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The Role of Carbon Pricing in Promoting Material Recycling: A Model of Multi-Market Interactions

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The Role of Carbon Pricing in Promoting Material Recycling: A Model of Multi-Market Interactions *

Xi Sun †

Abstract

Recycling of raw material can make a significant contribution to achieving climate neutrality by 2050. Carbon pricing can encourage material recycling by making it more competitive with waste incineration and primary material production. However, accounting for the interactions among different markets in a theoretical model, this paper finds that carbon pricing on material manufacturing alone does not necessarily promote material recovery, if the derived demand for material is elastic, the supply of primary materials inelastic, and the emission intensity for recycling relatively high. In contrast, extending the scope of this policy to the waste sector guarantees a positive effect of carbon pricing on material recovery, together with a strengthened effect on emission mitigation. Using a numerical simulation on plastic waste, this paper shows that implementing carbon pricing on both sources is able to save 37% of CO2e emissions, compared to a policy with a limited scope on production saving 10% less. It is important to consider the full range of impacts and interactions when designing climate policy to ensure that it effectively delivers the objectives for both climate mitigation and circular economy. **JEL classification** D62, H23, Q53

Keywords— carbon pricing, recycling, material production process emission, incineration, material efficiency.

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1 Introduction

Material recycling reduces global industrial emissions from fossil resource extraction, primary manufacturing processes, and end-of-life waste treatments like landfilling and incineration (CIEL, 2019). In 2019, the global industrial sector emitted a total amount of 12.3 Gt greenhouse gases (GHGs), with electricity and heat allocated to the consuming sectors. Of these emissions, 70% were generated by only three materials: steel, cement, and chemicals (IEA, 2021). Steel recycling can save up to 75% emissions compared to primary steel production, while cementitious materials recovered from construction waste can replace up to 30% limestone with current best practices. Additionally, 50 - 70% of plastic packaging waste can be mechanically recycled, which avoids emissions from steam cracking and waste incineration (Chiappinelli et al., 2021). Overall, a more circular economy could result in a 56% reduction in emissions from heavy industries in Europe alone, the majority of which would be achieved through raw material recirculation (Material Economics, 2018).

Economic theory rationalizes the internalization of environmental costs to economic activities through a Pigouvian policy, when the practitioners ignore the impact of their activities on other people in their decision-making. One example of such a policy is carbon pricing, which can be achieved through a carbon tax or an emission trading system. The policy practice of the European Union Emissions Trading System (EU ETS) incentivizes emission reduction or trading by requiring operators of energy-intensive manufacturing facilities to surrender a certain number of emission allowances at the end of each year. However, concerns about carbon leakage, or the possibility that operators may relocate to jurisdictions without such regulations, has led to the allocation of free allowances to manufacturers of industrial raw materials, including steel, cement, and chemicals.

Although the current policy still grants a significant portion of free allowances (See Commission Delegated Decision (EU) 2019/708), it also exempts emissions from municipal and hazardous waste incineration (See Directive 2003/87/EC). While only a few EU member states chose to follow the rules of the EU ETS for domestic incineration plants, there have been increasing policy efforts in recent years to regulate waste incineration in several European countries. For example, since 2020, incineration plant operators in Sweden have been required to pay a waste incineration tax on the wastes that are brought into their facilities. This tax is intended to complement the price signal from the EU ETS, which has been significantly lower than the Swedish carbon tax (Sweden Ministry of Finance, 2016). Two years later, in 2022, a similar waste incineration tax went into effect in Norway. In Germany, the waste incineration industry, which, as of 2023, is exempt from EU ETS liabilities, will be required to pay a national carbon price starting in 2024. This charge is being implemented through an amended Fuel Emissions Trading Act, which extends CO2 pricing to all fossil fuel emissions and includes waste incineration, with a one-year delay for the latter (the amended act went into effect in November 2022).

Most effectiveness studies on carbon pricing focus on the supply-side measures, including technological change and fuel switching (see Teixidó et al. (2019) for a review). However, far less attention is given to the demand side and how a more circular economy could reduce emissions through better use and reuse of existing materials (Material Economics, 2018). This paper aims to develop an analytical framework to answer the question: does a Pigouvian policy on GHG emissions from manufacturing and waste treatment motivate material recycling?

In the research field of waste policy, early studies show that a first-best Pigouvian policy, which accounts for the social cost of waste disposal through a garbage collection tax, can have a positive impact on waste reduction and recycling (Fullerton and Kinnaman, 1995; Palmer et al., 1997). A consumption tax, or an advanced disposal fee (ADF), charged on a product at the point of purchase, internalizes the environmental cost for consumers and motivates waste reduction at the source. A recycling subsidy, on the other hand, encourages this alternative waste treatment method (Palmer et al., 1997). Acuff and Kaffine (2013) also argues that waste policies that lead to the largest source reduction enable the highest level of upstream emission-saving. Furthermore, when an environmental externality is associated with the use of a material input, the theoretical alternative to a Pigouvian policy on the externality itself is to tax the material input and subsidize other, environment-neutral inputs such as labor (Walls and Palmer, 2001).

In contrast, the empirical effectiveness of an incineration tax is less straightforward. For instance, Sweden charged an incineration tax between 2006 and 2010 (Sweden Ministry of Finance, 2006). The tax, which was meant to increase material recycling and create a level playing field for energy generators, consisted of both an energy tax component levied at SEK 150 per tonne (16.5€/tonne) of fossil carbon and a carbon dioxide tax component levied at SEK 3374 per tonne (371€/tonne) of fossil carbon, based on the Swedish average fossil carbon content of the incinerated waste. Tax exemptions were granted to combined heat-and-power (CHP) production plants on the basis of electricity efficiency. However, the main effect of this incineration tax was an increase in CHP production from high calorific municipal wastes, while no significant improvement in material recycling was observed (Sahlin et al., 2007; Ekvall et al., 2014). After the incineration tax was abolished in 2010, waste incineration plants in Sweden became more cost-efficient and expanded to a capacity larger than domestic waste supply. As a result, one third of the heat produced in Swedish incineration plants is now produced from municipal waste collected abroad (Sweden Ministry of Finance, 2016). The large incineration capacity also discourages investment in recycling, due to the high opportunity cost of leaving the excess incineration capacity idle (Yamamoto and Kinnaman, 2022).

