Carbon Pricing and Power Sector Decarbonisation: Evidence from the UK

Marion Leroutier

PARIS SCHOOL OF ECONOMICS
ECOLE D'ECONOMIE DE PARIS

WORKING PAPER N° 2021 – 41

Carbon Pricing and Power Sector Decarbonisation: Evidence from the UK

Marion Leroutier

JEL Codes: D22, H23, Q41, Q48
Keywords: carbon tax, electricity generation, synthetic control method.

Funded by a French government subsidy managed by the ANR under the framework of the Investissements d'avenir programme reference ANR-17-EURE-001

Carbon Pricing and Power Sector Decarbonisation: Evidence from the UK

Marion Leroutier*

June 21, 2021

Abstract

Decreasing greenhouse gas emissions from electricity generation is crucial to tackle climate change. Yet, empirically little is known on the effectiveness of economic instruments in the power sector. This paper examines the impact of the UK Carbon Price Support (CPS), a carbon tax implemented in the UK power sector in 2013. Compared to a synthetic control unit built from other European countries, emissions from the UK power sector declined by 26 percent on an average year between 2013 and 2017. Bounds on the effects of potential UK confounding policies and several placebo tests suggest that the carbon tax caused at least 80% of this decrease. Three mechanisms are highlighted: a decrease in emissions at the intensive margin; the closure of some high-emission plants at the extensive margin; and a higher probability of closure than in the synthetic UK for plants at risk of closure due to European air quality regulations. This paper shows that a carbon tax on electricity generation can lead to successful decarbonisation.

Keywords: carbon tax, electricity generation, synthetic control method

JEL Codes: D22, H23, Q41, Q48

*Paris School of Economics, Universite Paris I, CIRED. Contact details: Email: marion.leroutier@psemail.eu - Postal: 48 Boulevard Jourdan, 75014, Paris, France. I am grateful to Philippe Quirion, Katheline Schubert, Nicolas Koch, Ulrich Wagner, Mirjam Kosch, Jan Abrell, Francois Libois, and three anonymous referees for their comments and suggestions. I thank seminar participants at PSE, LSE GRI, MCC, PIK and SSE, participants to the 2018 OECD environmental micro-data workshop, conference participants at Mannheim Energy conference, EAERE, FAERE, and participants to the Marseille Green Econ Spring School and CIRED summer school for useful feedback. I thank Ember (formerly Sandbag) for sharing their ETS data with me, Lorenzo Montrone for giving me access to the Global Coal Plant Tracker database, and Jan Abrell and Mirjam Kosch for sharing data on monthly power production.

1 Introduction

All governments in the world need to reduce their greenhouse gas emissions in order to tackle climate change. In the past two decades, they have implemented a variety of abatement policies to address this challenge, including economic instruments in the form of carbon taxes and markets (World Bank and Ecofys, 2018). Although carbon pricing is widely regarded by economists as the most cost-effective way to reduce emissions, ex-post evaluations of carbon pricing policies implemented in different sectors are still scarce (Green, 2021). This general observation is particularly true in the case of the power sector (Martin et al., 2016), which represent 25% of worldwide emissions in 2010 (IPCC, 2014).

In this paper, I estimate the impact of a carbon tax introduced in the UK power sector in 2013, the Carbon Price Support (CPS), on carbon emissions. At that time and during the period of analysis considered in the paper, the UK was part of the European carbon market (EU ETS) implemented in 2005. The carbon tax was introduced in response to the low prices prevailing on the European carbon market, while the UK was facing binding emission reduction targets under the 2008 Climate Change Act. The tax rate increased from around £5 (€5.9) per ton of equivalent carbon dioxide (hereafter tCO_2e) in 2013 to £18 (€26) in 2017. At the same period, the UK power sector experienced a remarkable transition: between 2012 and 2017, the share of coal in electricity generation decreased from 40% to 7%, gross electricity consumption decreased by 6%, and power sector greenhouse gas emissions decreased by 57% (Source: Eurostat). The rapid transformation of the UK power sector received significant coverage in the media and in policy reports (Evans, 2019; Brown, 2017), but how much the UK carbon tax contributed to such transformation is to date unclear.

