. As a result, we have

$$\begin{aligned}\varphi_r &= \varphi_{r,\alpha}(c, y) = \frac{1}{c}\varphi_c(c) + \varphi_p(y) + \varphi_{d,\alpha}(y) \\ &= \frac{1}{c}\varphi_c(c) + \varphi_p(y) + (\varphi_{l'}(1-\alpha) - \varphi_o\alpha)(1-y).\end{aligned}$$

To simplify the exposition, we use φ_r in place of $\varphi_{r,\,\alpha}(c,\ y)$ and $\varphi_d(y)$ in place of $\varphi_{d,\,\alpha}(y)$ when the meaning is clear from the context. We now discuss other members of the supply chain from Table 1. We use subscript o to denote the outside option and t to denote the transportation processes. We assume that $\varphi_v, \varphi_m, \varphi_u$ and φ_l are independent of c, y (in models with and without recycling).[5] Consumer transportation cost for recycling, φ_{t_c}, in general depends on the collection rate, c: that is, when more places accept post-consumer products for recycling, the consumer's transportation distance decreases. However, as our data indicates that both s_{t_c} and e_{t_c} represent a very small fraction of total operational costs and emissions (see Tables 2 and 3), these changes have little impact and are not considered here.

We next describe how we derive the long-term average emissions or costs. Starting with a product made from primary materials, after n recycling cycles, the total quantity of the same product generated through this process is given by $u(n) = 1 + cy + \cdots + c^n y^n = \sum_{i=0}^{n} c^i y^i$. The total impact function value (emissions/costs) generated during this process can be broken down as follows:

- From the cradle-to-gate cycle and from consumer usage (nodes v, m, u): $\varphi_v + \varphi_m + \varphi_u$.
- From product disposal (node l): $\varphi_l[(1-c) + \cdots + (1-c)c^n y^n] = (1-c)\varphi_l \cdot \sum_{i=0}^{n} c^i y^i$.
- From recycling (nodes t_c, r): $(\varphi_{t_c} + \varphi_r)(c + \cdots + c \cdot c^{n-1}y^{n-1}) = c(\varphi_{t_c} + \varphi_r) \cdot \sum_{i=0}^{n-1} c^i y^i$.
- From material process in the product manufacturing and from consumer usage (nodes t_m, m, u): $(\varphi_{t_m} + \varphi_m + \varphi_u)(cy + \cdots + c^n y^n) = cy(\varphi_{t_m} + \varphi_m + \varphi_u) \cdot \sum_{i=0}^{n-1} c^i y^i$.

Assuming that we can recycle products infinitely,[6] when $cy \neq 1$, the total quantity of product, the total value of φ, and the effective unit value of φ are:

- $u(\infty) = \lim_{n\to\infty} u(n) = \frac{1}{1-cy}$,
- $\varphi(\infty) = \lim_{n\to\infty} \varphi(n) = \varphi_v + \varphi_m + \varphi_u + (1-c)\varphi_l + [(1-c)cy\varphi_l + c(\varphi_{t_c} + \varphi_r) + cy(\varphi_{t_m} + \varphi_m + \varphi_u)]\frac{1}{1-cy}$,
- $\varphi_{eff}(\infty) = \frac{\varphi(\infty)}{u(\infty)} = (1-cy)\varphi_v + \varphi_m + \varphi_u + (1-c)\varphi_l + c(\varphi_{t_c} + \varphi_r) + cy\varphi_{t_m}$.

One can verify $\varphi_{eff}(\infty)$ also holds for $cy = 1$. We now introduce our first result; the proof is straightforward.

Proposition 1 (Minimum yield rate for recycling effectiveness). *When $\varphi_{v_{net}} := \varphi_v - \varphi_{t_m} > 0$[7], recycling improves the value of φ (i.e., $\varphi_{eff}(\infty) < \varphi(0)$) if and only if*

$$y > x_\varphi := \frac{\varphi_r + \varphi_{t_c} - \varphi_l}{\varphi_{v_{net}}}. \tag{1}$$

For simplicity, for our numerical analysis in Section 4 we assume closed-loop recycling, that is, $\alpha = 0$. As the use of outside options in general reduces costs (i.e., yields a lower value of φ_r), values of x_φ obtained when $\alpha = 0$ represent the maximum lower bound for which recycling is effective. In our analytical results (starting from Section 5), we assume a general open-loop framework and allow α to vary between 0 and 1.

4. Recycling effectiveness

In Sections 4.1–4.3, we discuss some practical implications of Proposition 1 from three perspectives—emissions, operational costs, and aggregate costs, respectively, for common materials shown in Table 2. To utilize the EPA emissions data EPA (2015a, 2015b, 2015c, 2015d), we follow the same definition of materials used by the EPA. Because glass material does not have a stable intermediate state before becoming manufactured into a product DOE (2002), we use "glass container" to represent both glass material and glass product.

[5] One may argue that the firm may invest in design to make products easier to recycle, which could increase the yield, therefore φ_m may depend on y. Although this may be true for complex products manufactured from multiple materials, for single-material products considered here (e.g., bottles and cans) we assume that manufacturing does not impact the yield.

[6] Technically, paper may not be recycled indefinitely, because in every recycling cycle the fibers become shorter and are mixed with new fibers during manufacturing. In the case of office paper, at the current yield rate of 60.29% (see Table 2), a marginal 2.9% of recycled fibers are retained after seven recycling cycles (https://archive.epa.gov/wastes/conserve/materials/paper/web/html/faqs.html).

[7] We verify that this condition holds for all of the common materials in our consideration in Tables 2 and 3.

Table 3
Minimum yield rates with respect to operational costs.

Product	Material	s_v	s_r	s_{t_c}	s_{t_m}	s_l	x_s (%)	Actual yield rate, y (%)
PET container	PET resin	\$1733.49	\$794.32	\$5.43	\$154.19	\$90	44.94	69.68
HDPE container	HDPE resin	\$1421.05	\$818.00	\$5.43	\$136.05	\$90	57.08	81.80
Office paper	Paper pulp	\$784.92	\$506.47	\$0.00	\$0.00	\$90	53.06	60.29
Aluminum cans	Aluminum ingot	\$1519.51	\$655.16	\$5.43	\$16.11	\$90	37.95	100.00
Steel cans	Tinplate	\$1098.28	\$641.90	\$5.43	\$298.01	\$90	69.64	98.00
Glass container	Glass container	\$1140.67	\$1148.04	\$5.43	\$17.33	\$90	90.67	85.00

4.1. Emissions

In this subsection, we assume $\varphi = e$. We illustrate Proposition 1 by using EPA emissions estimates to derive the minimum yield rates that assure recycling reduces emissions for a few common materials. This is presented in Table 2, in which the emissions unit is MTCO_2E (Metric Ton CO_2 Equivalent) per ton. Observe that, because paper becomes degraded in landfills by anaerobic bacteria and produces CH_4, office paper produces much higher GHG emissions than other materials during landfill operations, as compared to other materials. As a result, office paper recycling at any yield rate is environmentally beneficial.

We note that recycling yields additional benefits not considered in our analysis. For instance, Kuczenski and Geyer (2011) and Craighill and Powell (1996) evaluate the impact of recycling on acidification and eutrophication potential. This implies that recycling is, in fact, environmentally beneficial even at lower yield rates than those presented in Table 2.

4.2. Operational costs

We now consider the case $\varphi = s$ and analyze the operational cost of recycling, $s_r(c, y)$. The first component, *collection cost* $s_c(c)$, can include the recycling entity's cost of providing recycling stations, transporting materials to a recycling facility, advertising, enforcement, or a monitoring cost. Following some examples from literature (Atasu et al., 2013; Geyer et al., 2007; Savaskan et al., 2004), we assume that the cost can be represented by an increasing, strictly convex function.[8] This seems reasonable, as increasing the collection rate after reaching a certain threshold (say, from 70 to 80%) might require a significant cost increase, and it might be practically impossible to achieve a 100% collection rate. Similar logic applies to the second component, *processing cost* $s_p(y)$: in this case, achieving a yield rate close to 100% might be very costly or even impossible. The assumption for the last component, *disposal cost/diversion benefit* $s_d(y)$, implies that the unit landfill cost does not exceed the cost of handling the non-recycled portion of material, and that material not used in closed-loop recycling can be used in the manufacturing of outside products. Note that it might be desirable for the recycling entity to use a lower value of y (or even to set $y = 0$) when the outside option is profitable.

In Table 3, we present the values of lower bounds for some common materials, in which unit is USD per short ton (see Appendix C for details). While emission data were mostly obtained from EPA reports, no such aggregate source exists for costs, thus our numbers come from a variety of sources. For all common materials except glass, the actual yield rate exceeds the minimum yield rates, therefore, recycling is effective in reducing the operational cost. With glass, the unit operational cost of the primary material is very close to the corresponding cost of the recycled material, and this increases the lower bound. This phenomenon is confirmed in practice, in which some municipalities opt out from recycling glass (e.g., Spartanburg, SC; Brigham, UT; Gwinnett, GA[9]; and so forth.), while others collect glass, to prevent it from ending up as litter, and then send it to landfill (e.g., Denver, CO; Chattanooga, TN; Atlanta, GA[10]; and so forth).

Note that the abovementioned conclusion may change if the cost of primary materials decreases. The price of PET and HDPE fluctuates with oil prices, and an additional decrease in the primary materials cost can reverse the relationship, that is, recycling may cease to be cost-effective. It is easy to verify that when the cost of the primary PET material drops to \$1172.82 (a decrease of 32%), to \$1032.46 (a decrease of 27%) for HDPE, and to \$690.73 (a drop of 12%) for office paper, recycling of respective materials ceases to be profitable with the current yield rate. This is in line with several media reports (Daniels, 2016; Nash, 2016).

4.3. Aggregate costs—operational and emissions costs

We use ζ to denote the unit social cost caused by emissions (i.e., *social cost factor*), and use an estimate $\zeta = \$109$ per MTCO_2E U.S. Government (2013) to account for societal damages due to increased emissions. We denote the *aggregate cost* by $\varphi_i = \sigma_i := s_i + \zeta e_i$, where s_i denotes the operational cost at process i, and ζe_i denotes the emission-induced social cost from the same operation. Table 4 presents the minimum yield rates for this model.[11]

When $\zeta > 0$, we can show $\min\{x_e(c, y), x_s(c, y)\} < x_\sigma(c, y) < \max\{x_e(c, y), x_s(c, y)\}$ by Lemma A1 (see the online Appendix A) Note $x_s(c, y) = x_\sigma(c, y)$ when $\zeta = 0$, showing that the minimum yield rate for the aggregate cost model increases in ζ when $x_e(c, y) > x_s(c, y)$ and decreases when $x_e(c, y) < x_s(c, y)$. In other words, when $x_e(c, y)$ is low, the emission effect dominates the cost effect (i.e., recycling is beneficial for the environment even at a low yield), and a higher ζ reduces the minimum yield rate for the aggregate cost model—which holds for all considered materials. If the cost effect dominates the emission effect, then an increase in ζ makes recycling desirable at a higher yield rate.

8 Affine functions may also be reasonable, but in our setting lead to less interesting and less realistic corner solutions for optimal collection decisions. We also explored a family of more general functions $f(x)$, which achieves the minimum value of first derivative at some point, $x \in [0, 1]$ ($x = 0, 1$ each corresponds to convex, concave functions respectively); however, the main results and conclusions of our paper do not change.

9 http://americanrecycler.com/8568759/index.php/news/glass/1572-market-for-recycled-glass-remains-strong-despite-challenges, http://brighamcity.utah.gov/curbside-recycling.htm, http://www.gwinnettcb.org/glass-recycling-facts/.

10 http://www.westword.com/news/most-colorado-glass-doesnt-get-recycled-but-thats-starting-to-change-6833033, http://www.timesfreepress.com/news/local/story/2015/aug/14/glass-not-getting-recycled-chattanoogas-new-s/319805/, http://www.myajc.com/news/local-govt--politics/metro-atlanta-recyclers-reject-glass-ship-landfills/Nd82esxPLUTvCb6963WyWJ/.

