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Pricing Carbon Consumption: A Review of an Emerging Trend

Clayton Munnings, William Acworth, Oliver Sartor, Yong-Gun Kim,

and Karsten Neuhoff[1]

Abstract

Nearly every carbon price regulates the production of carbon emissions, typically at midstream points of compliance, such as a power plant. Over the last six years, however, policymakers in Australia, California, China, Japan, and Korea implemented carbon prices that regulate the consumption of carbon emissions, where points of compliance are farther downstream, such as distributors or final consumers. This article aims to describe the design of these prices on carbon consumption, understand and explain the motivations of policymakers who have implemented them, and identify insights for policymakers considering whether to price carbon consumption. We find a clear trend of policymakers layering prices on carbon consumption on top of prices on carbon production in an effort to improve economic efficiency by facilitating additional downstream abatement. In these cases, prices on carbon consumption are used to overcome a shortcoming in the price on carbon production: incomplete pass-through of the carbon price from producers to consumers. We also find that some policymakers implement prices on carbon in an effort to reduce emissions leakage or because large producers of carbon are not within jurisdiction. Since policymakers are starting to view prices on carbon consumption as a strategy to improve economic efficiency and reduce emissions leakage in a way that is compatible with local and international law, we expect jurisdictions will increasingly implement and rely upon them.

Key Words: Carbon pricing, Consumption based policy, Review

JEL: D12 - Consumer Economics: Empirical Analysis
H23 - Externalities; Redistributive Effects; Environmental Taxes and Subsidies
Q54 - Climate; Natural Disasters

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1. Introduction

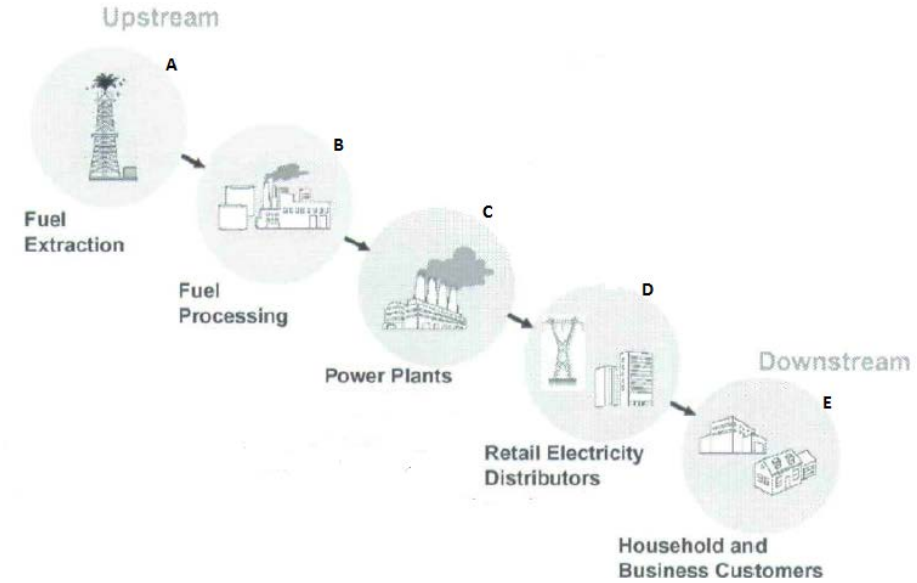
Nearly every carbon pricing scheme in existence around the world regulates the *production* of carbon emissions. Yet carbon *consumption* is receiving increased attention in part because of studies that have pointed out the substantial amount of global carbon emissions embodied in international trade (Davis and Caldeira 2010) and the significant role that increasing imports from developing countries plays in explaining the observed stabilization of carbon emissions in developed countries (Peters et al. 2011). In fact, policymakers in a number of jurisdictions—including California, China, Japan, and Korea—currently operate carbon prices that regulate the consumption of carbon emissions.[2] This article describes the design of these prices on carbon consumption, tries to understand and explain the motivations of the policymakers who have implemented them, and identifies insights for policymakers to consider when contemplating whether to price carbon consumption.[3]

How does a price on carbon consumption differ from a price on carbon production? To illustrate, we use Figure 1, which presents an abstract electricity system; however, note that prices on carbon consumption can, and do, cover other goods besides electricity. The point of compliance for many carbon prices is imposed midstream, at point C, where fossil fuels are combusted and carbon emissions are consequently produced. Upstream approaches place the point of compliance closer to where fossil fuels enter the economy, at point A (Burtraw 2008). This article focuses on prices that place the point of compliance downstream on electricity distributors and/or households and business customers (points D and E), which we collectively call "prices on carbon consumption." Figure 1 abstracts away from at least two important considerations: imports and exports of electricity. Some approaches to pricing carbon consumption would regulate only domestic consumption of carbon emissions associated with domestically produced electricity, while others may regulate consumption of carbon emissions from imports and perhaps exports as well.

[2] We choose to focus primarily on prices on carbon consumption in these jurisdictions, although other prices on carbon consumption exist. For example, as we describe later, Australia had operated a price on carbon consumption that is no longer in operation. As another example, the United Kingdom also has a levy that can be interpreted as a price on carbon consumption.

[3] We choose to explicitly focus consumption prices on carbon and largely ignore literature on taxing the consumption of nonclimate goods.

Figure 1. Points of Compliance for Carbon Prices



Source: Adapted from Burtraw (2008)

At first glance, pricing carbon consumption may seem straightforward. Virtually all goods and services we consume contain carbon emissions that result from manufacturing, packaging and transportation processes. Theoretically, policymakers can simply add a price to these goods and services based on their carbon content via an explicit price (e.g., a tax, charge, or fee) or the creation of a pollution rights market that reveals a price based on supply and demand for those rights (e.g., an emissions trading system or "ETS"). In practice, however, numerous barriers may exist. First, accurately identifying the carbon emissions associated with a substantial number of goods and services can become administratively burdensome (Helm 2012). Second, significant transaction costs may originate from monitoring and enforcing a potentially large number of regulated firms along the value chain (Mansur 2012). Third, a price on carbon consumption may have particular positive or negative implications for economic efficiency and emissions leakage (Bushnell and Mansur 2011). Fourth, a price on carbon consumption has to be designed in a manner that is consistent with relevant laws at the international and domestic levels that govern the pricing of goods and trade across borders. For these four reasons, a price on carbon consumption requires careful consideration.

Despite these barriers, several jurisdictions have decided that prices on consumption of carbon are worth pursuing and implementing. This paper focuses on understanding and explaining the motivations of policymakers who have attempted to overcome these barriers and maximize the benefits associated with pricing carbon consumption through careful design. It is beyond the scope of this paper to analyze each of the numerous prices on carbon consumption that we consider along administrative, economic, and legal dimensions; these aspects must be left for future research. Instead, our intention is to provide a thorough and insightful survey of prices on carbon consumption.

We believe that the importance of prices on carbon consumption will continue to increase, partly because jurisdictions continue to consider new proposals that would price carbon consumption. For example, policymakers in California are actively considering whether to price carbon consumption in the cement sector, and a group of academics has proposed pricing carbon consumption in sectors that produce carbon-intensive materials under the European Union's emissions trading system (EU ETS) (e.g., Neuhoff et al. 2014a, b, 2015). We therefore hope to inform the design of future prices on carbon consumption by providing an up-to-date analysis of ongoing and proposed efforts to price carbon consumption.

The remainder of this paper is organized as follows: Section 2 describes policies that price the consumption of carbon emissions, including ongoing, proposed, and defunct policies. Section 3 analyzes a selection of these policies in more depth by providing additional context, attempting to understand and explain the motivations of relevant policymakers, and making an initial assessment of the policies. Section 4 highlights key considerations and provides initial insights based on the cumulative experience with pricing carbon consumption to date. Section 5 concludes.

2. Experience with Policies That Price Carbon Consumption

In principle, prices on carbon consumption vary along numerous dimensions, including (1) whether they impose prices on goods *produced* in domestic and/or foreign regions; (2) whether they impose prices on goods *consumed* in domestic and/or foreign regions; (3) which goods they cover; (4) whether they impose a price explicitly or implicitly; and (5) which "downstream" entities they impose these prices on.