However, these studies do not consider the interaction in different markets. For example, Sahlin et al. (2007) estimate the net marginal cost for alternative waste treatment by adding up the costs of each actor involved in the system, ignoring the response of households to changes in waste fees, the response of product designers to variations in input prices, and the response of waste treatment facilities to price changes in both waste and its material or energy outputs.

First, pricing unsorted waste can drive significant reduction in waste generated by households and overall waste management cost (Valente, 2023; Kinnaman, 2006). Using a large panel dataset on waste generation and price adoption in Italian municipalities, Valente (2023) finds that the price elasticity of waste treatment demand is high when the price of waste is high and low when the price is low. On average, this study estimates a significant policy causal effect on unsorted waste reduction, driven by a 32% increase in recycling and a 5% reduction in waste. This estimate supports the review of Kinnaman (2006), who finds that a $1 charge per garbage bag can lead to a roughly 40% reduction in waste. Both studies also confirm an income effect, with low-income municipalities being more sensitive to waste prices than high-income municipalities.

Second, a Pigovian policy that internalizes the environmental costs associated with material manufacturing and waste disposal can influence the competitiveness of recycling businesses in both raw material and waste treatment markets. When the quality of recycled material is comparable to its primary counterpart, a cheaper price can drive material substitution in the production plans of producers (Demets et al., 2021). This competitiveness can be achieved through a positive price shock of fossil fuels, which are the main input for primary materials, or an effective carbon price. Meanwhile, energy price variations can also lead to delayed price changes for recycled materials. For example, as plastics are made from oil by-products, falling oil prices during the COVID-19 pandemic increased the cost of recycling by decreasing prices in recycled markets and reducing the cost of producing virgin resin (Issifu et al., 2021).

However, in the waste treatment market, a higher energy price will favor waste incineration plants, a significant portion of whose revenue comes from the sale of electricity. In the United Kingdom, approximately 70–80% of revenue for an incineration plant comes from the tipping fee and 20–30% is generated from the sale of electricity, while in China, income from electricity sales accounts for 70–85% of total incomes (Zhao et al., 2016). Therefore, a positive expectation of future revenue encourages investment in energy recovery from waste. Competition for waste feedstock and the opportunity costs associated with excess incineration capacity can further limit the growth potential of recycling and its opportunity to lower costs through economies of scale.[1]

In summary, understanding the full impact of carbon pricing on material recycling requires considering all three markets: the consumption good market, the material market, and the waste treatment market. This paper aims to fill this gap by constructing a three-market framework based on the multiple stakeholder analyses of Sahlin et al. (2007) and Ekvall et al. (2014). This allows us to examine the market mechanisms that drive the economic incentives of different stakeholders, providing a holistic foundation for analyzing the policy effect when an incineration tax is introduced. In the following sections, we propose an analytical framework that presents the economic incentives of the main stakeholders, conceptualize a theoretical model and derive the social optimal use of primary versus recycled materials, and compare the effects of carbon pricing with different scopes. Then, we present a simple simulation to compare the impact of a production process-oriented climate policy with a life-cycle-oriented policy on recycling, as well as a numerical example based on the consumption of polyethylenterephthalat (PET), one of the most widely used polymer in plastic packaging products. Finally, we summarize and discuss the key findings.

2 Conceptual Framework

In this section, we propose a conceptual framework (see Figure 1) to characterize analyze individual incentives of three stakeholders: waste manager, producer, and consumer, on which basis we can analyze the impact of an incineration tax. The next section derives the key parameters for this policy effect.

To begin with, the demand for waste treatment is determined by the amount of waste that is produced by waste generators (such as households, municipalities, and businesses). The supply of waste treatment, on the other hand, is determined by the capacity of treatment facilities, which is often fixed in the short term. A waste fee (P_w) can help balance the demand and supply of waste treatment by encouraging waste generators to reduce their waste production and by influencing the investment and operating decisions of waste treatment facilities. By setting the fee at an appropriate level, waste generators are incentivized to minimize their waste and treatment facilities are encouraged to optimize their operations and possibly expand their capacity.

The waste fee, often collected by a public waste manager, helps to offset the gate fees (P_i) that the waste manager must pay for downstream waste treatment. While the reception of waste at the gate makes up a significant portion of income for a waste treatment plant, the sale of outputs resulting from waste treatment, such as heat and electricity from energy recovery plants and sorted materials from recycling facilities, is becoming increasingly important for the profitability of these facilities. For instance, the Berliner Stadtsreinigung (BSR) reported a 26.3% increase in revenue from the sale of recyclable materials in 2021, which was the largest increase among all revenue

[1]Excess capacity of an incineration plant requires furnaces to run intermittently, which adds additional processes for storing waste temporarily and periodically igniting and extinguishing furnaces. Intermittence also complicates the process of removing pollutants and dioxins from the air stream (because burn temperatures must repeatedly pass through the dangerous 200 to 600-degree zone), increasing costs (Yamamoto and Kinnaman, 2022).

