The political economy of carbon pricing: a panel analysis

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Abstract

In virtually all jurisdictions that explicitly price carbon, its average (emissions-weighted) price remains low. Our analysis focuses on the political economy of its introduction as well as its stringency in an international panel of national and North American subnational jurisdictions. Results suggest that political economy factors primarily affected the former and that policy stringency is a highly persistent process. This has two important policy implications. First, successful passage of carbon pricing legislation will either come with *contemporaneous* compensation of incumbent, CO2-intensive, sectors or occur *after* their relative weakening. Second, if political economy constraints continue to prevail, a robust rationale for the design of climate change mitigation strategies with multiple instruments exists.

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1 Introduction

The agreement reached in Paris at the end of 2015 was a diplomatic success. Its environmental benefits are, however, much less clear. 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).[1]

As the IPCC Working Group II "reasons for concern" make clear, this level bears significant risks for human development and is likely to place unprecedented pressure on already stressed ecosystems (IPCC, 2014). Therefore, supplementary commitments to reduce Greenhouse Gases (GHG) emissions beyond existing INDCs are needed. This will, in turn, require the setting up of new (or the strengthening of existing) environmental policy tools. Historically, these tools took the form of "command-and-control" regulations, production quotas and subsidies for electricity from renewable energy sources and, more recently, carbon pricing instruments such as carbon taxes or cap-and-trade systems (Bennear and Stavins, 2007).[2] The focus of this paper is on the latter category.

While the earliest occurrences of these tools can be traced back to the experiences of Northern European states (Finland - 1990, Sweden - 1991), their development has only gained momentum in the last few years. According to World Bank (2018), thirty-eight new carbon pricing mechanisms started operations between 2005 and 2018, including the California Cap-and-Trade Program and 7 (sub-national) emissions trading schemes in China. These new schemes added to a group of existing carbon pricing tools such as the European Union Emissions Trading System or a range of taxes explicitly based on the carbon content of fossil fuels.

Yet, the introduction of such tools is often faced with strong political economy con-

[1] 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 CO2 eq (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] *Carbon* per se is not a greenhouse gas but carbon dioxide (CO2) is. We refer to instruments putting a price on CO2 emissions as carbon pricing instruments.

straints (Jenkins, 2014) that influence their design and prevent their full (i.e. socially optimal) implementation (Del Rio and Labandeira, 2009). Their influence on the implementation of carbon pricing policies is nonetheless currently under-researched. While substantial attention has been paid to the political economy of energy or renewable energy support (RES) policies, a relatively narrow set of studies have specifically focused on policies making use of carbon pricing mechanisms, be it in a specific national or subnational context, or in an international panel of countries. Furthermore, such studies often focus on policy outcomes as proxies for policy developments but do not directly study the policy tool itself (see, e.g. Cadoret and Padovano (2016) or Gassebner et al. (2011)).

Our study is a contribution to filling this gap. It aims at shedding light on the nature and working of political economy constraints on the development of carbon pricing policies. This allows us to address two fundamental questions pertaining to the design of climate mitigation strategies in the presence of political economy constraints. How should we adapt the policy design to a specific institutional and economic context? Do political economy constraints constitute a robust rationale in favour of a policy mix as opposed to a single instrument?

The paper is organized as follows. Section 2 reviews the relevant strands of the literature. Section 3 briefly discusses carbon pricing (in theory and practice). Section 4 introduces the Emissions-weighted Carbon Price and presents the hypotheses while section 5 presents the data and discusses the empirical methodology used in the analysis. Section 6 discusses the results and section 7 concludes.

2 Literature review

More often than not, economic policies resulting from the legislative and political bargaining process constitute sub-optimal social outcomes. Political economy theory provides a useful analytical framework to rationalise them. Olson (1965) highlights the role played by groups with shared interests in shaping policy outcomes and the factors that drive their behaviour. Building on Olson's conjecture, Stigler (1971) proposed the idea of regulatory capture, which views the State as a provider of regulation and the industry as an active seeker of regulation designed and operated for its own benefit.

The relevance of these theoretical insights has long been discussed in the context of environmental policy making (Pearce, 2005). Congleton (1992) takes an institutional perspective to the issue; proposing that political institutions, rather than resource endowments, determine a country's environmental regulation. More precisely, he argues that due to their focus on longer term outcomes, democratic institutions tend to deliver more stringent environmental regulations.[3] At the same time, democratic systems allow a plurality of, sometimes divergent, interests to be voiced. Hahn (1990) attempted to identify rationales for the emergence of incentive-based mechanisms and suggested that environmental policy is the result of a "struggle" between different interest groups. In the context of carbon pricing, the introduction of (economy-wide) schemes may induce profound changes in the magnitude and distribution of welfare. Therefore, even if the welfare of the polity as a whole is greater in an economic system constrained by environmental policies, one may expect strong opposition on the part of both consumers and producers. On the consumption side, some studies have shown that the willingness to pay for carbon emissions is low (Jenkins, 2014). Moreover, carbon pricing schemes have been found to be regressive, with varying degrees, in a wide range of institutional contexts (Wier et al., 2005; Grainger and Kolstad, 2010), with only some of them designed to alleviate this effect (Bowen, 2015). On the production side, sectors with assets whose value would be severely diminished in case of carbon pricing are expected to strongly oppose policy change; a possibility that Joskow and Schmalensee (1998) discuss in the case of the U.S. SO_2 market.

Analyses taking advantage of the availability of panel data have also shed light on political economy dynamics. Marques et al. (2010) analyse the drivers of the deployment of renewable energy in European countries. Using fixed effects (panel data) regression and vector decomposition, they find evidence that the conventional energy sector lobby and the level of CO_2 emissions impede the deployment of renewable energy sources for electricity production. Chang and Berdiev (2011) focus on the electricity and gas industries and seek to disentangle the effects of government ideology, political factors and

[3]This argument runs against the standard view that political representatives are self-interested and focused on short-term electoral cycles.

globalisation on energy regulation in 23 OECD countries over the period 1975-2007. They conclude that left-wing governments promote regulation in gas and electricity sectors and that less fragmented governments contribute to deregulation of gas and electricity industries. van Beers and Strand (2015), analysing data from 200 countries for the political determinants of fossil fuel pricing during the period 1991-2010, found that higher GDP levels lead to higher fuel prices (higher taxes or lower subsidy rates) and that a presidential system (unlike parliamentarian or proportional representation systems) could lead to significantly lower gasoline and diesel prices.

Lastly, insights drawn from analyses of the liberalisation of energy markets are also relevant to our investigation. Pollitt (2012) takes stock of the energy market liberalisation processes to draw lessons about the role of policy in energy transitions and argues that liberalisation *per se* will have little impact on the shift toward a low carbon energy mix. Rather, the willingness of societies to bear the cost of environmental policies will. Hence, liberalisation is not necessarily *neutral* for carbon pricing policy formulation as it has made the cost of those policies increasingly apparent to consumers (Pollitt, 2012). Evidence from the U.S. (Jenkins, 2014) suggests that citizens are indeed quite sensitive to the direct costs induced by carbon pricing policies, even if the net cost is brought (close) to zero via tax rebates or other fiscal mechanisms.

However substantial the discussion of political economy factors in environmental policy formulation has been, relatively less attention has been paid to the political feasibility of carbon pricing policies and, equivalently, to the variables that influence their implementation and strength. To our knowledge, only Jenkins (2014), Gawel et al. (2014) and Del Rio and Labandeira (2009) bring the issue to the fore. Shedding further empirical light on these dynamics is particularly important as we believe that they may differ in nature or in strength from those of: (i) excise duties, which in most occurrences constitute an indirect way to tax road transport; (ii) other climate policies, whose cost is less visible to the final consumer. In the absence of more refined assessment, suggestions about a way forward for the implementation of carbon pricing when faced with *political economy* constraints are, at best, incomplete. Before turning to that analysis we briefly review the rationale and tools for a carbon price.

3 Carbon pricing policies: theory and practice

In theory, provided that the public authority can credibly commit to a state-contingent carbon price path and in the absence of transaction costs, the carbon price signal should be economy-wide (Tirole, 2012).[4] Indeed, the externality associated with the release of GHG into the atmosphere is the same regardless of its source (i.e. sector of origin) or type of use. Any departure from this situation will inevitably introduce distortions between sectors and/or types of users.

Two market-based mechanisms (and hybrid combinations[5]) have emerged: carbon taxes and Emissions Trading Schemes. The former places a set price on each unit of CO2 emitted into the atmosphere, leaving an uncertainty about the resulting level of emissions; the latter sets an emissions cap and leaves to the market the creation of the price signal. Even though both mechanisms share the same underlying motivation and, under complete knowledge and perfect certainty, are theoretically equivalent and deliver the same environmental outcome, they relate to two slightly different views about carbon pricing.[6] The first view emphasises the use of carbon pricing mechanisms to internalise the externality associated with GHG emissions and hence is more sympathetic to carbon taxes. In that case, the price of carbon should closely track the Social Cost of Carbon (SCC). The second stresses the achievement of a set carbon budget over a given planning horizon in a cost-effective way, in which case the price will follow the dynamically cost-effective price path (Rubin, 1996).

Importantly for us, these schemes differ also in their practical implementations. On the one hand, most carbon taxes are based on the carbon content of fossil fuels. On the

[4]If transaction costs (i.e. costs of monitoring and verification) are positive, then optimal coverage may not be 100%. Additional emissions should then only be included if the marginal benefit in terms of enhanced cost efficiency outweighs the marginal cost of monitoring and verifying emissions.

[5]Hybrid schemes combine elements of price and quantity schemes by, e.g. setting floors and caps on the prices delivered by quantity schemes (Roberts and Spence, 1976; Weitzman, 1978).

[6]Outcomes may differ when there is uncertainty about either the marginal cost or benefit of abatement and the relative superiority of one instrument over the other depends on the relative slopes of the marginal abatement and cost curves around the optimum (Weitzman, 1974). Weitzman's original article considers only a static one-period model and so is more relevant to flow rather than stock pollutants like CO_2 but his conclusions were supported in theoretical settings closer to that of stock pollutants (Pizer, 2002; Hoel and Karp, 2002).

other hand, an Emissions Trading Scheme is based on actual verified emissions at covered (stationary) plants.[7] Therefore, an ETS can in theory include fugitive and industrial processes emissions in addition to emissions from fuel combustion.

In 2015, the last year of our panel(s), 35 national and 21 subnational jurisdictions had an operating Emissions Trading Scheme while 15 national and 1 subnational jurisdiction had a carbon tax targeting at least one type of fossil fuel (i.e. coal, oil or natural gas). Among jurisdictions operating an ETS at the time, 47 covered industry and 54 covered the power sector while the same sectors were included in 14 and 12 carbon tax schemes, respectively. Table 1 provides a summary of sectoral coverage per type of pricing mechanism.

Table 1: Sectoral coverage (2015) – # of jursidictions

	Carbon tax schemes (total: 16)	ETSs (total: 57)
Industry	14	47
Power	12	54
(Road) Transport	12	5
Aviation (domestic)	4	31
Buildings (residential and commercial)	12	8
Agriculture or Forestry	11	2
Waste	12	1

Note: The figures presented in this table count each jurisdiction participating in the EU-ETS as a separate scheme. A description of the sectoral nomenclature is available in appendix B and a complete list of the jurisdictions operating a carbon pricing mechanism in 2015 is available in appendix D.

4 Carbon pricing and its drivers

Following on the above discussion, we argue that introducing a carbon pricing mechanism involves two decisions. First, a decision on whether or not to enact a pricing scheme, regardless of the price level or the coverage. Second, a decision about the appropriate – or politically feasible – stringency (i.e. average price). The implementation of carbon pricing policy is recorded by a dummy variable taking value 1 if it is in force in a given country-year, 0 otherwise. The stringency is captured by the average (emissions-weighted) carbon price and is described in section 4.1. The hypotheses formulated about the drivers of implementation and stringency are presented in section 4.2.

[7]Emissions from the aviation sector, which have recently been included in some ETSs, are estimated based on the fuel consumption of each aircraft, multiplied by the appropriate emissions factor (European Commission, 2012).