Regarding the first two dimensions, a price on carbon consumption can, in principle, take at least four forms:

- a "domestic" price on carbon consumption, which involves pricing the domestic consumption of carbon emissions embedded in the production of domestic goods;
- an "import" price on carbon consumption, which involves pricing the domestic consumption of embedded carbon emissions resulting from the production of foreign goods (i.e., imports);
- an "export" price on carbon consumption, which involves pricing the foreign consumption of embedded carbon emissions resulting from the production of domestic goods (i.e., exports); or,

- any combination thereof.

Regarding the third dimension, currently operating prices on carbon consumption tend to impose a price on the use of electricity or transportation fuels. However, a defunct program in Australia imposed a price on the consumption of synthetic gases (e.g., hydrofluorocarbons). In addition, a proposal in California would impose a price on the embedded carbon emissions associated with consumption of cement, and a proposal in Europe would impose a price on embedded carbon emissions associated with the consumption of carbon-intensive materials.

Regarding the fourth dimension, some consumption prices come in the form of an explicit price (e.g., a tax, levy, or charge), whereas others result from ETSs with overall caps defined in absolute terms (e.g., tons of carbon dioxide) or rate terms (e.g., tons of carbon dioxide per unit of gross domestic product).

Regarding the fifth dimension, many of the prices on carbon consumption we consider place the point of compliance on the first importer of the good into the jurisdiction (e.g., California's first jurisdictional deliverer policy on electricity and fuel imports) or, more commonly, on large consumers, including office buildings, hotels, or public institutions, as is done in many of the Chinese pilot ETSs, Tokyo's ETS, and Korea's ETS.

Table 1 describes policies that price the consumption of carbon emissions, including ongoing, proposed, and defunct programs. In general, the policies described in Table 1 can be grouped into five broad categories:

- Category A: impose a domestic price on carbon consumption, including Tokyo's ETS.
- Category B: impose a domestic price on carbon consumption *and* a price on carbon emissions associated with the domestic production, including South Korea's ETS.
- Category C: impose domestic and import prices on carbon consumption, including Australia's defunct levy of synthetic greenhouse gases.
- Category D: impose import prices on carbon consumption and a price on carbon emissions associated with domestic production, including the treatment of electricity and transportation under California's ETS and perhaps its future treatment of cement.
- Category E: impose domestic and import prices on carbon consumption and a price on carbon emissions associated with domestic production, including the treatment of electricity in the Chinese ETS pilots and a proposal on how to treat energy in the EU ETS.

One important note is that the last three categories involve pricing imports, and therefore their implementation may require that domestic and foreign goods face similar prices (i.e., that prices are not discriminatory based on origin) in order to avoid lawsuits based on domestic laws (e.g., the Dormant Commerce Clause in the United States) or international laws (e.g., World Trade Organization principles).

A clear trend apparent in Table 1 is that prices on carbon emissions embedded in consumption of goods tend to occur simultaneously with prices on carbon emissions resulting from production of those goods.[4] In fact, all the prices on carbon consumption we consider in this paper—with the exception of Tokyo's ETS—price both consumption and production. Why would policymakers simultaneously price carbon production and consumption?

We identify at least two potential benefits that seem to motivate the policymakers represented in Table 1. First, a price on carbon emissions embedded in consumption can improve economic efficiency, if the costs associated with a price on carbon emissions resulting from production cannot be, or are not, passed along downstream to consumers. At first glance, this seems at odds with the "irrelevance of who pays" principle in textbook economics, which states, "If a market is in competitive equilibrium throughout, the effect of a tax upon relative prices and upon quantities is identical whether the tax is levied on the buyers or the sellers" (Layard and Walker 1978, 83). In practice, however, carbon pricing can occur in at least two contexts that violate the assumption of a market in competitive equilibrium, both of which muffle the pass-through of allowance costs from producers to consumers. The first context occurs when market structures are not competitive. One example is that in certain Chinese ETSs, electricity prices are determined by the national government and change infrequently, on the scale of years, thereby precluding pass-through of allowance costs to consumers—at least in the short term. Another example is that entire sectors that are covered by ETSs may be oligopolistic in nature (e.g., the cement industry), which may muffle the pass-through of allowance costs to consumers. The second context occurs when policymakers allocate allowances using output-based updating (e.g., in the EU ETS), which acts as a tax and a subsidy that can lead to imperfect pass-through of allowance costs to consumers. Both of these contexts prevent consumer prices from fully reflecting the carbon price. Therefore, a price on carbon emissions embedded in consumption can act to extend the price on carbon emissions resulting from production by exposing a larger portion of intermediary and final consumption choices to a price on carbon, thereby encouraging more efficient use of carbon-intensive goods and realizing additional cost-effective emissions reductions.

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[4] While this trend may be overstated because this paper focuses on carbon consumption prices imposed on goods already covered by ETSs, we are not aware of other examples (with the exceptions of Tokyo's ETS and the United Kingdom's carbon levy) of a carbon consumption price that exists without a carbon production price.

Second, a price on carbon consumption can reduce emissions leakage that a price on carbon production might cause, especially if a large portion of carbon emissions from the implementing jurisdiction originates from imports. Actually capturing these perceived benefits might at least require that policymakers spend significant effort and collect substantial amounts of data, issues we discuss in detail in the next section. Nonetheless, it is clear that policymakers tend to use prices on carbon consumption to complement prices on carbon production, with the possible intent of improving the economic efficiency and environmental effectiveness of their efforts to mitigate carbon emissions.

Table 1. A Summary of Policies Pricing Carbon Consumption

Region	Status	Good with price on consumption	Scope of coverage and relevant policy: Carbon from the production of domestic goods	Embedded carbon from domestic consumption of domestic goods	Embedded carbon from Domestic consumption of foreign goods (imports)	Embedded carbon from foreign consumption of domestic goods (exports)
Category A: “Domestic” price on carbon consumption						
Japan–Tokyo	Active since 2010	Electricity & fossil fuels (e.g., crude, heating, and gas oil)	None	Absolute C&T	None	None
Category B: “Domestic” price on carbon consumption and price on carbon emissions from production						
South Korea	Active since 2015	Electricity	Absolute C&T	Absolute C&T	None	None
Category C: “Domestic” and “import” price on carbon consumption						
Australia	Defunct	Synthetic gases	None	Levy	Levy	Rebate
Category D: “Import” price on carbon consumption with price on carbon production						
California	Active since 2012	Electricity	Absolute C&T	None	Absolute C&T	None
California	Active since 2015	Transportation fuels (e.g., gasoline)	Absolute C&T	None	Absolute C&T	None
Category E: “Domestic” and “import” prices on carbon consumption with price on carbon production						
China–Beijing	Active since 2013	Electricity	Absolute C&T	Absolute C&T	Absolute C&T	None
China–Chongqing	Active since 2014	Electricity	Absolute C&T	Absolute C&T	Absolute C&T	None
China–Guangdong	Active since 2013	Electricity	Absolute C&T	Absolute C&T	Absolute C&T	None
China–Hubei	Active since 2014	Electricity	Absolute C&T	Absolute C&T	Absolute C&T	None
China–Shanghai	Active since 2013	Electricity	Hybrid C&T	Hybrid C&T	Hybrid C&T	None
China–Shenzhen	Active since 2013	Electricity	Absolute and rate-based C&T	Absolute and rate-based C&T	Absolute and rate-based C&T	None
China–Tianjin	Active since 2013	Electricity	Absolute C&T	Absolute C&T	Absolute C&T	None
Proposed policies						
California	Proposed	Cement	Absolute C&T with OBA	None	A fee, possibly	Absolute C&T, possibly
EU ETS	Proposed	Cement and steel	Absolute C&T with OBA	Charge	Charge	Refunded

Notes: C&T = cap and trade; OBA = output-based allocation. Zhang et al. (2014) explain that all seven ETS pilots in China cover the indirect emissions from electricity (both generated within the pilot region and imported from other regions), as electricity is the major source of emissions associated with goods traded across provinces, and measuring the quantity of electricity flows is relatively straightforward. Duan et al. (2014)

explain that for all pilot systems in China, absolute emissions caps have been established, although the specific approaches used vary considerably. Munnings et al. (2014) elaborate on the cap designs in three pilots and describe the cap in Guangdong as absolute, in Shanghai as a hybrid, and in Shenzhen as both an absolute and rate-based cap.

