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INTERNATIONAL SPILLOVERS AND CARBON PRICING POLICIES

Geoffroy Dolphin Michael G. Pollitt

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Globally coordinated climate action has resulted in sub-optimal emissions reductions and unilateral (second-best) climate policies have so far provided the bulk of emissions reductions. This paper argues that the development of new unilateral carbon pricing policies was fostered by international signalling and technological spillover effects. The strength of both effects hinges, for each jurisdiction, on trade relations with other CO2-abating jurisdictions. We provide a stylised theoretical discussion in support of our proposition and investigate it using data on a panel of 121national jurisdictions over the period 1990-2014. Results show a strong positive association between import-weighted exposure to CO2-pricing partners and domestic environmental policy. The analysis also supports the technological spillover channel: trade-weighted installed capacity of wind and solar energy seems to prompt implementation of and more stringent carbon pricing policies.

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Keywords international spillovers, trade, carbon pricing

JEL Classification F18, Q56, Q58

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International spillovers and carbon pricing policies

G.G. Dolphin*[a] and M.G. Pollitt[a]

[a]Cambridge Judge Business School and Energy Policy Research Group, University of Cambridge, Trumpington St, Cambridge CB2 1AG, United Kingdom

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Abstract

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*Corresponding author (gd396@cam.ac.uk). The author gratefully acknowledges the support of the UK Economic and Social Research Council.

1 Introduction

Limiting the increase in Global Mean Temperature to 2°C compared to pre-industrial levels will require drastic reductions in greenhouse gas (GHG) emissions. Since CO_2 is a *global* pollutant, any environmentally effective solution requires a reduction in 'world' emissions. However, no World Government capable of enforcing worldwide reductions in GHG emissions exists. Instead, a multitude of sovereign states interact within the Westphalian system of International Relations and its founding principles (self-determination, legal equality of States and no third-party interference in internal affairs) make cooperation the only available option to efficiently address global public good problems like Climate Change (Barrett, 2003). It is precisely these principles – and their implications – that shaped the United Nations Framework Convention on Climate Change, formally established in 1992.

In line with these developments and following Carraro and Siniscalco (1993), a substantial body of research has explored the conditions for *climate coalition* formation. However, notwithstanding mechanisms to improve the stability of such coalitions (Nordhaus, 2015) and as predicted by standard game theoretical discussions of environmental agreement negotiation (Barrett, 1994), this *top-down* cooperative approach failed to deliver emissions reductions consistent with stated objectives of Global Mean Temperature increase.[1] At best, jurisdictions implement their Nash equilibrium strategy and commit to (very) low, globally sub-optimal, levels of emissions reductions.[2]

The urgency of the climate problem and the relative failure of the multilateral process justify renewed efforts to understand motivations for (and implications of) unilateral, second-best, GHG-emissions abatement. This paper contributes to that end by emphasising the role of abatement technology and its critical dependence on foreign technological and policy developments.

It therefore sheds light on the (external) effects of unilateral measures. As argued by Sim-

[1]If fully implemented, current Intended Nationally Determined Contributions (INDCs) submitted to the UNFCCC Secretariat place the world on an emissions path that is incompatible with least-cost 2 C scenarios, the goal stated in the Accord (United Nations/Framework Convention on Climate Change, 2015). Compared with the emission levels under least-cost 2 C scenarios, aggregate GHG emission levels resulting from the implementation of the INDCs are expected to be higher by 8.7 (4.5 to 13.3) Gt CO2eq (19 per cent, range 9-30 per cent) in 2025 and by 15.2 (10.1 to 21.1) Gt CO2 eq (36 per cent, range 24-60 per cent) in 2030 (United Nations/Framework Convention on Climate Change, 2016).

[2]Incentives for unilateral provision of global environmental quality beyond the Nash equilibrium outcome have so far proven relatively weak. These can be broadly grouped into altruistic (e.g. self-enforcing collective identity (Olson, 1965), rule utilitarianism (Harsanyi, 1977), different domestic preferences, or genuine care for the global environment) and self-interested (e.g. strategic innovation,. . .).

mons and Elkins (2004), such effects can lead to a process of policy diffusion through two main channels: one whereby mounting adoption of a policy by other jurisdictions alters the (net) benefits of adoption, another through which policy adoption provides information about the (net) benefits of policy implementation.

We apply this approach to the climate policy context and follow Mideksa (2016) in identifying two such effects: a *signalling* effect, whereby a jurisdiction's action signals the low cost of abatement to others; and a (technological) spillover effect, whereby (abatement) technology adoption by one jurisdiction, alters the cost of abatement (see, e.g., Heal (1993)). These two channels constitute powerful mechanisms of policy diffusion.

This paper focuses on those effects and argues that (recent) climate policy developments are directly related to them. Besides, we claim that their strength depends on the nature and intensity of the trade relationship with partners endowed with more advanced abatement technology and/or that put an explicit price on carbon. [For example, De La Tour et al. (2011), p.761, observe that "Chinese producers have acquired the technologies and skills necessary to produce PV products through two main channels: the purchasing of manufacturing equipment in a competitive international market and the recruitment of skilled executives from the Chinese diaspora who built pioneer PV firms".]

This approach offers a different perspective on the role of trade in the development of carbon pricing policies. While previous work has emphasised the potential environmental ineffectiveness of unilateral climate policy tightening (Babiker, 2005) and/or the additional domestic cost for trade-exposed sectors (Jaffe et al., 1995), we present a more optimistic view.[3]

We develop this view within the context of a static general equilibrium trade model with global pollution. Unlike Copeland and Taylor (2005), we do not rely on changes in world prices induced by the rich North to generate policy changes in the relatively poorer South. Rather, using an adapted version of the model in Copeland and Taylor (2003) we show how (trade-weighted) technological and demonstration spillovers can prompt a change in domestic pollution policy.

We find some evidence that technological development in abatement in neighbouring jurisdictions positively influences the probability of implementation of a carbon pricing mechanism.

[3]The only exception to this arguably pessimistic view in previous literature is Copeland and Taylor (2005) who suggest that changes in world prices resulting from unilateral action by a rich North can lead to self-interested policy tightening (i.e. increased abatement) in the South.

Overall, we find an especially strong effect of import channels in the diffusion of policy (and technology) across jurisdictions.

The paper is organised as follows: section 2 reviews the literature on trade and the environment, section 3 introduces the formal framework, section 4 presents the empirical methodology and hypotheses. Section 5 discusses the results. Finally, sections 6 and 7 highlights potential implications of the present work and conclude, respectively.

2 Trade and environmental quality: a review

The nexus between trade and environmental quality has long received close attention – not least because of its important policy implications. The related literature identifies two hypotheses (Copeland and Taylor, 2003). The *pollution haven hypothesis* states that, insofar as environmental regulation raises the cost of manufacturing goods, pollution-intensive economic activity will relocate to jurisdictions with lower environmental standards; and that international trade may exacerbate this effect by inducing a "race-to-the-bottom". The *factor endowment hypothesis* claims that standard forces such as factor endowments and technology determine the pattern of trade, not (only) environmental policy (Copeland and Taylor, 2003).

Most of the literature has either sought to strengthen the theoretical foundation of those hypotheses or to test their empirical validity. Several empirical studies have provided evidence in support of the second and, de facto, cast serious doubt on the first (Tobey, 1990; Grossman and Krueger, 1993; Jaffe et al., 1995). However, the existence of a pollution haven *effect* has been theoretically demonstrated and empirically tested (see, e.g., Levinson and Taylor (2008) for a partial equilibrium analysis).

These studies are part of a strand of literature concerned with how environmental regulations affect trade flows. Other studies ask whether trade flows affect environmental quality. Antweiler et al. (2001) develop a theoretical model that breaks down the effect of trade on the latter into scale, composition and technique effects and reach the conclusion that, when accounting for all three effects together, freer trade is good for the environment.[4]

[4]The scale effect is the channel whereby, for a given production technology and composition of the economy, an increase in the real output of a pollution producing sector will result in higher absolute amounts of pollution discharge. The composition effect refers to the fact that opening up to international trade increases the relative size of the dirty sector in countries endowed with the factor of production most intensively used in that sector while the technique effect points at the change in the technology of production. The question then becomes whether the negative impact of the scale effect is outweighed by the positive effect of the *composition* and

Within that context, a more positive hypothesis, the *gains from trade hypothesis* (Frankel, 2008), posits that trade may improve environmental quality: if trade allows countries to raise their GDP per capita then, under the assumption that environmental quality is a normal good, increased exposition to international trade flows might lead to improved environmental conditions. But Frankel and Rose (2005) also argue that there are at least three other ways in which trade may positively affect environmental outcomes, even for a given level of GDP per capita. First, trade might facilitate innovation, including environmentally friendly innovation. In this respect, Grossman and Helpman (1991) have previously argued that knowledge varies according to the number of contacts between domestic and foreign agents and that these contacts are directly proportional to trade flows. In the context of climate mitigation, this means that abatement technologies or policies developed in partner-jurisdictions can be "translated" into domestic technological and policy developments.[5]

Second, one may observe the international ratcheting of environmental standards: when a "significant" jurisdiction introduces more stringent environmental standards, others might follow suit.[6] Reasons for mimicking a jurisdiction abound. For example, environmental pollution abatement cost is often unknown or, at least, highly uncertain. Yet, some jurisdictions that tackled environmental problems in the past may have better knowledge about available abatement technologies and associated costs than others. In that case, *inaction* may simply reflect a lack of accurate information about actual abatement cost and abatement by better informed jurisdictions may serve as a signal about the (low) cost of abatement activities.

Third, multinational corporations can bring home country clean production techniques to host countries.

3 A two by two model of international trade

To support our empirical investigation we adapt a multi-country general equilibrium model of international trade ($n > 2$) with transboundary pollution as in Copeland and Taylor (2003). The model is static, productive factors are in inelastic supply and environmental quality is a

technique effects.