11 Our model can be modified to incorporate the case in which the social cost of emissions increases by a same factor, say $\delta > 1$, every recycling cycle, as long as $\delta cy < 1$: as the cost of, say, emissions from landfill in the first round is $\zeta \cdot (1-c)e_l$, in the second round $\delta\zeta \cdot (1-c)e_l \cdot cy$, and so on, the total social cost of emissions from landfill can be written as $\zeta(1-c)e_l \sum_{i=0}^{\infty} \delta^i c^i y^i$. Consequently, the total effective aggregate cost from landfill can be written as $(1-c)\left(s_l + \zeta \frac{1-cy}{1-\delta cy} e_l\right)$—the only modification required is to replace ζ by $\zeta \frac{1-cy}{1-\delta cy}$.

Table 4
Minimum yield rates with respect to aggregate costs.

Product	Material	σ_v	σ_r	σ_{t_c}	σ_{t_m}	σ_l	x_σ (%)	Actual yield rate, y (%)
PET container	PET resin	\$1974.38	\$862.30	\$7.35	\$163.46	\$94.36	42.81	69.68
HDPE container	HDPE resin	\$1588.91	\$859.46	\$7.35	\$144.23	\$94.36	53.47	81.80
Office paper	Paper pulp	\$842.69	\$564.30	\$0.00	\$0.00	\$280.75	33.65	60.29
Aluminum cans	Aluminum ingot	\$2332.22	\$684.81	\$7.35	\$17.20	\$94.36	25.82	100.00
Steel cans	Tinplate	\$1396.94	\$724.79	\$7.35	\$314.14	\$94.36	58.90	98.00
Glass container	Glass container	\$1190.70	\$1165.37	\$7.35	\$18.42	\$94.36	88.15	85.00

Table 5
Emissions, operational cost, and aggregate cost (per unit of product).

General function	Emissions	Operational cost	Aggregate cost	Supply chain member
φ_v	e_v	s_v	σ_v	Primary material manufacturing and transportation
φ_m	e_m	s_m	σ_m	Product manufacturing and transportation
φ_u	e_u	s_u	σ_u	Consumer's usage
φ_l	e_l	s_l	σ_l	Transportation to landfill and landfill
φ_{t_c}	e_{t_c}	s_{t_c}	σ_{t_c}	Consumer's transportation for recycling
φ_{t_m}	e_{t_m}	s_{t_m}	σ_{t_m}	Transportation to product manufacturing facility
$\varphi_{v_{net}} = \varphi_v - \varphi_{t_m}$	$e_{v_{net}} = e_v - e_{t_m}$	$s_{v_{net}} = s_v - s_{t_m}$	$\sigma_{v_{net}} = \sigma_v - \sigma_{t_m}$	See definitions of φ_v and φ_{t_m} above
$\varphi_{r,\alpha}(c, y)$	$e_{r,\alpha}(c, y)$	$s_{r,\alpha}(c, y)$	$\sigma_{r,\alpha}(c, y)$	Recycling operations
$\varphi_c(c)$	$e_c(c)$	$s_c(c)$	$\sigma_c(c)$	Collecting c-unit out of every 1-unit of post-consumer material
$\varphi_p(y)$	$e_p(y)$	$s_p(y)$	$\sigma_p(y)$	Processing each unit of collected material into y-unit of secondary materials
$\varphi_{d,\alpha}(y)$	$e_{d,\alpha}(y)$	$s_{d,\alpha}(y)$	$\sigma_{d,\alpha}(y)$	Disposal/diverting of remaining $(1-y)$-unit
$\varphi_{l'}$	$e_{l'}$	$s_{l'}$	$\sigma_{l'}$	Disposal of 1-unit of material
φ_o	e_o	s_o	σ_o	Diversion of 1-unit of material to outside option

Table 6
A summary of definitions for centralized recycling.

Notation	Definition
$e_{eff}(\infty)$	Long-run effective unit GHG emissions to society with recycling
$s_{eff}(\infty)$	Long-run effective unit operational cost to society with recycling
$\sigma(c, y)$	Long-run effective unit aggregate cost to society with recycling
$\sigma_{G/F/R}(c, y)$	Long-run effective unit aggregate cost when recycling is done by government/firm/recycler
$x_{e\|s\|\sigma}(c, y)$	Minimum yield rate for reduction of emissions/operational cost/aggregate cost

Optimal collection rate	Optimal yield rate	Recycling by	Optimum criteria
c^*	y^*	social planner (first-best)	Minimizing aggregate cost
$c^*_{G/F/R}$	$y^*_{G/F/R}$	government/firm/recycler	Minimizing aggregate cost

5. Centralized recycling decisions

In this section, we study optimal recycling decisions made by one entity: the social planner in Section 5.1, the government in Section 5.2, the firm in Section 5.3, and the recycler problem in Section 5.4. We then compare their optimal choices in Section 5.5.

We provide a summary of notation used to derive the effective cost to the society and carried over from previous sections in Table 5, where c and y denote collection rate and yield rate, respectively, and a summary of definitions on the long-run effective emissions and costs for different entities in Table 6.

5.1. Social planner's problem

We assume that the social planner is concerned with overall societal costs, which are incurred at every stage. Effective cost to the society can be calculated as

$$
\begin{aligned}
\sigma(c,y) &:= s_{eff}(\infty) + \zeta \cdot e_{eff}(\infty) \\
&= (1-cy, 1, 1, 1-c, c, c, cy) \\
&\quad \cdot \Big[(s_v, s_m, s_u, s_l, s_{t_c}, s_{r,\alpha}(c,y), s_{t_m})^{\mathsf{T}} \\
&\quad + \zeta \cdot (e_v, e_m, e_u, e_l, e_{t_c}, e_{r,\alpha}(c,y), e_{t_m})^{\mathsf{T}} \Big] \\
&= (1-cy, 1, 1, 1-c, c, c, cy) \\
&\quad \cdot (\sigma_v, \sigma_m, \sigma_u, \sigma_l, \sigma_{t_c}, \sigma_{r,\alpha}(c,y), \sigma_{t_m})^{\mathsf{T}} \\
&= \sigma_1(c) + c \cdot \sigma_2(y) + \sigma_3
\end{aligned}
\tag{2}
$$

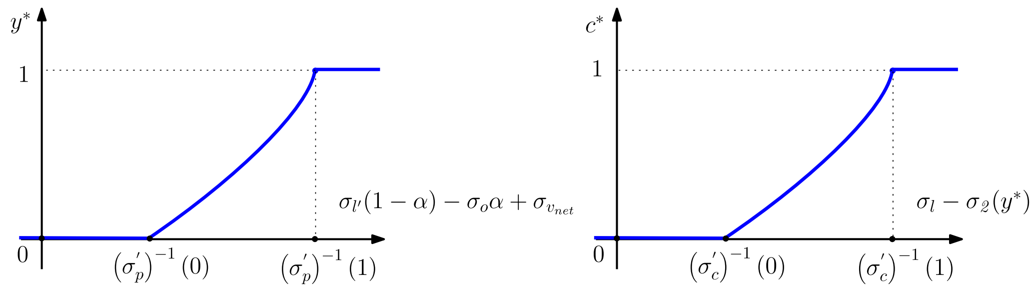
where $\sigma_1(c) := \sigma_c(c) + c[\sigma_{t_c} - \sigma_l]$, $\sigma_2(y) := \sigma_p(y) + \sigma_{d,\alpha}(y) - y\sigma_{v_{net}} = \sigma_p(y) + (\sigma_{l'}(1-\alpha) - \sigma_o\alpha + \sigma_{v_{net}})(1-y) - \sigma_{v_{net}}$, and $\sigma_3 := \sigma_v + \sigma_m + \sigma_u + \sigma_l$, which is independent of c and y. Based on (2), the first-best decision for the social planner's problem is obtained by solving

$$
\begin{aligned}
\min \quad & \sigma(c,y) = \sigma_1(c) + c \cdot \sigma_2(y) + \sigma_3 \\
\text{subject to} \quad & 0 \le c \le 1, \quad 0 \le y \le 1.
\end{aligned}
\tag{3}
$$

Under a simplifying assumption on the minimum cost of consumer's transportation for recycling, $\sigma_{t_c} := \frac{\eta}{c}$ for some $\eta > 0$, we have the following result.

Proposition 2 (Optimal collection and yield rates in the social planner's problem)**.** *There exist optimal collection and yield rates, c^* and y^*, that minimize the social planner's cost in problem (3), and each solution is unique. The optimal yield rate is non-decreasing with respect to $\sigma_{l'}(1-\alpha) - \sigma_o\alpha + \sigma_{v_{net}}$, and the optimal collection rate is non-decreasing with respect to $\sigma_l - \sigma_2(y^*)$.*

The details on how to derive the optimal collection rate, c^*, and yield rate, y^*, can be found in the proof of Proposition 2, given in Appendix A. In Fig. 2 below, we provide an illustration of the above results. Implications of Proposition 2 are rather intuitive. If



(a) Optimal yield rate, y^* (b) Optimal collection rate, c^*

Fig. 2. An illustration of optimal collection and yield rates in the social planner's problem.

the benefit from recycling the original product into outside products increases (larger σ_o), the social planner favors open-loop recycling; on the other hand, if the cost of primary materials increases, the social planner wants to increase the yield in the closed-loop recycling. Similarly, if disposal to landfill becomes less costly, or if the cost benefit of recycled materials decreases, the social planner has less incentive to collect material for recycling.

We first analyze costs in two local optimization models—the government's problem and the firm's problem. In both cases, although we assume that the entity responsible for recycling determines the collection and the yield rate, it can outsource the actual process to a third party. We then consider a scenario in which a recycler is in charge of recycling. As we assume that the recycling entities can outsource the actual process to a third party, we use the same costs in all cases and compare the recycling efforts in all scenarios.

Our models in this section are consistent with Savaskan et al. (2004) and Atasu et al. (2013). Both papers compare three models, with the manufacturer, the retailer, or the recycler responsible for recycling, respectively (they do not consider the government). Unlike our research, both of these papers assume price-dependent demand and analyze the model with remanufacturing. However, similar to our assumptions in this section, they assume that the recycling entity determines "the fraction of current generation products remanufactured from returned units," which corresponds to recycling rate cy in our model, and use the same cost functions across different models. In addition, while their models only consider operational cost, we also study the environmental impact.

5.2. Government's recycling problem

In this section we assume that the local government is in charge of recycling decisions. In theory, the government should have the same objective as the social planner (that is, to minimize the societal cost), but in practice the government is often concerned with minimizing its own cost (organizing the landfill and recycling operations—nodes l and r in Fig. 1b). This view is backed by Walls (2003), who conclude that local governments are primarily motivated by cost. Thus, the effective cost of government, when it is responsible for recycling, can be found as

$$\sigma_G(c, y) := (1-c, c)\cdot(\sigma_l, \sigma_{r,\alpha}(c, y))^{\mathsf{T}} = \sigma_{G,1}(c) + c\cdot\sigma_{G,2}(y), \quad (4)$$

where $\sigma_{G,1}(c) := \sigma_c(c) + (1-c)\sigma_l$ and $\sigma_{G,2}(y) := \sigma_p(y) + (\sigma_{l'}(1-\alpha) - \sigma_o\alpha)(1-y)$. Based on expression (4), to minimize the cost, the government solves

$$\begin{aligned} &\min && \sigma_G(c, y) = \sigma_{G,1}(c) + c\cdot\sigma_{G,2}(y) \\ &\text{subject to} && 0 \le c \le 1, \quad 0 \le y \le 1, \end{aligned} \quad (5)$$

which leads to the following result.