3. Analysis of Policies That Price Carbon Consumption

In this section, we discuss a selection of policies that price carbon consumption in further detail by providing additional context, attempting to understand and explain the motivations of relevant policymakers, and making an initial assessment of these policies. We split the discussion of policies into the five categories (A–E) introduced earlier and displayed in Table 1.

A. "Domestic" Prices on Carbon Consumption: Policies That Price Carbon Emissions from Domestic Consumption of Domestic Goods

1. Tokyo's Emissions Trading System

a) Introduction to Tokyo's Emissions Trading System

In 2010, the Tokyo Municipal Government (TMG) introduced an ETS that covers urban facilities such as office buildings. The TMG ETS is the world's first city-level ETS and Asia's first mandatory ETS. It represents the core of Tokyo's climate policy portfolio and covers 20 percent of total emissions. The system requires large urban facilities to reduce emissions between 6 percent and 8 percent in the first phase (fiscal year 2010–14) and by 15 percent to 17 percent in the second phase (fiscal year 2015–20). That is, the point of compliance is at point E in Figure 1. Emissions reductions are calculated at the entity level based on historic baselines.[5] Emissions permits are allocated freely. As of September 2015, the TMG ETS sustained the highest allowances prices of all global ETSs, at US$36 per ton of carbon dioxide equivalent (tCO_2e) (World Bank 2015).

The TMG ETS places obligations at the facility level. Specifically, rather than target electricity producers, the scheme covers almost 1,400 urban facilities that consume more than 1,500 kiloliters (kL) of crude oil equivalent per year. Four-fifths of the covered entities are office buildings, with the remainder made up of public institutions, commercial buildings, lodging, educational facilities, and medical facilities. The TMG ETS caps CO_2 emissions from both fossil fuel consumption (e.g., crude oil, heating oil, gas oil) and electricity usage, based on an estimate of the carbon intensity of the main power supplier, Tokyo Electric

[5] Constructed from any three consecutive years between 2002 and 2007 at the discretion of the covered entities.

Power Company (Rudolph and Kawakatsu 2012).[6] It can thus be classified as a downstream scheme that focuses on indirect emissions from consumption rather than production (Dabner 2014). In addition to emissions reductions, covered entities can use offset credits generated by small to medium-size office buildings not covered by the scheme or renewable energy certificates.

Reduction obligations are set at the beginning of each compliance period based on the baseline emissions, the compliance factor, and the number of years in the compliance period. Tradable credits result from excess emissions reductions beyond facilities' annual reduction obligations. Permits are awarded for reductions in excess to the facility-level emissions baseline and may be traded only after emissions reductions have been verified by the TMG. The system therefore places more emphasis on the "cap" than on the "trade" element of an ETS, with the market providing a backup strategy if emissions targets cannot be met. While trading can be done on the open market, such trades have been rare, with most taking place bilaterally between facilities. To facilitate trade, the TMG publicizes annual emissions and actual reductions of each facility on its information disclosure website (Rudolph and Kawakatsu 2012).

b) Motivation for Tokyo's Price on Carbon Consumption

The commercial sector dominates Tokyo's emissions profile, constituting 40 percent of total emissions—half of which stems from electricity use. Buildings within the commercial sector represent a significant opportunity for emissions reductions if, for example, lighting fixtures are replaced and insulation measures strengthened. Hence if climate policy in Tokyo (and indeed, any large metropolitan region) is to be effective, it must necessarily address emissions from electricity use in the commercial sector.

However, regulation of upstream electricity producers is difficult for two reasons. First, 90 percent of electricity production facilities are located outside the Tokyo prefecture. Therefore, from a legislative perspective, the TMG does not have the jurisdiction to impose emissions reduction targets on these facilities (Dabner 2014). Second, principal agent barriers in the commercial building sector limit the effectiveness of upstream regulation (IEA 2007). For example, split incentives in landlord-tenant contracts mean that landlords are focused on minimizing capital costs rather than energy use, as the tenant pays the final bill. Under this arrangement, if electricity prices were to rise because of an upstream carbon price, the

[6] For a list of fuels and conversion factors, see Appendix 1 of World Bank (2010). Note that while all commercial facilities are awarded the same benchmark for electricity use, fuels used in factories receive differentiated emissions factors.

landlord would have no incentive to upgrade the energy performance, since the tenant would be the one facing the increasing costs. This presents a significant barrier to improving the energy efficiency of a building, because much of the mitigation potential relies on investments in building fittings and structure. Placing the point of regulation on the building owner realigns incentives between the owners and tenants, as it is the building owner that is responsible for ensuring compliance.

c) Assessment of Tokyo's Price on Carbon Consumption

According to the TMG, the ETS achieved a 23 percent reduction compared with baseline emissions levels by the second year of operation. A major driver for these emissions reductions is thought to be the electricity savings introduced by covered facilities in the wake of the power crisis that followed the Great East Japan Earthquake. However, electricity reduction plans prepared as part of the TMG ETS reporting framework are also credited with facilitating reductions. Specifically, an open dialogue between tenants and building owners has helped result in reduced power demand (TMG 2013). A full report will be conducted at the end of the second trading period.

Numerous lessons can be learned from Tokyo's price on carbon consumption. First, the gradual phasing in of the policy was critical to its success because the nearly decade-long period of reporting that preceded the scheme allowed the regulator to build expertise on carbon emissions associated with consumption, gain credibility, and play an information-providing role. Second, Tokyo's price on carbon consumption serves as an important example where substantial emissions reductions can be achieved by only pricing domestic consumption of electricity and fossil fuels. The focus on urban electricity consumption also circumnavigates some competitiveness concerns, perhaps allowing companies to tolerate relatively high prices. Third, the idea that subjecting building owners to a carbon price may overcome principal agent (landlord-tenant) issues is important. Finally, the system allows the TMG to maintain a large degree of control, particularly through continued engagement with covered entities in setting their individual caps.

B. "Domestic" Prices on Carbon Consumption with Prices on Domestic Production: Policies That Price Carbon Emissions from Domestic Consumption of Domestic Goods and Domestic Production of Domestic Goods

1. South Korea's Emissions Trading System

a) Introduction to South Korea's ETS

The Republic of Korea started an ETS in January 2015. The ETS covers approximately 490 of the country's largest emitters, which account for about 67 percent of national greenhouse gas (GHG) emissions. The first and second phases have three-year commitment periods, and later phases have five-year commitment periods. The overall cap for the ETS has been set to be consistent with the economy-wide reduction target of 30 percent below business-as-usual emissions forecast for 2020. The first phase starts with 100 percent free allocation, and the second and the third phases limit the share of free allocations by 97 percent and 90 percent, respectively. Energy-intensive trade-exposed (EITE) sectors will receive 100 percent of their allowances free of charge. The criteria for EITE sectors are the same as those of EU ETS: production cost increases over 5 percent and trade intensity over 10 percent; production cost increases over 30 percent; or trade intensity over 30 percent.

b) Description of South Korea's Price on Carbon Consumption

The Korean ETS covers direct emissions of six Kyoto gases,[7] as well as indirect emissions from electricity consumption. Accordingly, the threshold for mandatory participation includes annual emissions of these gases and indirect emissions from power consumption. That is, the points of compliance are at points C and E in Figure 1. The level of this threshold is annual emissions greater than 125,000 tCO_2e by a company or annual emissions greater than 25,000 tCO_2e from an installation. The coefficient for the calculation of indirect emissions from the consumption of electricity (i.e., the emissions factor) is updated by the regulatory authority before each of the commitment periods, based on the average emissions intensity of power production. There is a risk that the actual emissions derived from electricity consumption may differ from the estimate of indirect emissions, though the gap between the two may be minimized through frequent updating of emissions factors.

c) Motivation and Assessment of South Korea's Price on Carbon Consumption

The regulatory authority introduced the inclusion of electricity consumption to circumvent the strict government control of retail electricity prices. The government determines the retail price of electricity on a "gross cost" basis, which is the variable cost of production plus a "reasonable" return on capital investment. In addition to this average pricing practice, various policy considerations, such as energy security, protection of industries, general price management, and income redistribution, are given to various end-user groups (Kim and Lim 2014).[8] The retail prices are updated irregularly, only once a year

[7] These six gases are carbon dioxide (CO_2), nitrous oxide (N_2O), methane (CH_4), hydrofluorocarbons (HFCs), perfluorocarbons (PFC), and sulfur hexafluoride (SF_6).