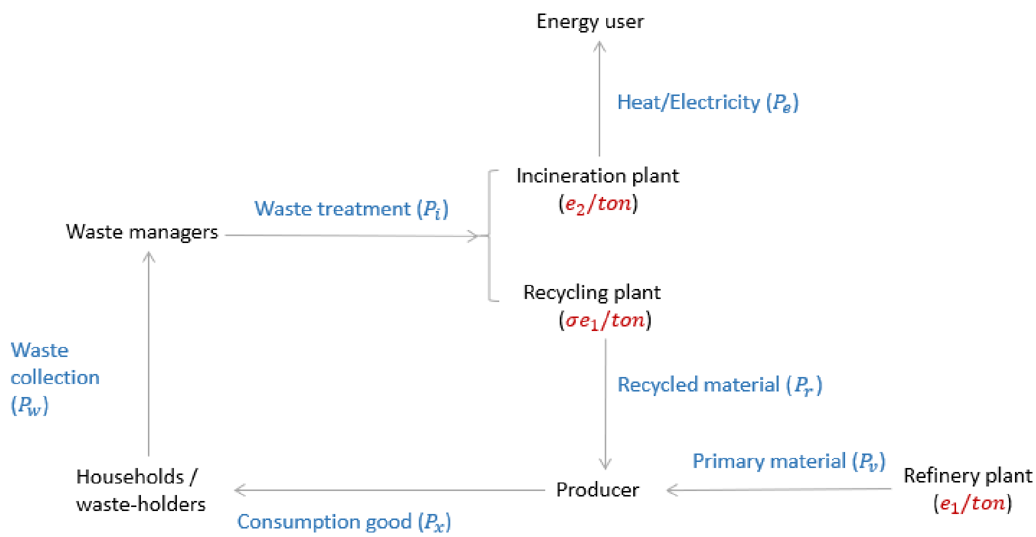


Figure 1: Analytical framework for key markets involved in a circular economy

sources (BSR, 2021). In countries like Norway and Sweden, some incineration plants generate as much or even more income from energy sales as they do from waste reception (Norwegian Environment Agency, 2012).

The prices of outputs from waste treatment processes, such as electricity (P_e) and sorted materials (P_r), are often determined by market forces if no price regulation is in place. Therefore, these prices are sensitive to various market conditions and influencing factors. For instance, the price of recycled material can be constrained by limited demand due to concerns about material quality and the investment required to adopt the material in production processes (Jones, 2021; Weerdt et al., 2022). Nevertheless, to the extent that the recycled material and primary material are substitutes, the price of recycled material (P_r) can also be influenced by the price of its primary counterpart (P_v). For example, the price of recycled plastics may increase after a positive shock to the price of crude petroleum (Weinhagen, 2006; Issifu et al., 2021).

Last, but not the least, demand elasticity and material substitution, among other factors, can influence the ability of cost passing through to consumers at a higher consumption price (P_x). If demand is elastic to price variations, meaning that the consumer has access to other consumption options or can avoid the consumption without significant welfare impairment, the producer will not be able to raise the consumption price to fully pass on the additional cost. Other factors, including imperfect competition and regulatory imbalance among jurisdictions, also influence the cost pass-through ability of producers (Neuhoff and Ritz, 2019). For instance, electricity market is estimated to have almost complete pass-through whereas the average pass-through rate across the US manufacturing sector is estimated at 70% (Ganapati et al., 2020; Fabra and Reguant, 2014). In addition, when alternative material inputs gain cost efficiency against the regulated primary material, producers will be able to minimize their cost burden from carbon pricing.

Now we use this framework to derive the theoretical impacts of a positive material price shock, due to e.g. supply shortage, on the upstream waste treatment and downstream commodity markets. In the long-run, increases in material prices stimulate the investment and expansion of recycling facilities. As a result, total waste treatment capacity increases, shifting the supply curve in the waste treatment market (See panel A in Figure 2) to the right. A lower waste fee P_w^2 and higher treatment quantity w^2 will be reached at the new long-run equilibrium, provided that the treatment cost per unit of waste (P_i) is positively correlated with the waste fee (P_w) paid by waste-holders. Under the condition that the reduced waste fee relaxes the budget constraint of waste-holders, its income effect would allow the households to relocate more resources for consumption. In this way, this income effect, regardless of its absolute scale, would drive up commodity demand, which leads to higher prices and quantities of consumption goods at the equilibrium (see panel B of Figure 2).

While a higher level of material evaluation motivates lower waste fees and relaxed consumption budgets, the downstream production confronts higher production costs provided that cost minimization via material substitution is negligible. Such a supply shock inflates the price of the consumption good, offsetting or even overrunning the previous budget relaxation. The distribution of this production cost increase across both consumers and producers depends on both demand and supply conditions. In case of a production function that performs decreasing return to scale, that is, the producing cost for the last unit product increases with total output level, the rise in material cost can be only partially passed on to the consumption price (see panel C of Figure 2). In contrast, if a constant return to scale is proven to be the right assumption, complete cost pass-through will become possible. The same applies to the condition of demand elasticity, as discussed above.

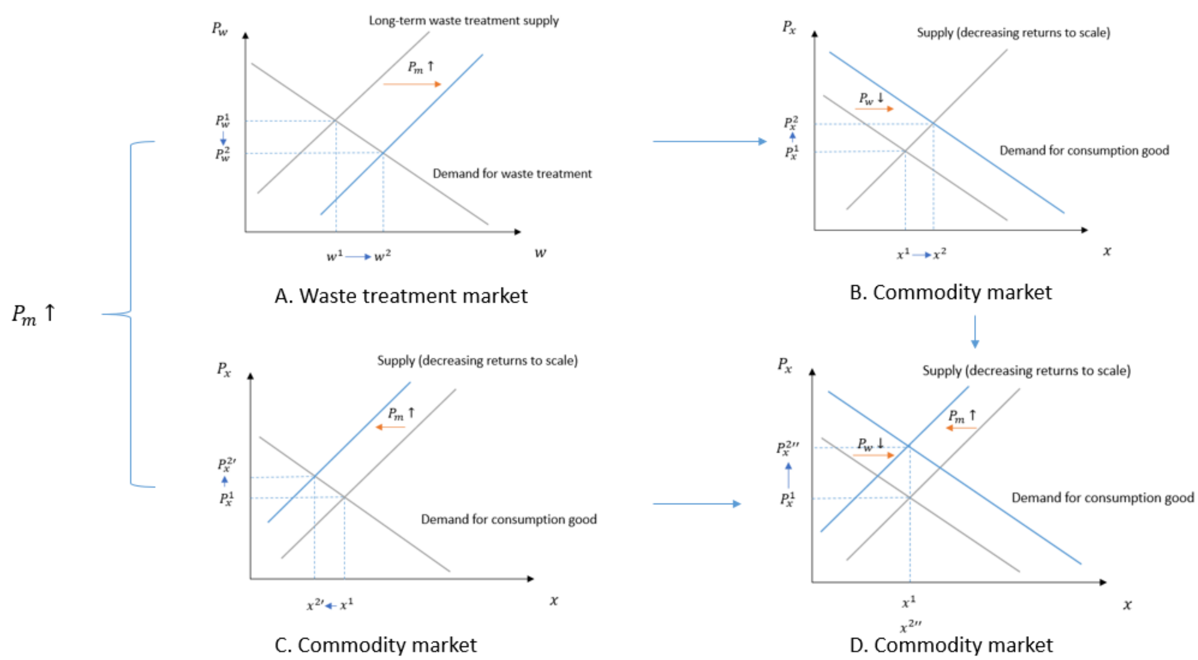


Figure 2: Influence of a material price shock on the three markets.