Proposition 3 (Optimal collection and yield rates in the government's problem)**.** *There exist optimal collection and yield rates, c_G^* and y_G^*, that minimize the government's cost in problem (5), and each solution is unique. The optimal yield rate is non-decreasing with respect to the disposal cost/diversion benefit, $\sigma_{l'}(1-\alpha) - \sigma_o\alpha$, and the optimal collection rate is non-decreasing with respect to the cost difference between landfill and recycling, $\sigma_l - \sigma_{G,2}(y_G^*)$.*

Our conclusions in this case are similar to those in the social planner's problem: if the benefit from diverting material into an outside product increases (larger σ_o), the government prefers to use open-loop recycling. If landfill costs decrease, or if the cost of recycling increases, the government has less incentive to collect material. One significant difference (compared to the social planner's choices) is that the government's decisions are not impacted by primary material cost.

5.3. Firm's recycling problem

We now assume that the firm is in charge of recycling decisions and that the relevant costs occur in nodes v, m, r and t_m in Fig. 1b. The effective cost can then be found as

$$\begin{aligned} \sigma_F(c, y) &:= (1-cy, 1, c, cy)\cdot(\sigma_v, \sigma_m, \sigma_{r,\alpha}(c, y), \sigma_{t_m})^{\mathsf{T}} \\ &= \sigma_{F,1}(c) + c\cdot\sigma_{F,2}(y) + \sigma_{F,3} \end{aligned} \quad (6)$$

where $\sigma_{F,1}(c) := \sigma_c(c)$, $\sigma_{F,2}(y) := \sigma_p(y) + (\sigma_{l'}(1-\alpha) - \sigma_o\alpha + \sigma_{v_{net}})(1-y) - \sigma_{v_{net}}$, and $\sigma_{F,3} := \sigma_v + \sigma_m$, which is independent of c and y. It follows from (6) that, to minimize its cost, the firm solves

$$\begin{aligned} &\min && \sigma_F(c, y) = \sigma_{F,1}(c) + c\cdot\sigma_{F,2}(y) + \sigma_{F,3} \\ &\text{subject to} && 0 \le c \le 1, \quad 0 \le y \le 1, \end{aligned} \quad (7)$$

which leads to the following result.

Proposition 4 (Optimal collection and yield rates in the firm's problem)**.** *There exist optimal collection and yield rates, c_F^* and y_F^*, that minimize the firm's cost for problem (7), and each solution is unique. The optimal yield rate is non-decreasing with respect to $\sigma_{l'}(1-\alpha) - \sigma_o\alpha + \sigma_{v_{net}}$ and the optimal collection rate is non-increasing with respect to $\sigma_{F,2}(y_F^*)$.*

The main difference between this result and Proposition 3 are parameters that impact the rate change. Similar to the social planner, the firm is concerned with the primary material costs and the outside option, which influence the firm's choice of yield rate. However, as the firm is not responsible for product disposal, its decisions about collection rate are not impacted by landfill cost changes.

5.4. Recycler's problem

We now assume that an independent party is in charge of recycling decisions. Driven purely by its benefit, this recycler shuts down operations if its operates at a loss. Suppose the firm is willing to pay a certain fraction, γ, of the primary material cost to obtain the secondary material and have it transported to the manufacturing facility. If the firm is willing to pay a premium for the secondary material, we have $\gamma > 1$; otherwise $\gamma \le 1$.[12] Recall that the unit cost of secondary material is $\frac{1}{y}\sigma_r(c, y)$, and its unit cost of transportation to the firm is σ_{tm}. Therefore, the recycler operates when $\frac{1}{y}\sigma_r(c, y) + \sigma_{tm} \le \gamma\sigma_v$. Unlike previous problems that considered multiple recycling cycles, we define the recycler's problem as a single period setting in the sense that at the beginning of each period, we have 1 unit of post-consumer product available, out of which c-unit is collected, and cy-unit of secondary material is generated. The firm then combines cy-unit of secondary material with $(1 - cy)$-unit of primary material to manufacture 1 unit of product, which goes to the consumer and, after usage and disposal/recycling, the second period begins. This contrasts with the previous scenarios in which the social planner and the firm were both concerned with a possible reduction in the primary material consumption due to the use of secondary material over multiple recycling cycles, or when the social planner and the government were concerned about the amount of material diverted from the landfill over multiple recycling cycles. In this scenario, the recycler is only concerned with the amount of secondary material it can obtain in any given recycling cycle, hence it is enough to consider a single period. The recycler, when it is responsible for recycling, wants to maximize its benefit, $cy(\gamma\sigma_v - \sigma_{tm} - \frac{1}{y}\sigma_r(c, y)) = cy\gamma\sigma_v - cy\sigma_{tm} - c\sigma_r(c, y)$. Note that

$$\begin{aligned}
&\max \quad cy\Big(\gamma\sigma_v - \sigma_{t_m} - \frac{1}{y}\sigma_r(c, y)\Big) = -\sigma_c(c) \\
&\quad - c\Big[\sigma_p(y) + \Big(\sigma_{l'} + \gamma\sigma_v - \sigma_{t_m}\Big)(1 - y) - (\gamma\sigma_v - \sigma_{t_m})\Big] \\
&\text{subject to} \quad 0 \le c \le 1, \quad 0 \le y \le 1,
\end{aligned}$$

hence the recycler solves

$$\begin{aligned}
&\min \quad \sigma_R(c, y) := \sigma_c(c) \\
&+c\Big[\sigma_p(y) + \Big(\sigma_{l'}(1 - \alpha) - \sigma_o\alpha + \gamma\sigma_v - \sigma_{t_m}\Big)(1 - y) - (\gamma\sigma_v - \sigma_{t_m})\Big] \\
&\text{subject to} \quad 0 \le c \le 1, \quad 0 \le y \le 1.
\end{aligned} \tag{8}$$

Proposition 5 (Optimal collection and yield rates in recycler's problem)**.** *There exist optimal collection and yield rates, c_R^* and y_R^*, that maximize the recycler's benefit in (8), and each solution is unique. The optimal solutions are non-decreasing with respect to $\sigma_{l'}(1 - \alpha) - \sigma_o\alpha + \gamma\sigma_v - \sigma_{tm}$.*

The implication for the recycler is thus very similar to the firm's recycling decisions discussed in Proposition 4.

5.5. Comparison of optimal recycling decisions

We now compare the optimal recycling efforts of all the abovementioned centralized decisions.

Proposition 6 (Optimal rates comparison)**.** *Let* $\rho := \frac{\sigma_p(y_R^*) - \sigma_p(y^*) - (y_R^* - y^*)\left(\sigma_{l'}(1-\alpha) - \sigma_o\alpha + \sigma_{v_{net}}\right)}{\sigma_v} > 0$. *Then, we have the following results:*

$$\gamma \begin{Bmatrix} >1 + \rho + \frac{\sigma_l}{\sigma_v} \\ \in \Big(1, 1 + \rho + \frac{\sigma_l}{\sigma_v}\Big] \\ \le 1 \end{Bmatrix}$$

$$\iff \begin{cases} y_R^* > y^* \equiv y_F^* > y_G^*; & c_R^* > \max\{c^*, c_F^*, c_G^*\}; \\ y_R^* > y^* \equiv y_F^* > y_G^*; & c^* \ge c_R^* > \max\{c_F^*, c_G^*\}; \\ y^* \equiv y_F^* \ge y_R^* > y_G^*; & c^* > c_F^* \ge c_R^*. \end{cases}$$

This result shows that the optimal yield rates chosen by the firm and by the social planner coincide and exceed the level chosen by the government; that is, $y^* \equiv y_F^* > y_G^*$. This is good news, as we need no additional mechanisms to induce an optimal yield level under the firm's problem. The relationship between optimal collection levels chosen by the firm, the government, and the social planner depends on landfill and primary materials costs. Most notably, we can show that when $\sigma_{v_{net}} \le \frac{\sigma_{G,2}(y^*) - \sigma_{G,2}(y_G^*)}{y^*}$, then $c_F^* < c^* \le c_G^*$; that is, when the cost of primary material is low, the government, which cares only about landfill and recycling costs, chooses a collection rate higher than the socially optimal one in order to curb its landfill cost. When the firm is willing to pay a premium for secondary materials, the recycler always chooses the highest yield rate; if the premium is high enough, the recycler also chooses the highest collection rate. In most settings (with the exception of office paper), the secondary materials are sold at a lower price then the primary materials (see Table 4), and the recycler attains lower recycle and yield rates than the firm. When a "green" firm pays too much for secondary materials to the recycler, the social planner may resort to taxing the secondary material to induce the recycler to lower its price, which may curb the optimal rates to their first-best level. This would effectively penalize the recycler for making excessive recycling efforts and discourage the firm from paying a premium for secondary materials.

We now compare our results to those from Savaskan et al. (2004).[13] Let us denote the retailer by the subscript *ret*. Savaskan et al. (2004) assume a convex collection cost and constant remanufacturing cost, and conclude that $c^*y^* > c_{ret}^*y_{ret}^* > c_M^*y_M^* > c_R^*y_R^*$. In our model, the processing cost is not constant, and this relationship does not always hold. More precisely, when the cost of secondary material is high enough (high γ), we can have $c_R^*y_R^* > c^*y^* > c_M^*y_M^*$.

As previously mentioned, our analysis assumes that the unit cost functions remain unchanged regardless of the entity responsible for recycling, as each recycling entity can outsource the actual recycling process to a third party. Now, if we assume that some of the recycling entities have more options than others and are able to achieve lower cost due to, say, economies of scale, the abovementioned results could change. For instance, if we assume that the firm cannot achieve the same economies of scale as the social planner, the firm would face a steeper processing cost and consequently select a lower yield rate to reduce this cost. In such a case, we can have $y_F^* < y^*$. This type of analysis is beyond the scope of this paper and therefore omitted.

6. Decentralized recycling decisions and coordination mechanisms

In this section, we assume that independent parties choose the collection rate and the yield rate. This scenario differs from Savaskan et al. (2004) and Atasu et al. (2013) and is closer to what can be observed in practice (compared with the centralized optimization problems discussed in the previous section). However, as will be seen in our analysis, we use results from centralized optimization models to obtain insights about decentralized cases.

[12] With the exception of office paper, discussed later, $\gamma > 1$ is rarely observed in practice. Our discussion with a PET recycling firm in California did reveal, however, that some companies pay a premium for recycled PET.

[13] Atasu et al. (2013) allow for economies/diseconomies of scale in collection costs (represented by concave and convex component of the collection cost, respectively), and show that with economies of scale, the optimal recovery rate (product of collection rate and yield rate) is always 0 or 1. As we do not consider concave costs, we limit our comparison to Savaskan et al. (2004), although the total collection cost may be convex even with economies of scale.

We assume that one entity (the social planer or local government) determines the desired minimum collection rate, and after observing this value, the entity responsible for recycling chooses its yield rate. In practice, it is more likely that the local government will determine the collection rate by making decisions about curbside recycling containers, drop-off recycling zones, recycling centers, and so forth, but we consider both options for completeness. We use backward induction and start with the second decision, the optimal yield rate. Recall that we assume that recycling cost is separable in c and y. As such, the optimal yield rates correspond to those obtained in centralized optimization models—y_G, y_F, and y_R—if the local government, the firm, and the recycler are responsible for recycling, respectively. The entity responsible for determining the collection rate anticipates this value when it makes its decision.

First, suppose that the social planner determines c. As we have previously shown, the firm always chooses the same yield rate as the social planner, and we obtain the following:

Theorem 1 (Social planner selects the collection rate)**.** *Suppose that the social planner chooses the collection rate. Then, selecting the firm as the entity responsible for recycling leads to socially optimal decisions without implementation of additional coordinating mechanisms.*

Thus, the social planner prefers to have firms responsible for recycling. This preference may not always be possible, so we discuss the social planner's options under other scenarios in the forthcoming subsections.