To estimate this contribution, I apply the synthetic control method (Abadie and Gardeazabal, 2003; Abadie et al., 2010, 2015) to build a counterfactual UK with a weighted combination of European countries having power sectors with characteristic similar to the UK. I use countries which, like the UK, were in the European Union at the period considered

(2005-2017) as potential candidates to enter the counterfactual UK, because all these countries were subject to the same European climate and energy policies as the UK before the introduction of the CPS, in particular the EU ETS and European air quality regulations.

I estimate that the introduction of the CPS is associated with emissions reductions - or abatement - of between 141 and 191 million tons of equivalent carbon dioxide (hereafter $MtCO_2e$) over the 2013-2017 period, implying emission reductions of between 20.5% and 26% on an average year. This range depends on the assumed effect for three UK-specific policies implemented around the same period - a subsidy to encourage the biomass conversion of coal plants, a new scheme for renewable support, and a capacity market - and on the magnitude of CPS-induced spillovers. The upper bound assumes that biomass conversion is a consequence of the CPS, that the other two policies have a negligible impact over the 2013-2017 period, and that emission leakage from the UK to other European countries is negligible. The lower bound is more conservative: it puts a bound on the effect of the three UK policies and calculates the amount of emissions from the synthetic UK which may be due to CPS-induced spillovers. Back-of-the-envelope calculations (Based on the lower bound) suggest that about a third of the impact was driven by UK plants facing a high carbon price reacting differently to European air quality regulation. Another third was caused by the closure of a few high-emitting plants and the last third by a decrease in emissions from plants staying in the market (likely due to fuel switching from coal to gas). A set of placebo tests suggest that the impact estimated is causal. The results are robust to several sensitivity analyses.

This paper contributes to several strands of the literature: first, it contributes to the growing empirical literature evaluating the impact of regional and national carbon pricing instruments (Martin et al., 2014; Rivers and Schaufele, 2015; Andersson, 2019; Colmer et al., 2020; Kim and Kim, 2016). Cropper et al. (2018) underline the challenge of finding a suitable control group for the retrospective analysis of environmental regulation, including carbon pricing. This is especially true for the power sector, as almost all power plants are subject to the policy examined, making it difficult to find a good counterfactual[1]. That both UK and non-UK power plants were subject to European-level energy policies but only UK plants were subject to the CPS offers an opportunity to compare the evolution of UK power sector emissions with that of an appropriately weighted average of European countries.

To my knowledge, two other recent papers examine the effectiveness of the CPS: Abrell et al. (2019) estimate counterfactual electricity generation for each power plant subject to the CPS in the absence of the CPS using machine learning. They find that the CPS induced a total abatement of 26 $MtCO_2e$ over the 2013-2016 period due to the short-term fuel switch from coal-fired to gas-fired plants[2]. Gugler et al. (2020) rely on a Regression-Discontinuity-in-Time (RDiT) approach and exploits the annual change in the tax rate of the CPS between 2013 and 2015. They estimate a cumulative abatement of 38.6 $MtCO_2$ over the 2013-2015 period compared to a no-policy scenario.

In contrast to these two papers, my paper uses less granular data but considers a longer post-treatment period and adopts a method allowing to take into account more mechanisms: a carbon tax on high-emitting input fuels may induce a decrease in emissions via fuel switching, but also via longer-term mechanisms such as plant closure and changes in demand or imports - although I find that demand and trade play a limited role role compared to changes in the emission intensity of domestic production. Using as a control group countries also part of the European Union (EU) at the time also enables me to control for the effect of the EU ETS and of an important environmental regulation affecting all EU power plants at the time, the Large Combustion Plant Directive, and highlight the interactions between the UK carbon price and these regulations. Finally, my work relies on open and freely accessible data, which facilitates replication.