[8] The wholesale price of electricity is determined by Cost Evaluation Committee based on the cost data submitted by the power-generating companies.

on average, and reflect a variety of political and economic considerations. The environmental regulatory authority was concerned that without liberalized market-based pricing of electricity, the carbon price on direct emissions would not be passed through to the price of electricity, so the demand of electricity would not be managed to an efficient level. Another motivation for the inclusion of electricity consumption in the Korean ETS is that it significantly enlarges the scope of covered entities, including large buildings, for example, which would otherwise be exempted from the ETS.

The inclusion of electricity consumption, however, raises the concern of double taxation of emissions, through the carbon pricing of both the production and consumption of electricity. Although the carbon cost might not pass through systematically to the consumption price of electricity, the retail price of electricity may change to reflect the carbon cost in the long term, since the government may not be able to perpetually subsidize the gap between the real cost of electricity production and the retail price of electricity. That is, the pass-through of carbon cost to electricity prices may be realized in the longer term. Facing strong opposition from industries, the regulatory authority loosened the caps for individual emitters, taking into consideration the shares of indirect emissions from electricity consumption. More specifically, via a strategy of allocating allowances via grandfathering, the emissions reduction ratios are set much higher for direct emissions than for indirect emissions from electricity consumption. For example, the government sets emissions reduction goals for direct emissions and indirect emissions differently, say 20 percent for the former and 5 percent for the latter.[9]

The concern over double taxation increases as the share of allowances that are auctioned increases. Under the current wholesale electricity price mechanism, power producers are compensated for their carbon cost through rate-of-return regulation, which effectively results in the pass-through of carbon costs only for the part not compensated by free allowances. As the share of auctioned allowances grows, as scheduled by the law, the pass-through of carbon costs to the retail price of electricity will increase, eventually leading to a higher cost of electricity consumption for ETS participants.

[9] For example, if an entity has historic direct emissions of 8 million tCO_2e and historic indirect emissions of 2 million tCO_2e (10 million tCO_2e in total), then the entity is given 8.3 million tCO_2e (8.3 = 8*(1 – 0.20) + 2*(1 – 0.05)) of allowances for free.

C. “Domestic” and “Import” Prices on Carbon Consumption: Policies That Price Carbon Emissions from Domestic Consumption of Domestic and Foreign Goods

1. Australia’s Price on Synthetic Gases

a) Introduction to Australia’s Price on Carbon Consumption

In June 2013, Australia implemented a carbon price as part of a broader legislative package known as the Australian Clean Energy Plan. This carbon price included a carbon tax of AU\$24/tCO_2eq, which was intended to transform into an ETS starting in July 2014. The price covered 260 large emitting entities in the power and industrial sectors and roughly 60 percent of Australia’s annual CO_2 emissions. However, the bulk of the Clean Energy Plan’s legislation was repealed following a change of government in July 2014.

Despite its repeal, the Australian carbon pricing experience contains another example of a governmental decision to use carbon pricing to target consumption of carbon emissions. Specifically, as part of the Clean Energy Plan, Australian policymakers decided to place an equivalent carbon levy on the imports of foreign produced or sale of domestically produced synthetic greenhouse gases (SGGs). This levy was legislated and administered outside the main carbon pricing scheme, although it was explicitly linked to the price.

SGGs are non-CO_2 greenhouse gases such as HFCs, PFCs, and SF_6, which tend to have very high global warming potentials (GWPs) relative to CO_2. SGGs were introduced as an alternative to CFCs and HCFC, which are ozone-depleting substances that have been or are being phased out under the Montreal Protocol. They are commonly found in refrigerant products and solvents, as well as downstream products such as aerosols, air conditioners, fire extinguishers, refrigerators, and some electrical and cleaning products. Emissions of SGGs typically occur as a result of leakage of the gases from refrigerant systems (leakage rates are believed to often be as high 25 percent in large industry applications, according to Bostock 2013) and from unsafe end-of-life disposal of products containing SGGs.[10] They are not currently covered by the Montreal Protocol, although it is now anticipated that the Montreal Protocol may be amended to include phaseout of these high-GWP GHG gases in the future, since safe, more energy efficient low-GWP alternatives now exist.

At the time of introduction of the carbon price–equivalent levy on SGGs, Australia had an existing reporting framework for domestic bulk sales, imports and exports of major

[10] In principle, SGGs that do not leak from equipment can be recovered at the end of product life if good disposal practices and regulations are established and the gases can then be destroyed, limiting their global warming impact. However, if end-of-life destruction is not done, then the gases eventually leak into the atmosphere, with a much larger global warming impact.

SGGs (HFCs and PFCs), established as part of its Ozone Protection and Synthetic Greenhouse Gas Management Act 1989 (OPSGG MA), introduced in response to the Montreal Protocol on Ozone Depleting Substances.[11] Under this legislation, bulk importers of ozone-depleting substances (ODSs) and SGGs, or of prefabricated products containing them, were subject to certain obligations. Specifically, they were required to obtain a license to sell these products in Australia, report sales to authorities on a quarterly basis, and pay a small levy to cover the administrative costs of the reporting and stewardship schemes.[12]

The legislation also made it an offense to dispose of products containing ODSs or SGGs in a way that could lead to their emission into the atmosphere. In practice, however, there is significant noncompliance. Many products that do not require a technician to uninstall (such as refrigerators or aerosols) often are illegally dumped by users at their end of life or collected by informal actors who do not comply with safe disposal regulations. Moreover, even some formal supply chain actors such as local government authorities and sellers that offer end-of-life collection and recycling services do not always comply with the law (KPMG 2014).

To implement the carbon price equivalent levy for SGGs, the government used the existing legislative vehicle of the OPSGG MA. It expanded reporting under the OPSGG MA (to include SF_6) and used the existing reporting and levy payment framework to apply an additional carbon levy to the importer each quarter (i.e., four times per year), whether or not goods were sold. Reporting and payment (with a two-month grace period for payment) were done quarterly to reduce cash-flow challenges for businesses involved. This allowed businesses to receive payment from the sale of their products and thus recover the cost of the charge from downstream consumers.

The carbon cost charge tracked the CO_2 allowance price level (and was intended to track the future ETS price). This was done by setting the carbon charge at the CO_2-equivalent level of GHGs contained in the products, multiplied by the annual tax level. After the start of the floating ETS price, the preceding year's average auction price of CO_2 permits was to be used. The policy reasons for this are explained below. To avoid carbon leakage for Australian exporters of products containing SGGs, the government also provided a rebate of the carbon costs paid during the manufacture of domestic products.

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[11] The Ozone Protection and Synthetic Greenhouse Gas Management Act 1989, Ozone Protection and Synthetic Greenhouse Gas (Import Levy) Act 1995, and Ozone Protection and Synthetic Greenhouse Gas (Manufacture Levy) Act 1995.

[12] Because the obligation applied to bulk importers, small retailers farther down the value chain were not required to hold licenses or be subject to the reporting obligations. Australia has no manufacturers of SGGs, and thus the obligations did not apply to them by default.