Next, suppose that the government chooses the collection rate. As we have shown in Proposition 6, government always selects the lowest yield rate (when compared with the firm or the recycler). As a result, if the firm or the recycler selects the yield rate (that is, when $y = y_F^*$ or $y = y_R^*$), then $\sigma_{G,2}(y) \geq \sigma_{G,2}(y_G^*)$. In response to this increase, the government would select the collection rate lower than c_G^* when it does not select the yield rate (that is, if $y > y_G^*$). Recall that our discussion of Proposition 6 concluded that the government chooses collection rate which exceeds the first-best when the cost of primary materials is low. Thus, allowing the firm or the recycler to select the yield rate in such a case reduces the collection rate chosen by the government and results in an improved systemwide performance. However, when $c_G^* < c^*$, choosing $y > y_G^*$ would move the collection rate chosen by the government even further away from the first-best, so it is better to let the government select the yield rate as well under this scenario. This is summarized in our second theorem below.

Theorem 2 (Government selects the collection rate)**.** *Suppose that the government chooses the collection rate. Then, when* $\sigma_{v_{net}} < \frac{\sigma_{G,2}(y^*)-\sigma_{G,2}(y_G^*)}{y^*}$, *systemwide performance can be improved if recycling is delegated to the firm or to the recycler; otherwise, society benefits if the government also takes over recycling responsibilities.*

Theorem 2 and the data collected (see Table 4) suggest that when the government determines the collection rate, paper and glass might be ideal for recycling by the firm or by the recycler.

For reader's convenience, we provide a summary of notations used in this section in Table 7.

Table 7
A summary of notations for decentralized recycling.

Notation	Definition
$I_{G,y}/I_{R,y}$	Incentive to government/recycler for improving the yield rate
$y_{G,I}/y_{R,I}$	Government's/recycler's yield rate in response to incentive
$I_{G,c}$	Incentive to government for improving the collection rate
$c_{G,I}$	Government's collection rate in response to incentive
$c_{G,d}$	Government's collection rate in response to deposit
d	Deposit/refund collected by the government
D	Deposit/refund collected by the social planner

6.1. Monetary incentives—taxes and rebates

We next analyze how to improve performance of the decentralized systems to ideally achieve the first-best rates. We consider both cases—in which the collection rate is determined by the social planner and by the government—and discuss incentive schemes that can induce the firm, the recycler, and the government to make socially optimal decisions. We first focus on the optimal yield rate.

Theorem 3 (Incentives for socially optimal yield rate)**.** *Suppose that the social planner determines the collection rate.*

- *If the government conducts the recycling, the social planner should offer an incentive equal to* $I_{G,y} = y_{G,I} \cdot \sigma_{v_{net}}$, *where* $y_{G,I}$ *is the government's choice of yield rate under incentive* $I_{G,y}$.
- *If the firm conducts the recycling, the social planner does not need to offer any incentives.*
- *If the recycler conducts the recycling, the social planner should offer an incentive equal to* $I_{R,y} = y_{R,I}(1-\gamma)\sigma_v$, *where* $y_{R,I}$ *is the recycler's choice of yield rate under incentive* $I_{R,y}$.

Under the mechanisms previously described, the social planner can induce the recycling entity to select the first-best yield rate. The decision on which entity to choose depends on the size of the required payout. Clearly, the easiest option is to have the firm responsible for recycling, as this does not require any incentives. If, however, this option is not supported with the current practice and existing infrastructure, the choice between the recycler and the government depends on the relationship between the value of the recycled material and the cost of transportation for manufacturing. That is, when the value of the recycled material is high enough ($\gamma \cdot \sigma_v > s_{t_m}$), it is cheaper to incentivize the recycler; otherwise, the government is the better choice. Note that $I_{R,y}$ can actually amount to a tax whenever secondary materials are costly and the recycler's yield exceeds the socially optimal one ($\gamma > 1$). Thus, our calculations indicate it is usually easier to incentivize the recycler to achieve optimal yield rate.

Next, we assume that the government determines the collection rate. As we have shown in Proposition 6, if the local government chooses both the collection and the yield rates, the optimal yield rate chosen by the government is always lower than the socially optimal one, and the optimal collection rate chosen by the government is usually lower than the socially optimal one. One exception is the case in which primary materials are very inexpensive and the government chooses a higher collection rate to reduce the amount of material sent to the landfill. As a result, when primary materials are cheap, the social planner might need to introduce taxes in order to reduce the government's collection rate. However, if the socially optimal yield rate is chosen, $y^* > y_G^*$, the government would in response reduce its collection rate, and the need for taxes never occurs. Thus, if the recycling entity is incentivized to choose the socially optimal yield rate, the social planner always need to incentivize the government[14] if the planner wants the government to implement the socially optimal collection rate.

Theorem 4 (Incentives for socially optimal collection rate)**.** *Suppose that the government determines the collection rate. If the recycling entity is incentivized to implement the socially optimal yield rate, a social planner who wants to achieve the socially optimal collection rate needs to offer to the government an incentive equal to*

[14] Although theoretically the social planner might need two mechanisms, taxes and incentives, to induce socially optimal collection rate by the government, the first case will never occur if the socially optimal yield rate is chosen.

$I_{G,c} = c_{G,I}y^{*}\sigma_{v_{net}}$, *where* $c_{G,\,I}$ *is the government's choice of collection rate in response to incentive* $I_{G,\,c}$.

As previously discussed, if the social planner wants to implement socially optimal collection and yield rates, the best option is to determine the collection rate itself, and let the firm be responsible for recycling. However, if this arrangement cannot be implemented, our result shows that the government would select an optimal collection rate when provided with an incentive proportional to the recovery rate, which is the product of the yield rate and the collection rate.

We note that, in practice, finding socially optimal yield and collection rates is a non-trivial problem, and that different entities can have different emissions and underlying costs (our model assumes that emissions and costs are equal for all parties). However, we hope that our model will encourage a social planner to incentivize the recycling entities to improve their choices. Our discussion with a PET recycling firm in California revealed that the California Recycle Market Development Fund (https://www.calrecycle.ca.gov/RMDZ/) provides an incentive for each pound of material recycled and sold to a California manufacturer, and indicated that even small incentives can make a big difference when primary material costs decrease.

6.2. Deposit/refund

Palmer et al. (1996) use a model of waste generation and recycling with price-dependent demand to analyze the impact of three policy interventions: deposit/refund, advance disposal fee, and recycling subsidy. They conclude that deposit/refund is the least costly mechanism for reducing MSW disposal. Unlike their model, our model assumes that demand is not price-dependent, which enables us to obtain some novel results. Although Palmer et al. (1996) see the impact of a tax/refund through a change in demand due to higher price, we show that this mechanism can improve the system performance even when the demand is not price-dependent.

In California, the state government implements a recycling policy for beverage containers. All manufacturers or importers are required to pay the recycling fee, say deposit d, to the state, which they later collect from their customers who then charge their customers, and so on. The customer who consumes the beverage can dispose of the container or return it for recycling and recuperate the deposit (indirectly) from the government. If we consider the government's problem described in (5), we can observe that the government obtains d for every unit of primary product sold, and then has to return $c \cdot d$ after the first cycle; $c \cdot cy \cdot d$ after the second recycling cycle, $c \cdot c^2y^2 \cdot d$ after the third recycling cycle, and so on. Consequently, the first part of the government's cost function changes to $\sigma_c(c) + (1-c)(\sigma_l - d)$. Let us denote the government's optimal collection rate in the model in which the government implements the deposit/refund model by c^{*}_{Gd}. We then have the following result.

Proposition 7 (Optimal collection rate in government's problem with deposit/refund)**.** *If the government chooses the collection rate and implements a deposit/refund policy, it will reduce the collection rate,* $c^{*}_{Gd} < c^{*}_{G}$. *The optimal collection rate is non-increasing with respect to deposit.*

The abovementioned result is not surprising: deposits reduce the government's cost, but when more material is collected, the government needs to issue more refunds, which then increases costs. Consequently, we have the following corollary.

Corollary 1 (Incentives for a socially optimal collection rate when the government implements the deposit/refund model)**.** *Suppose that the government determines the collection rate and uses the deposit/refund model. If the recycling entity is incentivized to implement the socially optimal yield rate, a social planner who wants to achieve the socially optimal collection rate needs to offer the government an incentive equal to* $I_{G,c} = c_{G,I}\left(y^{*}\sigma_{v_{net}} + d\right)$.

Thus, when the government uses the deposit/refund scheme, achieving the social optimum requires larger subsidies from the social planner. However, the social planner can reduce its cost by using the following model. Suppose that the social planner charges the government a deposit, D, for every unit sold under its jurisdiction, and returns a refund for every unit collected. The first part of the government's cost function in (5) then changes to $\sigma_c(c) + (1-c)(\sigma_l + D)$, and the government selects a higher collection rate in order to increase the refund obtained from the social planner. Let us denote the government's optimal collection rate in the model in which the social planner implements the deposit/refund model by c^{*}_{GD}. Our last major result establishes that, under this scheme, we can achieve a socially optimal collection rate without additional incentives.

Theorem 5 (Incentives for a socially optimal collection rate when the social planner implements the deposit/refund model)**.** *Suppose that the government determines the collection rate. If the recycling entity is incentivized to implement the socially optimal yield rate, a social planner who wants to achieve the socially optimal collection rate needs to implement a deposit/refund model in which it charges the government deposit* $D = y^{*}\sigma_{v_{net}}$ *for each product sold.*

If we apply $d = -D$ to Corrollary 1, we can see that the social planner needs to offer the government an incentive equal to $I_{G,c} = c_{G,I}\left(y^{*}\sigma_{v_{net}} - D\right)$; selecting $D = y^{*}\sigma_{v_{net}}$ would achieve a socially optimal collection rate without the need for government incentives. Using California as the social planner, and counties or cities as the government, the deposit/refund scheme should achieve better results if the state charged a deposit fee to local municipalities and refunded it for the recycled containers, instead of charging the firms directly.[15] According to our estimates, $\sigma_{v_{net}} = 1733.49–154.19 = \$1579.30$ per short ton of PET. Kuczenski and Geyer (2011) estimate that a typical bottle in California contains about 0.5 l of beverage, and that 1 kilogram of PET represents 27.9 l of beverage. Thus, we can obtain $27.9 \cdot 2 \cdot 907.17 = 50620$ beverage containers from a short ton of PET, hence $\sigma_{v_{net}} = \$0.031$ per average PET bottle in California. Consequently, by charging a modest per bottle deposit to local municipalities (lower than the current deposit/refund scheme charged to firms, which is \$0.05 for containers less than 0.7 liter (24 ounces) and \$0.10 for larger containers), California should be able to improve collection rates.

Walls (2011) discusses the use of deposit/refund systems for beverage containers, batteries, motor oil, and so forth. She concludes that both theoretical models and real-world application have shown that deposit/refund schemes outperform alternative waste disposal policies, such as advance disposal fees or recycled content standards. However, she also notices that many product upstream systems, in which recyclers receive the refund, may have lower costs and better environmental outcomes than downstream systems, in which the consumers receive the refund. Our results provide theoretical support for this conclusion. Walls and Palmer (2001) show that a traditional deposit-refund system alone cannot achieve full social optimum. Our results suggest that a careful choice of material-specific deposit/refund scheme implemented by a social planner with respect to local governments could achieve an optimal collection rate without the need for additional instruments, as required incentives could be offset by deposits/refunds.

[15] Recall that government's recycling model also applies to the case in which the recycler is responsible for both recycling and landfill.

7. A Case study: Minimum recycled content requirement

The implementation of minimum recycled content is another approach that may lead to an improvement in collection and yield rates. In March 2015, a California State Assembly bill (Alejo, 2015) was introduced, which would have required "every manufacturer of PET plastic packaging for sale manufactured in the state to include be manufactured with, and empty PET plastic packaging imported into the state to be filled with food or drink in the state for sale in the state to contain, a minimum of 10% of postfilled PET plastic in its PET plastic packaging." The bill did not pass.