It is important to note that this analysis assumes perfect substitution between the two types of materials. Under this assumption, the prices for both materials will rise to the same level, which will encourage an increase in the supply of recycled materials. The balance between demand and supply will lead to increased use of recycled materials for production at the new long-run equilibrium.

Ultimately, the net effect on consumption quantity and price will depend on whether the price shock has a greater impact on production costs or the consumer budget (see panel D of Figure 2). If the cost of waste payment is a small fraction of the budget, the increase in production costs will dominate, resulting in an overall reduction in total consumption.

Overall, the environmental impact of such a material price shock is determined by the amount of recycled material generated to replace primary material extraction and manufacturing as well as the net change in total consumption, all alongside the avoidance of waste incineration. Energy-related emissions from production processes (e_1/ton) originate from the use of fossil-based energy and electricity to convert fossil resources into primary materials. The use of recycled material avoids these emissions to the extent allowed by its own emission intensity ($\sigma e_1/ton$). The reduction of consumption avoids emissions from both processes. Moreover, to the extent that a recycling process gains cost efficiency against waste incineration, environmental impact from the latter will also be avoided (e_2/ton).

In this paper, we primarily examine greenhouse gas emissions, but it is important to recognize that economic activities have a range of environmental externalities beyond just greenhouse gas emissions. For example, extracting fossil resources can lead to land use changes and environmental leakage, which can harm biodiversity, human health, and the sustainability of local communities. Manufacturing and waste treatment processes can also produce harmful byproducts that can be difficult to contain and may be released into the atmosphere, water bodies, or soil. Additionally, the transportation and use of products and the mismanagement of waste can result in the release of non-degradable, harmful materials into the environment, causing damage to natural systems and human health. While our focus in this paper is on greenhouse gas emissions, it is possible to incorporate these other types of environmental externalities into our framework by clearly defining them.

Up to this point, we have not considered the mass balance constraint for total consumption and waste generation, which states that an increase in waste treatment at market equilibrium does not necessarily have to be matched by an equal increase in total consumption to provide material for the treatment. We incorporate this constraint in our theoretical model in the next section.

3 Model

In this section, we develop a representative agent model to characterize the economic relations in the circular framework presented in Figure 1. We apply several mass balance conditions for a material in different forms: consumption good x, waste w, primary material v, recycled material r, fossil feedstock f, and sorted waste x_r, as following:

$$w = x,\ x = v + r,\ v = f,\ r = x_r.$$

The latter three equations indicate two assumptions. First, we assume constant returns for the production of x using v and r, the production of v using f, and the production of r using x_r. Second, primary material and recycled material are assumed perfect substitutes. These assumptions are restrictive, but useful to reveal the basic trade-offs between the two types of materials and between material supply and demand, enabled by the cost patterns of fossil extraction and waste processing.

In particular, the cost function for waste processing (including waste collection, transport, and sorting; collectively referred to as "waste sorting") is assumed to be convex. This is to reflect the fact that easy-to-collect wastes are the first to be sourced for recycling, which are usually of good quality and easy to sort. When the uptake of sorted wastes goes up, especially when it approaches the limit of total waste provision, the cost for waste sorting will quickly grow given technology limits. We further assume the waste manager finances for this cost in two channels: a waste fee charged on waste generators (final consumers in this model) and revenue from selling sorted waste to a recycler.

Therefore, the waste manager aims to maximize the profit function: $\Pi_w = P_w w + P_{x_r} x_r - k(x_r)$, where $k(x_r)$ stands for waste sorting cost with $k'(\cdot) > 0$, $k''(\cdot) \geq 0$ and $k(0) = 0$. Similarly, we assume decreasing returns to fossil extraction: $\Pi_F = P_f f - c(f)$, where $c(f)$ stands for the fossil resource extraction cost with $c'(\cdot) > 0$, $c''(\cdot) \geq 0$ and $c(0) = 0$. Consumer surplus is defined by the inverse demand function, following from quasilinear utility from the consumption good:

$$CS = \int_0^x P(z)\,dz - P_x x - P_w w.$$

3.1 Social optimum

The negative environmental impact of GHG emission is quantified through a convex damage function $d(E)$. The total emission E comprises primary production process emission of e_1 per unit of primary material, recycling process emission of $\sigma \cdot e_1$ per unit of recycled material, with factor σ denoting production emission saving potential of recycling, as well as end-of-life disposal emission of e_2 for each unit of incinerated material.

After inserting the mass balance conditions, the social planner faces a subsequent objective function and optimizes it by choosing the level of fossil resource extraction and waste sorting:

$$\begin{aligned} SW &= CS + \Pi_X + \Pi_R + \Pi_V + \Pi_F + \Pi_W - d(E) \\ &= \int_0^{f+x_r} P(z)\,dz - k(x_r) - c(f) - d(f \cdot (e_1 + e_2) + x_r \cdot \sigma e_1)) \end{aligned} \tag{1}$$

The interior solution of social welfare maximization is determined by the first-order conditions:

$$\frac{\partial SW}{\partial f} = 0 \;\Leftrightarrow\; p = c'(f) + d'(E) \cdot (e_1 + e_2) \tag{2}$$

$$\frac{\partial SW}{\partial x_r} = 0 \;\Leftrightarrow\; p = k'(x_r) + d'(E) \cdot \sigma e_1 \tag{3}$$

$$p = P(f + x_r) \tag{4}$$

Lemma (Condition for Zero Fossil Extraction)**.** A corner solution of zero fossil extraction exist if and only if $c'(0) + d'(\sigma e_1 \hat{x_r})[e_1(1-\sigma) + e_2] \geq k'(\hat{x_r})$, with $\hat{x_r}$ the optimal use of sorted waste to satisfy consumption needs, i.e. $P(\hat{x_r}) = k'(\hat{x_r}) + d'(\sigma e_1 \hat{x_r})\sigma e_1$.