[5]See, e.g., De La Tour et al. (2011). Parrado and De Cian (2014) provide more evidence of such effects and use it to calibrate a CGE model, finding a small aggregate net effects of technological spillovers.

[6]The [legal] literature on environmental policy refers to this effect as the 'California' effect. See, e.g., Vogel (1995); Perkins and Neumayer (2012).

global public good. Jurisdictions are indexed by $i = 1, 2, \ldots, n$. Assume that n is large and that all countries have the same *relative* size so that each country cannot, individually, influence its terms of trade. In other words, each country is a price taker on world markets and prices are determined by the Rest of the World (ROW). Factor endowments vary across countries and determine trade patterns.

3.1 Technology

We distinguish between primary factors of production and consumption goods (Dixit and Norman, 1980). Our analysis will be conducted within a two factors $\mathbf{r} = (r_1 = K, r_2 = L)$ - two goods $\mathbf{t} = (t_1 = x, t_2 = y)$ model of international trade. Primary factors are non tradable while goods are. Labour is mobile across sectors but not across countries.

We assume constant returns to scale technology (CRS) for both goods. That is, the set of technologically feasible (r, t), T, is convex. The production of good x generates pollution as a by-product while the production of good y doesn't.[7] The production function of y is:

$$y = F(K_y, L_y) \tag{1}$$

where F is increasing, concave, and linearly homogeneous.

In industry x, firms produce potential output $B(K_x, L_x)$ and can choose to redirect a fraction $\phi \in [0, 1]$ of inputs to the abatement process, which will, in turn, reduce output of good x. In other words, the net production of x is the difference between potential production and production foregone due to the use of resources in abatement activity $(\phi K_x, \phi L_x)$. As a result, emission intensity in that sector is a choice variable. The joint production of x and e is given by

$$\begin{aligned} x &= B(K_x, L_x) - B(\phi K_x, \phi L_x) \\ &= (1 - \phi) B(K_x, L_x) \end{aligned} \tag{2}$$

$$e = \Omega \chi(\phi) B(K_x, L_x) \tag{3}$$

[7] This is without loss of generality and it can easily be extended to a context with $m > 2$ goods exhibiting different emissions intensities. See Levinson and Taylor (2008) for a partial equilibrium example and Copeland and Taylor (1994) for a General Equilibrium discussion.

The second line of equation (2) follows from the CRS assumption. More abatement efforts lead to less emissions, i.e. $\frac{d\chi}{d\phi} < 0$. In the absence of abatement ($\phi = 0, \chi(\phi) = 1$), each unit of good x produces Ω units of pollution; if ϕ is equal to 1 (and $\chi(\phi) = 0$), then all resources are devoted to abatement and no production takes place. Ω can be interpreted as a technological parameter for the abatement activity. A decrease in Ω then denotes an improvement in the abatement technology (Brock and Taylor, 2010).[8]

To simplify the analysis, we follow Copeland and Taylor (2003, 2004) and treat pollution as an input to the production process of good x. From (3), we note that $\phi = \chi^{-1}[e/(\Omega B(K_x, L_x))]$. It is then easy to see that

$$\begin{aligned} x &= (1-\phi)B(K_x, L_x) \\ &= \left(1 - \chi^{-1}\left[\frac{e}{\Omega B(K_x, L_x)}\right]\right) B(K_x, L_x) \end{aligned} \tag{4}$$

with $\partial\chi^{-1}(.)/\partial e < 0, \partial\chi^{-1}(.)/\partial B(.) > 0$.[9]

If we impose some more structure on $\chi(\phi)$ and define $\chi(\phi) = (1-\phi)^{1/\alpha}$ we can rewrite (4) as

$$x = \left(\frac{e}{\Omega}\right)^{\alpha} B(K_x, L_x)^{1-\alpha} \tag{5}$$

where e/Ω is the *effective emissions* input.

Three observations can be noted from equation (5). First, as emissions per unit of potential output (Ω) decrease, net output increases. That is, for a given e, as the abatement technology improves, the production of the dirty good expands. This is because improvements in abatement technology free up resources that were previously devoted to abatement and makes them available for actual production.

Second, as abatement technology improves, the emissions intensity of the economy decreases.

[8] In Copeland and Taylor (2003), Ω is constant and, by choice of units, set equal to 1.

[9] Define $C \equiv e/B(K_x, L_x)$. By the inverse function theorem, we know that $\chi^{-1}(.)$ satisfies $\partial\chi^{-1}(.)/\partial C < 0$. By definition of C, we have $\partial C/\partial e > 0$ and $\partial C/\partial B(.) < 0$. Hence we must have $\partial\chi^{-1}(.)/\partial e < 0, \partial\chi^{-1}(.)/\partial B(.) > 0$. This leads to the following observations: first, an increase in emissions raises total output of good x; second, an increase in potential output $B(.)$ affects total output via two channels, a production channel and an abatement channel. The first one straightforwardly tends to raise production, higher potential production leads to higher actual production. The second tends to lower actual production and is more indirect: $\chi(\phi)$ gives the abatement efforts as a function of the ratio of unabated to total potential emissions. Hence when potential production (and emissions) increases, that ratio decreases, for a given level of actual emissions. This requires an increase in abatement efforts which, in turn depresses actual output. [Whether one or the other effect dominates is an empirical question but it seems plausible to assume that the former outweighs the latter].

This observation uses a standard implication of Cobb-Douglas production functions, i.e. that the share of payments in total value added to a factor of production is equal to the associated output elasticity parameter.

$$\frac{\delta \frac{e}{\Omega}}{px} = \alpha \Leftrightarrow i \equiv \frac{e}{x} = \frac{\alpha \Omega p}{\delta} \tag{6}$$

where δ is the price of emissions and p is the relative price of good x (see section 3.3).[10]

The third observation is summarised in the following proposition.

Proposition 1. *The effect on the net output of x of a change in pollution emissions decreases in Ω. That is* $\left|\frac{\partial x}{\partial e}\Big|_{\Omega^{Low}}\right| > \left|\frac{\partial x}{\partial e}\Big|_{\Omega^{High}}\right|$.

Proof. The cost of tightening pollution policy in sector x is driven by the diversion of resources from actual production to abatement activities. From (5) it is easy to see how net output changes as a result of a change in allowed emissions:

$$\frac{\partial x}{\partial e} = \alpha \frac{e^{\alpha-1}}{\Omega^{\alpha}} B(K_x, L_x)^{1-\alpha} > 0 \tag{7}$$

which increases as Ω decreases. Although this might appear counter-intuitive, it reflects the increased opportunity cost of reducing emissions when the economy is already very efficient at abating. □

3.2 Abatement, signalling and technological spillovers

As integrated assessment modelling exercises show (e.g. Kriegler et al. (2014)) and as captured by equation (3), abatement technology is a key determinant of the economy's (optimal) level of emissions. Therefore, how this abatement technology is developed and accumulated domestically is crucial to understand the evolution of its CO2 emissions. In this paper, we focus on learning from *foreign* policy signals and technological developments.[11]

Signalling The signalling effect of emissions abatement or policy development will be most important when information about the abatement technology (and related cost) is private. In

[10] As is evident from equation (6), CO2-intensity depends on both policy (δ) and technology (Ω). Appendix B discusses that relationship further and presents the evolution of CO2-intensity for selected sectors and jurisdictions.

[11] As noted by Fankhauser et al. (2016), the process of policy diffusion can occur through knowledge spillovers, learning effects and peer pressure.

that case, the (successful) implementation of the abatement policy reveals that information to other jurisdictions which might, in turn, adjust their expectation about their own abatement cost. In the case of environmental policies, it is safe to assume that jurisdictions with prior experience in tackling environmental problems have better information about the abatement cost.[12] For example, at the international level, one can think of the EU-ETS (or RES policies) as signalling the cost of set abatement commitments to other countries; at the sub-national level, California's ETS might be thought of playing a similar role with respect to other US States.

Technology The signalling effect is, however, not the only one at play and is likely to be linked to abatement technology development and, hence, spillover. Unlike signalling, which is understood as affecting expectations about the domestic abatement technology stock, technological spillovers alter the domestic technological base. Evidence of such effects has been found both at a general level (Bloom et al., 2013) and for environmental technologies specifically (Dechezlepretre and Glachant, 2011).

We follow the literature on international R&D and knowledge spillovers (Grossman and Helpman, 1991) and assume that (the strength of) those effects are linked to bilateral trade relationships. There is strong theoretical support for and substantial empirical evidence of the importance of the trade channel for knowledge and technology diffusion. For example, Coe and Helpman (1995) identify four channels through which trade may promote growth. First a country can import intermediate goods which enhance productivity. Second, countries inside the technological frontier may imitate the products of frontier countries. Third, trade can encourage more efficient employment of resources through learning. Finally, international contacts can stimulate new indigenous technologies. We believe that similar channels can play a role in the diffusion of abatement technology and policy.

To account for (foreign) signalling and technological developments on the domestic abate-

[12]The strength of the signal might depend on the nature (and stringency) of the scheme: targets set in ETSs are as much a reflection of information about abatement technology cost as willingness to pay for abatement, whereas information provided by RES targets pertains more directly to abatement technology cost. This paper focuses mainly on the signalling effect of explicit carbon prices, set either trough taxes or ETSs, and hence does not discuss any potential specific signal arising from implementation of RES-policies.

ment technology, we alter the framework presented in section 3.1 and introduce a dependency between domestic abatement technology on the one hand, and the state of abatement technology and (carbon-pricing) policy in partner jurisdictions, on the other. That is, Ω now depends on jurisdiction i's exposure to its trading partners' *abatement technology* stock, denoted $\bar{\kappa}$, and *climate policy signal*, denoted σ. This jurisdiction-specific trade-related effect is denoted by ψ_i and is defined as a function of the trade-weighted aggregate of all learning from trading partners

$$\psi_i \equiv C\left(\sum_{h\in\Theta}\Gamma_{i,h}\sigma_h, \sum_{h\in\Theta}\Gamma_{i,h}\bar{\kappa}_h\right) \tag{8}$$

where Θ is the set of all trading partners, $\Gamma_{i,h}$ is the partner-specific trade-weight, σ_h is the signal received from partner h, and $\bar{\kappa}_h$ is the partner-specific abatement technology stock of trade partner h.