If the bill had passed and been implemented, what would have been the impact on collection and yield rates? Consider the following scenario. In the first period, the firm begins with $(1-cy)$ unit of primary materials and cy-unit of secondary materials to manufacture 1 unit of product for consumers. After recycling, the firm obtains cy-unit of secondary materials, which can be combined with $(1-cy)$ unit of primary materials in the next period to again manufacture 1-unit of the product. Therefore, each product contains cy-unit of recycled content in each period. Assume that the recycler is responsible for product recycling, and that the social planner requires at least θ-unit of recycled content. If $cy \geq \theta$, the legislation does not have any impact; if the opposite holds, there are not enough secondary materials available and the recycler needs to be incentivized to increase the rates. If secondary materials are cheaper than primary materials, the firm (instead of the social planner) can develop incentives similar to those described in Theorem 4 to achieve this goal. However, if the opposite is true, Proposition 6 implies that the recycler may already use rates that exceed the social optimum, and a push to increase those rates by implementing minimum recycled content may further worsen the situation. Thus, this type of legislation is only effective when $cy < \theta$ and the secondary materials are inexpensive.

As previously mentioned, PET recycling is undergoing a difficult period due to the low cost of oil. Table 7 in NAPCOR and APR (2015) shows that for US PET data, $cy = 22.6\%$ in 2013. Consequently, a minimum requirement of $\theta = 10\%$ is not useful; the recycled content should be set to a higher level, $\theta > cy = 22.6\%$, if we want to influence collection and yield rates. In the data we collected for early 2016 shown in Table 4, the aggregate cost of the secondary material (\$1237.56) is lower than that of the primary material (\$1974.38), so selecting a higher θ may be effective. However, if oil prices go down significantly and secondary material becomes costlier than primary material, the minimum recycled content requirement may actually worsen the societal outcome by increasing the rates to undesirable levels. When secondary material is costlier than primary material, there is a range of secondary material prices, $1 \leq \gamma \leq 1 + \rho + \frac{\sigma_l}{\sigma_v}$; however, in this scenario the optimal collection rate is lower than the social optimum, and a minimum recycle content can help keep the recyclers in business.

The impact of legislation is different if we focus only on beverage bottles consumed in California. According to Kuczenski and Geyer (2011), the collection rate of beverage containers in California is $c = 73\%$, while the recycled content in bottles is $cy = 3.9\%$, implying that $y = 5.34\%$.[16] If a minimum recycled content of 10% is imposed, it would represent an increase in the recycling rate, cy, of 156%. If we assume the collection rate remains unchanged, the legislation would require an increase in yield rate to $y = 13.7\%$. Similarly, if we assume that the collection rate could be increased to 85%, the legislation would require a yield rate of $y = 11.8\%$. This could be obtained by reducing the amount diverted to open-loop recycling.

[16] Kuczenski and Geyer (2011) estimate that 75% of collected material is diverted to open-loop recycling, hence 1kg of primary materials yields 0.547 kilogram of material sent to outside recycling, while 20% of collected material ends up in landfill.

8. Some extensions

8.1. Open-loop recycling: PET beverage containers in California

In our previous analysis, we considered PET in general and assumed $\alpha = 0$; we now present a brief open-loop analysis for PET beverage containers in California. Our collection and yield rates are based on Kuczenski and Geyer (2011); they assume $c = 73.3\%$ and $cy = 3.9\%$, which corresponds to $y = 5.32\%$. They further assume that $1 - y = 94.68\%$ is divided as follows: 60.01% of the total recycled quantity (or 63.38% of the 1-y amount) is sold to foreign markets, 9.20% (9.7% of the 1-y amount) goes to non-food use, 5.48% (5.79% of the 1-y amount) goes to non-bottle food use, and 19.99% ends up in landfills. Hence, $\alpha = 1 - \frac{(1-\alpha)(1-y)}{1-y} = 1 - \frac{19.99\%}{94.68\%} = 78.89\%$.

In order to estimate revenue that could be obtained from outside options, s_o, we use prices of different grades of pallet and flake material from PetroChem (2017) and estimate that we can obtain 32.9% of the value of bottle quality resin (estimated at \$1140 per short ton; see Appendix C) when selling to foreign markets (i.e., \$375), 53.2% (i.e., \$604.50) from non-food use, and 67.5% (i.e., \$769.50) from non-bottle food use, with landfill cost of $s_{l'} = \$94.36$ per short ton. Combining these values, we derive $s_d(5.32\%) = [\$94.36 \cdot 21.11\% - (\$375 \cdot 63.38\% + \$604.50 \cdot 9.7\% + \$769.50 \cdot 5.79\%)] \cdot (1 – 5.32\%) = -\$303.88$. In other words, a disposal cost/diversion benefit would generate benefits for PET beverage containers.

We want to obtain x_s for the open-loop case, so we first need to calculate s_r in this instance. For simplicity, we assume that the collection rate is the same as for the PET in general (hence $s_c(c)$ remains unchanged), and that $s_p(y)$ is linear in y, $s_p(y) = C \cdot y$ for some constant C. Environment and Pira (2003) estimate the cost of recycling 1 metric ton of PET bottles via curbside collection in Europe at around 1132 euros, which corresponds to around \$1200 per short ton, close to our estimate. They separate this recycling cost into various components and estimate the collection cost to be around 280 euros per tonne, or about 24.7% of total recycling cost.[17]

We now apply this estimate to our original closed-loop calculation for PET (with $y = 69.7\%$ and $\alpha = 0$) and obtain $s_c(c) = \$281.58$ and $C = \$694.82$. We can now calculate the cost for PET beverage containers, $s_r = \$281.58 + \$694.82 \cdot 5.32\% - \$364.58 \cdot 94.68\% = -\$26.64$. Thus, recycling of PET beverage containers in California generates revenue at any yield rate due to the profitability of open-loop recycling ($s_d < 0$) at the current collection cost, $s_c = \$281.58$. If the collection cost increases to \$481.16 per short ton, the current yield rate would become the lower boundary, $x_s = 5.32\%$. This example shows that open-loop recycling makes recycling desirable even when a lower fraction of primary material is recycled back into the original product.

8.2. Effect of the social cost factor, ζ

The social cost of emissions is expected to increase over time (U.S. Government, 2013), and recent research argues that the cost could be even higher (Moore & Diaz, 2016) than estimated. Therefore, we analyze the effect of ζ on optimal decisions, $y^*(\zeta)$ and $c^*(\zeta)$ for the social planner problem. Recall that $y^*(\zeta)$ and $c^*(\zeta)$ minimize aggregate costs. We introduce some new symbols, y_e^* and c_e^*, as minimizers of the social planner's problem emissions, and y_s^* and c_s^*, as minimizers of the social planner's problem operational costs.

[17] As an alternative, they consider recycling via bring-back collection, which further reduces collection costs to around 20% of the total recycling cost.

Proposition 8 (Impact of social cost factor on optimal rates)**.** *As the social cost factor, ζ, increases, the following holds:*

1. $\min\{y_s^*, y_e^*\} \le y^*(\zeta) \le \max\{y_s^*, y_e^*\}$. *In addition, if $y_e^* \le y_s^*$ (resp., $y_e^* \ge y_s^*$), then the optimal yield rate decreases (resp., increases) in ζ and $y^*(\zeta)$ converges to y_e^* as $\zeta \to \infty$.*
2. $\min\{c_s^*, c_e^*\} \le c^*(\zeta) \le \max\{c_s^*, c_e^*\}$. *In addition, $c_e^* \le c_s^*$ (resp., $c_e^* \ge c_s^*$), then the optimal collection rate decreases (resp., increases) in ζ and $c^*(\zeta)$ converges to c_e^* as $\zeta \to \infty$.*

Thus, as ζ increases over time, the optimal yield rate either monotonically increases or decreases, and it converges to the optimal yield rate for emissions, y_e^*; the same holds for the optimal collection rates. These results also hold for all other centralized and decentralized recycling decision problems we discussed in Sections 5 and 6 as the proof follow the same logic.

9. Concluding remarks and managerial insights

Whether recycling is overall (environmentally and/or financially) desirable has been a debatable topic for a long time. Our conclusion is that, for most common materials in the US, the answer is positive. Nonetheless, recyclers have to carefully select the yield rate of the underlying recycling processes, and a social planner might need to provide appropriate incentives to help them.

We first show that recycling is effective in reducing life cycle GHG emissions and operational costs for all the common materials under consideration, except for glass, which is not financially effective. In the case of office paper, recycling reduces the emissions at any yield rate. Our analysis sheds some light on the core reasons behind financial difficulties faced by recycling businesses, and shows that the recycling yield rate is a key metric in determining their financial feasibility. Investment in technology that improves the yield rate is desirable, yet the cost may be high.

One implication from our findings is that in areas with higher costs/penalties for disposal of non-recycled materials into landfills, air, or sewer systems, we can expect a higher yield rate. Further, we notice that the collection rate depends on the relative cost of primary materials and landfills. Different entities responsible for recycling will make different choices. When the social planner determines the collection rate, choosing the firm as the recycling entity is the best option, as it selects the same yield rate as the social planner. Among the remaining two options, we find that the recycler is a better choice when the value of the secondary material is high enough. If the government selects the collection rate, its choice approaches social optimum when primary materials are cheap and the recycler or the firm is responsible for recycling; otherwise, the government should undertake recycling as well.

One of the common instruments for increasing the collection rate is the deposit/refund model, which encourages the end consumer to recycle. Previous research has assumed price-dependent demand and found that this model is the cheapest instrument that can achieve desired goals. Our analysis shows that when government implements a deposit/refund model with constant demand, it may lead to a reduced collection rate; however, if the social planner implements this model with respect to the government, it may induce the government to make the socially optimal choice.

Supplementary material

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.ejor.2018.11.010.

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Online Supplement to "Recycling Common Materials: Effectiveness, Optimal Decisions, and Coordination Mechanisms"

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We provide the proofs in Appendix A, and detailed derivation of emissions and costs in Appendix B.

Appendix A: Proofs

The following Lemma is used in Section 3.3; the proof is easy and is omitted.

Lemma A1. *Let $b > 0, d > 0$, and $\zeta > 0$. Then, $\frac{a}{b} > \frac{a+\zeta c}{b+\zeta d}$ iff $\frac{a}{b} > \frac{c}{d}$, and $\frac{c}{d} > \frac{a+\zeta c}{b+\zeta d}$ iff $\frac{a}{b} < \frac{c}{d}$.*

The following Lemma is used to prove Propositions 2 to 7; the proof is easy and is omitted.