The money collected from the imposition of the levy was partially earmarked to strengthen incentives for safe end-of-life destruction of products containing SGGs. Specifically, a new payment was introduced to industry actors who could prove safe destruction had taken place, with the government to set up a reinforced stewardship scheme in consultation with industry. This was funded using revenue collected from the imposition of the levy. The remaining funds raised appear to have been returned to the federal budget without earmarking, although the budget funded a significant package of measures to compensate consumers for new costs associated with the introduction of the carbon price and other complementary policies. The final CO_2-equivalent content of the product was not made visible to the final consumer via labeling or a line on a receipt. Indeed, in general, the final cost was small for the final consumer—at AU$24/t$CO_2$, the increase in cost of a refrigerator to the final consumer was around AU$10 and of a car was around AU$25. The price was more visible (because of its impact on costs) to industrial users of refrigeration equipment and to maintenance service providers.

b) Motivation for Australia's Price on Carbon Consumption

The decision to apply a carbon levy to SGGs reflected a desire by the Australian Federal Treasury Department to make coverage of the Australian CO_2 price as broad as possible.[13] This apparently reflected a prevailing view at the Treasury that the effectiveness and economic efficiency of the carbon pricing policy would be maximized by a carbon price that was as broad-based as was politically feasible. Indeed, only the transport and agriculture sectors were exempted from the initial Australian carbon pricing scheme, and only after strong political pushback from those sectors.

Given the diffuse nature of emissions of SGGs, such as from product leakage, the government's view was that it would be infeasible to directly apply emissions pricing to the emissions point source. But it was believed that importers would inevitably pass on the carbon price to purchasers of refrigerants and products containing them. In turn, if downstream purchasers of refrigerants were faced with a carbon price–equivalent levy based on the embedded CO_2-equivalent content of SGG in the product, it was believed that they would be incentivized to undertake a range of abatement actions. These actions were expected to include switching to alternative existing technologies (such as low-GWP synthetic or natural refrigerants), reducing leakage rates, and increasing rates of recovery and reuse of refrigerants at the end of product life.[14]

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13 Telephone interview with Greg Picker, Refrigerants Australia, May 20, 2015.
14 Email correspondence with Patrick McInierney, director of ozone and synthetic greenhouse gas policy, Australian government, April 28, 2015.

The decision not to include SGG emissions in the main carbon pricing scheme was made for two reasons. First, as noted above, a preexisting scheme for licensing companies to import and sell SGGs, for reporting, and for paying levies was already in place. It was therefore deemed to be administratively more efficient to work within the existing framework for SGGs. Second, the high global warming potentials of SGGs means that the product cost after the carbon price was applied would be several multiples of the initial product cost and potentially an order of magnitude higher than the margins that importers made on resale. This raised concerns within the industry that once Australia moved to the ETS, a floating carbon price would potentially be disruptive to the market for refrigerants in Australia. For example, it was feared that refrigerant sellers would have a strong incentive to become "carbon market speculators," purchasing large amounts of refrigerants when prices fell, and then withholding supply until prices were high. This also helped motivate a decision to have the carbon price applied to refrigerants done outside the ETS and with a less volatile price regime. Hence once the floating price scheme began, it was intended that the SGG levy would be equal to the GWP of the specific gas multiplied by the volume imported and the annual average of the preceding year's auction prices.

c) Assessment of Australia's Price on Carbon Consumption

It is interesting to note that the combination of downstream stewardship (collection for safe disposal scheme) and the carbon price is an instance of double regulation to some extent. This is because the CO_2 levy overlapped with an existing policy of regulating end-of-life disposal of SGGs, which is when a significant part of the emissions either occur or can be avoided. Thus in a sense, it was charging consumers for emissions that may not fully occur.[15]

The SGG carbon levy was in place for only about 18 months, so there has been insufficient time to gather data and assess policy effectiveness in a comprehensive and robust way. Nevertheless, interviewees from industry, NGOs, and government have reported several anecdotal impacts of the policy. First, it was clear that a perverse incentive was created at the start of the scheme for importers to stockpile massive amounts of refrigerants just before the scheme came into effect, in order to avoid the liability. These refrigerants reportedly were sold once the scheme had started with the carbon levy included in the price, leading to a loss of revenue for government and a windfall profit for the importers. Therefore, as with any policy, one needs to think through and react to possible perverse incentives that are created.

[15] On average, between 5% and 25% of refrigerant gases leak during their use over their productive lifetimes. However, up to 100% can leak when products are not correctly handled at disposal. This is a frequent occurrence in Australia despite laws guiding safe disposal and existing stewardship programs.

Second, it seems that policy clearly did lead to price pass-through and to corresponding behavioural changes, although sometimes in unexpected ways. For example, some large industrial users shifted technologies to lower-GWP alternatives, and the policy had some impact on management practices for large users (such as reducing leakage). In addition, recovery of refrigerants decreased, suggesting greater reuse within the industry, but in some cases refrigerants were not being brought up to manufacturer quality specifications before being reused, thus potentially undermining the energy efficiency of the systems in which they were used. However, the policy had no impact on smaller users, because the low amount of refrigerant in consumer products (such as refrigerators or cars) meant that the carbon levy remained small relative to the final consumer product price. There were also concerns that higher prices were leading to some recharging of refrigerants at less than 100 percent of the total capacity of the refrigeration device to save on cost.[16] Another concern was that the lack of domestic manufacturers in Australia made it harder for the incentives for material efficiency along the value chain to be fully exploited.

Third, the imposition of the levy on importers did not lead to any backlash or retaliation from exporting countries toward Australia. This probably reflects the expectation among industry participants that high-GWP alternatives would be phased out given the existence of new, lower-GWP and more energy efficient alternatives.

D. "Import" Prices on Carbon Consumption with Prices on Domestic Production: Policies That Price Carbon Emissions from Imported Goods and Emissions from Domestic Production

1. California's Treatment of Electricity under Its Emissions Trading System

a) Introduction to California's ETS

In 2006, the California legislature passed the Global Warming Solutions Act, hereafter referred to as Assembly Bill (AB) 32, which authorizes the California Air Resources Board (CARB) to craft regulations that implement a statewide ETS. Leveraging this authority, CARB started an ETS in 2012 that initially covered the electricity, oil, and industrial sectors. California's ETS has since expanded in 2015 to also cover transportation fuels, making it the most comprehensive ETS in the world by regulating 85 percent of the state's greenhouse gas emissions, equivalent to 394 million tCO_2e. The program imposes a price floor of roughly US$10, and allowance prices have ranged from US$11 to US$23 over the last several years.

[16] This has the negative impact of lowering energy efficiency of the refrigerator, thus potentially offsetting the emissions benefits from the lower refrigerant use.

The California ETS stands alone in the western United States. Although the governors of Arizona, New Mexico, Oregon, and Washington joined California in starting the Western Climate Initiative—the main goal of which was to implement a regional ETS in the five states—only California has actually followed through with carbon pricing. This may change in the near future, as Washington and Oregon have again started to consider carbon pricing, and the forthcoming Clean Power Plan may incentivize other neighbors to also consider carbon pricing. However, at this time, the California ETS is particularly susceptible to emissions leakage to neighboring regions that do not price carbon.

b) Motivation for California's Price on Carbon Consumption

Regulating carbon emissions from the electricity sector is a central component to virtually all ETSs, and California's is no exception. Electricity consumption in California accounts for approximately 20 percent of the state's emissions. While the production of electricity within California is relatively clean, electricity imports from outside California are substantial and relatively dirty. For example, in 2013, electricity imports accounted for only one-third of total electricity consumption in California but constituted 44 percent of the carbon emissions from the state's electricity consumption (CARB 2015; CEC 2015a). The importance of carbon emissions from out-of-state producers complicates the design of California's ETS, since the state has no explicit jurisdiction to directly regulate these entities.