Taking the above into account, we alter the modelling framework and equation (3) is adapted accordingly:

$$e = \chi(\phi)\Omega(\psi)B(K_x, L_x) \tag{9}$$

with $0 < \Omega(\psi) \leq 1$ and where, for the sake of clarity, we have dropped the subscript i.[13] We assume that higher access/exposure induces an improvement in abatement technology, i.e. $\frac{\partial\Omega}{\partial\psi} < 0$. For given levels of production and abatement, a decrease in Ω induces a decrease in emissions. As before, $\chi(0) = 1$, $\chi(1) = 0$.[14] Note that if a country has no carbon-pricing trading partner to learn from or if it does not participate in international markets, $\psi = 0$ and we are back in the standard model.

3.3 Production decision and pollution demand

Equipped with these technological priors, we can now look at the production decisions of firms.[15] Good y is the numeraire with price p_y normalised to 1. We denote the relative price of good

[13] Restricting $\Omega(\psi)$ to values below or equal to 1 ensures that emission intensity is below or equal to 1 and avoids complications in the firm's profit maximisation problem.

[14] Note that adopting this specification is equivalent to assuming an explicit pollution abatement function. To see this, define the abatement technology as $A(e^P, v^A)$ where e^P is the potential amount of pollution produced and v^A is the (absolute) amount of resources allocated to abatement. $A(.)$ is a CRS activity. Then, $e = e^P - A(e^P, v^A) \Leftrightarrow e = e^P(1 - A(1, v^A/e^P))$. Now, recall that without abatement activity, $e^P = x = \Omega(\psi)B(.)$ and that $v^A/B(.) = \phi$. Hence $e = \Omega(\psi)B(.)(1 - A(1,\phi))$ where we have defined $(1 - A(1,\phi))$ as $\chi(\phi)$.

[15] The detailed production decision problem of firms in sectors x and y is presented in appendix A.

x in terms of good y as p. The optimal output vector $\mathbf{t} = (x, y)$ will depend on primary input endowments, $\mathbf{r} = (K, L)$, output prices, $\mathbf{p} = (p, 1)$ and, for pollution emitting sector(s), emissions e. That is, the firms' problem is

$$\max_{\mathbf{t}}\{\mathbf{p}.\mathbf{t} \mid (t, r) feasible\}$$

Since input factors (K,L) are supplied inelastically, the firms' decision determines the relative allocation of inputs to each sector. In the dirty good sector, the firm faces the additional decision of how much of these resources to devote to abatement. The solution to this problem defines the optimum (technologically feasible) vector of output

$$\hat{\mathbf{t}} \equiv t(\mathbf{p}, \mathbf{r}) \tag{10}$$

Consequently, the (maximum) revenue function can be defined as

$$g\left(p, K, L, \frac{e}{\Omega(\psi)}\right) = \mathbf{p}.t(\mathbf{p}, \mathbf{r}) \tag{11}$$

The revenue function is convex in $\mathbf{p}$, $\bigtriangledown_{pp} g(\mathbf{p}, \mathbf{r}) > 0$, but concave in $\mathbf{r}$, $\bigtriangledown_{rr} g(\mathbf{p}, \mathbf{r}) < 0$.[16] In addition, it is increasing and concave in e ($\partial g(\mathbf{p}, \mathbf{r})/\partial e > 0$, $\partial g(\mathbf{p}, \mathbf{r})/\partial^2 e < 0$) but decreasing and concave in Ω ($\partial g(\mathbf{p}, \mathbf{r})/\partial \Omega < 0$, $\partial g(\mathbf{p}, \mathbf{r})/\partial^2 \Omega < 0$).[17] That is, as the abatement technology deteriorates, revenue falls at an increasing rate.

If we further assume that profit-maximising firms maximise national income, this revenue function can be interpreted as the *national income function*, $G(p, \delta, K, L, \frac{e}{\Omega(\psi)})$ Copeland and Taylor (1994, 1995). Hence we write

$$I \equiv G\left(p, K, L, \frac{e}{\Omega(\psi)}\right) = \max_{x,y}\left\{\mathbf{p}\cdot\mathbf{t} : \mathbf{t} \in T(K, L, \frac{e}{\Omega(\psi)})\right\} \tag{12}$$

The national income function preserves all the properties of the revenue function.

[16]For an informal justification of this statement, see Dixit and Norman (1980), p.31.

[17]From (5) we know that for a given level of emissions, e, and capital & labour, $K\&L$, firms in the dirty sector can expand production when $\Omega(\psi)$ decreases. In appendix A, we show that this expansion in potential output leads to an increase in net output through a reallocation of resources from the clean to the dirty sector. Moreover, the technological improvement will reduce pollution demand and depress equilibrium price of emissions which, in turn, will reduce resources allocated to abatement. Both effects work toward an increase of the net output in the dirty sector.

It is useful to note the relationship between the national income function and the price of emissions. For given prices and factor endowments, we can define the value of a pollution permit as the marginal effect on national income of additional pollution:

$$\delta \equiv \frac{\partial G(\mathbf{p}, \mathbf{r})}{\partial e} \tag{13}$$

δ is the opportunity cost of emissions or, put differently, the cost (in terms of lost national income) of reducing emissions by one – infinitely small – unit; equation (13) gives the demand schedule of firms for pollution which, since $G(.)$ is concave in e, is decreasing.

Proposition 2. *For a given scale of the dirty good sector, an improvement in the abatement technology reduces pollution demand. That is,* $\frac{\partial G(\mathbf{p},\mathbf{r})}{\partial e \partial \Omega} > 0$

Proof. First, note from equation (6) that the demand for pollution can be expressed as the emissions intensity times the production of good x, i.e. $e = i(p, \delta, \Omega) \times x(p, K, L,)$. Now, using equation (6) again, it is easy to note that an improvement in abatement technology (i.e. a decrease in Ω) leads to a decrease in emissions intensity. Hence, for a given level of production in the x sector, an improvement in abatement technology decreases demand for pollution. □

3.4 Consumers

Let us assume the existence of N identical consumers in each country.[18] Consumers derive utility from the consumption of both goods and incur disutility – i.e. damage – from global pollution E. The utility function is strongly separable with respect to consumption goods and environmental quality. Each consumer of jurisdiction i has the following utility[19]

$$U^i \equiv U^i(x, y, E) = u^i(x, y) - D(E) \tag{15}$$

[18] That is, we abstract from political economy and distributional considerations. See Copeland and Taylor (2004) for such an approach.

[19] Note that equation (15) assumes that the consumer does not derive any utility from global environmental quality. One could take this form of altruism into account by attributing a strictly positive weight to the damage that domestic emissions impose on other jurisdictions. That is, e.g.,

$$U^i \equiv U^i(x, y, E) = u^i(x, y) - [\alpha D_1(E)] + \beta D_2(E)] \tag{14}$$

where $\beta = 1 - \alpha < 1$ and D_1 and D_2 denote domestic and foreign (or world) environmental damage, respectively. Care for the global environment will reduce equilibrium emissions level.

where $E = \sum_i e_i$ and e_i denotes the emissions of jurisdiction i. $u^i_x(x,y)$, $u^i_y(x,y) \geq 0$, $u^i_{xx}(x,y), u^i_{yy}(x,y) < 0$ and $D'(E) > 0, D''(E) > 0$. Note, in addition, that $u^i(x,y)$ is homothetic.[20] Consumers maximise utility given goods prices – which determine the revenue function specified by (11) – and (global) pollution levels.

Using duality, we can write consumer i's indirect utility function, which gives the maximum utility attainable for given prices and income, as:

$$V^i \equiv V(\mathbf{p}, I, E) = v(\mathbf{p}, I) - D(E) \tag{16}$$

Consumers earn their revenue from their ownership of factors of production, capital and labour, which are remunerated at the equilibrium market rate. In a perfectly competitive economy, the total value of payments to all factors of production is equal to the maximum value of production. It will thus depend on the composition of the economic production, the price at which said production is sold and environmental policy.

Eventually, using the homotheticity assumption, function $v(.)$ can be written as a function of real income.

$$V^i(\mathbf{p}, I, E) = v(\mathbf{p}, I) - D(E) = v(1, I/\omega(\mathbf{p})) - D(E)$$

$$V^i(R, E) \equiv v(R) - D(E) \tag{17}$$

where $\omega(\mathbf{p})$ is a price index.

3.5 Pollution supply

As in Copeland and Taylor (2003), we adopt the point of view of a small country whose environmental policy has no influence on world prices. There is thus no leakage effect due to changes in world prices. We start our investigation by assuming $\psi_i = 0$. That is, the free-riding problem is only driven by the direct, strictly positive, costs of emissions reductions and the benefits consumers derive from domestic abatement.