Lemma A2. *Let $f(x)$ be an increasing, strictly convex function for $x \in [0,1]$. Then, the following unique minimizer x^* is non-decreasing (increasing when interior solution) with respect to s.*

$$x^* = \underset{x\in[0,1]}{argmin}\ f(x) - sx = \begin{cases} 1, & \textit{if } s > \left(f'\right)^{-1}(1), \\ \left(f'\right)^{-1}(s), & \textit{if } s \in \left[\left(f'\right)^{-1}(0), \left(f'\right)^{-1}(1)\right], \\ 0, & \textit{if } s < \left(f'\right)^{-1}(0). \end{cases}$$

Proof of Proposition 2: Because the optimization problem (3) is separable in c and y, we first minimize $\sigma_2(y) = \sigma_p(y) + \sigma_{d,\alpha}(y) - y\sigma_{v_{net}} = \sigma_p(y) + (\sigma_{l'}(1-\alpha) - \sigma_o\alpha)(1-y) - y\sigma_{v_{net}}$, and obtain $y^* = \underset{y\in[0,1]}{\operatorname{argmin}}\ \sigma_p(y) - y\left(\sigma_{l'}(1-\alpha) - \sigma_o\alpha + \sigma_{v_{net}}\right)$. Because $s_p(y), e_p(y)$ are both strictly increasing convex functions, the same is true for $\sigma_p(y) = s_p(y) + \zeta e_p(y)$ for $\zeta > 0$. Therefore, Lemma A2 implies that we can write the optimal yield rate of the social planner's problem as

$$y^* = \begin{cases} 1, & \text{if } \sigma_{l'}(1-\alpha) - \sigma_o\alpha + \sigma_{v_{net}} > \left(\sigma_p'\right)^{-1}(1), \\ \left(\sigma_p'\right)^{-1}(\sigma_{l'}(1-\alpha) - \sigma_o\alpha + \sigma_{v_{net}}), & \text{if } \sigma_{l'}(1-\alpha) - \sigma_o\alpha + \sigma_{v_{net}} \in \left[\left(\sigma_p'\right)^{-1}(0), \left(\sigma_p'\right)^{-1}(1)\right], \\ 0, & \text{if } \sigma_{l'}(1-\alpha) - \sigma_o\alpha + \sigma_{v_{net}} < \left(\sigma_p'\right)^{-1}(0). \end{cases}$$

Second, we use y^* derived above and minimize $\sigma_1(c)+c\cdot\sigma_2(y^*) = \sigma_c(c)-c\left[\sigma_l-\sigma_{t_c}\right]+c\cdot\sigma_2(y^*)$, which yields $c^* = \underset{c\in[0,1]}{\operatorname{argmin}}\ \sigma_c(c) - c\left[\sigma_l - \sigma_{t_c} - \sigma_2(y^*)\right] = \sigma_c(c) - c\left[\sigma_l - \sigma_2(y^*)\right] + \eta$, given $\sigma_{t_c} = \frac{\eta}{c}$. Since $\sigma_c(c)$ is a strictly increasing convex function, we can use the same argument as above for $\sigma_c(c)$ and derive the optimal collection rate of the social planner's problem as

$$c^* = \begin{cases} 1, & \text{if } \sigma_l - \sigma_2(y^*) > \left(\sigma_c^{'}\right)^{-1}(1), \\ \left(\sigma_c^{'}\right)^{-1}(\sigma_l - \sigma_2(y^*)), & \text{if } \sigma_l - \sigma_2(y^*) \in \left[\left(\sigma_c^{'}\right)^{-1}(0), \left(\sigma_c^{'}\right)^{-1}(1)\right], \\ 0, & \text{if } \sigma_l - \sigma_2(y^*) < \left(\sigma_c^{'}\right)^{-1}(0). \end{cases}$$

Notice that we allow a rather general functional form for collection, $\sigma_c(c)$, and production, $\sigma_p(y)$, cost functions—our requirements were only that they should be strictly increasing and convex. Thus, we cannot derive explicit closed-form solutions for the optimal collection rate, c^* and the optimal yield rate, y^*. However, when the social planner has the information about the collection and production cost functions—for example, $\sigma_c(c) = \beta_1 c^2$ and $\sigma_p(y) = \beta_2 y^2$—one may derive the corresponding closed-form solutions, by plugging in $\left(\sigma_c^{'}\right)^{-1}(x) = \frac{x}{2\beta_1}$ and $\left(\sigma_p^{'}\right)^{-1}(x) = \frac{x}{2\beta_2}$ into the above expressions for y^* and c^*. □

Proofs of Propositions 3, 4, 5 are omitted because they follow the same logic as in the above proof.

Proof of Proposition 6: Our proof is based on two parts below.

Part I: We claim the optimal recycling decisions of social planner, government and firm as below.

- The social planner and the firm always choose the same optimal yield rate, and $y^* \equiv y_F^* \geq y_G^*$.
- The social planner always chooses a higher collection rate than the firm. In addition,

$$\sigma_{v_{net}} \left\{ \begin{array}{l} \leq \frac{\sigma_{G,2}(y^*)-\sigma_{G,2}(y_G^*)}{y^*} \\ \in \left(\frac{\sigma_{G,2}(y^*)-\sigma_{G,2}(y_G^*)}{y^*}, \frac{\sigma_l}{y^*} + \frac{\sigma_{G,2}(y^*)-\sigma_{G,2}(y_G^*)}{y^*}\right) \\ \geq \frac{\sigma_l}{y^*} + \frac{\sigma_{G,2}(y^*)-\sigma_{G,2}(y_G^*)}{y^*} \end{array} \right\} \Longleftrightarrow \left\{ \begin{array}{l} c_F^* < c^* \leq c_G^*; \\ c_F^* < c_G^* < c^*; \\ c_G^* \leq c_F^* < c^*. \end{array} \right. \tag{A1}$$

Proof: To compare optimal solutions of (3), (5), (7), we use the technique from the proof of Proposition 2. One can verify that we compare optimal yield rates below:

$$y^* \equiv y_F^* = \underset{y\in[0,1]}{\operatorname{argmin}}\ \sigma_p(y) - y\left(\sigma_{l'}(1-\alpha) - \sigma_o\alpha + \sigma_{v_{net}}\right),\ y_G^* = \underset{y\in[0,1]}{\operatorname{argmin}}\ \sigma_p(y) - y\left(\sigma_{l'}(1-\alpha) - \sigma_o\alpha\right).$$

and obtain $y^* \equiv y_F^* \geq y_G^*$ because $\sigma_{v_{net}} > 0$. Because $\sigma_2(y) = \sigma_{F,2}(y) = \sigma_{G,2}(y) - y\sigma_{v_{net}}$, one can verify that we next compare optimal collection rates below:

$$
\begin{aligned}
c^* =& \operatorname*{argmin}_{c\in[0,1]} \sigma_c(c) - c\left[\sigma_l - \sigma_{G,2}(y^*) + y^*\sigma_{v_{net}}\right], \\
c_G^* =& \operatorname*{argmin}_{c\in[0,1]} \sigma_c(c) - c\left[\sigma_l - \sigma_{G,2}(y_G^*)\right], \quad \text{and} \\
c_F^* =& \operatorname*{argmin}_{c\in[0,1]} \sigma_c(c) - c\left[-\sigma_{G,2}(y_F^*) + y_F^* \cdot \sigma_{v_{net}}\right].
\end{aligned} \tag{A2}
$$

Because $\sigma_l > 0$ and $y^* = y_F^*$, we claim $c^* \geq c_F^*$ by Lemma A2. Thus, when $y_F^* > 0, y^* > 0$, we have

$$
\begin{aligned}
&c_F^* > c_G^* \Longleftrightarrow -\sigma_{G,2}(y_F^*) + y_F^* \cdot \sigma_{v_{net}} > \sigma_l - \sigma_{G,2}(y_G^*) \Longleftrightarrow \sigma_{v_{net}} > \frac{\sigma_l}{y_F^*} + \frac{\sigma_{G,2}(y_F^*) - \sigma_{G,2}(y_G^*)}{y_F^*} \quad \text{and} \\
&c^* > c_G^* \Longleftrightarrow \sigma_l - \sigma_{G,2}(y^*) + y^*\sigma_{v_{net}} > \sigma_l - \sigma_{G,2}(y_G^*) \Longleftrightarrow \sigma_{v_{net}} > \frac{\sigma_{G,2}(y^*) - \sigma_{G,2}(y_G^*)}{y^*}.
\end{aligned} \tag{A3}
$$

Part II: We use the result in Part I to prove Proposition 6.

Proof: With some algebra, we can show that

$$
y_R^* = \operatorname*{argmin}_{y\in[0,1]} \sigma_p(y) - y\left(\sigma_{l'}(1-\alpha) - \sigma_o\alpha + \gamma\sigma_v - \sigma_{t_m}\right), \text{and} \tag{A4}
$$

$$
c_R^* = \operatorname*{argmin}_{c\in[0,1]} \sigma_c(c) - c\left[-\sigma_{G,2}(y_R^*) + y_R^*(\gamma\sigma_v - \sigma_{t_m})\right]. \tag{A5}
$$

Assuming interior solutions (recycling firm stays in business), we have $\gamma\sigma_v - \sigma_{t_m} \geq \frac{1}{y_R^*}\sigma_r(c_R^*, y_R^*) > 0$, and $\sigma_{l'} + \gamma\sigma_v - \sigma_{t_m} > \sigma_{l'}$ and $y_R^* > y_G^*$. Hence, we notice that $y_G^* < \min\{y_R^*, y^* \equiv y_F^*\}$ and

$$
\gamma \lesseqqgtr 1 \Longleftrightarrow \sigma_{l'} + \gamma\sigma_v - \sigma_{t_m} \lesseqqgtr \sigma_{l'} + \sigma_v - \sigma_{t_m} \Longleftrightarrow y_R^* \lesseqqgtr y^* \equiv y_F^*. \tag{A6}
$$

Consequently, we have the following results on the optimal collection rates comparison:

1. When $\gamma < 1$, then $c_R^* < c_F^*$, and since $c_F^* < c^*$ we conclude $c_R^* < c_F^* < c^*$. However, we may have different relationships for c_R^* and c_G^*.

   – First, we show $c_R^* < c_F^*$ as below:

$$
\begin{aligned}
&\sigma_{l'} + \gamma\sigma_v - \sigma_{t_m} < \sigma_{l'} + \sigma_v - \sigma_{t_m} \\
\Longrightarrow\ &\sigma_p(y_R^*) - y_R^*\left(\sigma_{l'} + \gamma\sigma_v - \sigma_{t_m}\right) > \sigma_p(y_F^*) - y_F^*\left(\sigma_{l'} + \sigma_v - \sigma_{t_m}\right) \\
\Longrightarrow\ &-\sigma_p(y_R^*) + y_R^*\left(\sigma_{l'} + \gamma\sigma_v - \sigma_{t_m}\right) - \sigma_{l'} < -\sigma_p(y_F^*) + y_F^*\left(\sigma_{l'} + \sigma_v - \sigma_{t_m}\right) - \sigma_{l'} \\
\Longrightarrow\ &c_R^* < c_F^*.
\end{aligned}
$$

– Second, we compare c_R^* and c_G^*. Recall the recycling and disposal cost at yield rate y was defined as $\sigma_{g,2}(y) := \sigma_p(y) + (1-y)\sigma_{l'}$ and $y_G^* = \underset{q\in[0,1]}{\text{argmin}}\ \sigma_p(y) + (1-y)\sigma_{l'}$. Assuming interior solutions, we must have $\gamma\sigma_v - \sigma_{t_m} > \frac{\sigma_{g,2}(y_R^*)}{y_R^*}$, because

$$\gamma\sigma_v - \sigma_{t_m} \geq \frac{1}{y_R^*}\sigma_r(c_R^*, y_R^*) \geq \frac{\frac{1}{c_R^*}\sigma_{r,1}(c_R^*) + \sigma_p(y_R^*) + (1-y_R^*)\sigma_{l'}}{y_R^*} > \frac{\sigma_p(y_R^*) + (1-y_R^*)\sigma_{l'}}{y_R^*} = \frac{\sigma_{g,2}(y_R^*)}{y_R^*}$$

As a result, we have the following two cases:

Case 1 : If $\gamma\sigma_v - \sigma_{t_m} > \frac{\sigma_{g,2}(y_R^*) - \sigma_{g,2}(y_G^*) + \sigma_l}{y_R^*}$, then $-\sigma_p(y_R^*) + y_R^*\left(\sigma_{l'} + \gamma\sigma_v - \sigma_{t_m}\right) - \sigma_{l'} > -\sigma_p(y_G^*) + y_G^* \cdot \sigma_{l'} - \sigma_{l'} + \sigma_l$ and $c_R^* > c_G^*$.

Case 2 : If $\gamma\sigma_v - \sigma_{t_m} \leq \frac{\sigma_{g,2}(y_R^*) - \sigma_{g,2}(y_G^*) + \sigma_l}{y_R^*}$, then $c_R^* \leq c_G^*$. Notice that this scenario is only possible when $-\sigma_{g,2}(y_G^*) + \sigma_l > 0$, because otherwise $\gamma\sigma_v - \sigma_{t_m} \leq \frac{\sigma_{g,2}(y_R^*) - \sigma_{g,2}(y_G^*) + \sigma_l}{y_R^*} \leq \frac{\sigma_{g,2}(y_R^*)}{y_R^*}$, which causes a contradiction.