The substantial carbon emissions originating from electricity imports inspired the California legislature to also direct CARB via AB 32 to account for carbon emissions from imports and to minimize emissions leakage in the policies it chose to implement. Emissions leakage is defined as an increase in carbon emissions outside the jurisdiction pricing carbon caused by that carbon price. Emissions leakage can occur through various means, including physical relocation of electricity generation and a more subtle mechanism called "reshuffling," in which importers shuffle their sale of clean electricity to the jurisdiction pricing carbon and reshuffle their sale of dirty electricity to jurisdictions without a carbon price without changing the overall carbon emissions of their portfolios. Bushnell et al. (2007, 14) write that such phenomena "are likely to overwhelm any meaningful impact" of California's carbon price if the state acts unilaterally.

In response to such concerns over emissions leakage, the Market Advisory Committee to CARB recommended the implementation of a first jurisdiction deliverer (FJD) approach to regulate the production of electricity in California, as well as imports of electricity into California consumed in the state (MAC 2007). An electricity generation facility in California and an importer of electricity into California are both defined as FJDs, and both must retire allowances for the carbon emissions associated with the generation of electricity they sell (Parlar et al. 2012). In this way, the points of compliance are at points C and D in Figure 1.

Electricity generators and importers trade allowances in the same market. Electricity importers and the quantities of electricity associated with individual transactions can typically be identified through North American Electric Reliability Corporation (NERC) e-tags because California's electricity market is essentially the same as the state's borders (Murtishaw 2011).[17] Assigning carbon emissions levels to those imports is more difficult. In order to do so, California distinguishes between specified and unspecified electricity transactions. The former is typically tied to a specific generating facility with a contract to import electricity into California, where a facility-level emissions rate is known and assigned. The latter refers to electricity imports where the originating generator is unknown and therefore an accurate emissions rate is not known. Under this circumstance, California assigns these electricity imports a default emissions rate of 0.428 million metric tons per megawatt hour—a value that represents a fairly clean natural gas power plant and was constructed to reflect a marginal unit in the western electricity grid (Bushnell et al. 2014). In 2014, 85 percent of total electricity consumption in California was specified, and 15 percent was unspecified (CEC 2015b). California uses a similar FJD approach in regulating emissions from the transportation sector.

First and foremost, the FJD approach must be legal in order to remain in implementation. The most relevant law is the Dormant Commerce Clause (DCC) of the US Constitution, which limits states' ability to impose burdens on interstate commerce. Courts ask two questions when considering cases related to the DCC. First, does the regulation facially or effectively discriminate against out-of-state interests? If so, the court applies a strict scrutiny test, which sets a high bar for the regulation. Second, does the regulation regulate extraterritorially? If so, the regulation is invalidated. If the regulation is found not to discriminate against out-of-state regulations or extraterritorially, then the court applies a *Pike* balancing test, which upholds a regulation if, inter alia, it effectuates a legitimate local public interest, affects interstate commerce only incidentally, and does not impose a burden on interstate commerce that is clearly in excess of the putative local benefits (Parlar et al. 2012).

The FJD approach must also be effective at stemming emissions leakage in order to serve its purpose. California has supplemented the FJD approach with a suite of additional regulations. For example, California attempts to mitigate underreporting of carbon emissions and incentivize investment in cleaner generation through a variety of strategies, including the placement of strict requirements on contracts of specified imports below the default rate (Burtraw 2008). These imports must come from a source that has historically exported to

[17] Importantly, California does not require imports that are not ultimately consumed and would not require imports from states with ETSs to retire allowances (Murtishaw 2011).

California, have originated from federally owned hydro facilities, or be generated from new facilities or new capacities at existing facilities pursuant to a written power contract (Murtishaw 2011). In addition, California has proposed an explicit prohibition of reshuffling, which remains under consideration (Bushnell et al. 2014). These additional regulations aim to facilitate the effectiveness of the FJD approach.

c) Assessment of California's Price on Carbon Consumption

The FJD approach in California has yet to be challenged in court, and it is unlikely that a challenge against the FJD approach in general would be successful, at least according to some legal scholars. For example, Parlar et al. (2012) find that an FJD approach similar to California's would not discriminate against out-of-state interests because it subjects electricity importers to a carbon price that is also faced by domestic producers. In addition, they find that such an approach would not likely be considered to regulate extraterritorially, since electricity importers are located within the state imposing the regulation. In conclusion, they argue that an FJD approach would probably pass a *Pike* balancing test, primarily because the regulations "do not present burdens disproportionate to their benefits and would not obviously result in a shift in demand from out-of-state electricity to in-state electricity" (Parlar et al. 2012, 4). It therefore seems that an FJD approach is capable of complying with the DCC.

The FJD approach, unlike some other prices on carbon consumption we consider, has been the subject of significant academic debate from the legal and economic communities. This research has offered at least two key insights. First, the FJD is carefully crafted to comply with the DCC. The fact that the policy has not been challenged to date shows that policymakers can achieve the important goal of crafting a legally sound price on carbon consumption. The legality of California's FJD approach might be called into further question, however, if local producers of electricity were not already subject to a carbon price.

Second, strong additional regulatory measures bolster the effectiveness of California's FJD approach at stemming emissions leakage. Specifically, policies that improve the accounting of carbon emissions associated with electricity imports and stem perverse economic incentives (such as reshuffling) to create emissions leakage seem particularly salient. This highlights the complexity of regulating electricity imports and the potential utility of relying on a mixture of market-based and more command-and-control regulatory measures to facilitate the effectiveness of a price on carbon consumption. It is also clear that the institutional flexibility of regulated polluters (in this instance, electricity contracts) should be considered when designing a price on carbon consumption.

2. California's Treatment of Cement

A significant amount of cement is produced in California, and these producers are covered under the ETS. They receive some assistance in the form of free allocation. However, a substantial amount of cement is also imported into California from other jurisdictions. CARB considers the cement industry to be prone to high leakage and therefore intends to regulate it to prevent leakage, in accordance with the CARB's interpretation of AB 32. In fact, CARB passed a resolution in 2010 to prioritize efforts to prevent leakage from the cement sector.

To regulate emissions from imported cement, CARB must first measure those emissions and is actively considering three approaches to data collection: using default emissions from an unspecified source, using default emissions based on technology and fuel mixes, or full reporting of emissions associated with cement imports.

To reduce leakage from the cement sector, California is currently considering four approaches, all of which involve regulating the FJDs of cement into the state from importers located in California that deliver non-California cement to California. The first option is to include FJDs of cement into California's ETS. This may be attractive to CARB because it replicates California's treatment of imported electricity, but California's current cap would need to expand an estimated 1–6 million metric tons to cover importers. The second option is to essentially tax FJDs for emissions associated with their imports, with the tax equal to the tons of emissions associated with imports multiplied by the current allowance price in California's ETS. This may be attractive because it is simple, but a tax approach has both political and legal drawbacks in California. For example, Proposition 26 states that an increase in taxes must be approved by two-thirds of both houses of the state legislature (Next 10 2012). The third option is to construct a separate ETS program for FJDs of cement. Allowances would not be traded with California's larger market. This would, according to CARB, require a cumulative cap to be set out to 2020, which may be hard to predict. The fourth option is to set an updating cap with permits auctioned to FJDs of cement based on the price floor of California's larger program.