We consider a noncooperative Nash Equilibrium where pollution policy is endogenous and decided by a self-interested government, which maximises the utility of a representative con-

[20] With homotheticity, the analysis is simplified in two ways. First, the indirect utility function can be written as an increasing function of real income. Second, it ensures that relative consumption patterns do not change with income which, in turn, makes trade patterns dependent on factor endowments and relative costs only (Copeland and Taylor, 2003).

sumer given world prices and Rest Of the World (ROW) emissions. Government policy is cast in terms of pollution targets, e_i. The problem of the government is as follows:

$$\max_{e_i} \quad V^i(R,E) \tag{18}$$

$$s.t.: \qquad R = [G(p,K,L,e_i)]/\omega(\mathbf{p}) \tag{19}$$

$$E = E_{-i} + e_i \tag{20}$$

where E_{-i} is the total aggregate emission of all jurisdictions bar the emissions of jurisdiction i. The optimality condition of this maximisation problem is:

$$\underbrace{V_R R_e}_{(1)} + \underbrace{V_R R_p p_e}_{(2)} + \underbrace{V_e}_{(3)} = 0 \tag{21}$$

That is, the government's decision reflects the tradeoff between the direct effect of emissions change on the nation's real income (1), the effect of emissions change on the consumer's utility (3) and the effect of the induced change in the price of the dirty good on real income (2). However, with exogenous world prices, (2) is equal to zero because there is no real income effect of a change in domestic prices. Hence

$$R_E = \underbrace{-V_E/V_R}_{\equiv MD(R,E)} \tag{22}$$

with $V_e < 0$ and $V_R > 0$.

Equation (22) defines the optimal level of emissions e_a^*: the optimal level of pollution is such that the marginal benefit of increased emissions (i.e. the resulting increase in real income) is equal to the domestic marginal damage of pollution [Samuelson rule].

It is then straightforward to see that if there is only one (World) jurisdiction, the environmental externality is fully internalised and the supply of emission permits is such that the price of emissions is equal to the global marginal damage of emissions.

Allowing for spillovers in abatement technology ($\psi_i \geq 0$) will affect the pollution demand schedule which, recalling proposition 2, shifts downwards.[21] That is, as emissions per unit of output of the dirty good decreases, and for a given level of endowment, goods and emissions

[21]Equation (19) in the government's planning problem becomes $R = [G(p,K,L,\frac{e_i}{\Omega(\psi_i)})]/\omega(\mathbf{p})$.

prices, the emission intensity of the economy declines and hence, for a given 'scale' of domestic production of x, so too do total emissions.[22] Hence, in the context of a small open economy, equation (22) defines a lower emissions equilibrium.

Proposition 3. *Assuming that the scale effect is smaller than the technique effect, an improvement in abatement technology reduces equilibrium emissions.*

Proof. From proposition 2 and appendix A, we know that an improvement in abatement technology induces both a *technique* and *scale* effect. For a given price of emissions, the former lowers total emissions in the dirty sector whereas the latter raises them. Formally, these two effects are apparent in $e = i(p, \delta, \Omega) \times x(p, K, L,)$. Assuming that the decrease in emissions intensity more than outweighs the rise in dirty good production, total emissions will fall. $\square$

4 Empirical strategy

4.1 Hypotheses

Section 3.2 introduced a formal relationship between the domestic abatement technology stock on the one hand, and foreign technological and policy developments on the other – see equation (8) – while sections 3.3, 3.4, and 3.5 enable us to relate those changes to the optimal level of emissions – equation (22). Indeed, signalling and technological spillovers improve domestic abatement technology. This, in turn, leads to reduced demand for emissions in the dirty sector and, ultimately, to a downward adjustment of equilibrium emission level.

The empirical analysis relates foreign signalling and technological developments (eq. (8)) to domestic policy developments (eq. (22)). This section discusses the spillover channels in greater details and formulates the related hypotheses.

Signalling Based on the discussion above, it is hypothesised that foreign climate policy development and, in particular, carbon pricing, encourages the development of domestic carbon pricing policy, either through peer pressure (Fankhauser et al., 2016) or (low) abatement cost signalling.[23] Regarding the former, partner jurisdictions that price CO2-emissions signal a

[22] As it keeps the scale of the dirty good sector constant, this does not account for GE effects.

[23] As detailed further in section 4.2.2, the former is captured by a binary variable capturing the presence/absence of a carbon pricing scheme (which only effect is to signal a strictly positive willingness to pay for carbon) whereas the latter is measured by the level of the price signal.

strictly positive willingness to pay for emissions reductions and might use this [positive signal] to induce (trade) partners to follow suit. Hence, when more partner jurisdictions price carbon or when trade with carbon pricing jurisdictions increases, so too does the signal that the home country receives which, in turn, is expected to increase both the probability of implementation and the stringency of (domestic) carbon pricing schemes. However, one could also expect to observe a form of free riding effect whereby policy initiatives by other (significant) jurisdictions reduces the incentive to act. In that case, jurisdictions exporting to partners with more stringent environmental policy might be induced to weaken their own in order to strengthen their comparative advantage. We do not, a priori, expect the impact of the signal to differ significantly depending on whether it is channeled via imports or exports.

Regarding the latter, the observed price signal leads the home jurisdiction to revise its belief about the cost/quality of abatement technology. [Indeed, in combination with observed/nationally determined emissions (reductions), the price provides information about the stringency of the implemented pricing scheme and the related abatement cost.[24] That is, for a given emissions (reduction) target, a higher price implies a higher (marginal) abatement cost.]

Technological spillover The main hypothesis related to technology is that improvements in foreign abatement technology spill-over to the domestic economy and induce more stringent domestic environmental policy. Early literature on trade and international knowledge spillovers has emphasised the importance of the import channel but, as noted by Falvey et al. (2004), there is no reason why spillovers should be limited to imports. However, the nature of the spillover mechanism associated with each channel differs. Imports (of intermediate goods) embody foreign knowledge that is extracted by the recipient country and contributes to the domestic stock of abatement technology. Exports, on the other hand, emphasise "learning-by-doing" and the "pure idea exchange and knowledge spillovers gained from formal and informal contacts"(Funk, 2001). Besides, competition in international markets might drive domestic exporters to acquire and adapt foreign technologies.[25]

Given the above, we expect our trade-weighted proxies for foreign technological development

[24] Another, equivalent, interpretation of the effect of the signal on the part of the domestic economy is that it starts mimicking the technology of the partner jurisdiction.

[25] Evidence of a 'trading up' effect, i.e. the fact that greater exports to jurisdictions with more stringent (environmental) regulations leads to a strengthening of domestic regulations, has been provided by Perkins and Neumayer (2012) for the automotive industry.

to relate positively to the probability of implementation of a carbon pricing scheme.

To test those hypotheses, we use variables that capture the signalling and technological spillover effects, respectively. These variables are described in the following subsections. Our dataset covers the period 1990-2014 in four sectors of the economy for 121 national jurisdictions.

4.2 Data

4.2.1 Policy developments – dependent variable

We investigate the above hypotheses on two policy decisions, implementation and stringency. To that end, we consider two approaches. First, we consider a logistic regression model and analyse the effect of the above variables on the probability of implementation of carbon pricing policies

$$\mathcal{I}_{i,t} = \beta \mathbf{X}_{i,t} + \gamma \mathbf{Z}_{i,t} + \epsilon_{i,t}$$

where $\mathcal{I}_{i,t}$ denotes the presence (1) or absence (0) of a carbon pricing scheme in any sector of jurisdiction i in year t, $\mathbf{X}$ is the set of variables capturing the signalling and technological spillover effects whereas $\mathbf{Z}$ is the set of 'control' variables.

Second, we test these hypotheses for the stringency of carbon pricing using our Emissions-weighted Carbon Price (ECP) as dependent variable. Indeed, the price signal per se, while providing useful information regarding the abatement behaviour in each sector, is unhelpful when trying to gauge the stringency of the policy for the economy as a whole. For this, total CO2-emissions in each sector is needed. Hence, to capture carbon pricing policy developments at the economy level, we use an Emissions-weighted Carbon Price (ECP) – see Figure 1.[26]

We estimate the following equation

$$ECP_{i,t} = \beta \mathbf{X}_{i,t} + \gamma \mathbf{Z}_{i,t} + \epsilon_{i,t}$$

where $ECP_{i,t}$ is the emissions-weighted average carbon price in jurisdiction i at time t and $\mathbf{X}$ and $\mathbf{Z}$ are as before.

[26]The methodology used to compute the ECP is described in Dolphin et al. (2016). The sectoral classification follows the International Standard Industrial Classification of All Economic Activities (ISIC), Rev.4.

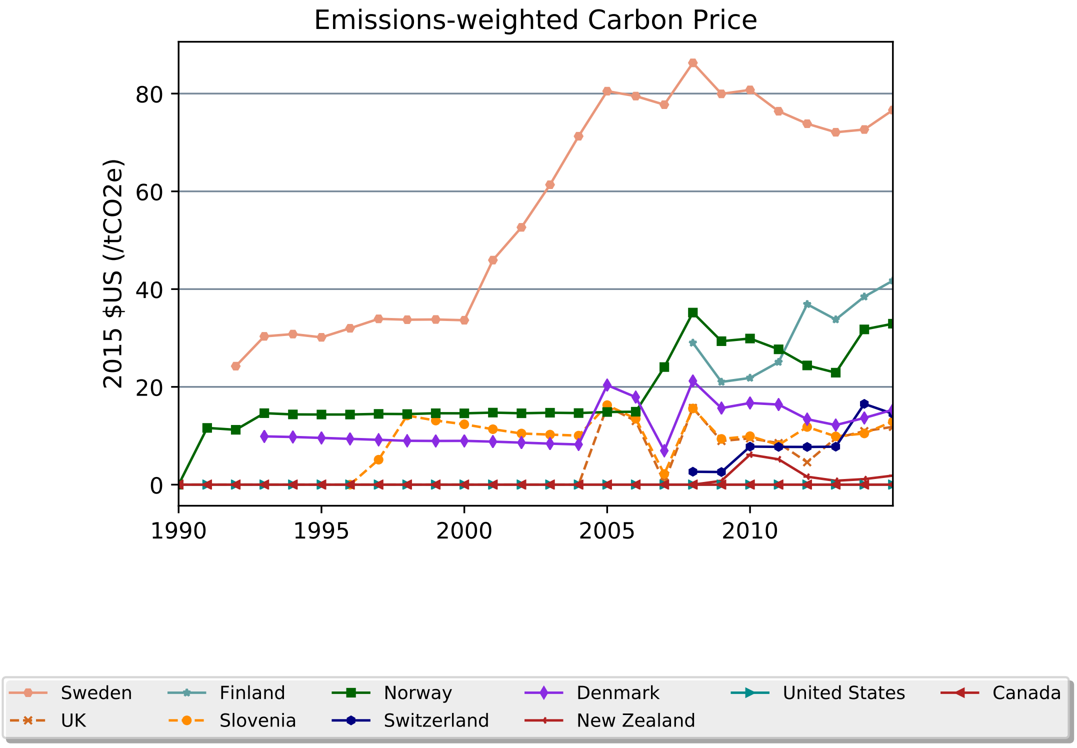


Figure 1

4.2.2 Signalling

The informational signal that each jurisdiction sends by implementing climate policies is captured by three variables. The first variable records the cumulative number of Climate Laws passed in a given jurisdiction up to year $t-1$ and is constructed using the Climate Change Laws of the World database (2016).