4.1 Carbon price and coverage: an Emissions-weighted Carbon Price

Following 'first-best' theoretical prescriptions, applied macroeconomic integrated assessment models often assume a single, economy-wide (100% coverage) carbon price. Yet, experience with carbon pricing policies suggests that their implementation has rarely, if at all, followed such prescriptions. For example, most of the schemes under consideration entailed low coverage at time of introduction – due to, e.g., sectoral or fuel-based exemptions, or a combination of the two – and their coverage remained partial over their lifetime. Moreover, careful observation of policy developments shows little consistency between the stated environmental goals (and implied GHG budgets) and carbon prices. Therefore, the price tag alone cannot appropriately reflect the stringency of a carbon pricing scheme. It has to be analysed together with its coverage. Moreover, as will be shown in section 4.1.2, the carbon price is usually not unique within jurisdictions, let alone across them.

In order to accurately account for these two dimensions of carbon pricing mechanisms and reflect their stringency, we introduce the concept of an Emissions-weighted Carbon Price (ECP).[8] This price, computed on a yearly basis, is a weighted average of all carbon price signals present in an economy at a point in time where the weights are the quantity of emissions covered as a share of that jurisdiction's total GHG emissions. To our knowledge, this is the first attempt at capturing the stringency of carbon pricing policies in a consistent and standardised way.[9] Before turning to a discussion of the underlying methodology, we review its two underlying components: coverage and price.

[8]The methodology behind the computation of the Emissions-weighted Carbon Price is similar to that suggested for the Effective Carbon Rate (OECD, 2015). However, the OECD methodology accounts for both explicit carbon prices and energy duties that indirectly price carbon, which we believe is misleading since, as we have emphasised and as the OECD itself acknowledges (OECD, 2015), the motivations behind their introduction are often unrelated to climate change concerns.

[9]Measuring policy stringency is inherently difficult, even more so when the metric needs to be comparable across jurisdictions. Most studies rely on indirect measures of policy stringency such as private-sector cost measures, measures based on pollutant emissions and environmental policy enforcement expenditures (Brunel and Levinson, 2013).

4.1.1 Coverage

The coverage of carbon pricing schemes is usually defined at the sectoral level although carbon taxes can be defined per fuel type too. The main difference between emissions trading schemes and carbon taxes lies in that the former sometimes cover multiple gases whereas the latter only apply to the carbon content of fossil fuels and, by extension, to CO2 emissions. The present paper focuses exclusively on CO2. Provided that accurate measurement of sectoral CO2 emissions is available, sectoral coverage of a scheme can easily be translated into "covered" CO2 emissions as a share of total, CO2 equivalent, GHG emissions. Based on this information, coverage figures were calculated for 135 national and 63 subnational (50 US States and 13 Canadian Provinces and Territories) jurisdictions as well as a hypothetical 'World' jurisdiction between 1990 and 2015.[10] Figure 1 provides an overview of the coverage of carbon pricing mechanisms in selected jurisdictions.[11] Panel (a) clearly shows that there is significant variation in coverage of carbon tax schemes across jurisdictions. Between 1992 and 2005, Denmark's scheme covered roughly 70% of its GHG emissions, the highest share among all jurisdictions considered, while Finland's coverage was only 30%. It is also striking to see that if those schemes imply a significant coverage in terms of respective national emissions, they mean very little in terms of world GHG emissions, as illustrated by the "World" coverage. Except for the "structural break" observed in 2005 for some countries, which reflects the fact that they adapted their legislation to avoid an overlap with the EU-ETS, coverage of GHG emissions by tax schemes is, for each country individually, relatively stable over time. Similarly, the coverage induced by the ETS in the selected countries does not show significant variation over time. Yet, one notes that all countries that are part of the EU-ETS exhibit different coverage figures, despite the ETS being harmonised across all

[10] See appendix B for a description of the methodology. At the national level, although our initial intention was to cover all jurisdictions, the cross-section dimension of our panel has been constrained by IEA emissions data availability while the time dimension has been constrained by CAIT data availability. At the subnational level, our focus on North America is driven by the fact that, up to 2015, that region concentrated most of the subnational carbon pricing schemes and that we were unable to gather robust data on the Chinese ETS pilot schemes or the Tokyo, Saitama, and Kyoto schemes.

[11] Besides the 'World' jurisdiction, the jurisdictions for which information is presented in Figures 1 to 4 are among the earliest adopters of carbon pricing mechanisms – exclusive of the EU-ETS countries – and for which the information is therefore available for a number of years.

countries. A potential explanation for these cross-country differences is that they reflect the differences in economic structure across participating countries.

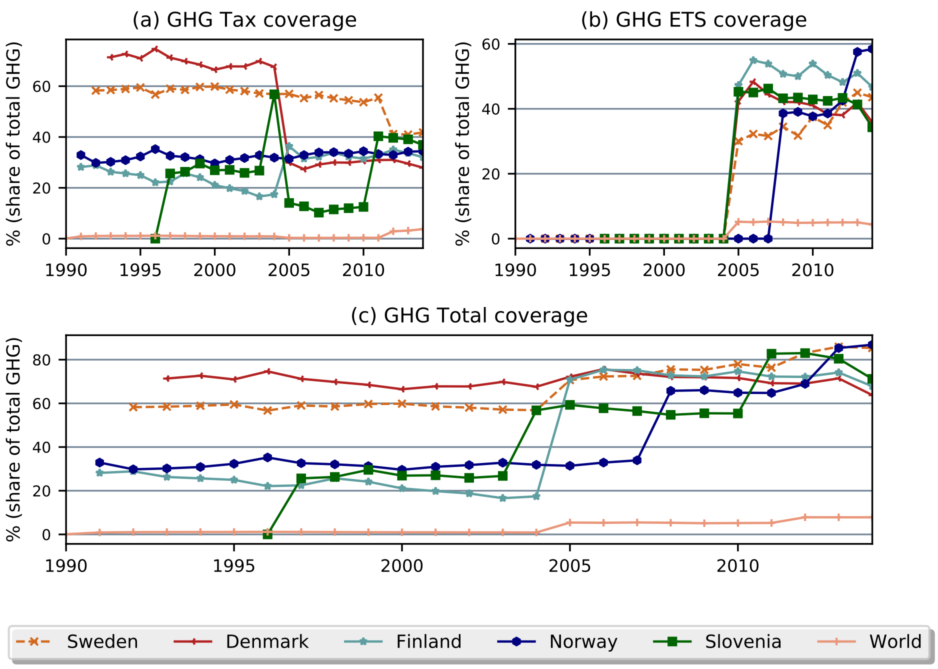

Figure 1: Carbon pricing coverage – selected (national) jurisdictions

In addition to the jurisdictions presented in Figure 1 several others have introduced carbon pricing policies. Switzerland introduced a carbon tax in 2008 covering about 28% of its total emissions. The coverage remained relatively stable over time, with the scheme covering 27% of emissions in 2012. In that same year, Japan introduced a carbon tax covering 69% of its emissions. Other jurisdictions opted for ETSs. This is the case of New Zealand, which introduced its scheme in 2010 with a coverage of 43% (gradually increased to 54% following inclusion of waste treatment activities in the scheme).

At the subnational level, another group of jurisdictions can be identified: US States participating in the Regional Greenhouse Gases Initiative (RGGI). This scheme is a regional initiative gathering initially 10 (but now 9: the state of New Jersey pulled out of the scheme in 2012) North-Eastern US States. Figure 2 shows the implied coverage of the scheme in the 10 participating states over the period 2009 (start year)-2012. It is again striking to see that substantial cross-state variation characterizes coverage. New Hampshire exhibits the highest coverage over the entire period, oscillating between 36.41% in 2009 and 34.42% in 2014. The coverage in all other participating states is between

13.19% (New Jersey - 2009) and 34.42% (New Hampshire - 2012). Outside the RGGI initiative, British Columbia launched its own carbon tax scheme in 2008, covering roughly 70% of its total GHG emissions while, in 2013, California introduced a Cap-and-Trade (CaT) mechanism covering approximately 32% of its emissions.[12]

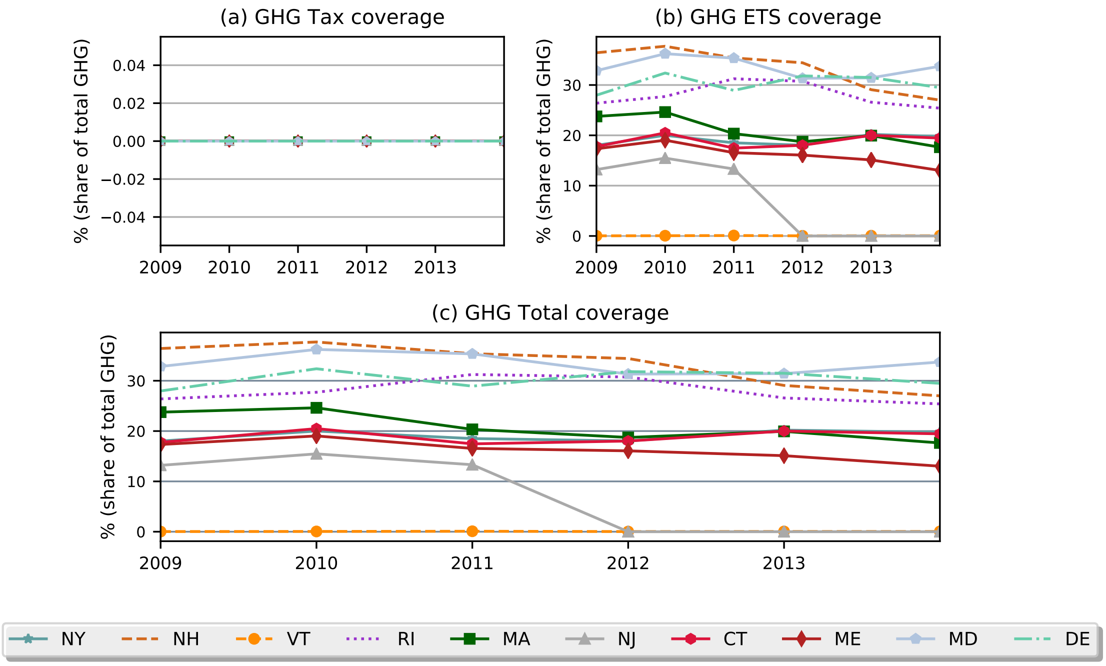

Figure 2: Carbon pricing coverage – US Regional Greenhouse Gas Initiative

4.1.2 Price

Coverage is only one side of the coin. The other is the price level. Countries that have introduced carbon pricing policies have experimented with different strengths of the price signal, which varies mainly along three dimensions: time, jurisdiction, and sector(-fuel). In other words, the price signal varies both across and within countries, introducing distortions between countries as well as between sectors of a given country. Importantly, however, distortions introduced by Emissions Trading Schemes are only between covered and non-covered sectors (since the price signal is the same across all covered sectors and fuels) whereas a carbon tax scheme also introduces distortions at the sector-fuel level.

Figure 3 displays the total (i.e. the sum of the tax rate and the ETS allowance price, as applicable) price of CO2 (in 2015 \$US/tCO2e) in selected sectors of selected countries

[12] As of January 1^{st}, 2015, new activities were added to the California CaT, increasing coverage to about 85% of California's total GHG emissions.