E. "Domestic" and "Import" Price on Consumption with Price on Production: Policies That Price Carbon Emissions from Domestic and Imported Goods and Emissions from Domestic Production

1. Shanghai's Price on Carbon Emissions from Electricity Consumption through Its Emissions Trading System

a) Introduction to Shanghai's ETS

The ETS in Shanghai is one of China's seven ongoing pilots that will inform the design of a national Chinese ETS slated to take effect in 2017. The Shanghai pilot started in

November 2013 and covers roughly 200 entities that emit about 160 million tCO_2e annually. The pilot covers the power and industrial sectors, which emit over 20,000 tons of carbon per year, as well as commercial and transportation companies, which emit over 10,000 tons of carbon per year. The cap that governs the emissions of these sectors is a hybrid between an absolute and an intensity-based approach, and regulators have not defined clear caps for any of the (at least) three years the pilot will operate. All allowances are freely allocated, although the corresponding regulations do leave the possibility of an auction later on (Munnings et al. 2014). Allowance prices hover between US$2 and US$10 with low market liquidity (Tanpaifang 2015).

b) Motivation for Shanghai's Price on Carbon Consumption

The Shanghai pilot regulates the electricity sector by covering "direct" emissions (primarily from fossil fuel combustion to produce electricity) and "indirect" emissions (associated with the purchase of electricity and determined by multiplying consumed electricity by a default grid emissions factor). The indirect emissions originate from covered entities in the industrial and commercial sectors, the latter of which includes large buildings such as hotels. That is, the points of compliance are at points C and E in Figure 1. Shanghai regulators then freely allocate the number of allowances they estimate to be appropriate for each regulated firm's direct and indirect emissions. Generally, the power sector receives allowances according to benchmarking, and the industrial and commercial sectors receive allowances according to historic emissions (Munnings et al. 2014). The direct and indirect allowances are fungible, such that an electricity producer with no indirect emissions is allowed to trade allowances with a large hotel with no direct emissions (Wu et al. 2014). At the end of a compliance period, direct emitters must retire allowances for their production of carbon emissions, and indirect emitters must retire allowances equal to the emissions associated with the electricity they purchased.

c) Assessment of Shanghai's Price on Carbon Consumption

The main motivation for Shanghai's policymakers to regulate indirect emissions is to encourage emissions reductions from lower demand of electricity (Wu et al. 2014). In China, the pass-through rate for electricity producers arguably equals zero because of national government controls on electricity prices. As an example, the government has changed electricity prices twice in the last five years (NDRC 2009, 2011). If China were to price direct emissions only, the carbon price would be communicated to only a small part of the economy (just the electricity producers), while the remainder of the economy (namely, electricity consumers) would act without a carbon price. Such an arrangement would prevent cheap reductions from coming online—for example, from emissions reductions from lower demand of electricity—and lead to a higher carbon price, all else equal (Munnings et al. 2014). By covering electricity producers and consumers in the presence of a controlled electricity market, the policymakers in Shanghai approximate the pass-through of allowance costs from electricity producers to consumers, allowing electricity prices to rise and increasing the overall economic efficiency of the ETS. However, the precise extent to which this policy increases economic efficiency depends on the quantity of demand-side reductions that the policy induces and the relative cost-effectiveness of those reductions, both of which are currently unknown.

At first glance, this policy may seem to tax indirect emitters twice. It does represent double counting in that local governments allocate two allowances for one unit of carbon emissions; therefore, the overall cap does not represent total emissions, because some tons are counted twice: once as a direct emission and again as an indirect emission. However, this policy does not represent double taxation, because no single entity has to retire two allowances for one ton of carbon emissions—so long as the pass-through rate from electricity producers effectively equals zero, which seems to be the case at least in the short run.

The effectiveness of this policy hinges on how well its design accounts for three important aspects. First, this policy could lead to overallocation or underallocation. Under free allocation, the number of allowances electricity consumers receive depends on the product of two variables: the megawatt hour (MWh) of electricity consumed and the emissions factor for the grid, expressed in tons of carbon dioxide per MWh (Munnings et al. 2014). An inaccurate representation of either variable may lead to misallocation of allowances (i.e., too many or too few allowances). In the context of Shenzhen's pilot, Chai (2013) presents evidence that the grid emissions factor used to allocate allowances was higher than the actual grid emissions factor, which led to overallocation of allowances. It is uncertain whether the remaining pilots, including Shanghai's, have accurately chosen their grid emissions factors, but Chai's analysis warrants a degree of skepticism.[18] Second, dependent on the precise allocation rules, free allocation to consumers of emissions that are embedded in electricity consumers may lead to windfall profits. In some of these sectors, passing costs along to consumers may be quite easy. For example, large buildings, hotels, and retail stores have no direct emissions but may receive an allowance allocation and can pass along costs to their customers (Munnings et al. 2014). Third, it is possible that the national government may allow electricity producers to pass along allowance costs in the future, especially if producers decide to lobby on this issue (Li et al. 2014). To the extent that upstream regulation and incentives from free allocation result in some pass-through of carbon

[18] This points to the value of either setting top-down emissions targets that ensure scarcity or facilitating an auction of at least a portion of the allowances to create a buffer against scarcity.

prices toward power prices, the liability of consumers for indirect emissions needs to be reduced to avoid double taxation.

The coverage of direct and indirect emissions under Shanghai's ETS offers two key insights. First, coverage of direct emissions alone would not induce efficient abatement, because China's electricity sector acts to prevent cost pass-through from electricity producers to consumers. Second, a price on indirect emissions was seen as a potential solution to this problem and motivated policymakers to craft a price on carbon consumption. Notably, economic efficiency remained the primary motivation (Wu et al. 2014) although Shanghai imports 20 percent of its electricity (Jotzo 2013) and therefore emissions leakage represents a potentially significant threat to the program's environmental effectiveness. The Shanghai example therefore highlights a recent but growing trend of pricing carbon consumption as a complement to pricing carbon production and for economic efficiency reasons.

The coverage of direct and indirect emissions under China's ETSs more generally provides at least two lessons learned. First, the design of a price on carbon consumption will tend to remain effective if it updates key parameters and reacts to the larger institutional contexts within which it is nested. Specifically, evidence from Shenzhen underlines the importance of keeping updated data on electricity consumption and the grid emissions factor in order to accurately report carbon consumption. Second, the coverage of direct and indirect emitters is likely effective only in an electricity regime that allows no (or little) cost pass-through of allowances; if the wider electricity regime switches to a competitive one in China, the Shanghai pilot and perhaps others will need to adapt their policies to avoid double taxation of carbon emissions from electricity.

2. Proposal to Price Consumption of Carbon Emissions Embedded in Carbon-Intensive Materials in Europe

A group of academics (including authors of this paper) have proposed a consumption charge on the use of carbon-intensive materials such as cement, steel, or aluminum (Neuhoff et al. 2015). The motivation for this proposal comes from two related factors.

First, iron and steel, cement, and aluminum together account for at least 10 percent of total emissions in many European countries (Neuhoff et al. 2015). Within these sectors, the majority of emissions originate from the production of the primary material; for example, iron accounts for approximately 85 percent of emissions related to steel, and clinker accounts for approximately 90 percent of emissions related to cement. It is increasingly clear that the EU's 2050 objectives to reduce emissions by 80 percent to 95 percent below 1990 levels will not be met in these sectors by improvements in production efficiency alone. Instead, these sectors will need to exploit a multitude of additional abatement options, including greater material efficiency, enhanced substitution in end uses, and increases in investments for breakthrough technologies such as carbon capture and storage and low-carbon cements (Neuhoff et al. 2014b).

Second, it is uncertain whether the EU ETS can incentivize these types of abatement in carbon-intensive sectors, at least as it is currently designed. In terms of material efficiency, the EU ETS currently allocates free allowances to installations producing cement, steel, and aluminum in order to allow them to avoid passing on the cost of their CO_2 allowances through their product prices—all in an effort to maintain competitiveness with importers that are not subject to similar constraints. Such levels of free allocation may not appropriately incentivize abatement from carbon-intensive sectors through pathways such as material efficiency, end use substitution, and investments in low-carbon products. Regarding the third category, the EU ETS would likely have to provide not only a high carbon price to encourage such investments but also confidence to industries that they will be able sell their free allowances over a long period of time to recover the incremental costs of new technologies. Yet levels of free allocation are politically contentious, and the EU ETS is revised relatively quickly (every 5–10 years), making it especially unclear whether it will encourage investments in transformative technologies for carbon-intensive sectors such as steel, cement, and aluminum.