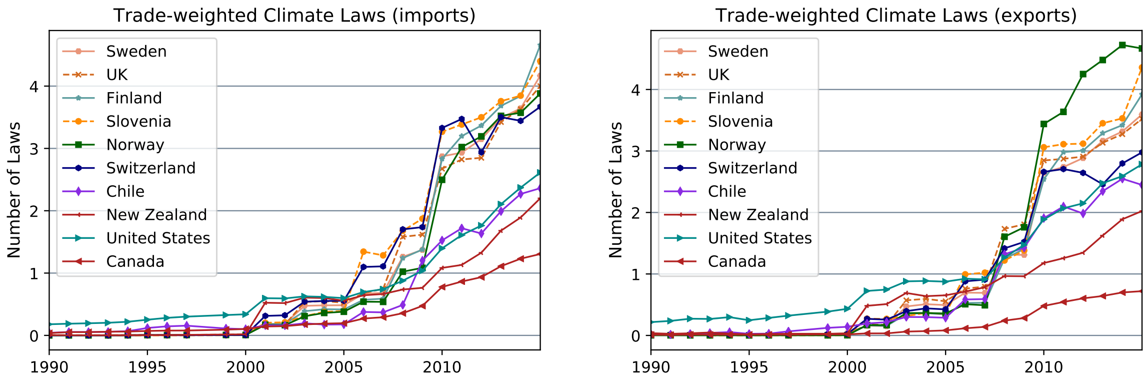
The second and third variables capture the signalling effect of carbon pricing policies specifically. The former is an author-created (sector-level) dummy variable that takes value 1 if the jurisdiction has established a carbon pricing scheme in the sector, and 0 otherwise. The latter is the price of CO2-emissions. As noted in the formal discussion, the price of polluting emissions relates directly to abatement efforts, i.e. the share of resources devoted to abatement.[27] A strictly positive price provides an informational signal regarding the amount of resources that

[27] As acknowledged in section 3.2, it might seem difficult to derive from prices any information about the state of the abatement technology. First, in a quantity based system, (low) polluting emissions price can reflect either "low ambition" and poor abatement technology or "high ambition" and good abatement technology. Second, in a price-based system, a (low) polluting emissions price induces a set level of abatement effort but can result in different amounts of total emissions depending on the state of the abatement technology. Yet, combined with information about CO2 emissions, both pricing mechanisms do reveal information about the cost of achieving a given emissions target.

each jurisdiction allocates to abatement activity and can be interpreted as evidence of a positive willingness to pay for climate mitigation and the presence of (cost-effective) abatement technology.

Trade-weighted signal As mentioned in section 4.1, for each "recipient" jurisdiction, the strength of those effects depends on the relative importance of each bilateral trade relation and can arise through export and import channels. To account for either of these channels and the relative importance of each trading partner, the above variables are export- or import-weighted.

The first measure is the trade-weighted cumulative number of climate laws. As we can see, little climate-related legislative activity takes place before the year 2000 and most additions to the current stock of climate legislation occurs from 2005 onwards.



(a) (b)

Figure 2

The second proxy represents the share of imports or exports coming from carbon pricing jurisdictions. It is defined as

$$IM_{i,t}^{carb} = \sum_h \sum_k \left[\mathbb{I}_{t,h,k} \times \frac{IM_{t,h,k,i}^{carb}}{IM_i^{Tot}} \right] \qquad EX_{i,t}^{carb} = \sum_h \sum_k \left[\mathbb{I}_{t,h,k} \times \frac{EX_{t,h,k,i}^{carb}}{EX_i^{Tot}} \right] \tag{23}$$

where i denotes the *home* jurisdiction, $h \in H_{-i}$ denotes the *trading partner*, k denotes the sector and t denotes the time period. That is, for each jurisdiction and each year, we multiply the trade flow $IM_{t,h,k,i}^{carb}$ – normalised by total imports of jurisdiction i (IM_h^{Tot}) – with an indicator variable ($\mathbb{I}_{t,h,k}$) recording whether the corresponding sector in the jurisdiction of origin is covered by a carbon pricing scheme. Figures 3a and 3b show the evolution for selected countries for imports and exports, respectively.

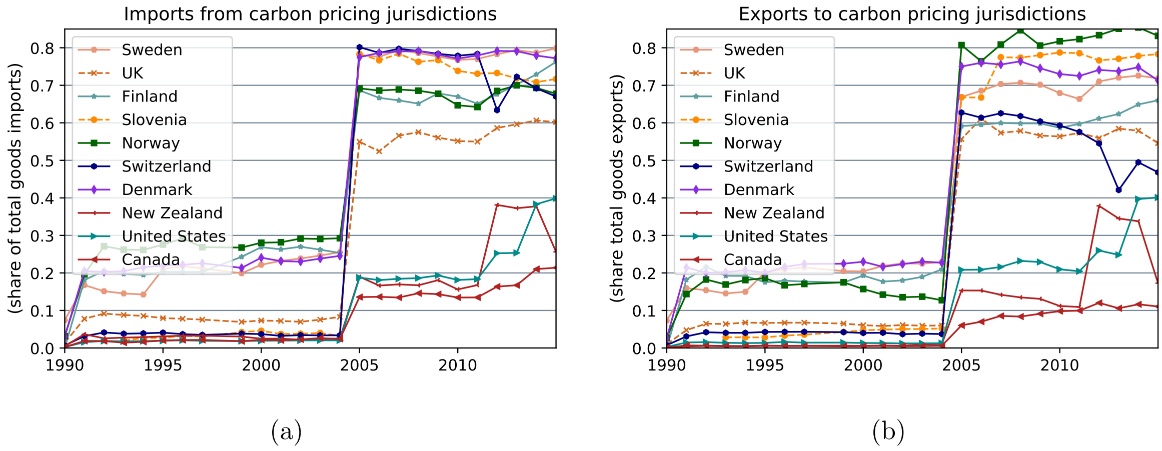


Figure 3

Prior to 2005, only Northern European countries were explicitly pricing carbon and no "big" jurisdiction had introduced a carbon pricing scheme. This explains the relative stability of the figures in panels 3a and 3b. From 2005 onward, the situation changes dramatically for all jurisdictions in the sample, not only those taking part in the EU-ETS. For example, after the start of the EU-ETS, the United States have just under 20% of their imports covered by a carbon pricing scheme in the jurisdiction of origin. The situation changes again in 2012 when Japan and Australia introduce their carbon pricing scheme. The impact of those changes is even clearer for New Zealand, which saw a surge in its imports from and exports to carbon pricing jurisdictions in 2012 and then a sudden drop in 2015 when Australia repealed its legislation.

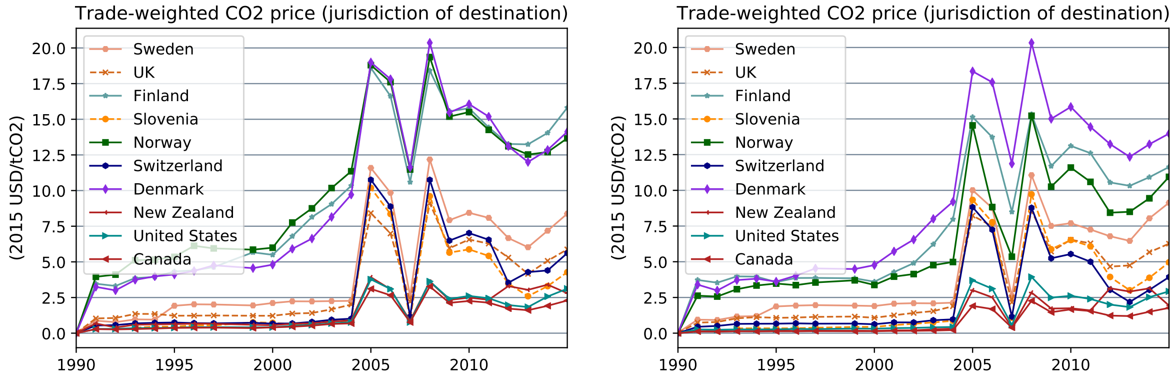
The third measure is the trade-weighted average price of CO2 in partner jurisdictions.

$$Price^{imp}_{i,t} = \sum_{h}\sum_{k}\left[\delta_{t,h,k} \times \frac{IM^{carb}_{t,h,k,i}}{IM^{Tot}_{i}}\right] \qquad Price^{exp}_{i,t} = \sum_{h}\sum_{k}\left[\delta_{t,h,k} \times \frac{EX^{carb}_{t,h,k,i}}{EX^{Tot}_{i}}\right] \tag{24}$$

Figures 4a and 4b provide this metric for selected jurisdictions. This sheds light on the *external* effect of CO2 pricing and the significant role played by the EU-ETS for non EU-ETS jurisdictions.