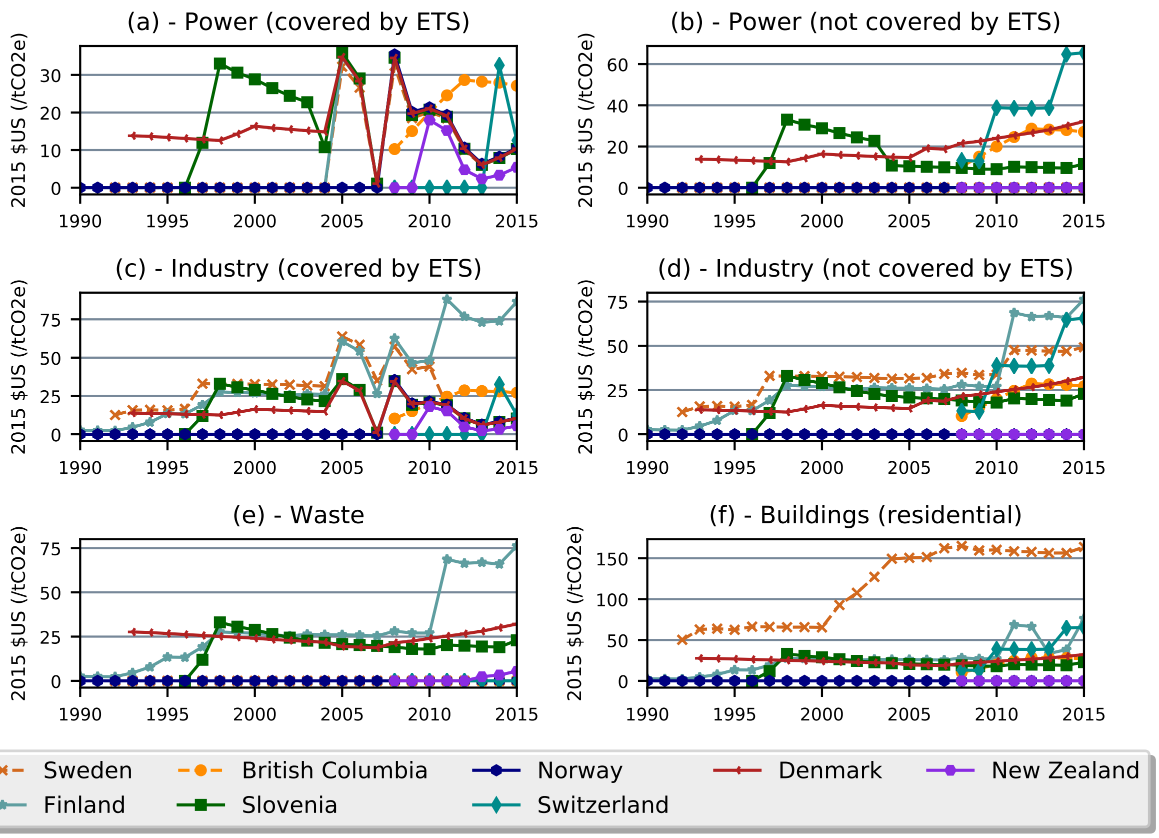

Figure 3: Total carbon price over time – coal/peat

for coal.[13] The carbon price does not vary much across fuels, suggesting that most tax schemes apply the same tax rate to all fossil fuels. The most significant variations arise across countries and, hence, across sectors within those countries. A look at panel (b) of Figure 3 shows that, among the selected countries, the power sector in Sweden is confronted to the highest price signal whereas the sectors in the other countries face much lower carbon prices.

4.1.3 The Emissions-weighted Carbon Price (ECP)

Combining sector- or sector-fuel-level coverage and price information allows for the calculation of an economy-wide Emissions-weighted Carbon Price (ECP). To compute the ECP each emitted ton of GHGs is attributed the corresponding total price signal. That is, emissions covered by either a tax or an ETS receive the associated tax rate or permit price as price tag whereas emissions of a sector covered by both schemes receive

[13] Figures A.1 and A.2, available in appendix A, show the total price for oil and natural gas, respectively.

the sum of the tax rate and the permit price.[14]

The evolution of the ECP in selected countries over the period 1990-2015 is presented in Figure 4. One observes that among all selected countries, only Sweden's ECP has increased steadily over time. All other countries exhibit either constant (e.g. Norway) or decreasing (e.g. Denmark) ECPs. Moreover, contrary to what is generally understood, the ECP varies across countries that are part of the EU-ETS. This is partly due to the presence of carbon taxes in some – but not all – countries, which create an additional price signal for some emissions. It is also, perhaps more importantly, due to differences in the relative size of sectors and their respective CO_2 intensity, as mentioned in section 4.1.1. This feature is particularly well illustrated by the ECP of states participating in the US RGGI (Figure 5).

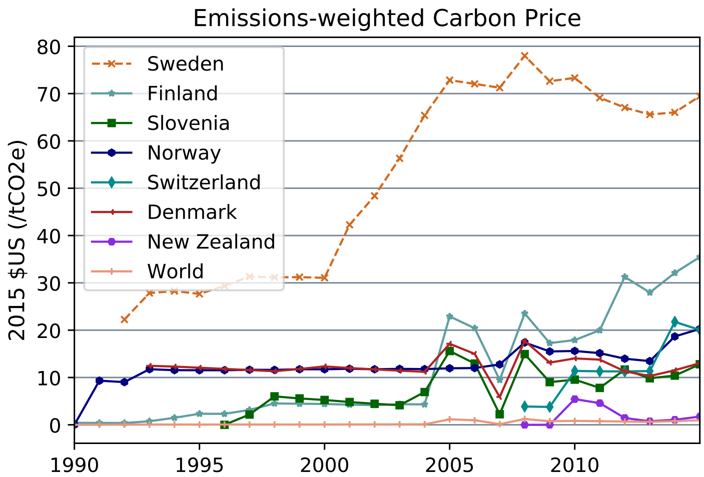

Figure 4: ECP – selected jurisdictions

Lastly, note that some countries' ECP exhibit more variability than others. For this specific group of countries, this is due to the relative importance of emissions covered by the EU-ETS as opposed to those covered by the respective national carbon taxes. Indeed, the (futures) price of EUAs, i.e. EU-ETS emissions allowances, exhibited strong variability over the sample period.

[14]Note that the ECP can be computed using time-varying or fixed weights. In the former case, weights (i.e. sector-fuel emissions share) are the year-specific emissions share; in the latter, we use 2013 emissions share. The latter is used in the empirical analysis as it is not subject to changes in emissions shares (i.e. weights) that might have been the result of the policy itself. See appendix C for a formal presentation.

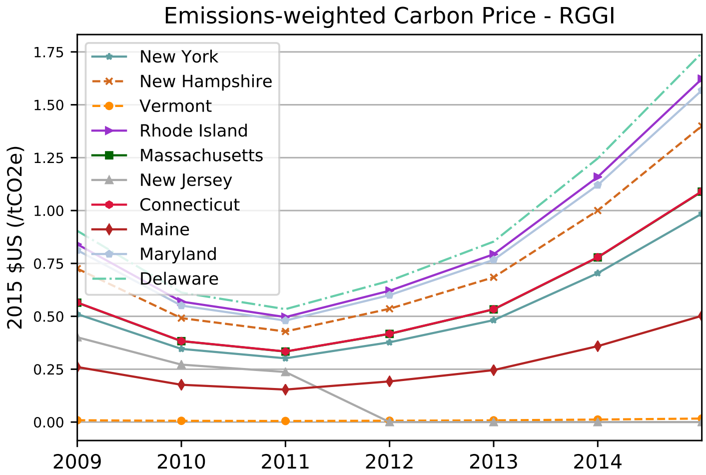

Figure 5: ECP – RGGI

4.2 Hypotheses

We now formulate hypotheses about the determinants of policy implementation and stringency. These are grouped as follows: (i) regulatory capture; (ii) political institutions; and (iii) macroeconomic determinants.

4.2.1 Regulatory capture

Power sector Any form of carbon pricing that includes the power sector might impose costs (e.g. reduced profits or capital losses) on those electricity producers that produce electricity from fossil fuels. We expect these costs to be higher the larger the share of electricity produced from fossil-fuelled power plants which, following Olson (1965), would weaken the political feasibility of carbon pricing regulation. This argument needs to be nuanced, however. First, the extent to which carbon pricing policies affect the value of covered firms depends on their capacity to *pass* the additional cost *through* to consumers. Under perfect competition and 100% pass through, electricity producing firms' profits will remain largely unaffected. With less than 100% pass through the change in equilibrium market price will not entirely reflect the increase in cost and firms' profit will be affected. Second, one does not necessarily expect the electricity generating sector to react in the same way to a carbon tax and a cap-and-trade system. In the case of the former, the sector will, at best, remain unaffected whereas in the case of an ETS, the possibility of capturing significant "windfall profits" exists if emissions permits are freely allocated.

Such a possibility has probably played a significant role in dampening the opposition of affected sectors to the introduction of such schemes. Several studies have examined that possibility and the associated rent-seeking behaviour both theoretically (Rode, 2013) and empirically (Markussen and Svendsen, 2005). The empirical evidence suggests that powerful (and CO2-intensive) sectors were successful in influencing the design of GHG trading systems. In fact, except for the US RGGI, all emissions trading schemes have been introduced with close to 100% free allocation of emission permits (World Bank, 2014). It is difficult to explicitly account for such effects in an econometric investigation but we note at this stage that it is likely to reduce the magnitude of the coefficients on the variables accounting for the role of CO2-intensive sectors, including the power sector.

Industry Besides the power sector, other energy-intensive sectors, broadly defined as "industry" are likely to oppose a carbon pricing scheme on the grounds that it holds the potential to increase production costs. There are two channels via which costs to industry could be pushed upward by a carbon pricing policy. A direct channel whereby CO2-intensive industries that fall within the scope of a carbon pricing scheme will have to pay for their own CO2 emissions; and an indirect channel whereby the introduction of carbon pricing policies covering the electricity generating sector leads to an increase in wholesale (and retail) electricity prices (as has been observed after the introduction of the EU-ETS (Sijm et al., 2008)) which, in turn, might raise the production cost of electricity-intensive industries. This argument closely follows Cadoret and Padovano (2016).

International competitiveness As emphasised by Aldy and Pizer (2012), sectors of the economy that are export-oriented should be more reluctant to the introduction of a carbon price as it risks putting them at a competitive disadvantage in international markets. Care is usually taken to design the schemes in ways that minimise the international competitive disadvantage that domestic firms may suffer from but jurisdictions that are very exposed to international markets may nonetheless be less inclined to implement carbon pricing policies.

4.2.2 Political institutions

Political regime Congleton (1992) argues that autocrats' time horizon is shorter than that of democratic planners and they therefore set weaker environmental targets. Yet, Hahn (1990) also argues that liberal democracies offer the possibility for different interest groups to express their views and "weigh" on the legislative process, in which case regulatory outcomes will be a balancing act that reflects the relative bargaining power of the different interest groups. This could work both in favour or against the introduction of carbon pricing policies, depending on interest groups' relative lobbying strengths.

Government ideology Prior studies have found left wing governments to implement more stringent environmental policies (Chang and Berdiev, 2011; Cadoret and Padovano, 2016). Fankhauser et al. (2015), however, found the political orientation of the government to be irrelevant to the number of climate laws passed in their sample of jurisdictions. We test whether the orientation of the executive branch of government with regard to economic policy affects the implementation and/or stringency of carbon pricing schemes.

Institutional capacity A relatively high degree of institutional capacity is a prerequisite for the introduction of any form of regulation and, a fortiori, to introduce a carbon pricing scheme. We expect institutional capacity to be positively correlated with the presence of a carbon pricing scheme but not necessarily with the level of the ECP. Indeed, the "institutional burden" arises from the creation of such a scheme, irrespective of the level of the price associated with it.

International dynamics Membership of international organisations (such as the OECD or the EU) or international institutional frameworks (such as the Annex-I countries of the Kyoto Protocol) plays a significant role in the presence and development of carbon pricing policies. For example, the EU, a club of countries cooperating on a wide range of issues – including the environment, has implemented an organisation-wide emissions trading system. Several EU Member countries currently part of the system were "dragged in" and implemented it only because it was part of the preexisting legislative *acquis* (Robinson and Stavins, 2015). 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. Having committed to a reduction of their GHG emissions, these countries may have had an additional incentive to develop climate mitigation strategies, including carbon pricing policies.

4.2.3 Macro(economic) determinants of environmental policy

Finally, besides the sector-specific stance towards carbon pricing and the political institutions of a jurisdiction, we note two further factors potentially affecting implementation and stringency of a carbon pricing scheme. First, under the assumption that environmental quality is a normal good, the willingness to pay for CO2 emissions abatement rises with income. Therefore, we expect the income level (per capita) to be positively associated with the probability of implementation of a carbon pricing policy as well as the Emissions-weighted Carbon Price. Second, given the direct economic cost that pricing carbon entails, larger emitters (per capita) may be less prone to introduce pricing policies.