[1]In the case of the ETS, the only installations exempted are those with a rated capacity of less than 20 Megawatt thermal input (MWth). In the UK those installations represent 0.2% of the installed capacity in 2015 (Source: Digest of United Kingdom Energy Statistics)

[2]Fuel switching arises because carbon pricing increases the relative marginal cost of coal-fired plants compared to gas-fired plants due to the higher carbon intensity of the former. This change in costs modifies the short-term electricity supply curve, defined by the ranking of power plants by ascending marginal cost (the so-called "'merit order"). As a result, the hourly output from high-emitting coal-fired plants' increases while the hourly output from lower-emitting gas-fired plants decreases.

Second, this paper contributes to the scarce literature examining the rapid decarbonisation of the UK power sector. Staffell (2017) links this decrease in emissions to the evolution of electricity demand, capacity, prices, the fuel mix, imports and exports in a descriptive approach. Wilson and Staffell (2018) insist on the role played by coal to gas fuel switching and underline the likely role of the CPS but do not quantify it. In contrast, this paper builds a comparison group and carefully examines potential confounding factors in order to recover a plausibly causal impact of the CPS on emission reduction.

Third, this paper is linked to a recent literature applying the synthetic control method to estimate the impact of environmental policies. It is close to the paper by Andersson (2019) examining the impact of the Swedish carbon tax on transport sector emissions, but examines power sector emissions, where the carbon tax is levied on producers. Kim and Kim (2016) similarly examine the impact of carbon pricing in the power sector using the SCM approach, by they do so in the context of the US regional carbon market RGGI and observe fuel switching rather than emission. Other recent work include Lee and Melstrom (2018), who estimate the impact of RGGI on electricity imports, and Isaksen (2020), who evaluates the effectiveness of international pollution protocols. A distinctive feature of my paper is to build my outcome variable at the country level starting from disaggregated plant-level emission data. This enables me to take into account shocks experienced by individual plants, and to document the channels through which the UK carbon tax may operate, an under-investigated area of research in the SCM literature according to Abadie (2021).

Beyond its academic contribution, this paper is relevant from a policy perspective. To be in line with the 2015 Paris Climate Agreement and net-zero commitments, OECD countries need to be coal-power-free by 2030 (Rocha et al., 2016). The means necessary to achieve such transition are still under discussion. Some European countries are considering adopting a carbon price floor to hedge against variations in the ETS price (Newbery et al., 2019). Lessons can probably be drawn from the UK experience analysed here.

The paper is organized as follows: Section 2 presents the background, potential effects of a carbon tax in the power sector, and descriptive evidence; Section 3 describes the empirical strategy; Section 4 presents the main results; Section 4.4 discusses them; Section 6 concludes.

2 The UK carbon tax: context and expected effects

2.1 The UK Carbon Price Support

The Carbon Price Support was introduced in April 2013. The carbon tax was proposed in a double context of low prices on the EU carbon market, and the obligation for the UK to meet national targets for greenhouse gas emissions as defined in the 2008 Climate Change Act. The Climate Change Act set an emission target for 2050 and implemented a system of 5-year carbon budgets. Under the second carbon budget running from 2013 to 2017, the UK had to reduce its total emissions by 236 $MtCO_2e$ compared to the first carbon budget (running over 2008-2012). Low prices on the EU carbon market were perceived as limiting the potential for high emissions reductions among industrial installations covered by the EU ETS. In this context, the UK Government announced in March 2011 that a Carbon Price Floor (CPF) would be implemented in the power sector in the 2013/2014 budget year[3]. Under this price floor, power installations located in Great Britain (GB)[4] would have to pay a tax called the Carbon Price Support (CPS), which yearly rate would reflect the difference between the desired level of carbon price floor and the expected carbon price on the EU ETS. The announced goal of the CPF was to tackle price uncertainty on the EU ETS and encourage investment in low-carbon technologies in the generation sector; in official communication documents, the CPF was labelled "support and certainty for low-carbon investment"(Hirst, 2018). The price floor was expected to increase over time, with a total carbon price target of £30 (around €35) by 2020.