4.2.3 Abatement technology

To account for the state of (abatement) technology in each jurisdiction, two ways are available. First, as equation (6) shows, one possibility is to use the share of (CO2-)pollution payments in



(a) (b)

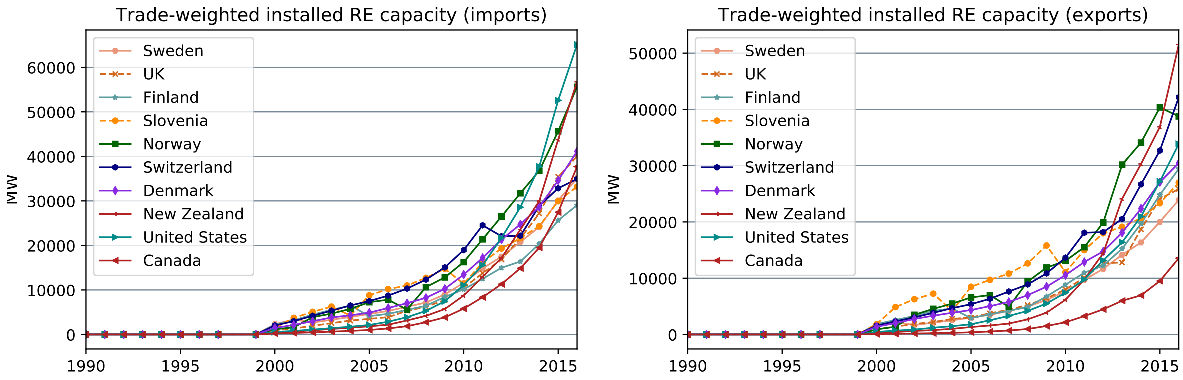
Figure 4

value-added, which relates directly to the state of the domestic abatement technology. However, this raises two issues. First, it would *de facto* restrict our informational set to partner jurisdictions with explicit carbon prices and leave out any information about abatement technology in other jurisdictions. Second, to constitute a valid proxy of the state of abatement technology, the pollution payments as a share of value added should, under perfect competition, be independent of pollution prices. As is discussed further below, this is not the case.

Therefore, we use an indirect measure of the state of abatement technology: the cumulative installed electricity generation capacity from wind and solar energy. This emphasises the 'learning by doing' process that characterises technological development (Arrow, 1962) and its implications for the (unit) cost at which the technology can be provided. In the case of solar photovoltaics, for example, (IRENA, 2012) finds that costs decline by 22% for every doubling of capacity.[28]

Trade-weighted technological spillovers Figure 5 shows the import- and export-weighted installed capacity of wind and solar (in MW). Any significant effect of the import-weighted measure would provide support for the embodied technology assumption whereas the export-weighted metrics would emphasise the pure exchange of ideas and/or, adjustment brought by competition in export markets.

[28]While pollution payments are subject to caution due to variations which are directly linked to variations in polluting emissions prices, installed capacity of wind and solar electricity generation isn't.



(a) (b)

Figure 5

4.2.4 Other variables/hypotheses

GDP per capita To control for the general economic environment, we use GDP per capita (Purchasing Power Parity, in constant 2011 USD) and trade openness. GDP per capita captures the standard income effect and, assuming that environmental quality is a normal good, should have a positive impact on both policy implementation and stringency.

Openness The effect of trade openness, however, is slightly more subtle. In our framework, trade is considered as a catalyst of carbon-pricing policies, and should therefore have a positive effect on the implementation of carbon pricing schemes. However, carbon leakage and international competitiveness concerns might induce governments to soften the stringency of their respective carbon pricing schemes. Real GDP per capita and trade openness data come from **?**.

Altruism Finally, we test our main hypothesis against an alternative rationale for pricing carbon: altruism. Our main working assumption so far has been that jurisdictions were self-interested and implementing Nash equilibrium carbon pricing policies. That is, we explicitly excluded care for the global environment as a motivation for undertaking emissions reductions. Yet, there is evidence [ref] that several jurisdictions (e.g. Norway) are taking relatively stringent emissions reductions commitment at home and actively supporting other jurisdictions in theirs. These commitments might well reflect care for the global environment, not only the domestic one. To gauge altruistic behaviour on the part of jurisdictions we use Official Development Aid as a share of Gross National Income (**??**). We account differently for its effect on donor and recipient countries. For donor countries, it seems reasonable to assume that more altruistic jurisdictions tend to give a larger share of their national income. This is less clear on the recipient side for several reasons. On the one hand, jurisdictions that receive a larger ODA (as a share of GNI) might also be poorer, and hence have a lower WTP for carbon. On the other hand, larger receivers might be prompted to introduce carbon pricing policies.

EU We account for the influence of EU membership by using an author-created dummy variables. The EU, a club of countries cooperating on a wide range of issues – including the environment, has implemented an organisation-wide emissions trading scheme. Several EU Member countries that are currently operating such a scheme were "dragged in" and did not willingly sign up for it. This is the case, for instance, of current EU Member States that joined the Union in 2004, i.e. a year before the start of the EU-ETS but a few months after Directive 2003/87/EC, which implemented the EU-ETS, was passed.[29] It is relatively clear that some Eastern European countries that joined the EU at the time had little (if any) say on the development of the legislation pertaining to the creation of the EU-ETS and implemented it only because it was part of the preexisting legislative *acquis* (Robinson and Stavins, 2015).

Table 1: Summary of variables

Category	Variable	Expected sign Carbon Price (Y/N)	Expected sign ECP (Level)
Signalling	Climate Laws (imports)	+	+
	Climate Laws (exports)	+	+
	Pricing (imports)	+	+
	Pricing (exports)	+/-	+/-
	CO2 Price (imports)	+	+
	CO2 Price (exports)	+/-	+/-
Tech. spillovers	RE CAP (imports)	+	+
	RE CAP (exports)	+	+
Economic env	GDP per capita ($1000)	+	+
	Trade Openness (% of GDP)	-	+
Other	ODA (% GNI) – donor	+	+
	ODA (% GNI) – receiver	-	-
	EU	+	+

[29]The Accession Agreement of the 10 countries meant to join the EU in 2004 was signed in April 2003. This agreement acknowledges compliance with the *acquis communautaire*.

Table 2: Data sources

Series	Source
ECP	Dolphin et al. (2016)
CO2 price	Dolphin et al. (2016)
Sectoral CO2 emissions	IEA
Wind+Solar inst. Cap.	IRENA
GDP per cap., PPP	WB-WDI
EU	Author
Trade openness	WB-WDI
Climate laws	GLOBE
Bilateral Trade	IMF Direction of Trade

Table 3: Summary statistics

Variable	**Mean**	**Std. Dev.**	**Min.**	**Max.**	**N**
Pricing	0.11	0.313	0	1	3699
ECP(2013)	1.053	5.142	0	77.83	3699
Pricing - imports	0.172	0.224	0	0.857	3112
Pricing - exports	0.178	0.244	0	0.929	3112
CO2 Price - imports	2.029	2.712	0.001	20.362	3112
CO2 Price - exports	1.915	2.797	0.001	21.364	3112
Climate Policy - imports	0.132	0.269	0	2.627	3112
Climate Policy - exports	0.142	0.337	0	5.673	3112
RE cap. - imports	5398.981	8918.339	0	109125.453	3112
RE cap. - exports	4778.052	9520.35	0	124159.953	3112
Trade Openness	80.562	47.378	0.021	441.604	3525
GDP (per cap.), thousand USD	19.474	19.962	0	129.35	3531
EU member	0.132	0.338	0	1	3699
ODA, % GNI	-2.737	6.020	-72.06	7.997	3349

5 Results

The regression analysis presented below discusses the signalling and technology spillover effects on the implementation and stringency of domestic policy. We treat the two variables pertaining to the signalling effect in separate regressions to avoid multicollinearity problem. The estimation results are presented in tables 4 and 5.[30]

Table 4: (Random-effects) logistic regression

Pricing (1=yes,0=no)	(1)	(2)
Pricing - imports$_{t-1}$	17.536*	
	(9.0445)	
Pricing - exports$_{t-1}$	15.915**	
	(6.3015)	
CO2 Price - imports$_{t-1}$		2.281**
		(0.9293)
CO2 Price - exports$_{t-1}$		2.143***
		(0.9285)
Climate Policy - imports	0.764	1.852
	(2.2066)	(2.7517)
Climate Policy - exports	0.225	-3.507
	(2.6119)	(3.8053)
RE cap. - imports$_{t-1}$	0.000285	0.000666***
	(0.0002)	(0.00024)
RE cap. - exports$_{t-1}$	0.0000129*	0.000073
	(0.0001373)	(0.000153)
Trade Openness	-0.0375	-0.0137
	(0.035)	(0.02119)
GDP, thousand USD	0.133***	0.042
	(0.0496)	(0.076)
ODA, % GNI (donor)	0.817	0.372
	(1.24)	(1.0435)
ODA, % GNI (receiver)	-14.426***	-21.023***
	(3.8282)	(5.0801)
EU member	27.233***	35.93***
	(3.7367)	(2.236)
Constant	-36.2573***	-43.11***
	(4.2025)	(2.028)
Observations	2699	2699

Standard errors in parentheses
* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$

Economic and institutional environment First, the results suggest that there is a positive effect of GDP per capita: a \$1000 increase in GDP per capita increases the odds of implemen-

[30] All regression include year fixed effects.