Table 2: Summary of hypotheses

Category	Variable	Expected sign Carbon Price (Y/N)	Expected sign Carbon Price (Level)
Regulatory capture	Power-coal	-	-
	Power-oil	-/0	-/0
	Power-gas	-/0	-/0
	Industry	-	-/n.a.
	International competitiveness	+/-	+/-
Political institutions	EU	+	+
	Annex-I	+	+
	Institutional capacity	+	n.a.
	Level of democracy	+	n.a.
	Left	+	+
Macro determinants	GDP per capita (WTP)	+	+
	CO2 emissions per capita	-	-

5 Data and identification strategy

5.1 The dataset

The analysis is performed on three different panels: 124 national jurisdictions – panel A, 50 US States – panel B – and 13 Canadian Provinces – panel C.[15] Panel A runs

[15] Although the ECP is calculated for 135 national jurisdictions, data availability for some of our covariates constrains the panel dimension of our sample to 124 – models (I) and (III) – and 110 units –

over the period 1990-2015; panel B starts in 2008 and ends in 2015; panel C covers the years 2005-2015. This represents (a maximum of) 3224 country-year observations, 400 (US)State-year observations and 143 (Canadian) Province-year observations. The actual emissions-weighted carbon price is only observed for those jurisdictions that have selected into a pricing mechanism (either ETS or tax, or both). In 2015, 35 national jurisdictions, 11 US States and 2 Canadian Provinces had had a carbon pricing mechanism in force in at least one year. That is, the ECP is observed for 420 country-year, 70 (US)State-year and 11 (Canadian) Province-year pairs. This particular structure of the data has implications for our empirical analysis – see section 5.3.

5.2 Covariates

This section introduces the variables used to investigate the hypotheses presented in section 4.2.

Regulatory capture Previous literature has proxied the strength of the lobbying exercised by the power/energy sector in at least two ways. First, Fredriksson et al. (2004) and Fredriksson and Vollebergh (2009) use the share of value added of the energy industry in total GDP. Second, Marques et al. (2010) and Marques and Fuinhas (2011) disentangled the specific role played by different fossil fuel sources using the contribution of each of them to total electricity production. Since coal, gas and oil electricity generation would not be similarly affected by the introduction of a carbon pricing scheme, we follow this last approach and use their share in total electricity generation as proxies for their influence on policy developments. To capture the lobbying activity of CO2/energy-intensive industries, we follow Cadoret and Padovano (2016) and use the value added of industry (as a share of GDP). Finally, the effect of trade openness is captured by the sum of a jurisdiction's exports and imports (as a share of GDP).

Political institutions We introduce an institutional capacity indicator, constructed as the simple average of the World Bank's "Government Effectiveness" and "Regulatory

models (II) and (IV). For panel A, each estimator is therefore presented for two alternative specifications because the *Left* variable is not available for all panel units. In addition, panel A is unbalanced.

Quality" indicators (World Bank, WGI, 2016). This follows Steves et al. (2011).[16] The first year of these series is 1997 but they only became available on an annual basis in 2002. Therefore, years 1998 and 2000 are filled using a linear interpolation method. Our main proxy for the state of democracy (*Dem*) comes from the Center for Systemic Peace, Polity IV project (2015). As a robustness check, we also perform the analysis with two variables taken from the Varieties of Democracy Database. The first (*Polyarchy*) measures to what extent the ideal of electoral democracy is achieved whereas the second (*Libdem*) measures performance regarding the achievement of principles of liberal democracy (Varieties of Democracy, 2018).[17]

To investigate the effect of the political orientation of the executive with respect to economic policy in national jurisdictions we create, based on Cruz et al. (2018) in Varieties of Democracy (2018), a variable (*Left*) which takes value 1 whenever the ruling party is identified as left wing party and 0 otherwise. The '0' therefore lumps together right-wing and centre parties as well as parties whose political platform does not take a clear stance regarding economic policy. For panel B, the variable used (*Ideology*) captures the median ideology in the state's house of representatives, as defined in Shor and McCarty (2011). No such variable is available for subnational Canadian jurisdictions. Finally, the effect of EU membership (*EU*) is tested with the use of a dummy variable that takes value 1 whenever a country is a member of the EU, and 0 otherwise.[18]

Macro determinants To control for general economic and environmental conditions, we use GDP (PPP, $US 2011) and CO2 emissions (metric tonnes), both per capita.

5.3 Model and identification strategy

Our objective is to identify some of the determinants of carbon pricing policy implementation as well as stringency. To that end we introduce two (sets of) models. One that relates our covariates to a binary outcome variable recording the presence of a carbon

[16] Correlation with the World Bank *Control of Corruption* estimate is also investigated, see section 6.3.

[17] Such variables are not available for subnational jurisdictions but their inclusion would most likely add little to the model as there would be little cross-section and/or time variability.

[18] Earlier work tested the role of being listed in Annex-I and Annex-II of the Kyoto Protocol. Both institutional features turned out to have negligible impact on our outcome variables.

Table 3: Variable sources and summary statistics

Variable	Jurisdiction	Source	Mean	Std. Dev.	Min.	Max.	N
Pricing	National	Author (see appendix)	0.12	0.33	0	1	3510
	US States	Author (see appendix)	0.17	0.37	0	1	400
	Can. Prov./Terr.	Author (see appendix)	0.12	0.32	0	1	143
ECP	National	Author (see appendix)	12.84	16.67	0.002	95.21	420
(time-invariant weights)	US States	Author (see appendix)	0.88	1.38	0.01	9.66	70
	Can. Prov./Terr.	Author (see appendix)	16.24	10.18	2.19	29.48	11
Elec. generation-coal,	National	World Bank, WDI (2016)	16.78	25.87	0	100	3477
% of total	US States	U.S. EIA (2015)	39.53	29.11	0	97.79	400
	Can. Prov./Terr.	Statistics Canada (2016a)	15.88	24.77	0	69.93	143
Elec. generation-gas,	National	World Bank, WDI (2016)	22.63	30.12	0	100	3477
% of total	US States	U.S. EIA (2015)	24.48	23.64	0	98.51	400
	Can. Prov./Terr.	Statistics Canada (2016a)	6.82	9.52	0	38.37	143
Elec. generation-oil,	National	World Bank, WDI (2016)	19	27.96	0	100	3477
% of total	US States	U.S. EIA (2015)	2.32	10.34	0	76.21	400
	Can. Prov./Terr.	Statistics Canada (2016a)	3.25	7.46	0	34.65	143
Industry VA,	National	UN data	32.11	11.67	6.3	84.65	3452
% of GDP	US States	US BEA (2016)	21.6	7.3	0	46.5	400
	Can. Prov./Terr.	Statistics Canada (2016b)	29.83	11.26	15.37	58.56	143
EU	National	Author-created	0.15	0.35	0	1	3510
Institutional capacity	National	World Bank, WGI (2016)	0.1	0.98	-2.19	2.26	2538
Democracy	National	Polity IV project	3.7	6.74	-10	10	3247
Left	National	Varieties of Democracy (2018)	0.33	0.47	0	1	2894
Ideology	US States	Shor and McCarty (2011)	0.08	0.72	-1.47	1.23	338*
GDP per capita,	National	World Bank, WDI (2016)	17810.61	19575.93	354.28	129349.9	3338
PPP $2011 USD	US States	US BEA (2016)	47662.9	9118.25	31565.52	74417.54	400
	Can. Prov./Terr.	Statistics Canada (2016c)	44510.85	13280	28806.69	95355.49	143
Trade openness,	National	World Bank, WDI (2016)	83.14	49.01	0.02	441.60	3348
% of GDP	US States	U.S. Census Bureau (2016)	18.99	9.64	4.04	59.05	350*
	Can. Prov./Terr.	Statistics Canada (2016c)	57.56	18.55	15.78	111.71	143
CO2 emissions, t/cap	National	World Bank, WDI (2016)	5.62	7.24	0.017	70.14	3283
	US States	CAIT (2015)	23.76	19.51	8.47	130.7	350*
	Can. Prov./Terr.	Statistics Canada (2018)	20.14	13.23	6.47	52.81	143
Polyarchy	National	Varieties of Democracy (2018)	0.45	0.28	0.01	0.916	3176
Libdem	National	Varieties of Democracy (2018)	0.56	0.27	0.017	0.95	3176
Corruption	National	World Bank, WGI (2016)	0.02	1.05	-2.06	2.59	3150

*The *Ideology* covariate is unavailable for the states of Massachusetts and Nebraska.

*The *Trade openness* and *CO2 emissions* are unavailable for 2015.

pricing scheme for a given jurisdiction-year entry, another relating some of these same covariates to the stringency of the scheme. A general representation of each of them is

$$\mathbb{1}_{it} = \alpha + \boldsymbol{\psi'X_{it}} + \boldsymbol{\gamma'Z_{it}} + \boldsymbol{\eta'W_{it}} + d_t + u_{it} \tag{1}$$

$$ECP_{it} = \eta + \delta ECP_{it-1} + \boldsymbol{\psi'X_{it}} + \boldsymbol{\gamma'Z_{it}} + \boldsymbol{\eta'W_{it}} + d_t + \phi_i + \epsilon_{it} \tag{2}$$

where $\mathbb{1}_{it}$ is an indicator variable capturing the operation of a carbon pricing scheme, ECP_{it} is the Emissions-weighted Carbon Price, $\boldsymbol{X_{it}}$ is the vector of regulatory capture variables, $\boldsymbol{Z_{it}}$ is the vector of political and institutional variables and $\boldsymbol{W_{it}}$ is the vector of macro(-economic) variables. ϕ_i is the unobserved jurisdiction fixed-effect while d_t is the vector of time dummy variables; $\boldsymbol{\psi'}$, $\boldsymbol{\gamma'}$ and $\boldsymbol{\eta'}$ are vectors of dimensions m, n and p, respectively, each element of which corresponds to the estimated parameter of the associated explanatory variable. u_{it} and ϵ_{it} are the observation specific error terms.

In estimating (1) and (2), two potential problems may arise. First, as much as economic structure and electricity generation mix may affect policy implementation and stringency, the latter can also affect the former, creating a reverse causality problem and causing standard estimation approaches to fail. Second, there may be endogenous selection into the policy, which would bias the coefficient estimates in equation (2).[19]

Reverse causality The potential presence of simultaneity bias prompts us to: (i) note the features of the data that make our analysis less prone to it, (ii) describe the steps taken to minimise and subsequently address this issue. First, note that $\mathbb{1}_{it}$ records *implementation*, not *passage* of the legislation. In most cases, the year of implementation differs from the year the legislation is passed, the former following the latter by a lag of 1 (in the case of the EU-ETS) to 3 years (in the case of, for example, Chile's carbon tax). This provides a rationale for the use of lagged values of all the variables included as regressors in equation (1) – see Table 4 – and prevents the possibility that its outcome variable would determine the covariates – at least in a contemporaneous manner. That

[19]Note, however, that a selection bias would only be present insofar as the population of interest is the entire set of jurisdictions initially present in our panels.

is, some regressors may only be pre-determined.

Second, an endogeneity problem only arises if the policy, once implemented, works as intended. Carbon pricing policies were primarily designed to affect jurisdictions' CO2 emissions through altered technological choices or structural changes in the composition of the economy. While it cannot be ruled out that some of these policies (especially the most stringent ones) did have the intended effects, it is worth noting that: (i) these policies affect the economy only slowly, (ii) technological advances in abatement technology may reduce structural shifts in economic composition, (iii) except for a few jurisdictions, most carbon pricing policies introduced over the period covered in the sample have been relatively weak, (iv) several jurisdictions introduced said schemes towards the end of our sample period. All this suggests that it is unlikely that the value added of industry, which we use to proxy for the lobbying intensity of energy-intensive sectors, will be determined by policy implementation or stringency. However, the same argument does not hold as strongly for the electricity mix variables or CO2 emissions, which tend to be more sensitive to carbon prices.