[3]The budget year over which the annual tax rate is set runs from 1 April to 31 March of the next calendar year

[4]power generators located in Northern Ireland are integrated in a separate wholesale electricity market with the Republic of Ireland and are not subject to the policy.

The CPF was introduced as planned on 1 April 2013. It was part of a broader reform called the Electricity Market Reform, which includes three other components described in more details in the next section: a capacity market aiming at securing production capacity to back up intermittent renewable capacity; support to investments in renewable power capacity in the form of Contracts for Difference (CfDs)[5]; and Emission Performance Standards banning new coal-fired plants not fitted with Carbon Capture and Storage (CCS). The first rate of the CPS was set at around £5/tCO_2e. However, in 2014 the Government decided to freeze the CPS rate to £18/tCO_2e (€22 in 2016) until 2019/2020, after business representatives expressed concerns over the competitiveness of energy-intensive industries because of generators passing on the tax costs (Ares and Delebarre, 2016). Furthermore, actual ETS carbon prices turned out to be much lower than expected over the period. Because of the freeze and the difference between expected and actual carbon prices, the nature of the Carbon Price Support changed compared to what was initially envisioned: it is a carbon tax which rate is set several years in advance. Tax receipts go to the general budget.

The CPS applies to almost all power generators located in GB.[6]. The only exemptions are for stand-by generators used to provide emergency electricity supplies if a building's usual power supply is cut, and generators with a rated thermal input smaller than 2 MWth.

Table 1 shows the level of the tax rate confirmed for each period in 2016. Figure 1 overlays annual CPS rates with annual ETS carbon prices converted to British pound since 2009. The sum of the two gives the total carbon price paid by GB generators, which departs significantly from the level of carbon price floor initially envisioned. The CPS component nevertheless implies that GB power generators pay a much higher carbon price than non-GB

[5]CfDs guarantee a flat payment to low-carbon electricity generators: auctions determine the strike price, which reflects the long-term cost of generating low-carbon electricity for the awarded generators; then, when the electricity market price is below the pre-determined strike price, contracted generators are paid the difference between the strike price and market price; and when it is above the strike price, contracted generators must pay this difference - See https://www.emrsettlement.co.uk/about-emr/contracts-for-difference/ for more details.

[6]This includes conventional power plants, Combined Heat and Power (CHP) plants producing both electricity and heat (they only pay the CPS on the amount of fuel used to produce electricity for the grid), and auto-generators producing electricity for their own use (HM Revenue & Customs, 2017). Both CHP plants and auto-generators represent a negligible share of power production and emissions.

power generators (only subject to the ETS price). In 2016, the relative difference is the highest and is five-fold. The rate of the tax depends on the carbon content of the input fuel used for power generation. The CPS rate on coal is about 70% higher than the tax on natural gas, in line with the much higher emission factor of coal. The CPS rate thus substantially increases the relative cost of coal-fired generation compared to gas-fired generation.

Table 1: Level of CPS rate for each period in pound per ton of CO_2e

Period	CPS rate in £/tCO_2e
April 2013/March 2014	4.96
April 2014/March 2015	9.55
April 2015/March 2016	18.08
April 2016/March 2017	18
April 2017/March 2018	18
April 2018/March 2019	18

Source:Ares and Delebarre (2016)