Table 5: Fixed-effects panel regression

ECP	(1)	(2)
Pricing - imports	0.05303***	
	(0.0151)	
Pricing - imports$_{t-1}$	0.0057	
	(0.0145)	
Pricing - exports	0.0146	
	(0.0095)	
Pricing - exports$_{t-1}$	-0.0097	
	(0.0062)	
CO2 Price - imports		0.548***
		(0.0412)
CO2 Price - imports$_{t-1}$		0.0884*
		(0.0412)
CO2 Price - exports		0.268***
		(0.0416)
CO2 Price - exports$_{t-1}$		0.00611
		(0.0423)
Climate Policy - imports$_{t-1}$	0.446***	0.466**
	(0.191)	(0.154)
Climate Policy - exports$_{t-1}$	-0.080	-0.338*
	(0.001)	(0.133)
RE cap. - imports	-0.000022	-0.0000327*
	(0.00002)	(0.0000131)
RE cap. - exports	-0.0000003	0.00000817
	(0.00000008)	(0.00000793)
Trade Openness	-0.0017	0.00145
	(0.00384)	(0.00288)
GDP, thousand USD	0.102*	0.0604***
	(0.0487)	(0.0154)
EU member	4.804***	4.318***
	(1.002)	(0.308)
Constant	-2.295*	-1.848***
	(1.076)	(0.390)
Observations	3042	3042
R^2	0.328	0.397

Standard errors in parentheses

* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$

tation of a carbon pricing scheme and leads to a \$0.1 increase in the ECP.[31] This is in line with standard environmental economics literature suggesting that environmental quality is a normal good (i.e. that the willingness to pay for a clean environment increases with income). We find, however, no Environmental Kuznets Curve effect: the introduction of the square of GDP in the regression turned out to be statistically non significant for both implementation and stringency.

Openness to international trade influences policy implementation and stringency differently.

[31]The coefficients reported in table 4 are expressed on a log-odds scale, i.e. indicate the effect of the variable on the logarithm of the odds ratio. To interpret these in terms of probabilities, the following transformation is applied: e^{β_i}.

Results indicate that it tends to have a positive effect on the probability of implementation but a negative (albeit small and not statistically significant) impact on policy stringency.

Unsurprisingly, EU membership has had a positive effect on both implementation and stringency, increasing the emissions-weighted price by about \$3.6. Results also provide some support for an altruism narrative, with the level of Official Development Aid (as a share of GNI) positively affecting the odds of implementation of a carbon pricing scheme.

Signalling The results provide some support for the signalling hypothesis. Overall, they indicate a strong relationship between the policy implementation/stringency of a jurisdiction and the introduction of carbon-pricing schemes by "close" neighbours. The "import channel" seems to be particularly important, as it both increases the probability of a pricing scheme being implemented and its stringency. For example, a one percent increase in imports from carbon pricing jurisdictions is associated with a \$0.045 increase in the emissions-weighted domestic price of carbon.

The level of the carbon price (either import- or export-weighted) does not, however, appear to affect the probability of implementation of a carbon pricing scheme. It does, however, affect the carbon price level: a 1\$ increase in the import-weighted price of CO2 leads to a \$0.31 increase in the domestic emissions-weighted price, i.e. increased effort to abate emissions on the part of your trading partners is associated with an increased ECP (and hence abatement efforts) at home. Interestingly, the lagged import-weighted CO2 price is also significant, although the effect is much smaller (\$0.036). Accounting for the CO2 price rather than a signalling effect increases the explanatory power of the model.

Export-weighted variables do not add to the explanation of policy implementation but have a negative effect on policy stringency. That is, higher (non-zero) carbon price in export markets tends to lower the emissions-weighted price in the domestic jurisdiction. However small this effect is (-\$0.0014 for a 1% contemporaneous increase in exports to carbon pricing jurisdictions, -\$0.00245 for every 1% increase in the lagged variable), it might lend some support to the pollution haven hypothesis as jurisdictions exposed to carbon pricing in their export markets might soften their own environmental policy or, at the very least, not strengthen it. This effect is present in the price of CO2 as well (-\$0.02).

Lastly, climate policy developments in import markets is associated with more stringent domestic carbon price (\$0.38-\$0.51).

Spillover effects The (lagged) import-weighted cumulated installed capacity of wind and solar affects positively the odds of implementation of a carbon pricing policy. The effect is, however, relatively small. We were unable to identify any export-related effect.

Finally, the (import-weighted) share of pollution payments is negatively related to policy stringency, which strengthens support for the technology transfer hypothesis: home jurisdictions learn more from jurisdictions with better technology.

If we exclude EU countries from the sample, several relationships identified above are no longer supported. This suggests that the identified relationships have, so far, been mainly driven by EU (and potentially EU-ETS) countries. While this clearly points to the observation that identified historical relationships mostly hold for Western Europe, it does not undermine their relevance for future carbon pricing development. Indeed, we believe that a lesson drawn from the European experience is that general economic integration, be it through trade or broader institutional arrangements, fosters cooperation by reducing transaction costs and, most importantly, facilitates access to technological advances within the integrated group.

Conducting the analysis running with 1990-2004 observations does not alter the conclusions drawn on the basis of the full sample. The only difference is, again, one of magnitude. For example, a 1% increase in imports from carbon pricing jurisdictions leads to a \$0.09 increase in the [ECP] while a \$1 increase in the (import-weighted) carbon price leads to a \$0.4 increase in the ECP. These effects are stronger compared to the full sample results. Export-weighted pollution payments affect negatively the ECP (-\$0.15).

Over the period 2005-2014, the effect of imports from and exports to carbon pricing jurisdictions vanishes but the effect of import-weighted carbon price is significant and close to the result obtained with the full sample (\$0.22). Pollution payments exhibit a negative and significant effect on the ECP during that period. Import and export-weighted pollution payments raise the ECP by \$0.29 and \$0.13 respectively.

6 Discussion: trade & environmental coalition formation

The development of carbon pricing policies over the last quarter century owes much to the ability of policymakers to overcome domestic political economy barriers to their implementation. In particular, they had to find ingenious ways around the resistance opposed by CO2-intensive and trade-exposed sectors. The implications of carbon pricing policies for open economies have aroused much interest in the academic and policy literature, either emphasising the potential leakage effects of unilateral policy tightening – and hence the environmental ineffectiveness – or the direct cost to trade-exposed sectors.

In any case, bar a few exceptions, both literatures have pointed to the difficulties that trade might pose to the development of such policies. This paper offers a different and more positive perspective. There are reasons to believe that trade is not only the source of environmental ineffectiveness or increased domestic cost, but also a catalyst in the development of carbon pricing policies. First, it provides information about the exposure of domestic agents to foreign policy developments. Second, trade may constitute an important channel of technology diffusion between jurisdictions. Third, strong trade relationships not only account for geographical but also political and cultural proximity, which might foster cooperation on carbon pricing policies.

Hence, while this study is primarily concerned with unilateral climate policies, it could also advance the understanding of bottom-up climate coalition formation. By casting light on the diffusion of abatement technologies and the policy developments that it triggers, it provides a new perspective on the development of grassroots, regional, carbon pricing policies. From a dynamic perspective, the development of initially uncoordinated carbon pricing schemes might lead to coordination and scaled up ambitions in subsequent periods.

Moreover, the results and general reasoning behind the study suggest that domestic carbon pricing policy development should go hand in hand with technological transfers to "close" trading partners as they are more likely to reap the full benefits of it.[32] This may also mean that the offer of technological transfers conditional on abatement commitment will be more effective on countries that are close to the domestic economy. Closeness, in this case, increases the credibility of the proposal.

[32] This is in line with the (recent) empirical literature on the international diffusion of knowledge that supports the idea that knowledge spillovers are trade-related and, hence, localised (Coe and Helpman, 1995; Fracasso and Vittucci Marzetti, 2015).

7 Conclusion

The last quarter century has witnessed the development of numerous carbon pricing policies. While a lot of effort has been spent on understanding the implications of carbon pricing for an economy's CO2 emissions, there has been little systematic discussion of the factors underlying such development. Yet, if pricing carbon is to become a non-marginal policy tool for addressing climate change, building a clear understanding of the dynamics behind these policy developments is of the essence.

This paper seeks to contribute to that understanding. Taking the perspective of a (small) jurisdiction considering implementation of a carbon pricing scheme, it argues that its policy decision is linked to (foreign) signalling and technological spillover effects. In other words, a jurisdiction looking to price carbon will assess its access to abatement technology and the policy signal of its various (trading) partners. Crucially, we claim that access to abatement technology and the nature/strength of the signal received depends on the nature of the trade relationship with its carbon pricing/abating partners.

We believe that the above analysis lends support to that view. First, there is strong evidence that the implementation and stringency of carbon pricing policies is related to import flows from carbon pricing partners. This echoes earlier studies on trade-related knowledge spillovers and evidence about the development of renewable energy industries in some jurisdictions. Secondly, it appears that not only do jurisdictions learn more from their carbon pricing partners but, among them, learn more from the best.

In that respect, we believe that the European experience holds particularly strong insights for future carbon pricing developments. Indeed, integration, be it through trade or broader institutional arrangements, seems to foster facilitate policy diffusion by enhancing access to technological advances within the integrated group and strengthening the policy signal.

These results are particularly important as they cast a new light on the external (positive) effects of carbon pricing policies. It is very likely that instigators of such schemes incurred some cost that are sunk in nature, reducing barriers to implementation in *close* jurisdictions. In a world where globally coordinated action has failed to deliver environmentally efficient outcomes, it is crucial to understand the external effects of unilateral policy development. This work contributes to that understanding by looking at carbon pricing policy development over the last quarter century.