Selection We only observe the stringency for the jurisdictions that have a scheme in operation in any given year. If there is correlation between the selection/participation and level/stringency decision, then the process is best modelled as a model with incidental truncation (or selection) where both a level and a selection equation are specified and a correction for the selection bias is applied (Gronau, 1974; Heckman, 1976). That is, equation (2) should be complemented with the introduction of a latent variable (see e.g. Semykina and Wooldridge (2010)):

$$s_{it} = 1[z_{it}\delta_t + \phi_{i2} + v_{it} > 0] \tag{3}$$

where 1[.] is an indicator function and s_{it} is a selection indicator that equals 1 if ECP_{it} is observed and 0 otherwise.

Given the above, we implement the following econometric approach, applied to each panel separately. First, we estimate (1) with random effect logit and probit models. All regressors in estimations of equation (1) are introduced with a two-period lagged value to account for the lag between passage of legislation and policy implementation. Second, we provide results of (FE) OLS and system GMM estimations for equation (2). In addition to controlling for unobservable time-invariant fixed effects, the GMM estimator allows us to account for potential endogeneity of the regressors and model the persistence of the stringency variable. The GMM approach has been taken in Marques and Fuinhas (2011) to study the relationship between a very similar set of covariates and renewable energy deployment.

We correct for the selection bias following by introducing a sample selection correction term. This term is calculated for each ECP observation using a Heckit approach adapted from Semykina and Wooldridge (2010). In a first step, a probability of occurrence of that observation is obtained by estimating, for each t, equation (3) using a probit regression on the entire cross-section of jurisdictions. In a second step, this term is then included in the estimation of equation (2) as a regressor. For panel A, the set of regressors for the selection equation includes the electricity generation mix variables, the value added of industry, CO2 emissions and GDP per capita, trade openness, *Dem* and *EU*.[20] All variables are introduced with a two-year lag to reflect our theoretical assumption that operation of a carbon pricing scheme ($s_{it} = 1$) is dictated by past economic and institutional structure. For panel B, the same variables are included, except *EU*.[21]

6 Estimation results

We comment separately on the estimation of equations (1) and (2). The results are presented for panels A, B and C. All estimations include year fixed-effects, and all estimations of equation (2) include country fixed-effects.

6.1 Implementation

Panel A – National jurisdictions Table 4 presents the results of both RE logit and RE probit estimations, which lead to convergent conclusions about the effects of the co-

[20] Institutional capacity and *Left* are not part of the selection equation since the former is not available until 1997 and the latter further reduces the panel dimension.

[21] Given the paucity of ECP observations for Canadian Provinces & Territories (2 panel units, for a total of 9 observations) no estimation is presented.

variates on the decision to implement a carbon pricing scheme. First, estimates suggest that a larger share of electricity generated from gas and oil fired power plants lowers the probability of subsequent introduction of a carbon pricing scheme. This is in line with our expectation that jurisdictions whose electricity generation system relies more heavily on fossil fuels would face greater opposition to the introduction of such schemes. The estimates of the coefficient on the share of coal in the electricity system, however, do not indicate a consistent pattern of influence on the implementation of carbon pricing mechanisms, which runs against the understanding that jurisdictions with coal fired electricity systems would fiercely oppose the introduction of carbon pricing policies. One potential explanation for this is that 'dirty' electricity producers have been granted significant compensation in the schemes introduced so far. For example, in the case of the EU-ETS, CO2-intensive electricity producers and heavy industries were "bought in" by grandfathering emissions allowances in the first two phases of the operation of the system. Second, within the sample of national jurisdictions, we find little evidence supporting the hypothesis that larger industry or manufacturing sectors hindered the introduction of carbon pricing policies. Although coefficient estimates are negative across all estimations, they are only weakly statistically different from zero. One potential explanation for this observation is that the set of panel units includes both strongly industrialised and less industrialised jurisdictions, with most carbon pricing policies having been introduced within the former group. Lastly, trade openness does not seem to have played a determining role in the introduction of carbon pricing policies. Again, one can plausibly suggest that it is related to the fact that existing schemes have: (i) covered non-traded sectors; (ii) provided sectoral exemptions/compensation for industries exposed to international competition. Third, the institutional environment does play a role in the adoption of carbon pricing policies. Results suggest a consistent pattern of introduction of carbon pricing policies among jurisdictions that rank higher on the Polity IV democracy index and have a stronger institutional capacity as calculated in this paper. This partially supports the hypothesis formulated by Congleton (1992) that democratic institutions are conducive to more stringent environmental regulations (*a* carbon price, regardless of its level, is a more stringent policy than *no* price at all) and lends support to Hahn's conclusions (1990) that environmental regulation is a balancing act between a variety of interests (the actual stringency is determined by the relative weight of each interest group and not by the 'democratic' nature of a political system). This might also suggest that the "green" lobby is effectively given some weight in the policy making process. The economic orientation of the executive does not seem, on the contrary, to play a significant role in the adoption of carbon pricing policies, suggesting that, in the sample currently considered, such policies have received support from parties across the political spectrum. This result is in line with Fankhauser et al. (2015). The results also highlight the (international) institutional dynamics at play in the development of carbon pricing policies as EU membership is found to strongly affect the probability of introduction of a carbon pricing scheme, an result that is likely to be mainly driven by the introduction of the EU-ETS in 2005. The results also indicate that, all else equal, larger emitters (per capita) have been more to likely introduce carbon pricing mechanisms, reflecting the fact that, until now, carbon pricing mechanisms have been introduced in more economically advanced jurisdictions with large CO2/capita emissions, possibly following international commitments. Finally, GDP per capita has a positive effect on the introduction of a carbon pricing mechanism ((I) and (III)), although the statistical significance of this effect vanishes when institutional capacity is accounted for ((II) and (IV)). Its magnitude changes depending on the econometric specification but the direction of the induced change is stable across all estimated models, strongly suggesting that economic agents are more likely to support the introduction of such policies if they are relatively better off.

Panels B & C – US States & Canadian Provinces The results for the subnational jurisdictions considered are broadly consistent with those based on panel A, although we note some interesting differences.[22] In the US, results indicate that electricity generation from fossil fuels negatively impacted the development of carbon pricing mechanisms, with the largest absolute effect associated with electricity generation from oil. This latter

[22]Panel B contains only *Ideology* as political variable and panel C contains no variable reflecting the state of political institutions.

Table 4: Implementation – Outcome: $\mathbb{1}$

Panel	Category	Variable	RE Logit		RE Probit	
			(I)	(II)	(III)	(IV)
A - National	Regulatory capture – $\boldsymbol{X}$	Power-Coal$_{t-2}$	-0.014	-0.016	0.01	0.043
			(0.0355)	(0.0358)	(0.0165)	(0.0184)
		Power-Gas$_{t-2}$	-0.065*	-0.096	-0.029	-0.063*
			(0.0467)	(0.0616)	(0.0207)	(0.0359)
		Power-Oil$_{t-2}$	-0.286***	-0.188***	-0.109***	-0.132***
			(0.0812)	(0.0775)	(0.0332)	(0.0434)
		Industry, VA$_{t-2}$	-0.141	-0.114	-0.06	-0.054
			(0.1319)	(0.2163)	(0.0706)	(0.0943)
		Trade Openness$_{t-2}$	0.023	0.008	0.01	0.01
			(0.0198)	(0.023)	(0.0087)	(0.0147)
	Political institutions – $\boldsymbol{Z}$	Level of Democracy$_{t-2}$	1.756***	0.82***	0.759***	0.834***
			(0.2229)	(0.4216)	(0.0972)	(0.2363)
		EU$_{t-2}$	14.449***	17.9***	5.643***	11.249***
			(4.1656)	(3.2755)	(1.5721)	2.095
		Left$_{t-2}$		-1.209		-0.564
				(1.4653)		(0.8598)
		Institutional capacity$_{t-2}$		7.423**		5.925***
				(3.0055)		(1.6916)
	Macro(economic) environment – $\boldsymbol{W}$	GDP per cap.$_{t-2}$	0.397***	0.093	0.167***	0.125**
			(0.0489)	(0.1014)	(0.0306)	(0.0599)
		CO2 Em$_{t-2}$	0.153	0.199	0.045	0.036
			(0.1575)	(0.2003)	(0.0812)	(0.1103)
		Constant	-46.352***	-42.879***	-20.257***	-33.924***
			(7.3768)	(9.8192)	(3.7665)	(4.8143)
		Year dummies	Yes	Yes	Yes	Yes
	Observations		2726	1793	2726	1793

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

Note: (i) The institutional capacity variable is observed for the first time in 1997, shortening the time dimension of models (II) and (IV). Together with the lower number of panel units available for *Left* explains the lower number of observations; (ii) all time dummies from 2005 onwards were statistically significant at the 1% level in models (I) to (IV). We also took a different approach and introduced a time trend. It did not, however, exhibit any significance.

effect is, however, only significant in model (VI). The results also suggest that carbon pricing mechanisms were less likely to be introduced in states with high CO2 emissions per capita as well as states for which (CO2-intensive) industry represents a large share of total economic activity – consistent across models (I) and (II). Conversely, states with relatively higher income per capita were more likely to introduce carbon pricing. The insights obtained from panel C need to be interpreted with caution as only two Canadian Provinces introduced a carbon pricing mechanism over our sample period. Nonetheless, results suggest that richer (per capita) jurisdictions were more likely to introduce carbon pricing mechanisms whereas larger CO2 emitters (per capita) or jurisdictions with relatively larger industry were less likely to do so. In that regard, it is interesting to note the contrasted dynamics between national and subnational jurisdictions. For example, national jurisdictions with large CO2 emissions (per capita) seem to have taken the lead in pricing carbon (perhaps due to international commitments) whereas large (per capita) subnational emitters have not.

Table 5: Implementation – Outcome: $\mathbb{1}$

Panel	Category	Variable	RE Logit	RE Probit
			(V)	(VI)
B – US States	Regulatory capture – $\boldsymbol{X}$	Power-Coal$_{t-2}$	-0.176	-0.056
			(0.1259)	(0.0915)
		Power-Gas$_{t-2}$	-0.041	-0.024
			(0.0746)	(0.0448)
		Power-Oil$_{t-2}$	-0.369	-0.311*
			(0.3602)	(0.1603)
		Industry, VA $_{t-2}$	-2.27***	-1.435***
			(0.7442)	(0.4267)
		Trade Openness$_{t-2}$	0.304	0.202
			(0.202)	(0.1521)
	Political institutions – $\boldsymbol{Z}$	Ideology$_{t-2}$	1.128	0.318
			(3.3663)	(2.3116)
	Macro(economic) environment – $\boldsymbol{W}$	GDP per cap.$_{t-2}$	0.691***	0.387**
			(0.2381)	(0.1765)
		CO2 per cap.$_{t-2}$	-1.794**	-1.387**
			(0.84)	(0.5495)
		Constant	29.839	23.549
			(21.1233)	(14.739)
		Year dummies	Yes	Yes
	Observations[a]		249	249
C – Canadian Prov. & Territories	Regulatory capture – $\boldsymbol{X}$	Power-Coal$_{t-2}$	0.544	0.208
			(0.8818)	(0.4137)
		Power-Gas$_{t-2}$	1.093	0.481
			(0.6613)	(0.3694)
		Power-Oil$_{t-2}$	-5.5328	-1.918
			(4.0517)	(1.8411)
		Industry, VA$_{t-2}$	-0.193	-0.092
			(1.1889)	(0.3729)
		Trade Openness$_{t-2}$	-0.405	-0.159
			(0.3713)	(0.1843)
	Macro(economic) environment – $\boldsymbol{W}$	GDP per cap.$_{t-2}$	0.556	0.271
			(0.748)	(0.3140)
		CO2 per cap.$_{t-2}$	-0.994	-0.338
			(2.0007)	(0.9613)
		Constant	-14.086	-4.699
			(19.9356)	(10.8357)
		Year dummies	Yes	Yes
	Observations[a]		117	117

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

[a] Panel A: from a number of observations of 400, we lose 2x50 observations given that we use two-years lag. Panel B: 2x13 observations are lost.