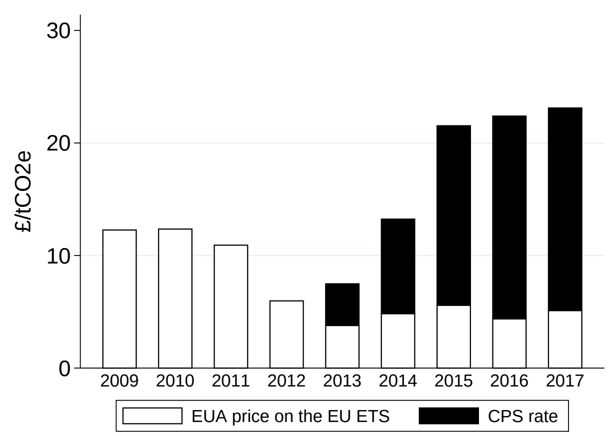


Figure 1: The Carbon Price Support and EUA price on the EU ETS

Notes: EUA stands for European Union Allowance and EUAs are carbon allowances traded on the ETS. Source for EUA price data: Ember website. Source for CPS prices: Hirst (2018). CPS prices adjusted with appropriate weights to reflect the calendar year rather than the April to March period. EUA price data converted to £ using yearly averages of monthly market exchange rates.

2.2 Descriptive evidence

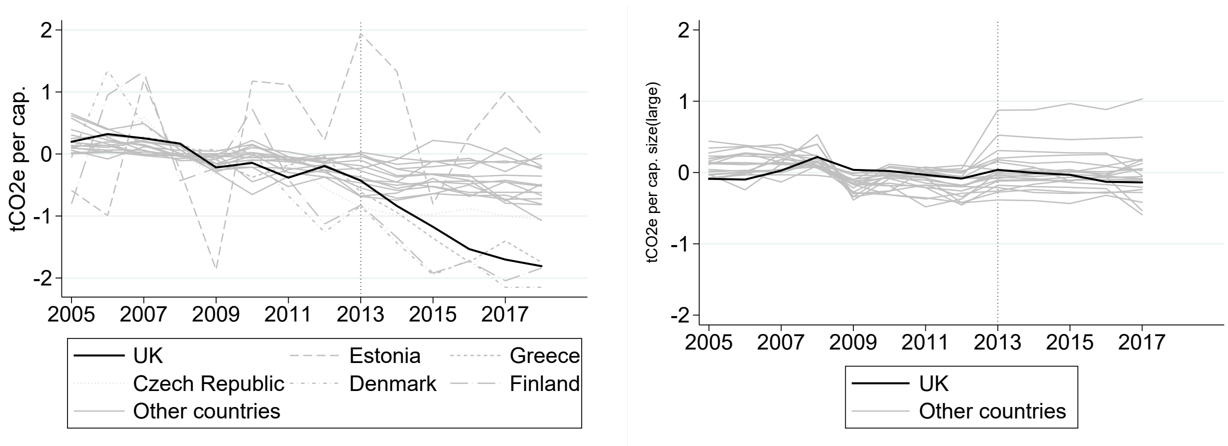
Power sector emissions are the main outcome of this analysis. I define a variable of per capita power sector emissions at the country level to facilitate country comparison[7], expressed in tons of carbon dioxide equivalent or tCO_2e. Figure 2a shows de-meaned per capita power sector emissions for each European country between 2005 and 2017, using emission data described in section 3.2. I demean emission values by taking the difference between annual per capita power sector emissions and per capita power sector emissions averaged over the 2005-2012 period, which is the pre-treatment period before the introduction of the CPS in 2013. Most countries tend to have stable emissions per capita, except for a few outliers, which emissions are shown in dashed or dotted lines[8]. After 2012, UK emissions are falling compared to most other countries. Three other countries have decreasing emissions: Finland, Denmark and Greece. Emissions from Finland and Denmark experience large variations across the whole period (see Figure A.1 in Appendix). The decrease in emissions in Greece cannot be traced back to a specific policy. It may be due to the large reforms implemented in all economic sectors around that period following the Greek debt crisis, combined with the deployment of a large amount of solar PV in 2011, 2012 and 2013 under an appealing Feed-in-tariff that was subsequently retroactively cut in 2014[9]. In contrast to the strong decrease in power sector emissions, UK per capita emissions in other ETS sectors follow the same path as other European countries, as shown on figure 2b.