2. When $\gamma > 1$, one can easily check that the first result changes direction to $c_R^* > c_F^*$, but the second result $c_R^* > c_G^*$ remains. Next, we compare c_R^* and c^*. Recall $y_R^* > y^* \equiv y_F^* > y_G^*$ from (A6), and therefore

$$\sigma_{l'} + \gamma\sigma_v - \sigma_{t_m} > \sigma_{l'} + \sigma_v - \sigma_{t_m} \Longleftrightarrow \sigma_p(y_R^*) - y_R^*\left(\sigma_{l'} + \gamma\sigma_v - \sigma_{t_m}\right) < \sigma_p(y^*) - y^*\left(\sigma_{l'} + \sigma_v - \sigma_{t_m}\right).$$

Recall that $y^* = \underset{q\in[0,1]}{\text{argmin}}\ \sigma_p(y) + (1-y)\left(\sigma_{l'} + \sigma_v - \sigma_{t_m}\right)$, thus, $\sigma_p(y_R^*) + (1-y_R^*)\left(\sigma_{l'} + \sigma_v - \sigma_{t_m}\right) > \sigma_p(y^*) + (1-y^*)\left(\sigma_{l'} + \sigma_v - \sigma_{t_m}\right)$, and $\frac{\sigma_p(y_R^*) - \sigma_p(y^*) - (y_R^* - y^*)\left(\sigma_{l'} + \sigma_v - \sigma_{t_m}\right)}{\sigma_v} > 0$. We distinguish the following two cases:

Case 1: When $\gamma > 1 + \frac{\sigma_p(y_R^*) - \sigma_p(y^*) - (y_R^* - y^*)\left(\sigma_{l'} + \sigma_v - \sigma_{t_m}\right)}{\sigma_v} + \frac{\sigma_l}{\sigma_v}$, then $-\sigma_p(y_R^*) + y_R^*\left(\sigma_{l'} + \gamma\sigma_v - \sigma_{t_m}\right) - \sigma_{l'} > -\sigma_p(y^*) + y^*\left(\sigma_{l'} + \sigma_v - \sigma_{t_m}\right) - \sigma_{l'} + \sigma_l$, and $c_R^* > c^*$.

Case 2: When $\gamma \in (1, 1 + \frac{\sigma_p(y_R^*) - \sigma_p(y^*) - (y_R^* - y^*)\left(\sigma_{l'} + \sigma_v - \sigma_{t_m}\right)}{\sigma_v} + \frac{\sigma_l}{\sigma_v}]$, then $-\sigma_p(y_R^*) + y_R^*\left(\sigma_{l'} + \gamma\sigma_v - \sigma_{t_m}\right) - \sigma_{l'} \leq -\sigma_p(y^*) + y^*\left(\sigma_{l'} + \sigma_v - \sigma_{t_m}\right) - \sigma_{l'} + \sigma_l$ and $c_R^* \leq c^*$.

3. When $\gamma = 1$, the recycler's problem and the firm's problem collapse and we have identical results. □

Proofs of Proposition 7 and Theorems 1 to 5 follow from Propositions 2—6 and are omitted.

Lemma A3. *Let $f_1(x), f_2(x)$ be both increasing, strictly convex functions for $x \in [0,1]$. Define $x_1^* = \underset{x\in[0,1]}{argmin}\ f_1(x) - s_1 x$, $x_2^* = \underset{x\in[0,1]}{argmin}\ f_2(x) - s_2 x$. Then, for $\zeta > 0$, the following unique minimizer $x^*(\zeta) = \underset{x\in[0,1]}{argmin}\ f_1(x) - s_1 x + \zeta(f_2(x) - s_2 x)$ satisfies the properties below:*

- $\min\{x_1^*, x_2^*\} \leq x^*(\zeta) \leq \max\{x_1^*, x_2^*\}$.
- *If $x_1^* \lesseqgtr x_2^*$, then $\frac{\partial x^*(\zeta)}{\partial \zeta} \gtreqless 0$, and $\lim_{\zeta \nearrow \infty} x^*(\zeta) = x_2^*$, $\lim_{\zeta \searrow 0} x^*(\zeta) = x_1^*$.*

Proof of Lemma A3: Notice that $f_1(x) - s_1 x + \zeta(f_2(x) - s_2 x) = f_1(x) + \zeta f_2(x) - (s_1 + \zeta s_2)$, where $f_1(x) + \zeta f_2(x)$ is an increasing, strictly convex function for $\zeta > 0$. When $x_1^* = x_2^*$, it is trivial that $x^*(\zeta) = x_1^* = x_2^*$, is independent of ζ, and satisfies both properties. For ease of presentation, we will use x^* below instead of $x^*(\zeta)$ as needed.

When $x_1^* < x_2^*$, we prove the first property. We claim $x^* \leq x_2^*$. When $x_2^* = 1$, our claim holds trivially. When $x_2^* < 1$, suppose $x^* > x_2^*$. Since $f_1(x) - s_1 x$ increases when $x > x_1^*$ due to strict convexity, $f_1(x^*) - s_1 x^* > f_1(x_2^*) - s_1 x_2^*$. Because x_2^* is the unique minimizer of $f_2(x) - s_2 x$, we have $f_2(x^*) - s_2 x^* > f_2(x_2^*) - s_2 x_2^*$. Therefore, $f_1(x^*) - s_1 x^* + \zeta(f_2(x^*) - s_2 x^*) > f_1(x_2^*) - s_1 x_2^* + \zeta(f_2(x_2^*) - s_2 x_2^*)$, which contradicts the definition of x^*. Thus, $x^* \leq x_2^*$, and by symmetry, $x^* \geq x_1^*$. Hence, the first property holds when $x_1^* < x_2^*$; the proof $x_2^* < x_1^*$ follows by symmetry.

Next, we prove the second property. By definition, $x^*(\zeta)$ simultaneously solves equations (A7) and (A8) below, and depends on the parameter ζ.

$$FOC : f_1'(x) - s_1 + \zeta \left(f_2'(x) - s_2 \right) = 0, \tag{A7}$$

$$SOC : f_1''(x) + \zeta f_2''(x) > 0. \tag{A8}$$

When $x_1^* < x_2^*$, we have $x^* \in (x_1^*, x_2^*)$ by the first property. Since $x^* > x_1^*$, we have $f_1'(x^*) - s_1 > 0$ which leads to $f_2'(x^*) - s_2 < 0$ from (A7). By using the formula for derivative of implicit function, (A7) gives $\frac{dx^*}{d\zeta} = -\frac{f_2'(x^*) - s_2}{f_1''(x^*) + \zeta f_2''(x^*)}$, and it then follows from (A7) that $\frac{dx^*}{d\zeta} > 0$. Hence, $\lim_{\zeta \nearrow \infty} x^* = x_2^*$, $\lim_{\zeta \searrow 0} x^* = x_1^*$. Similar analysis can be performed when $x_1^* > x_2^*$. Notice that regardless of whether $x_1^* < x_2^*$ or $x_1^* > x_2^*$, we always have $\lim_{\zeta \nearrow \infty} x^* = x_2^*$ and $\lim_{\zeta \searrow 0} x^* = x_1^*$. □

Proof of Proposition 8: Follows directly from Lemma A3 after replacing $f_1(x)$ by corresponding operational cost functions, and after replacing $f_2(x)$ by corresponding emissions functions.

Appendix B: Derivation of Emission Levels and Costs

Emissions estimates

Unless otherwise specified, we assume the emissions unit is MTCO_2E per short ton, as in the EPA emissions reports. All emissions data (unless otherwise stated) are obtained from EPA emissions reports: EPA Plastics (2015), EPA Paper products (2015), EPA Metals (2015), and EPA Glass (2015).

Emissions from the use of primary materials, e_v

Except for PET and HDPE containers, total EPA emissions for product made of primary materials (third column in Table B1) include emissions related to product manufacturing (fourth column), emissions due to retail transportation (fifth column), in addition to emissions related to the actual acquisition and processing of primary materials (e_v, last column). Thus, to obtain e_v we subtract emissions in the fourth and fifth column from one in the third column, or directly estimate e_v as in tinplate. Note that the emissions in the fourth and fifth columns add up to manufacturing emissions, which we denote by e_m. Since EPA's definitions of products and primary materials slightly differ from one product category to another, further analyses are required to derive estimates for e_v based on EPA emissions reports.

Product	Material	Total EPA emissions for product made of primary materials	Product manufacturing emissions	EPA retail transportation emissions	e_v
PET container	PET resin	2.25	0	0.04	2.21
HDPE container	HDPE resin	1.58	0	0.04	1.54
Office paper	Paper pulp	1	$1 \cdot 0.45$	0.02	0.53
Aluminum cans	Aluminum ingot	11.09	3.61	0.02	7.46
Steel cans	Tinplate	3.66	0.90	0.02	2.74
Glass container	Glass container	0.60	$0.37 \cdot 0.3$	0.03	0.46

Table B1: Emissions from primary materials

PLASTICS: EPA Plastics (2015) states that *"...Due to the large number of end applications for plastics (e.g., bags, bottles and other consumer products) and the lack of data specific to the U.S., EPA models HDPE, LDPE and PET as resin form."* Therefore, we assign the product manufacturing emissions of PET and HDPE containers to be zero as in the fourth column.

PAPER: D'Antonio (2003) estimates that paper production emissions account for 45% of total emissions for paper manufacturing process.

METALS: According to EPA Metals (2015), aluminum cans are made from aluminum ingot with two additional processes: aluminum sheet rolling and aluminum can and lid fabrication. Therefore, for aluminum cans, we consider aluminum ingot as primary material. For steel cans, we assume the primary material is tinplate (tin-coated steel) based on JFE Steel Corporation (2014). We estimate the emissions of tinplate as 2,486 CO_2g/kg or equivalently 2.74 MTCO_2E per short ton based on Figure 5.5 of Beer et al (2003), and further obtain an estimate of 0.90 for steel can manufacturing, which we use in Table to estimate recycling emissions for steel cans.

GLASS: We estimate that the production emissions of glass are approximately 30% of total process energy based on Venditti (2015). We define *Glass container* as a mixture of raw materials for primary input, which we discuss in more details in operational cost analysis for s_v.

Emissions from consumer's transportation for recycling, e_{t_c}

Franklin Associates (2001) estimates that consumer's transportation of PET and HDPE on average requires $(0.81 + 1.24) \div 2 = 1.027$ gallons of gas per 1,000 lbs (see Table 2-7 on p. 29). Using the estimated 18.95 lbs of CO_2E per gallon of gas (EIA 2016), we derive $1.027 \cdot 18.95 = 19.46$ lbs of CO_2E per 1,000 lbs, or 0.018 MTCO_2E per short ton.

We observe that PET, HDPE, aluminum and steel cans are commonly accepted for recycling (with or without deposit) in the U.S. ("CRV" 2014, NAPCOR and APR 2015, "All US Bottle Bills" 2014), whereas office paper can be put in a recycling bin in places such as offices, libraries, and homes. Hence, we estimate $e_{t_c} = 0.018$ for PET, HDPE, aluminum and steel cans, and glass (without differentiation between these materials), and $e_{t_c} = 0$ for office paper.