Year dummies from 2012 onwards are significant in Panel B estimations. None of the year dummies for panel C were significant.

6.2 Stringency

This section discusses the estimation results of equation (2). Tables 6 and 7 present the results for national jurisdictions and US States, respectively. Models (VII) and (IX) do not account for potential sample selection bias whereas models (VIII) and (X) do. The results clearly demonstrate a very high persistency of policy stringency. This effect is present across all panels, consistent across estimators and of the order of 0.8-0.9, indicating that carbon pricing policy stringency changes very slowly over time. The evidence regarding the role of the electricity generation portfolio is mixed. In panel A, electricity generation from coal is found to have a negative (significant) impact on the stringency of implemented schemes only in model (X). The associated estimated coefficient suggests that an increase of 10% in the share of electricity generated from coal would lead to a 1.2USD/tCO2e decrease in the average carbon price. Model (IX) identifies a similarly negative effect, albeit of roughly half the magnitude identified in model (X). Similar mechanisms are identified for the share of gas in the electricity generation mix, which weighs negatively on the stringency of the carbon pricing policy. A 10% increase in this share would reduce the average carbon price by between 0.9USD/tCO2e and 1.8USD/tCO2e. Next, coefficient estimates of all models except model (IX) point at a negative effect of the relative strength of industry on the policy stringency. The magnitude of this negative effect varies from -0.17USD/tCO2e in model (X) to -0.04USD/tCO2e in model (VIII) for each 1% increase in the share of industry in total GDP. Results do not support the existence of a clear relationship between trade openness and policy stringency, most likely for the same reasons that it did not seem to decisively affect implementation. Interestingly, results indicate that being part of the EU had a positive yet not statistically significant impact on stringency. The magnitude of the estimated effect varies greatly across models (VII)-(X), with the largest effect is estimated in model (X), at about 10USD/tCO2e. Finally, the effect of CO2 emissions per capita is not clearly identified, being negative in models (VII) and (VIII) where estimates may suffer from simultaneity bias, and positive yet with weak statistical significance in models (IX) and (X).

Insights from panel B point a similarly persistent stringency process, with a coefficient estimate for ECP_{t-1} of 0.87 in model (XIV). Estimates in models (XI) and (XII) are most likely severely biased due to the short time dimension of Panel B. The results also suggest that larger CO2 emitters and more industrious (CO2-intensive) US States had

Table 6: Stringency – ECP (2015 $US/tCO2e)

Panel	Category	Variable	FE OLS (VII)	FE OLS, Heck (VIII)	Syst GMM (IX)	Syst GMM, Heck (X)
A – National		ECP_{t-1}	0.875***	0.879***	0.824***	0.824***
			(0.0438)	(0.052)	(0.1126)	(0.1402)
	Regulatory capture – $\boldsymbol{X}$	Power-Coal$_t$	0.068**	0.064*	-0.0554	-0.123**
			(0.0261)	(0.0324)	(0.0773)	(0.111)
		Power-Gas$_t$	0.022	0.0304	-0.088*	-0.181
			(0.0274)	(0.0268)	(0.0508)	(0.0911)
		Power-Oil$_t$	-0.016	-0.013	0.095	0.061
			(0.1084)	(0.1089)	(0.1265)	(0.1609)
		Industry, VA$_t$	-0.084	-0.041	0.068	-0.173
			(0.1021)	(0.0918)	(0.2857)	(0.4142)
		Trade Openness$_t$	-0.014	-0.021	0.02	0.028
			(0.0239)	(0.0242)	(0.026)	(0.0427)
	Political institutions – $\boldsymbol{Z}$	EU$_t$	0.621	0.127	3.801	10.41
			(1.0813)	(0.8366)	(6.101)	(7.9493)
	Macro(economic) environment – $\boldsymbol{W}$	CO2 Em$_t$	-0.533**	-3.352*	0.458	0.453
			(0.2058)	(1.9516)	(0.5451)	(0.6637)
		Constant	6.186	7.861**	5.499	9.101
			(4.0966)	(3.6756)	(16.577)	(20.9732)
		Year dummy	Yes	Yes	Yes	Yes
		Sample selection term	No	Yes	No	Yes
		Instruments			GMM-sys	GMM-sys
		Observations	349	349	349	349
		AR(1) test			-3.38***	-2.79***
		AR(2) test			0.64	0.16
		Sargan (χ^2)			10.12	3.95

Standard errors in parentheses * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$
Time dummies for years 2005 and 2007 to 2010 were significant in models (VII) and (VIII) but not in models (IX) and (X).
Including GDP in the stringency equation does not qualitatively nor quantitatively change the results.
The sample selection terms are significant in regressions (VIII) and (X)

more stringent carbon pricing mechanisms. These effects are, however, not statistically significant when estimated with the GMM estimator.

6.3 Robustness checks and discussion

To test the robustness of our results, we perform a series of additional estimations. First, to address concerns that the coefficient estimates of equation (1) may be biased due to correlation between past residuals and current regressors, we repeat its estimation with a sample where, for each panel unit, the time dimension is (right)-curtailed at the year of policy adoption. That is, for each panel unit in which a carbon pricing scheme was introduced, we keep only one observation for which $\mathbb{1} = 1$. The results are qualitatively similar, although statistical significance of the estimated coefficients decreases. Second, we note that for equation (1), i.e. regression models (I) to (VI), the results do not change qualitatively if we choose a different (meaningful) lag structure. In particular,

Table 7: Stringency – ECP (2015 $US/tCO2e)

Panel	Category	Variable	FE OLS (XI)	FE OLS, Heck (XII)	Syst GMM (XIII)	Syst GMM, Heck (XIV)
B – US States		ECP_{t-1}	1.042***	1.025***	0.88***	0.873***
			(0.0815)	(0.0824)	(0.0462)	(0.0526)
	Regulatory capture – $\boldsymbol{X}$	Power-Coal$_t$	-0.014*	-0.015*	-0.004	-0.004
			(0.0076)	(0.0079)	(0.0051)	(0.0047)
		Power-Gas$_t$	-0.001	-0.0036	0	0.003
			(0.0037)	(0.0047)	(0.0045)	(0.0064)
		Power-Oil$_t$	0.025	0.027	0.142***	0.138***
			(0.0316)	(0.0372)	(0.045)	(0.0497)
		Industry, VA$_t$	0.018***	0.022**	0.011	0.011
			(0.0057)	(0.0078)	(0.0072)	(0.0067)
		Trade Openness$_t$	0.032***	0.029***	0.006	0.013
			(0.0077)	(0.0082)	(0.0132)	(0.0171)
	Macro(economic) environment – $\boldsymbol{W}$	CO2 Em(Mt)$_t$	0.025***	0.023	0.074	0.046
			(0.0254)	(0.0334)	(0.0585)	(0.0557)
		Constant	-0.573	-0.924**	-	-
			(0.4375)	(0.39)		
		Time dummies	Yes	Yes	Yes	Yes
		Sample selection	No	Yes	No	Yes
		Instruments			GMM-sys	GMM-sys
		Observations	49	49	49	49
		AR(1) test			1.51	1.03
		AR(2) test			1.27	0.99
		Sargan (χ^2)			39.41***	25.56***

Standard errors in parentheses: * $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$
All year dummies are statistically significant at (at least) the 5% level in models (XI) and (XII). In models (XIII) and (XIV), year dummies 2010 and 2011 are not significant; all others are.

they are robust to the use of one-period lagged values of the regressors. Third, it can be argued that, over the period considered, there exists a structural break in carbon pricing policy developments as many jurisdictions introduced such policies after 2005. To test whether the dynamics presiding over these developments differ prior and after 2005, we re-estimate models (II) and (IV) over two different time periods: 1990-2004 and 2005-2015. For the period 1990-2004, the share of electricity generated from fossil fuel is negatively associated with the probability of implementation of carbon pricing policies. This observation is made for all fossil fuel types, although, surprisingly, the share of electricity produced from coal seems to have had the smallest impact of the three fuels. This may be driven by the fact that Denmark, which was in 1990 heavily reliant on coal for its electricity generation, managed to introduce a carbon pricing mechanism in 1992 (albeit with substantial sectoral exemptions and rebates). The more general observation that electricity generation from fossil fuels weighted negatively on carbon pricing policy developments may reflect the fact that until 2004, only northern European jurisdictions, which have achieved a lower share of electricity generated from fossil fuel over time, had introduced a carbon pricing scheme. In addition to this, the coefficient of *EU* is an order of magnitude smaller than in the full sample models. This is likely explained by the fact that carbon pricing activity took place in jurisdictions that were not EU members at the start of the scheme. Over the period 2005-2015, the results for these models indicate that EU membership was a much more significant determinant of the introduction of carbon pricing mechanisms. This echoes the fact that a lot of these jurisdictions' first carbon pricing mechanism was the EU-ETS. GDP per capita, institutional capacity and the level of democracy are both found to positively affect implementation over the period whereas the share of industry, CO2 emissions per capita and the orientation of the executive with respect to economic policy have a negative impact on implementation. The effects identified exhibit, however, weak statistical significance.

We also checked whether the results were robust to the use of different indicators of democratic institutions, namely *Polyarchy* and *Libdem*, as well as with another indicator of governance, *Corruption*. Estimations of models (I) to (IV) with these variables provided very similar results, which is somewhat unsurprising given that these are highly positively correlated with our main indicator of democratic institutions ($\rho = 0.9$ and $\rho = 0.85$, respectively). Estimations with the (control of) corruption variable suggested that implementation of carbon pricing policies was more likely to occur in less corrupt jurisdictions. Finally, we repeated the estimation of regressions (VII) to (X) with a different version of our outcome variable in equation (2), one where the weights are varying year-to-year. The results are, for all models considered, qualitatively similar.

7 Conclusions

Carbon pricing policies have re-emerged in the policy making arena as potential tools to achieve (some) reductions in GHG emissions. This renewed political appetite for carbon pricing mechanisms is apparent in the number of new schemes brought online over the last decade. However welcome these developments are, they should not be understood to mean that carbon pricing policies are on track to expand quickly to new jurisdictions nor to reach the stringency that achieving the Paris Agreement target requires. Indeed, this article shows that, when weighted for the share of covered emissions, these policies are, in most jurisdictions, much weaker than typically assumed. Moreover, because the jurisdictions with carbon pricing policies represent a small share of world emissions, the world average price of emissions remains extremely low, at about 1USD/tCO2e in 2015. In light of the statistical results discussed in this study, this is an uncomfortable state of affairs. First, because structural political and economic forces continue to hinder the introduction of new schemes beyond jurisdictions for which the political and economic cost of pricing carbon is comparatively low. For example, carbon pricing is unlikely to appeal to jurisdictions with low GDP per capita and/or, oil-fuelled electricity generation, for which other domestic policies, possibly complemented by international technology and financial transfers would prove more palatable. Second, because all implemented schemes exhibit strong persistency in their stringency, which is particularly problematic given that most of the schemes introduced so far are associated with weak (average) price signals.

The present analysis does, however, also offer lessons – and cautious optimism – for future policy developments. First, even if such developments are hindered by political economy factors, their effect is not as strong as one might have initially expected and suitable policy designs have been found to overcome them. Nonetheless, the difficulty with which carbon pricing schemes can be introduced, together with the weakness of most existing ones continues to provide a rationale for the development of climate mitigation strategies with multiple GHGs abatement tools and stresses the need to carefully consider the private and public cost of (early) retirement of the existing capital stock. This also highlights the importance of the sequence of introduction of the climate change mitigation policy package. One way to weaken incumbents' lobbying power is to weaken their relative influence *prior* to the introduction of carbon pricing policies.

Second, even in the presence of other climate policies, this analysis suggests that there is room for further strengthening of carbon pricing mechanisms. This can be done through extension of coverage or a price increase in currently covered sectors. But when considering implementation or strengthening of carbon pricing schemes, jurisdictions ought to pay close attention to the factors discussed here as it may save them from spending time and political capital on policy proposals bound to be met by fierce opposition.

References

Aldy, J. and Pizer, W. (2012). The competitiveness impacts of climate change mitigation policies. *Duke Environmental Economics Working Paper Series*, (EE 12-01).