Proof. As $c''(\cdot) \geq 0$ and $d''(\cdot) \geq 0$, assume a marginal increase in f by ϵ, we get $c'(\epsilon) + d'(\epsilon(e_1 + e_2) + \sigma e_1 \hat{x_r})[e_1(1-\sigma) + e_2] > k'(\hat{x_r})$. In the meantime as $P'(\cdot) < 0$, $P(\epsilon + \hat{x_r}) < k'(\hat{x_r}) + d'(\epsilon(e_1 + e_2) + \sigma e_1 \hat{x_r})\sigma e_1$. Hence, it is not efficiency-improving to increase fossil extraction above zero. □

This Lemma says that if the emission-saving potential of a recycling process is sufficiently large and the marginal cost for waste sorting sufficiently low, it will be desirable for an economy to extract zero new fossil resource. In this case, consumption will be fully supplied by recycled materials generated from the sorted wastes of the last period. However, if waste sorting costs increases to infinity as the processed waste approaches total consumption, this corner solution will fail. This is particularly the case in the long-run. Here, our analysis proceeds with a focus on the short-run, where waste could also be sourced from accumulated waste stock ('waste mining'). Moreover, the waste to be treated within one period is usually a small fraction of the final products generated in that period due to product durability. Under these conditions, it would be safe to assume a bounded cost function for waste sorting.

3.2 Carbon pricing with different regulatory scopes

Now we derive the effect of a first-best policy - carbon pricing - and analyze the difference if this policy only regulates emissions from production processes or if it is extended to end-of-life treatment processes as well. To begin with, if a unit emission price τ is only charged on production processes, both the primary material manufacturer and the recycler would internalize the social cost of carbon into their production plan, respectively subject to market prices $P_v = P_f + \tau e_1$ and $P_r = P_{x_r} + \tau\sigma e_1$. P_f stands for the market price for fossil resources and P_{x_r} for sorted wastes.

In this case, the market equilibrium quantities for both types of materials would derive from the following first-order conditions:

$$c'(f) + \tau e_1 = k'(x_r) + \tau\sigma e_1 \tag{5}$$

$$c'(f) + \tau e_1 = P(f + x_r) \tag{6}$$

If, on the other hand, this pricing scheme is extended to involve the waste treatment sector, the social cost of carbon would also be internalized to the decision of the waste manager on the recycling level. To allow for an explicit analysis on the effect of this extension, we denote the second part of unit emission charge as t. The incentive of the waste manager is fully characterized by:

$$\begin{aligned}
&\max_{w,x_r} \ \Pi_W = P_w w + P_{x_r} x_r - k(x_r) - t \cdot e_2(w - x_r) \\
&FOC: \ \ P_w = t \cdot e_2 \\
&\qquad\quad\ P_{x_r} = k'(x_r) - t \cdot e_2
\end{aligned}$$

With this extension, the first-order conditions for market equilibrium solutions become:

$$c'(f) + \tau e_1 = k'(x_r) + \tau\sigma e_1 - te_2 \tag{7}$$

$$c'(f) + \tau e_1 = P(f + x_r) - te_2 \tag{8}$$

On the basis of a comparative static analysis, we now can derive the market equilibrium quantities for primary vs. recycled materials under the difference scopes of carbon pricing. To begin with, by using the implicit function theorem on equations (5) and (6), we obtain the effect of carbon pricing only levied on production process emissions (see Appendix for derivation):

$$\frac{\partial f}{\partial \tau} = \frac{e_1 - (1-\sigma)e_1 \frac{P'(x)}{k''(x_r)}}{P'(x) \cdot \frac{c''(f)}{k''(x_r)} + P'(x) - c''(f)} \tag{9}$$

$$\frac{\partial x_r}{\partial \tau} = \frac{\sigma e_1 + (1-\sigma)e_1 \frac{P'(x)}{c''(f)}}{P'(x) \cdot \frac{k''(x_r)}{c''(f)} + P'(x) - k''(x_r)} \tag{10}$$

Given $P'(\cdot) < 0$, $c''(\cdot) > 0$, and $k''(\cdot) > 0$, we can easily see that the RHS of equation (9) is negative, whereas the sign of equation (10) is uncertain. In fact, the sign of the latter depends on three components: the emission factor of recycling σ, demand elasticity $P'(x)$, and supply elasticity of fossil extraction $c''(f)$. This result is in line with intuition. On the one hand, an emission charge will reduce the uptake of primary material, thus the use and extraction of fossil feedstock. On the other hand, this emission charge leads to an increase in the uptake of recycled material, only if (1) the emission from recycling is significantly less than its counterpart, (2) consumption demand is inelastic, and (3) the marginal cost of fossil feedstock is relatively flat - supply being elastic. This result is graphically illustrated with a simple numerical simulation in the next section.

The overall influence of a production process emission tax τ on total emission is:

$$\frac{\partial E}{\partial \tau} = e_1 \cdot \frac{\partial f}{\partial \tau} + \sigma e_1 \cdot \frac{\partial x_r}{\partial \tau} - \frac{\partial x_r}{\partial \tau} e_2 + \frac{f}{w}(\frac{\partial f}{\partial \tau} + \frac{\partial x_r}{\partial \tau})e_2$$

The first and second terms reflect the direct impact of such an emission charge on process emission, with the last two terms characterizing the indirect impact of this charge on end-of-life emission. Specifically, the third term states that, while this emission charge is targeted at production process emission, end-of-life emission saving is also possible if the tax induces a higher demand for recycled material ($\frac{\partial x_r}{\partial \tau} > 0$). Implicitly, this gives the waste manager an incentive (from higher price of waste P_{x_r}) to process more waste for recycling. However, if the tax fails to encourage material substitution due to elastic demand or inelastic supply of primary material ($\frac{\partial x_r}{\partial \tau} \leq 0$), the demand for recycled material continues to be weak and the waste manager lacks incentive to change their waste treatment method. The fourth term states that this restricted material emission charge could contribute to overall waste reduction, hence also reducing end-of-life emissions ($\frac{\partial f}{\partial \tau} + \frac{\partial x_r}{\partial \tau} < 0$).[2]

Hence, due to the omission of end-of-life emission, this carbon pricing policy cannot fully restore the social optimal use of materials with insufficient level of recycling.