Emissions from transporting secondary materials to a manufacturing facility, e_{t_m}

Product	Material	EPA total transportation emissions	EPA retail transportation emissions	e_{t_m}
PET container	PET resin	0.21	0.04	0.085
HDPE container	HDPE resin	0.19	0.04	0.075
Office paper	Paper pulp	0	0	0
Aluminum cans	Aluminum ingot	0.04	0.024	0.008
Steel cans	Tinplate	0.32	0.024	0.148
Glass container	Glass container	0.05	0.03	0.010

Table B2: Emissions from Transportation of secondary materials

We estimate that emissions from transportation of secondary materials to the production facility account for 50% of the total transportation emissions of secondary materials excluding retail transportation (i.e., we assume emissions from recycling stations to recycling facilities are approximately the same as emissions from recycling facilities to manufacturing facilities). Our results are given in Table B2.

Emissions from secondary materials and e_r

The secondary materials emission estimates from EPA reports include emissions from production of the final product and transportation of secondary materials from recycling stations to the production facility, in addition to the emissions related to the actual processing of secondary materials. We consider emissions from production of the final product and transportation of secondary materials from the recycling facility to the production facility separately (as e_m and e_{t_m}, resp.), as shown in Table B3. Recall that the unit emissions of secondary materials processing is $\frac{1}{y}e_r(c,y)$, therefore, $e_r = e_r(c,y) = y \cdot$ secondary materials emissions.

Product	Material	Total EPA emissions for product made of secondary materials	Product manufacturing emissions	e_{t_m}	Emissions from recycling operations	Actual yield rate, y	e_r
PET container	PET resin	0.98	0	0.085	0.90	69.7%	0.62
HDPE container	HDPE resin	0.54	0	0.075	0.47	81.8%	0.38
Office paper	Paper pulp	1.33	$1 \cdot 0.45$	0	0.88	60.3%	0.53
Aluminum cans	Aluminum ingot	0.28	0	0.008	0.27	100.0%	0.27
Steel cans	Tinplate	1.82	0.90	0.148	0.78	98.0%	0.76
Glass container	Glass container	0.28	$0.37 \cdot 0.3$	0.01	0.16	85.0%	0.14

Table B3: Emissions from secondary materials and e_r

Emissions from transportation to landfill and landfill, e_l

Emissions for transportation to and from the landfill, denoted as $e_{landfill}$, can be directly obtained from EPA emissions reports.

Operational costs estimates

We now describe the methodology for cost calculations, in USD $ per short ton of materials.

Costs of primary materials, s_v

We first derive cost estimates for the acquisition and manufacturing of primary materials; we will derive secondary materials costs in corresponding months for a fair comparison later.

PLASTICS: We obtain $1,670.00 for PET resin and $1,285.00 for HDPE resin from from “Current Pricing: Commodity TPs” (2016) by averaging over Volume I and II for April 2016.

PAPER: We use a January 2014 cost estimate of $784.92 from “Wood Pulp Monthly Price” (2014) for primary paper pulp. The reason we use January 2014 data is to be consistent with the recycled paper pulp data, for which we could not obtain more recent data than January 2014.

METALS: For aluminum cans, we estimate primary material cost as \$1,473.20 for from "Aluminum Prices" (2016) (March 2016). For steel cans, we use electro-zinc coil cost, \$760.00 from "MEPS steel price" (2016) (April 2016) to approximate its primary material tinplate cost. Tin costs a few times more than zinc based on "Tin Prices" (2016), however, the fraction of tin in tinplate is negligible — estimated as 0.055% by weight (see Chapter 25 in Rollett 2008), we consider \$760.00 is a reasonable approximation for tinplate.

GLASS: First, we estimate glass container cost as \$1,140.67 per ton by considering two products: 12oz beer bottle and 750ml wine bottle. We estimate that a 12oz beer bottle weights 170g and costs \$0.20 (Satran 2014) to \$0.23 (IBISWorld 2017) which translates to \$1,067.29 to \$1,227.37 for an average of \$1,147.33 per ton; and a 750ml wine bottle weights 400g and costs \$0.40 (Hesser 2003) which corresponds to \$1,134.00 per ton. Second, we estimate the cost of raw ingredients \$37.98 (April 2016) using cost estimates of silica (71% of volume, \$7.81, Sandorfi 2006), limestone (14% of volume, \$2.80, "Limestone and Fill Sand Price List" 2016), and soda ash (11% of volume, \$25.85, Bolen 2015). Third, we estimate the transportation cost as \$34.65 by transportation distance (see more details in the next paragraph.) Finally, we can derive the glass container manufacturing cost as \$1,140.67 - \$37.98 - \$34.65 = \$1,068.04, which will use to estimate the recycled glass container cost later.

Product	Material	Total transportation emissions	Retail transportation emissions	Transportation emissions for manufacturing	Transportation distance (miles)	Transportation cost	Acquisition and manufacturing cost	s_v
PET container	PET resin	0.11	0.040	0.035	435	\$63.49	\$1,670.00	\$1,733.49
HDPE container	HDPE resin	0.19	0.040	0.075	932	\$136.05	\$1,285.00	\$1,421.05
Office paper	Paper pulp	0.02	0.020	0	0	\$0.00	\$784.92	\$784.92
Aluminum cans	Aluminum ingot	0.07	0.024	0.023	317	\$46.31	\$1,473.20	\$1,519.51
Steel cans	Tinplate	0.36	0.024	0.168	2,317	\$338.28	\$760.00	\$1,098.28
Glass container	Glass container	0.07	0.030	0.020	237	\$34.65	\$1,106.02	\$1,140.67

Table B4: Virgin Material Manufacturing and Transportation Costs

Next, we estimate the cost of transporting primary materials to the firm by using travel distances based on EPA emissions reports. There are three components of transportation involved in EPA emissions reports for primary materials: transportation of raw ingredients (e.g., derivatives from petroleum and natural gas) to the manufacturer of primary materials (e.g., plastic resins), transportation of primary materials from their manufacturer to the manufacturer of end product (e.g., plastic bottle), and transportation of finished products to the retailer. We assume that the first two types of transportation emissions are approximately the same, while the last type can be explicitly derived from the EPA emissions reports (referred to as "retail transportation"). The abovementioned costs only include the cost for acquisition and manufacturing of primary materials (including transportation of raw ingredients to the primary material manufacturer); therefore, we need to add the costs incurred for transportation of primary materials to the firm. To do this, we estimate emissions for transportation of primary materials to the firm by first subtracting emissions from retail transportation from total transportation-related emissions, and then dividing the resulting number by 2 (because of the assumption that transportation emissions to and from the manufacturer of primary material are approximately the same). We then compute the travel distance by dividing the emissions by the emissions factor (0.00008 MTCO_2E per mile per short ton; see Exhibit 5 in (EPA Plastics 2015). Finally, we derive the transportation costs by multiplying the travel distance by a cost estimate using \$0.146 per mile per short ton (Austin 2015). All results are given in Table B4.

Cost of consumer's transportation for recycling, s_{t_c}

Franklin Associates (2001) shows a comprehensive analysis of different ways in which consumers transport products for recycling, and estimate that it takes on average 1.027 gallons of of gas per 1,000 lbs of PET and HDPE. Using the average gas price of \$2.642 per gallon ("Annual Gasoline Price Outlook" 2015), we estimate consumers' transportation costs as \$5.43 per short ton. By applying arguments similar to those used in the emissions analysis, we assume the same cost for aluminum and steel cans and glass, as these are typically widely accepted for recycling, and zero cost for office paper, which can be put in recycled bins in offices and homes.

Costs of transportation of secondary materials to product manufacturing facility, s_{t_m}

Product	Material	e_{t_m}	Distance (miles)	s_{t_m}
PET container	PET resin	0.085	1,056	\$154.19
HDPE container	HDPE resin	0.075	932	\$136.05
Office paper	Paper pulp	0.000	0	\$0.00
Aluminum cans	Aluminum ingot	0.008	110	\$16.11
Steel cans	Tinplate	0.148	2,041	\$298.01
Glass container	Glass container	0.010	119	\$17.33

Table B5: Cost for Transportation of secondary materials to Product Manufacturing Facility

We first compute the travel distance, by dividing the transportation emissions from Table B3, e_{t_m}, by the emissions factor (0.00008 MTCO_2E), and then derive transportation costs by multiplying the travel distance by cost estimate (\$0.146 per mile per short ton; results are shown in Table B5).

Cost of secondary materials and s_r

PLASTICS: We obtain \$1,140.00 for PET resin and \$1,000.00 for HDPE resin from "Current Pricing: Recycled Plastics" (2016).

PAPER: We estimate the recycled paper pulp cost as \$840.00 for office paper from Venditti (2015) for January 2014 data, as this is the latest data we could obtain.

METALS: For aluminum cans, note that the cost of manufacturing aluminum cans is estimated as \$1,103.20 = \$1,473.20 - \$370.00, where \$1,473.20 is the cost of primary material–i.e., s_v from "Aluminum Prices" (2016) (March 2016)–and \$370.00 is the cost of alumina from Bray (2015). We estimate the cost of recycling operation as 5% of primary materials manufacturing, based on the energy consumption from Environmental Benefits of Recycling (2016), thus $5\% \cdot \$1,103.20 = \$55.16$. We now estimate cost of scrap metal as \$600.00 from "Scrap Metal Prices" (2016) and add a \$55.16 recycling cost to obtain a total of \$655.16 as the secondary materials cost. For steel cans, we obtain the cost of recycled tinplate as \$655 from RIM (2016) for April 2016.

GLASS: We estimate recycled container manufacturing cost, s_r, as \$1,148.04 as a sum of two costs: cost of secondary material, i.e., cullet as \$80 per ton, and cost of glass container manufacturing \$1,068.04 from primary material analysis. According to Janes (2013), it costs between \$70 and \$90 to process a ton of glass, but then it is sold only for about \$10 per ton. It should be noted that this recycling process does not include the actual glass manufacturing process (GPI 2017). Therefore, the market price \$10 here is a subsidized cost and should not be considered an actual cost. For this reason, we use \$80 (as an average of \$70 and \$90) as cullet cost.

Recall that the unit cost of secondary materials is $\frac{1}{y} s_r(c, y)$, therefore, $s_r = s_r(c, y) = y \cdot$ secondary materials cost (except for glass.) Using estimates of secondary materials costs and the actual yield rate, we estimate s_r in Table B6.

Product	Material	Cost of secondary materials	Actual yield rate y	s_r
PET container	PET resin	\$1,140.00	69.7%	\$794.32
HDPE container	HDPE resin	\$1,000.00	81.8%	\$818.00
Office paper	Paper pulp	\$840.00	60.3%	\$506.47
Aluminum cans	Aluminum ingot	\$655.16	100.0%	\$655.16
Steel cans	Tinplate	\$655.00	98.0%	\$641.90
Glass container	Glass container	NA	85.0%	\$1,148.04

Table B6: Cost of secondary materials and s_r for Each Material

Cost of transportation to landfill and landfill, s_l

We denote the costs of transportation to landfill and the actual landfill by $s_{landfill}$, and use A. Goldsmith Resources (2014) to estimate it as the sum of a \$45 transportation cost and a \$45 tipping fee for a total of \$90 as a national average.

Summary

Finally, we provide a comparison of minimum yield rates from the perspective of emissions, operational costs, and aggregate costs, along with the actual yield rates and collection rates.

Product	Material	x_e (emissions)	$x_s(c,y)$ (operational cost)	$x_\sigma(c,y)$ (aggregate cost)	Actual yield rate,	Actual collection rate,
PET container	PET resin	28.29 %	44.94%	42.81%	69.68%[2]	31.00%[2]
HDPE container	HDPE resin	24.44%	57.08%	53.47%	81.80%[2]	33.60%[2]
Office paper	Paper pulp	≥ 0	53.06%	33.65%	60.29% [3]	68.00%[3]
Aluminum cans	Aluminum ingot	3.35%	37.95%	25.82%	100.00%[4]	66.70%[4]
Steel cans	Tinplate	28.48%	69.64%	58.90%	98.00%[4]	70.00%[4]
Glass container	Glass container	25.12%	90.67%	88.15%	85.00%[5]	50.00%[5]

Table B7: Minimum Yield Rates vs. Actual Yield and collection rates

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