Bennear, L. and Stavins, R. (2007). Second-best theory and the use of multiple instruments. *Environmental and Resource Economics*, 37(1):111–129.

Bowen, A. (2015). Carbon pricing: how best to use the revenue? *Policy brief – Grantham Research Institute on Climate Change and the Environment*.

Brunel, C. and Levinson, A. (2013). Measuring environmental regulatory stringency. Technical Report 05, OECD.

Cadoret, I. and Padovano, F. (2016). The political drivers of renewable energies policies. *Energy Economics*, 56:261–269.

Chang, C. P. and Berdiev, A. (2011). The political economy of energy regulation in OECD countries. *Energy Economics*, 33:816–825.

Congleton, R. (1992). Political institutions and pollution control. *The Review of Economics and Statistics*, 74(3):412–421.

Del Rio, P. and Labandeira, X. (2009). Barriers to the introduction of market-based instruments in climate policies: an integrated framework. *Environmental economics and policy studies*, 10:41–60.

European Commission (2012). *COMMISSION REGULATION (EU) No 601/2012 of 21 June 2012 on the monitoring and reporting of greenhouse gas emissions pursuant to Directive 2003/87/EC of the European Parliament and of the Council*.

Fankhauser, S., Gennaioli, C., and Collins, M. (2015). The political economy of passing climate change legislation: Evidence from a survey. *Global Environmental Change*, 35:52–61.

Fredriksson, P. and Vollebergh, H. (2009). Corruption, federalism, and policy formation in the oecd: the case of energy policy. *Public Choice*, 140:205–221.

Fredriksson, P., Vollebergh, H., and Dijkgraaf, E. (2004). Corruption and energy efficiency in oecd countries: theory and evidence. *Journal of Environmental Economics and Management*, 47:207–231.

Gassebner, M., Lamla, M., and Sturm, J.-E. (2011). Determinants of pollution: what do we really know? *Oxford Economic Papers*, 63(3):568–595.

Gawel, E., Strunz, S., and Lehmann, P. (2014). A public choice view on the climate and energy policy mix in the EU - how do the emissions trading scheme and support for renewable energies interact? *Energy Policy*, 64:175–182.

Grainger, C. and Kolstad, C. (2010). Who pays a price on carbon. *Environmental and Resource Economics*, 46:359–376.

Gronau, R. (1974). Wage comparisons - a selectivity bias. *Journal of Politcal Economy*, 82:1119–1143.

Hahn, R. (1990). The political economy of environmental regulation: Towards a unifying framework. *Public Choice*, 65:21–47.

Heckman, J. (1976). The common structure of statistical models of truncation, sample selection and limited dependent variables and a simple estimator for such models. *Annals of Economic and Social Measurement*, 5:475–492.

Hoel, M. and Karp, L. (2002). Taxes versus quotas for a tax pollutant. *Resource and Energy Economics*, 24:367–384.

IPCC (2006). *2006 IPCC Guidelines for National Greenhouse Gas Inventories*. IGES, Japan.

IPCC (2014). Summary for policymakers. In Field, C., Barros, V., Dokken, D., Mach, K., Mastrandrea, M., Bilir, T., Chatterjee, M., Ebi, K., Estrada, Y., Genova, R., Girma, B., Kissel, E., Levy, A., MacCracken, S., Mastrandrea, P., and White, L., editors, *Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*, pages 1–32, Cambridge, United Kingdom and New York, NY, USA. Cambridge University Press.

Jenkins, J. (2014). Political economy constraints on carbon pricing policies: What are the implications for economic efficiency, environmental efficacy, and climate policy design. *Energy Policy*, 69:467–477.

Joskow, P. and Schmalensee, R. (1998). The political economy of market-based environmental policy: The u.s. acid rain program. *Journal of Law and Economics*, 41(1):37–83.

Markussen, P. and Svendsen, G. (2005). Industry lobbying and the political economy of ghg trade in the european union. *Energy Policy*, 33:245–255.

Marques, A. and Fuinhas, J. (2011). Drivers promoting renewable energy: a dynamic panel approach. *Renewable and Sustainable Energy Reviews*, 15:1601–1608.

Marques, A., Fuinhas, J., and Pires Manso, J. (2010). Motivations driving renewable energy in european countries: A panel data approach. *Energy Policy*, 38:6877–6885.

OECD (2015). Effective carbon rates on energy. Technical report, OECD.

Olson, M. (1965). *The Logic of Collective Action*. Harvard Univeristy Press, Cambridge, Massachussetts.

Pearce, D. (2005). The political economy of an energy tax: The united kingdom's climate change levy. *Energy Economics*, 28:149–158.

Pizer, W. (2002). Combining price and quantity controls to mitigate global climate change. *Journal of Public Economics*, 85(3):409–434.

Pollitt, M. (2012). The role of policy in energy transitions: Lessons from the energy liberalisation era. *Energy Policy*, 50:128–137.

Roberts, M. and Spence, M. (1976). Effluent charges and licenses under uncertainty. *Journal of Public Economics*, 5:193–208.

Robinson, A. and Stavins, R. (2015). Linkage of greenhouse gas emissions trading systems: learning from experience. *Climate Policy*, pages 1–17.

Rode, A. (2013). Rent seeking over tradable emissions pemits: theory and evidence. *University of California Santa Barbara Working Paper*.

Rubin, J. (1996). A model of intertemporal emission trading, banking and borrowing. *Journal of Environmental Economics and Management*, 31:269–286.

Semykina, A. and Wooldridge, J. (2010). Estimating panel data models in the presence of endogeneity and selection. *Journal of Econometrics*, 157:375–380.

Shor, B. and McCarty, N. (2011). The ideological mapping of american legislatures. *American Political Science Review*, 105(3):530–551.

Sijm, J., Hers, S., Lise, W., and Wetzelaer, B. (2008). The impact of the eu ets on electricity prices.

Steves, F., Treisman, D., and Teytelboym, A. (2011). *The Low Carbon Transition*, chapter four, Political economy of climate change policy in the transition region. European Bank for Reconstruction and Development, London.

Stigler, G. (1971). The theory of economic regulation. *The Bell Journal of Economics and Management*, 2(1).

Tirole, J. (2012). Some political economy of global warming. *Economics of Energy and Environmental Policy*, 1(1):121–132.

United Nations/Framework Convention on Climate Change (2015). Adoption of the paris agreement. 21st Conference of the Parties, Paris.

United Nations/Framework Convention on Climate Change (2016). Aggregate effect of the intended nationally determined contributions: an update. 22nd Conference of the Parties - Item X of the provisional agenda.

van Beers, C. and Strand, J. (2015). *Political determinants of Fossil fuel pricing*, chapter 5. The MIT Press, Cambridge, Massachussetts and London, England.

Weitzman, M. (1974). Price vs. quantities. *The Review of Economic Studies*, 41(4):477–491.

Weitzman, M. (1978). Optimal rewards for economic regulation. *American Economic Review*, 68(4):683–691.

Wier, M., Birr-Pedersen, K., Jacobsen, H., and Klok, J. (2005). Are CO_2 taxes regressive? Evidence from the Danish experience. *Ecological Economics*, 52:239–251.

World Bank (2014). *State and trends of carbon pricing.* Washington, DC.

World Bank (2018). *State and trends of carbon pricing.* Washington, DC.

Data sources

CAIT (2015). Climate data explorer.

Center for Systemic Peace, Polity IV project (2015).

IEA (2016a). *Energy prices and taxes*, volume 2015/4. OECD Publishing, Paris.

IEA (2016b). IPCC fuel combustion emissions (2006 guidelines).

OECD (2016). Database on instruments used for environmental policy.

Shor, B. and McCarty, N. (2011). The ideological mapping of american legislatures. *American Political Science Review*, 105(3):530–551.

Statistics Canada (2016a). Table 128-0014 Electricity generated from fossil fuels, annual. CANSIM (database).

Statistics Canada (2016b). Table 379-0028 Gross Domestic Product (GDP) at basic prices, by North American Industry Classification System (NAICS), provinces and territories, annual (percentage share). CANSIM (database).

Statistics Canada (2016c). Table 384 - 0038 Gross Domestic Product, expenditure-based, provincial and territorial, annual. CANSIM (database).

Statistics Canada (2018). National inventory report.

US BEA (2016). Regional economic accounts.

U.S. Census Bureau (2016). State Export and Import Data, by North American Industrial Classification (USA Trade Online).

U.S. EIA (2015). State-level generation and fuel-consumption data.

Varieties of Democracy (2018). Version 8.

World Bank, WDI (2016).

World Bank, WGI (2016).

World Resources Institute (2015). *GHG Protocol tool for stationary combustion. Version 4.1.*

A Carbon prices - data sources and details

For each jurisdiction and each year we collect carbon price data in nominal local currency. Most jurisdictions quote the price of greenhouse gases (including CO2) per tonne of CO2e; others (essentially those with carbon taxes) express the carbon price per natural unit of the fuel. In the latter case, we convert the price to express it per tCO2e using conversion factors from the World Resource Institute (World Resources Institute, 2015). All values are then converted into 2015 $US using the Official Exchange Rate (Local Currency Unit/$US) and inflation rate from the World Bank, WDI (2016).

A.1 Emissions Trading Schemes

Table A.1: ETSs prices – details

Jurisdiction	*Price information*
EU-ETS	European Union emissions Allowances (EUA) futures price. Annual average of daily prices. Source: Bloomberg
Korea, Rep.	The market for Korean Allowance Units (KAUs) has been characterised by high illiquidity due to the absence of sellers amid concerns that the market is under-allocated. The last trade took place on March 15, 2016 at a price of $15.53. Source: South Korea Exchange
New Zealand	Annual average of daily spot prices of New Zealand Allowances (NZU). Source: Bloomberg.
Switzerland	As of 2015, no transaction of Swiss emissions allowances (CHU) had taken place over a centralised platform. Consequently, the price quoted in this study is the volume-weighted average price at auction. Source: Swiss Emissions Registry
California(-Quebec)	Annual average of daily California Carbon Allowances (CCA) futures contract price. Source: California Carbon Dashboard
RGGI	Volume-weighted annual average of spot transactions. Source: RGGI CO2 Allowance Tracking System (COATS).

A.2 CO_2 taxes

Information on sectoral fuel tax rates has been retrieved from a wide range of sources. A full list of sources is available upon request. These sources include (but are not limited to): OECD Database on Instruments used for Environmental Policy (OECD, 2016), International Energy Agency Energy Price and Taxes publication (IEA, 2016a), jurisdictions' budget proposals (as in the case of, e.g., Norway or Denmark), customs' agencies documentation, academic journal articles, policy assessment reports.