Now, when the carbon pricing scheme is extended to the waste sector, the comparative static analysis follows from equations (7) and (8):

$$\frac{\partial f}{\partial t} = \frac{e_2 \frac{1}{P'(x)} - e_2 \frac{1}{k''(x_r)}}{\frac{c''(f)}{k''(x_r)} + 1 - \frac{c''(f)}{P'(x)}} \tag{11}$$

$$\frac{\partial x_r}{\partial t} = \frac{e_2 \frac{1}{k''(x_r)}}{\frac{c''(f)}{k''(x_r)} + 1 - \frac{c''(f)}{P'(x)}} \tag{12}$$

The first equation indicates that this additional emission charge further discourages the use of fossil feedstock. First, this tax directly nudges less consumption, as we model the waste fee to be burdened on product consumption, with the scale of this effect hinging on demand elasticity $P'(x)$. Second, higher incineration costs reward the alternative waste treatment method - recycling in this model. As a result, the waste manager gains an incentive to source for more sorted wastes, which, in the long-run, will equilibrium substitute an equal amount of fossil feedstock ($e_2 \frac{1}{k''(x_r)}$). This extension then fully restores the first-best result.

4 Numerical Illustration

4.1 A Simple Simulation

Now we illustrate the role of a carbon pricing policy on the use of primary vs. recycled materials with a numerical simulation. To allow for the derivation of closed form solutions, we adopt simple functional specifications as the following:

[2]The collective effect of a process emission charge on end-of-life emissions can be negative even if the third term is positive (i.e. $\frac{\partial x_r}{\partial \tau} < 0$), as this sign depends on $\frac{f}{w} e_1 k''(x_r) + \frac{x_r}{w} \sigma e_1 c''(f) + \frac{x_r - f}{w}(1-\sigma) e_1 P'(x)$, which indicates a negative sign for the collective effect, even if the conditions for $\frac{\partial x_r}{\partial \tau} < 0$ remain.

$$\begin{aligned}
&Inverse\ demand\ function: \ P(x) = A - B \cdot x \\
&Cost\ function\ for\ fossil\ extraction: \ c(f) = 0.5 \cdot c_f \cdot f^2, \ c_f > 0 \\
&Cost\ function\ for\ waste\ sorting: \ k(x_r) = 0.5 \cdot c_{x_r} \cdot {x_r}^2, \ c_{x_r} > 0 \\
&Damage\ function\ of\ emission: \ d(E) = 0.5 \cdot d \cdot E^2, \ d > 0
\end{aligned}$$

Parameters in these functions describe demand and supply features in a stylized way. In the linear demand function, parameter B characterizes how elastic the demand responds to price changes, with a higher value of B indicating more inelastic demand.[3] Parameters c_f and c_{x_r} respectively describe how fast the marginal costs for primary production and waste sorting increase, which also feature the supply elasticities of outputs from these two processes. Finally, the marginal social cost of emission is captured by parameter d and total emissions by capital E, the same as above.

The optimal consumption levels for both primary and recycled materials, as well as the optimal carbon pricing for both manufacturing and waste disposal that allows for these consumption levels, are calculated as the following:

$$\begin{aligned}
x_r^* &= \frac{A[c_f + d(e_1+e_2)(e_1+e_2-\sigma e_1)]}{c_{x_r}c_f + d[c_{x_r}(e_1+e_2)^2 + c_f(\sigma e_1)^2] + B[c_{x_r} + c_f + d(e_1+e_2-\sigma e_1)^2]} \\
f^* &= \frac{A[c_{x_r} + d\sigma e_1(\sigma e_1 - e_1 - e_2)]}{c_{x_r}c_f + d[c_{x_r}(e_1+e_2)^2 + c_f(\sigma e_1)^2] + B[c_{x_r} + c_f + d(e_1+e_2-\sigma e_1)^2]} \\
t^* = \tau^* &= \frac{dA[c_{x_r}(e_1+e_2) + c_f\sigma e_1]}{c_{x_r}c_f + d[c_{x_r}(e_1+e_2)^2 + c_f(\sigma e_1)^2] + B[c_{x_r} + c_f + d(e_1+e_2-\sigma e_1)^2]}.
\end{aligned}$$

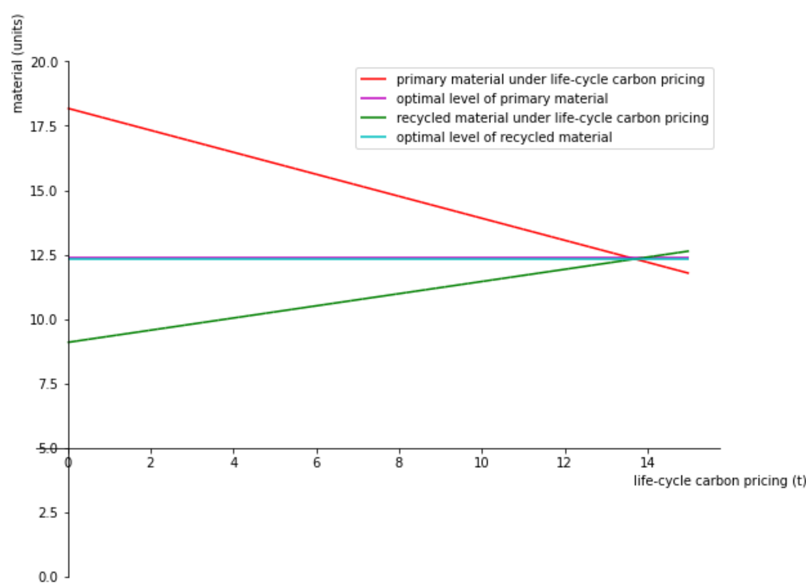
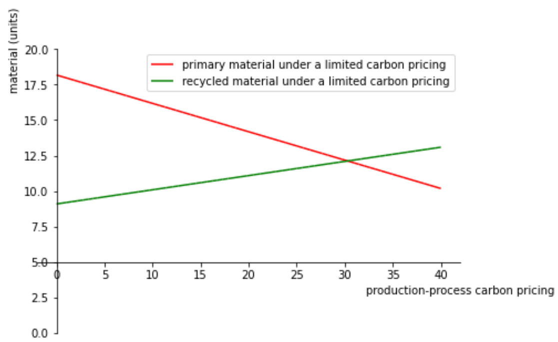