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A Firm's profit maximisation

$$\begin{aligned}
\pi^x &= pX(K_x, L_x) - wL_x - rK_x - \delta e \\
&= pX(K_x, L_x) - wL_x - rK_x - \delta i X(K_x, L_x) \\
&= (p - \delta i) X(K_x, L_x) - wL_x - rK_x \\
&= \left(p - \delta \frac{\alpha p \Omega(\psi)}{\delta}\right) X(K_x, L_x) - wL_x - rK_x \\
&= \underbrace{p(1 - \alpha\Omega(\psi))}_{\text{net producer price}} X(K_x, L_x) - wL_x - rK_x
\end{aligned} \tag{A.1}$$

The second and fourth equalities result from $e = ix$ derived from (6). Recalling that

$$\frac{\delta \frac{e}{\Omega(\psi)}}{px} = \alpha \tag{A.2}$$

and that $0 < \alpha < 1$ and $0 < \Omega(\psi) \leq 1$ it is easy to see that $\alpha\Omega(\psi)$ represents the share of pollution payments in total value added. We note two observations. First, assuming constant α, a decrease in the share of pollution payments can be interpreted as reflecting a decrease in $\Omega(\psi)$, i.e. an improvement in abatement technology. Second, as $\Omega(\psi)$ decreases, the emission intensity of the economy decreases and (net) revenue (i.e. revenue net of pollution permit payment) increases. Quite straightforwardly then, the firm optimally increases its output of good x.

The firm in the Y sector does not pollute and profit function is thus

$$\pi^y = pF(K_y, L_y) - wL_y - rK_y \tag{A.3}$$

The key question that remains is then: how does the economy allocate its resources between the two sectors? This crucially depends on the relative price of the good and the share of pollution payments. Indeed, recalling our perfect competition assumption, Euler's theorem, and the fact that labour and capital are inelastically supplied, we have

$$\begin{aligned}
F_K &= p(1 - \alpha\Omega(\psi))X_K = r \\
F_L &= p(1 - \alpha\Omega(\psi))X_L = w
\end{aligned}$$

where X_K, X_L and F_K, F_L denote the marginal productivity of factors in sectors X and Y, respectively. That is, factors of production are remunerated at the value of their marginal product which, since both sectors trade inputs in the same markets, is equalised across sectors. Rearranging the above yields,

$$\frac{F_K}{X_K} = \frac{F_L}{X_L} = p(1 - \alpha\Omega(\psi)) \equiv S \tag{A.4}$$

When international spillovers increase, reducing "payments to pollution", then more inputs are diverted toward the dirty good sector and production expands.

It is interesting to put this in the broader context of our general equilibrium model. The total effect of a (positive) technological change in abatement comes in two steps: first, for a given (equilibrium) price of emissions, demand for emissions decreases, reducing pollution payments, inducing a shift of inputs from the clean to the dirty sector and hence stimulating production; second, the subsequent (downward) adjustment in emissions price induces a reduction in resources devoted to abatement – hence an increase in pollution demand – and further stimulates production in the dirty sector. The emission intensity of the dirty sector nevertheless decreases.

Lastly, equation (A.4) provides an interesting result: the effect of a change in relative price on (between sector) resource allocation varies with abatement technology, $\Omega(\psi)$. That is, define Ω^{high} and Ω^{low}, denoting *poor* and *good* abatement technology, respectively. Then

$$\left.\frac{\partial S}{\partial p}\right|_{\Omega^{high}} < \left.\frac{\partial S}{\partial p}\right|_{\Omega^{low}} \tag{A.5}$$

When a jurisdiction has good abatement technology, a change in the relative price of the dirty good will induce a larger reallocation of resources from the clean to the dirty sector.

B Emission intensity and abatement efforts

It now becomes possible to derive an expression of ϕ in terms of prices. Using (6) to note that total emissions are equal to $e = ix$, we can rewrite the production function (4) as

$$x = \left(\frac{ix}{\Omega(\psi)}\right)^{\alpha} B(K_x, L_x)^{1-\alpha}$$

Yet, we also know that $x = (1-\phi)B(K_x, L_x)$. Hence

$$i = (1-\phi)^{(1-\alpha)/\alpha}\Omega(\psi) \tag{B.1}$$

which suggests that the emission intensity of the economy decreases in two cases: when more resources are devoted to abatement and when the abatement technology improves. Now, substituting i for its expression in equation (6) yields

$$\frac{\alpha\Omega(\psi)p}{\delta} = (1-\phi)^{(1-\alpha)/\alpha}\Omega(\psi)$$

and we can therefore write

$$\phi = 1 - \left(\frac{\alpha p}{\delta}\right)^{\alpha/(1-\alpha)} \tag{B.2}$$

As it turns out, abatement effort is independent from Ω, the abatement technology quality. However, a change in abatement technology will affect equilibrium abatement effort through its effect on equilibrium emissions price.

As equation (6) indicates, changes in both emissions price and abatement technology should lead to changes in the emissions intensity of the economy: a higher price of emissions (or a lower share of pollution payments) is associated with lower emissions per unit of value-added. Empirically, emissions intensity is defined as

$$INT_{i,t,k} \equiv \frac{E^{Tot}_{i,t,k}}{VA_{i,t,k}}$$

where $E^{Tot}_{i,t,k}$ is total emissions of sector k in jurisdiction i at time t and $VA_{i,t,k}$ is the constant [2005] USD Value Added. Figure 6 represents CO2-intensity for a selection of jurisdictions over the period 1990-2014.

The first observation is that the CO2-intensity of the four sectors under consideration has declined steadily over time. This likely reflects technological change and accompanying efficiency gains as well as policy intervention. Figure 6 also shows that in all jurisdictions represented, the power sector is the most CO2-intensive sector, followed by Mining, Manufacturing and Agriculture, in that order. This ranking is unsurprising and partly explains why it has been so difficult for industrialised countries to introduce tighter environmental regulations [reference].

It also helps understand why most manufacturing sectors and the power sector were granted exemptions and/or free allocation of emissions allowances in jurisdictions that introduced carbon pricing.

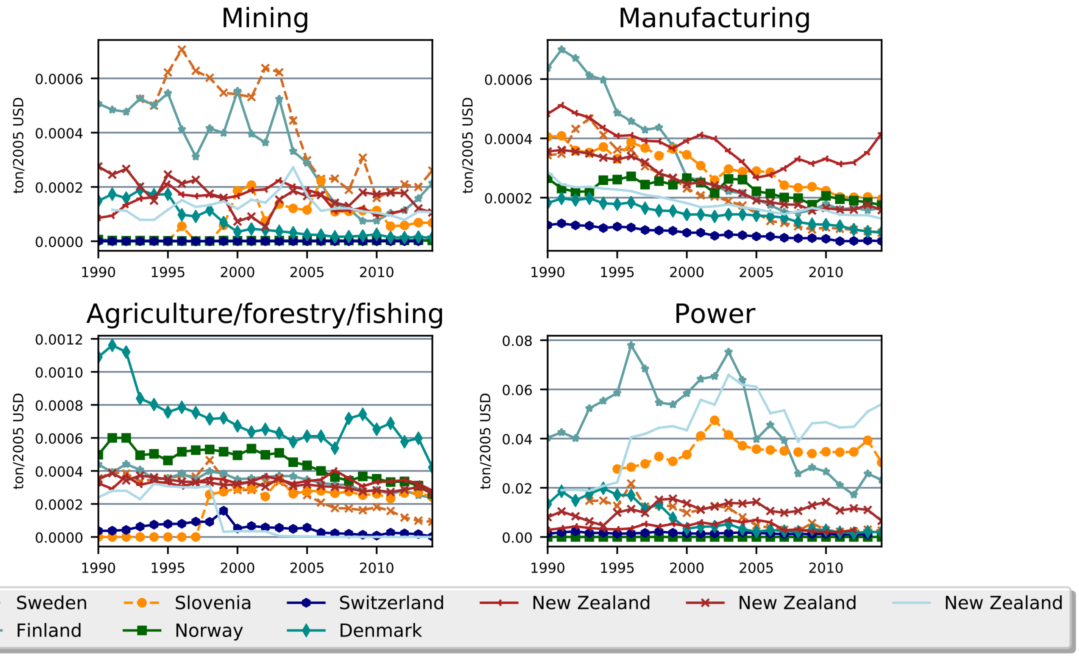

Figure 6: CO2-intensity in 'traded' sectors

C Data

Sectoral CO2 emissions data is from **?**. It contains emissions data by sector and fuel type arranged according to the International Standard Classification of All Economic Activities (ISIC), Revision 4, expressed in Million ton for the period 1971-2014. The CO_2 price data has been collected from various sources and presented in detail in Dolphin et al. (2016). It has been structured to match IPCC (2006) sectoral disaggregation (i.e. ISIC, Rev. 4) in order to ensure consistency with the emissions and value added data.[33]

The data on sector-level current Value Added is constructed from a combination of **?** and **?**. The value added is expressed in current local currency units, which is then converted into [2015] U.S. dollars using World Bank DEC conversion factor and inflation rates.[34] Data on constant

[33]Agriculture/forestry/fishing, Mining [and quarrying], Manufacturing, and Power. These sectors correspond respectively to Sections A,B,C and D of the International Standard Industrial Classification System (Rev.4) (ISIC) or (sub-)Sectors 11,212,22 and 31-33 of the North American Industrial Classification System (NAICS).

[34]Importantly, **?** is based on ISIC, Rev. 3 while **?** is based on ISIC, Rev. 4, thereby creating a potential

value added, used to compute CO2-intensity, is from **?**. The bilateral trade data comes from the **?**. Hence, for each sector and for each jurisdiction, we have the value of exports and imports to and from all trading partners. Data on installed wind and solar electricity capacity comes from **?**.

source of discrepancy in the Value Added series. However, because we are focusing on aggregate sector, i.e. division categories (Agriculture, forestry and fishing, Manufacturing, Power and Mining and Quarrying), we do not expect changes in classification systems to alter the validity of the constructed time-series. We are in fact able to retain the consistency of sector definitions across tables. The value added of the electricity sector is lumped together with gas and water supply (Rev. 3) and with gas, steam and air conditioning supply (Rev. 4). Hence, given that we use emissions from Electricity and Heat production, the CO2-intensity of the Power sector is likely to be slightly underestimated.