A.3 Total CO_2 price (oil, natural gas)

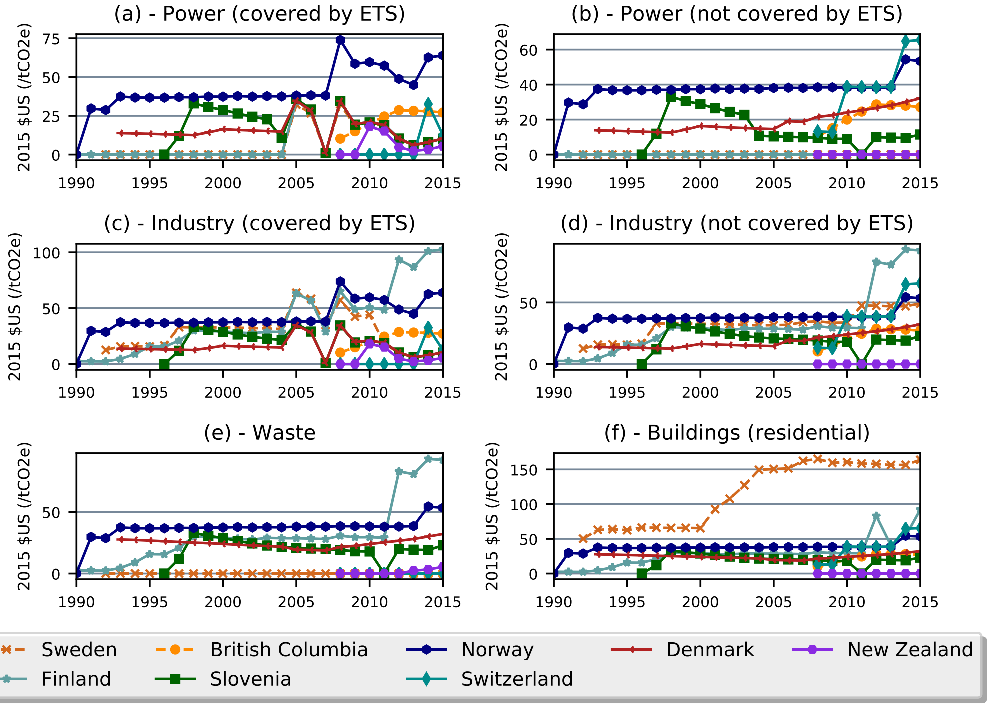

Figure A.1: Total carbon price over time – oil

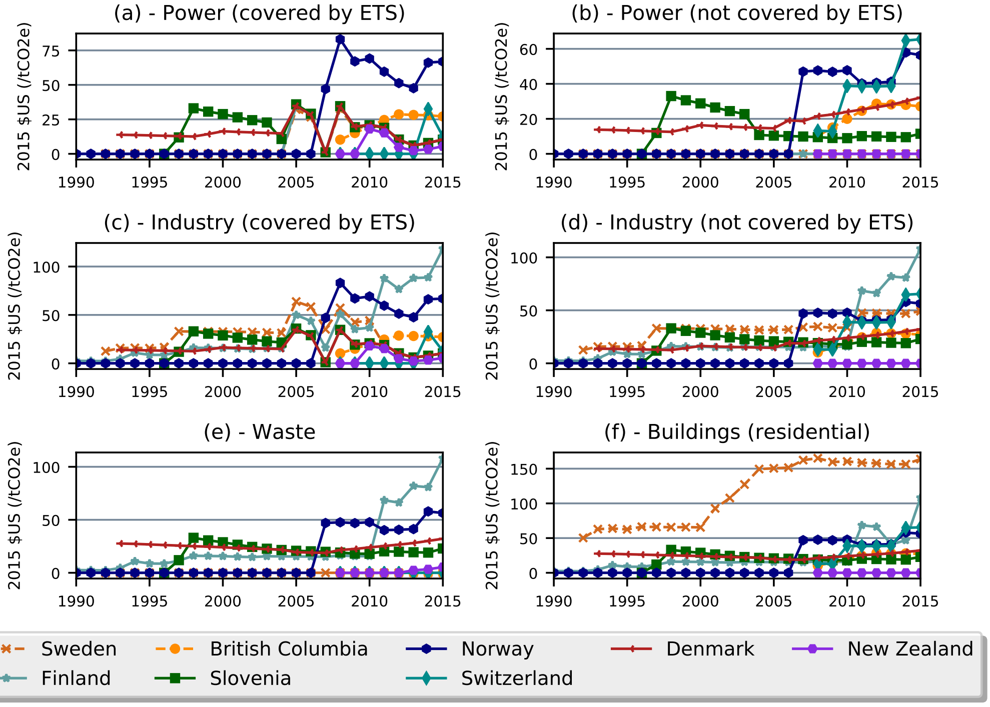

Figure A.2: Total carbon price over time – natural gas

B Scheme's coverage

This methodological appendix further details the steps involved in the computation of the coverage figures. Computing coverage figures requires defining a sectoral disaggregation of the economy. For the sake of consistency with IEA (2016b) and CAIT (2015) data, we adopt the sectoral disaggregation recommended by the IPCC (2006) Guidelines for National Greenhouse Gases Inventories, which is itself based on the United Nations International Standards Industrial Classification (ISIC), Revision 4. Table B.1 summarises the sectoral disaggregation.

Table B.1: IPCC 2006 Sectoral disaggregation

IPCC sector name	**IPCC sector label**
Electricity Generation*	1.A.1.a.i
Combined heat and Power Generation*	1.A.1.a.ii
Manufacturing industries and construction*	1.A.2
Domestic Aviation	1.A.3.a.i
Road Transportation	1.A.3.b
Commercial and public services	1.A.4.a
Residential	1.A.4.b
Agriculture/forestry	1.A.4.c
Industrial Processes – cement	2.A.1
Waste	5

*In some countries and in some years, these sectors are covered by a tax and an emissions trading system. Sometimes, however, the tax schemes are designed to exempt those installations that are covered by the relevant ETS. Since CO2 emissions data is disaggregated at the sector-fuel level and does not, within it, distinguish between those covered by the ETS and those that are not, it is not possible to account for this unless one makes an assumption about the proportion of emissions represented by the installations covered by the ETS.

The scope of an emissions trading scheme is defined at the sectoral level regardless of the fuel from which CO_2 – and other GHG – emissions originate. Therefore, an emissions trading scheme requires the measurement of GHG emissions at the point of emission. The design of carbon (or any other GHG)-taxes is different in that they can applied to specific fuel(s) within particular sectors. The sectors subject to it are determined independently. The relevant physical unit to be measured in the case of a carbon tax is therefore the fuel consumption (and associated CO2 emissions) at the user-fuel level. The fuel categories used in this study are: Coal/peat, Oil, Natural Gas.

The coverage information is recorded, for each jurisdiction and year, at the sector-fuel level as a binary variable (0 if the sector-fuel is not covered, 1 if it is). This coding is based on various sources, which vary from one country to the other. As for the carbon prices, a complete list of sources used to create the data points is available upon request. Table B.2 summarises the information recorded.

Table B.2: Institutional design

	Carbon Tax	Emissions Trading System
Price signal	Tax rate (nominal - local currency)	(Spot/Futures) Allowance price (nominal - local currency)
Sectoral coverage	✓	✓
Fuel coverage	✓	n.a.
GHG-gas coverage	*	✓
Sector-fuel exemptions	✓	n.a.

*The only GHG covered by carbon taxes is obviously CO_2.
Note: For each jurisdiction and year, except price, all information is coded as a binary entry.

Calculating total coverage (as a share of total GHG emissions) of carbon pricing schemes at the level of a jurisdiction is then performed according to the following formula

$$Coverage_{i,t} = \frac{\sum_j \sum_k q_{i,t,j,k} \times \mathbb{1}_{i,t,j,k}}{q_{i,t}^{GHG}} \tag{B1}$$

where $q_{i,t,j,k}$ represents jurisdiction i's CO2 emissions from sector j arising from the combustion of fuel k in year t; $\mathbb{1}_{i,t,j,k}$ is an indicator variable taking value 1 if fuel k in sector j of country i in year t is covered by the scheme, 0 otherwise; $q_{i,t}^{GHG}$ is the total greenhouse gases emissions in jurisdiction i in year t. Note that in the case of ETSs, the aggregation starts at the sector level, since all fuels are, by definition, covered.

The calculations make use of sector and sector-fuel CO2 emissions data. National jurisdictions: IEA (2016b); US States: CAIT (2015); Canadian Provinces and Territories: Statistics Canada (2018). Total GHG emissions (excluding land use change) are taken from the CAIT (2015) of the World Resources Institute.

C Emissions-weighted Carbon Price

Equipped with this information, the emissions-weighted price (ECP) can be computed at the sectoral or economy-wide level. In the former case, the weights are the emissions as a share of a sector's total GHG emissions; in the latter, the weights are the emissions as a share of the jurisdiction's total GHG emissions. Formally, the ECP of sector j of country i in year t is expressed as

$$ECP_{i,t,j} = \frac{\sum_k [\tau_{i,t,j,k} \times (q^{tax}_{i,t,j,k} + q^{ets,tax}_{i,t,j,k}) + p_{i,t,j,k} \times (q^{ets}_{i,t,j,k} + q^{ets,tax}_{i,t,j,k})]}{q^{GHG}_{i,t,j}} \tag{C1}$$

where $\tau_{i,t,j,k}$ is the carbon tax rate applicable to fuel k in sector j of country i at time t, $q^{tax}_{i,t,j,k}$ is the amount of CO2 emissions covered by a tax only, $p_{i,t,j}$ is the price of an emission permit, $q^{ets}_{i,t,j,k}$ is the amount of CO2 emissions covered by an ETS, $q^{ets,tax}_{i,t,j,k}$ is the amount of CO2 emissions covered by both an ETS and a tax and $q^{GHG}_{i,t,j}$ is the quantity of GHG emitted by sector j of country i in year t.

An economy-wide ECP is then computed as a weighted average of the carbon rates across sectors, where the weights are the quantity of emissions subject to each individual carbon rate:

$$ECP_{i,t} = \sum_j (ECP_{i,t,j} \times \gamma_{i,t,j}) \tag{C2}$$

where $\gamma_{i,t}$ represents the GHG emissions of sector i as a share of the economy's (jurisdiction's) total GHG emissions, i.e. $\frac{q^{GHG}_{i,t,j}}{q^{GHG}_{i,t}}$. For the purpose of the present study, only the economy-wide ECP is computed and both a time-varying and fixed weights version of the ECP are calculated. The fixed-weights ECP uses 2013 emissions data.

D Jurisdictions with carbon pricing

Table D.1: Jurisdictions with implemented carbon pricing schemes as of 2018

Jurisdiction	Emissions Trading	Carbon tax	ECP - 2015 (2015 $US)	ECP-2018 (2015 $US)
Austria	2005	-	4.04	7.9
Belgium	2005	-	3.15	6.18
Bulgaria	2007	-	5.64	11.12
Cyprus	2005	-	4.7	9.8
Czech Republic	2005	-	6.06	11.89
Denmark	2005	1992	16.07	8.09
Estonia	2005	2000	8.48	13.34
Finland	2005	1990	35.53	43.95
France	2005	2014	6.49	17.32
Germany	2005	-	5.07	9.96
Greece	2005	-	5.51	11.04
Hungary	2005	-	3.28	6.45
Iceland	2008	2010	6.68	19
Ireland	2005	2010	10.29	12.81
Italy	2005	-	3.89	7.66
Japan	-	2012	1.24	1.87
Kazakhstan	2013	-	0.01	†
Korea, Rep.	2015		6.78	14.11
Latvia	2005	1995	2.34	4.61
Liechtenstein	2008	-		
Lithuania	2005	-	2.73	5.42
Luxembourg	2005	-	1.44	2.81
Malta	2005	-	5.73	11.15
Mexico	-	2014	1.42	1.42
Netherlands	2005	-	4.61	9.02
New Zealand	2008	-	1.91	6.32
Norway	2007	1991	40.85	39.39
Poland	2005	1990	5.4	10.73
Portugal	2005	2015	7.36	12.59
Romania	2007	-	4.15	8.23
Slovak Republic	2005	-	4.86	9.61
Slovenia	2005	1996	12.87	16.93
Spain	2005	-	4.21	8.32
Sweden	2005	1991	87.86	91.3
Switzerland	2008	2008	17.3	
United Kingdom	2005	2013	12.16	16.32
Alberta*	2007	-		
Beijing	2013	-	n.a.	
British Columbia	-	2008	16.85	19.66
California	2009	-	9.66	11.54
Chongqing	2014	-	n.a.	
Connecticut	2009	-	1.09	0.75
Delaware	2009	-	1.74	1.20
Guangdong	2013	-	n.a.	
Hubei	2013	-	n.a.	
Kyoto	2011	-	†	
Maine	2009	-	0.5	0.35
Maryland	2009	-	1.57	1.08
Massachusetts	2009	-	1.09	0.75
New Hampshire	2009	-	1.4	0.97
New York	2009	-	0.99	0.68
Quebec	2013	-	6.7	7.93
Rhode Island	2009	-	1.62	1.12
Saitama	2011	-	†	
Shanghai	2013	-	n.a.	
Shenzhen	2013	-	n.a.	
Tianjin	2013	-	n.a.	
Tokyo	2010	-	†	
Vermont	2009	-	0.02	0.01

†: missing information at the time of writing – Chile: 2017; South Africa: 2016; New Jersey's scheme was discontinued in 2011, Australia's in 2012.