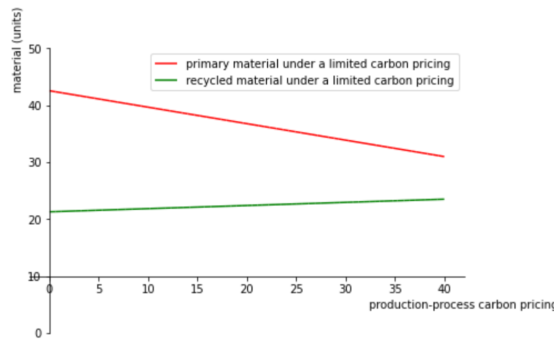
Figure 3: Material use impact of a carbon pricing policy on both production and incineration.

[3] Demand elasticity at a given combination of price and quantity (P_1, x_1): $-\frac{1}{B} \cdot \frac{P_1}{x_1}$, $B > 0$.

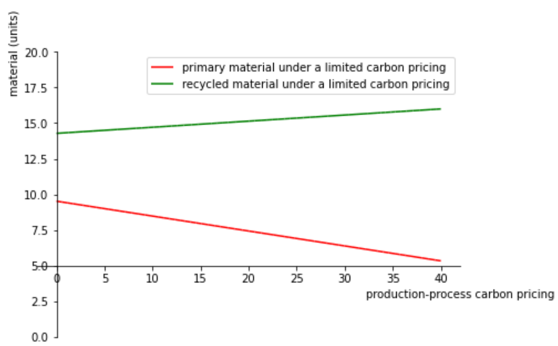
Figure 3 illustrates how a more complete policy coverage of carbon pricing changes the amount of the materials used in an economy.



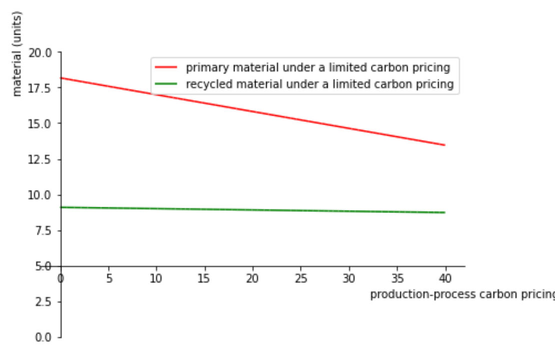
(a) $B = 3, c_f = 1, \sigma = 0.2$



(b) Elastic demand: $B = 0.9$



(c) Costly primary production: $c_f = 3$



(d) Emission-intensive recycling: $\sigma = 0.8$

Figure 4: Policy scope matters: a sensitivity check on three structural parameters.

If emissions from waste disposal are exempted from carbon pricing, a much higher level of the charge is required to meet the same results of material use as under a carbon pricing policy on both production and incineration (Figure 4a). The limited policy scope would also lead to an ambiguous effect on material recycling, depending on the values of the structural parameters that shape material demand and supply. In particular, this emission charge can stimulate material substitution only when the parameters satisfy certain conditions: relatively inelastic demand B, relatively elastic supply of primary resources c_f, and low emission intensity σ for the recycling process.

This result can be intuitively interpreted as following. First, the levy of an emission charge on material increases the average price of products made out of it, thereby reducing the total derived demand for materials. If demand is elastic, the demand for both types of material will decline; however, if demand is inelastic, a higher amount of the material less affected by the emission charge will be needed to satisfy the demand and substitute the more emission intensive type. Similarly, if the supply of the primary resource is completely inelastic (here, equivalent to a high level of marginal cost to generate f, which renders the relative scale of the emission charge small), the impact of the emission charge will be absorbed by the primary material provider, leaving the market price of materials untouched and no further market incentive for material recovery. Lastly, the more emission efficient the recycling process is, the more it will benefit from the emission charge that is based on the emission units for the generation of one unit material.

4.2 An Example: PET

Now we apply this model to a real-life example: the plastic polymer Polyethylenterephthalat (PET), one of the most common synthetic material made from fossil fuels. Accounting for 8.4% of total plastic demand in Europe (Plastics Europe, 2021), PET is a suitable example for closed-loop recycling. For one thing, PET separate collection, which is important to avoid pollution of inputs to the recycling process, has been established in several countries. For instance, 97.5% of PET drinking bottles covered under the German deposit-refund system were sent for mechanical recycling in 2021; out of the recycled PET polymers, 37.7% is currently used to produce new PET bottles, while the theoretical potential for bottle-to-bottle recycling approaches 97% (GVM, 2020). Furthermore, the mechanical and physical properties of PET allow multiple-time recycling and achieving quality criteria sufficient for use in food-contact bottles, if the recycling process is designed properly (Eriksen et al., 2019).

The production of primary PET has a relatively high emission intensity, with an estimated 2.696 tonnes of CO2 equivalent (tCO2e) per tonne of production. In contrast, the mechanical recycling process for PET has a much lower emission intensity, at 0.273 tCO2e per tonne of recycled PET (Dormer et al., 2013). Incinerating one tonne of plastic leads to an additional tonne CO2 emission, after accounting for substituted electricity generation (CIEL, 2019). Overall, these factors highlight the potential of increasing the closed-loop recycling rate for this material to reduce greenhouse gas emissions and contribute to climate neutrality.