These observations have given rise to a proposal for a different approach to free allocation and carbon charging in these sectors. Carbon-intensive materials would be subject to a consumption charge that would be levied per ton of cement or steel once that good entered the consumption sphere. More specifically, each cement or steel producer would still receive a free allocation of allowances for each ton of cement or steel it produced multiplied by a best available technology benchmark (i.e., an output-based allocation). But in addition, each ton of product would immediately be subject to a CO_2 charge (based on a standard benchmark of CO_2 per ton and the prevailing CO_2 price in the EU ETS). However, the cement producer would be allowed to defer the liability of the charge to the next entity in the value chain purchasing the unit of cement or steel. Each actor in the value chain would be allowed to choose to either pay or similarly defer the charge liability, until the good reaches the final consumer.

The motivation for this approach is that it would introduce a carbon price along the full value chain for purchasers of CO_2-intensive materials, since all would face the liability of the charge and would have to either pay it or effectively pass it on in their sales prices. This would send a price signal that would encourage material efficiency and also ensure that customers of carbon-intensive materials paid for the additional CO_2 cost of these goods. In addition, it would ensure that final consumers bear the cost of carbon externality, and resulting revenues can be used to directly fund innovation toward lower-carbon materials and processes or policy objectives otherwise funded with auction revenue from allowances to upstream emitters.

4. Insights, Motivations, and Lessons Learned

The experience with pricing of carbon consumption, although relatively limited, is significant and provides a number of initial insights, an understanding of policymakers' motivations in implementing prices of carbon consumption, and lessons learned.

From a high-level perspective, it is clear that prices on carbon consumption can be designed in numerous ways. Policies and proposals to date indicate the following.

- Prices on carbon consumption can create incentives to reduce carbon emissions associated with a variety of goods. While most prices on carbon consumption address electricity, other examples already impose prices on carbon emissions associated with transportation fuels (e.g., under California's ETS) and synthetic gases and products containing them (e.g., in Australia), and one could also envisage imposing prices on carbon emissions associated with consumption of carbon-intensive materials (e.g., the recent proposal to do so in Europe).
- Prices on carbon consumption have taken different forms that vary between a pollutions right and tax approach, including Australia's *levy* on carbon emissions associated with consumption of synthetic greenhouse gases; a consumption *charge* for carbon-intensive materials in Europe; the potential proposal to impose a *fee* on carbon emissions associated with the consumption of cement in California; and a *price* on carbon that emerges from an emissions trading system (e.g., the prices on carbon emissions associated with electricity consumption in the South Korean ETS and the Chinese pilot ETS or the price on carbon emissions associated with electricity and transportation fuel imports under California's ETS).
- Prices on carbon consumption can be applied to different flows of goods. Most apply to consumption of domestically produced or foreign goods (imports) in order to supplement preexisting prices on domestic production. One important exemption is Tokyo's ETS, which applies to the domestic production of domestic goods without a corresponding price on carbon production.

From a more idiosyncratic perspective, our review of prices on carbon consumption allows us to make some cursory observations regarding the motivations of policymakers who implement such prices. Specifically, we find the following:

- Policymakers who have priced carbon consumption tend to do so in order to complement a price on carbon production.

- Policymakers have partially justified a price on carbon consumption by arguing that such a price might improve the economic efficiency of carbon emissions abatement, with three notable arguments rising to the top. First, programs that price carbon production in certain parts of Asia (e.g., the Korean ETS or China's pilot ETSs) tend to operate in a context with low or no cost pass-through from electricity producers to consumers, and a price on carbon emissions associated with electricity consumption is viewed as restoring part of that cost pass-through and therefore as improving economic efficiency. Second, Tokyo's price on carbon emissions associated with electricity was justified in part by an argument that placing the burden of compliance on facilities that consume electricity (rather than relying on cost pass-through from electricity producers or making consumers liable) would better incentivize abatement by overcoming principal agent issues (e.g., landlord-tenant issues in large buildings). Third, Australia's price on carbon emissions associated with synthetic gases was justified in part because including consumption effectively increased the portion of the economy facing a carbon price, which in turn increased the economic efficiency of abatement.
- Policymakers have partially justified a price on carbon consumption by arguing that such a price might reduce emissions leakage caused by a price on carbon production. This line of reasoning is especially apparent with California's ETS (which prices carbon emissions associated with electricity and transportation fuels), as it is particularly susceptible to emissions leakage, and the state legislature has tasked the agency implementing the ETS reduce associated emissions leakage. A concern about emissions leakage may also motivate prices on carbon consumption in other ETSs (including some in China) that cover emissions associated with imports.
- No price on carbon consumption explicitly regulates the production of foreign goods. This is likely because no jurisdiction that imposes a price on carbon consumption has legal authority to directly regulate foreign jurisdictions. Laws largely prevent this type of policy intervention on the domestic level (e.g., the DCC of the US Constitution) and perhaps on the international level (e.g., the World Trade Organization). Instead, approaches to pricing carbon emissions associated with domestic production are taken, such as regulating the first jurisdictional deliverers of imported goods (e.g., in the California ETS).

Finally, here are some lessons learned from the experiences of policymakers who have crafted prices on carbon emissions, along with potential best practices:

- Prices on carbon consumption require up-to-date data on relevant goods. For a price on carbon consumption to be effective, the quantity of relevant goods consumed by a regulated firm and the emissions associated with that consumption must be known with reasonable certainty.

- Prices on carbon consumption that are motivated by an attempt to improve economic efficiency may need to adapt to broader economic trends to stay relevant and avoid double taxing regulated firms. A primary example is the Shanghai pilot ETS, which requires that producers and consumers of electricity each retire an allowance for the same ton of carbon emissions, under the premise that cost pass-through from electricity producer to consumer is effectively zero. If China ends up further liberalizing its electricity market, which is under consideration, this might allow electricity producers to pass through costs to electricity consumers—meaning that consumers could be taxed twice for consumption of one ton of carbon emissions. Similar considerations apply under Korea's ETS and may apply to proposals that would price carbon emissions associated with consumption of cement and steel if the nature of the global markets for these products changes to allow significant cost pass-through. However, policymakers can at least partially avoid this double taxation quite easily by moving to either output-based allocation or auctioning of allowances.
- Prices on carbon consumption that are partly motivated by an attempt to reduce emissions leakage (in cases such as California's FJD policy for electricity) may benefit from layering on additional regulatory measures. These additional measures might help stem the impact of perverse incentives that cause foreign firms to attempt to avoid the impact of the price on carbon consumption (e.g., by reshuffling, in the context of California's ETS). However, such policies must be carefully designed to comply with relevant laws (e.g., the DCC in California or the WTO if trade occurs between countries), since they effectively regulate imports.

5. Conclusions

This paper has surveyed a selection of prices on carbon consumption and attempted to describe their designs, understand and explain the motivations of the policymakers who have implemented them, and identify potential best practices and key insights for policymakers considering whether to price carbon consumption in the future.

One clear trend is that prices on consumption are used in conjunction with prices on production by most of the jurisdictions we surveyed. In these jurisdictions, layering a price on carbon consumption on top of a price on carbon production is seen by regulators as a way to improve economic efficiency, increase abatement, and reduce emissions leakage. Thus the increase in prices on carbon consumption at least partially reflect a shortcoming of prices on carbon production: that prices on carbon production do not end up being passed through to consumers for some reason, thereby failing to reduce additional and cost-effective emissions reductions. In addition, our survey suggests that prices on carbon consumption can be used in a way that abides by the law to aid in reducing carbon emissions, help improve economic efficiency, and potentially ameliorate emissions leakage. We expect jurisdictions to increasingly implement prices on carbon consumption, since policymakers view these prices as a strategy to improve economic efficiency and potentially reduce emissions leakage that is compatible with the law.

The in-depth assessment of individual prices on carbon consumption is a subject for future research. Some unanswered questions include estimating the administrative costs of collecting data on carbon emissions embedded in different goods and ensuring enforcement, issues that will differ depending on the jurisdiction (e,g., the precise extent of cost pass-through from producers to consumers) and the degree to which prices on carbon production can improve economic efficiency and reduce emissions leakage relative to alternative policy prescriptions.

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