We first formulate the assumption on PET demand function, based on an estimated market value of four billion euros in the European single market.[4] Moreover, we assume here a constant elasticity demand function to allow for balanced solutions:

$$P(x) = (\frac{4 * 10^9}{x})^{-\frac{1}{B}}$$

, where $B = -1$ indicates the price elasticity for PET demand.

Consistent with the simulation, we assume that the marginal cost of recycling increases at a rate that is double the marginal cost of primary production. This reflects the fact that higher quality and easily sourced waste is typically processed first, while waste of lower quality becomes more expensive to process as it becomes harder to source:

$$c(f) = 5 * 10^{-5} f^2, k(x_r) = 1 * 10^{-4} x_r^2.$$

Under these assumptions, the total demand for PET polymer without carbon pricing is 7.74 million tonnes, including 5.16 million tonnes from primary production and 2.58 million tonnes from recycling. This process results in a total emission of 20.99 million tCO2e. If an emission charge were to be imposed on all sources in the value chain, the optimal charge would be equal to the marginal social damage caused by this emission. If we assume that this marginal social cost is constant at 100 euros per tonne of emissions, PET consumption would decrease by 23% to a total of 5.98 million tonnes, with 2.99 million tonnes coming from primary production (a 42% reduction) and 2.98 million tonnes coming from recycling (a 15% increase). As a result, overall emissions would be reduced by 37% to 13.27 million tCO2e compared to the unregulated market solution.

If the emission charge were only levied on emissions from production processes, the resulting overall emissions would be 15.31 million tCO2e, about 2 million tonnes or 10% more than in the

[4] This assumption is based on a total European PET demand of 4.14 million tonnes in 2020 (Plastics Europe, 2021), and a stylized assumption of 1000 euros per metric ton for both primary and recycled PET, while noting a soaring price especially for recycled PET that approached 2000 euros in 2022.

full coverage scenario. Total consumption would decrease to 6.36 million tonnes, with 3.59 million tonnes of primary PET (a 30% reduction) and 2.77 million tonnes of recycled PET (a 7% increase). This means that exempting waste incineration leads to an additional demand for 600,000 tonnes of primary resources and a decrease in material recovery of 210,000 tonnes compared to the full coverage scenario.

Lastly, if the policy goal is to restore material efficiency and reach the optimal levels of primary resource use and recycling, the necessary level of emission charge would be 141 euros per tonne of emissions. This is higher than the level of marginal social damage. In this case, the additional emission charge would result in an undesired loss of social welfare, while still failing to fully restore the optimal level of recycling.

5 Conclusion and Discussion

This paper proposes a multi-market framework to investigate the role of carbon pricing in motivating material substitution in production and end-of-life material recovery. Using a partial equilibrium model, we show that implementing a carbon price on production creates an economic advantage for recycled materials over the primary counterparts, while extending the policy to waste disposal makes recycling more competitive against waste incineration. Both processes generate a significant share of emission savings. Using the emission intensities associated with the plastic polymer PET, we estimate that a carbon pricing policy on both PET production and incineration at the social cost of carbon of 100 euros leads to a 37% reduction in total emissions, contributed by a 23% reduction in total PET consumption and a 15% increase in PET recycling.

In addition, this paper highlights the ambiguous effect of carbon pricing on material recycling, if the scope of this policy is limited to material manufacturing. Various parameters, including the price elasticity of demand for final products, the relative cost of a primary production process compared to its recycling counterpart, and the emission-saving potential of a recycling process compared to primary production and incineration, can cause this ambiguity. In particular, if the derived demand for materials is elastic, the supply of primary materials relatively inelastic, or the emission intensity of recycling relatively high, such a policy alone is insufficient to drive a desired improvement in material recovery.

The equilibrium model of this paper assumes constant return to scale for all technologies, which leads to complete pass-through of carbon cost to the demand prices for consumption good, waste, and materials. This assumption allows an emphasis on the demand-side of the markets, who receive the full price signal to choose a low-carbon option, while leaving the supply-side responding with perfect elasticity. In practice, however, the cost pass-through rate is seldom 100%, especially for the manufacturing sector (Ganapati et al., 2020). This indicates a potential empirical and simulation extension of this paper. Furthermore, the constant return of waste treatment allows the carbon pricing on waste incineration to increase both the waste fee paid by consumers and the price of recycled materials. Thus, this scope for policy extension brings about a double incentive of consumption reduction and recycling subsidy, the theoretical effect of a deposit-refund system (Palmer et al., 1997; Acuff and Kaffine, 2013).

Appendices

A Derivation of comparative static effects

Given the decentralized market solutions under a process emission charge τ prescribed by equations (5) and (6), the implicit function theorem states that these equations can be solved at (f^*, x_r^*, τ^*) by implicitly defined functions of f and x_r with respect to τ that are continuously differentiable. Hence, we obtain the impact of τ on market solutions of f and x_r by taking total derivatives of both sides of the equations:

$$c''(f)\frac{\partial f}{\partial \tau} + e_1 = k''(x_r)\frac{\partial x_r}{\partial \tau} + \sigma e_1$$
$$c''(f)\frac{\partial f}{\partial \tau} + e_1 = P'(x) \cdot (\frac{\partial f}{\partial \tau} + \frac{\partial x_r}{\partial \tau})$$

By rearranging these equations to solve for the first derivatives of f and x_r with respect to τ, we obtain the equations (9) and (10).

$$\frac{\partial f}{\partial \tau} = \frac{e_1 - (1-\sigma)e_1\frac{P'(x)}{k''(x_r)}}{P'(x) \cdot \frac{c''(f)}{k''(x_r)} + P'(x) - c''(f)}$$
$$\frac{\partial x_r}{\partial \tau} = \frac{\sigma e_1 + (1-\sigma)e_1\frac{P'(x)}{c''(f)}}{P'(x) \cdot \frac{k''(x_r)}{c''(f)} + P'(x) - k''(x_r)}$$

Equations (11) and (12) can be derived in a similar way from conditions (7) and (8).

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