Different channels may trigger such decrease in power sector emissions. The following decomposition helps to understand the channels - for ease of reading, there are no indices,

[7]The advantage of taking emissions per capita rather than per MWh of electricity output is twofold: first, population is more stable over time than gross electricity production, so that the time variation in the outcome is mostly due to variations in emissions; second, decomposing emission per capita allows to analyse what happens to electricity demand and trade, rather than simply focussing on the emission intensity of domestic production.

[8]Estonia's emissions are both high on average and with a high variance; Czech Republic has the highest average after Estonia; Greece has decreasing emissions after 2012; Finland and Denmark's emissions have a high variance, likely due to the inter-annual variation in available hydro resources in Finland, and hydro and wind resources in Denmark.

[9]https://www.pv-magazine.com/2014/03/11/greece-brings-new-retroactive-measures-cuts-fit-by-30_100014491/



(a) De-meaned per capita power emissions (b) De-meaned per capita non-power emissions

Figure 2: Evolution of per capita power and non-power sector emissions in European countries

Notes: For figure a (resp. figure b), per capita emission values were obtained by aggregating plant-level emission data for ETS participants identified as power installations (resp. non-power) at the country level, and dividing by annual country population. De-meaned per capita emissions were obtained by taking the difference between the annual value and the 2005-2012 average. “Other countries” include twenty European countries: Austria, Belgium, Czech Republic, Denmark, Estonia, Finland, France, Germany, Greece, Hungary, Ireland, Italy, Latvia, Lithuania, the Netherlands, Poland, Portugal, Slovakia, Spain, Sweden.

but all the variables should be interpreted as values for a given country c in a given year t. Calling P the country population and Q_{CO_2e} the quantity of emissions from the domestic power production[10], $\frac{Q_{CO_2e}}{P}$ are per capita power sector emissions, and are the product of per capita electricity generation $\frac{Q_{elec}}{P}$ and the emission intensity of domestic power generation, $\frac{Q_{CO_2e}}{Q_{elec}}$:

$$\frac{Q_{CO_2e}}{P} = \frac{Q_{elec}}{P}\frac{Q_{CO_2e}}{Q_{elec}} \tag{1}$$

Q_{elec} can be rewritten as the difference between domestic gross electricity consumption C_g and net electricity imports, defined as electricity imports M minus electricity exports X, $(M - X)$. Gross electricity consumption is itself the sum of net consumption C_n (equivalent to demand), the amount of network losses, and the amount of electricity used by power generators. Grouping these two last components in the variable L, this leads to the following

[10]Only fossil fuels used for electricity generation generate emissions, so Q_{CO_2e} is the sum of emissions from coal-, gas- and oil- fired power plants.

equation:

$$\frac{Q_{CO_2e}}{POP} = (\frac{C_n}{POP} + \frac{L}{POP} - \frac{(M-X)}{POP})\frac{Q_{CO_2e}}{Q_{elec}} \tag{2}$$

From the right-end side of the equation, four different channels may lead to a decrease in per capita emissions: a decrease in consumption per capita $\frac{C_n}{POP}$ (the *demand channel*), a decrease in the amount of network losses and self-consumption of electricity by power generators $\frac{L}{POP}$ (the *network efficiency* channel), an increase in net imports per capita $\frac{(M-X)}{POP}$ (the *trade channel*), and a decrease in the average emission intensity of the domestic power sector (the *emission intensity channel*).

In appendix A.1, I show the evolution of the demand, trade and emission intensity channels[11]. I find that UK electricity demand has been declining steadily since 2005 (Figure A.2a), and UK net electricity imports per capita tend to increase, but they remain very low compared to other countries (Figure A.2b). In contrast, the UK emission intensity of domestic power production follows a similar pattern to total per capita emissions with a strong decrease after 2013 not witnessed in other European countries. These graphs suggest that the decrease in power sector emissions in the UK after 2013 is mostly due to a marked decline in the emission intensity of domestic production.

2.3 Potential confounders

Isolating the contribution of the CPS to this change in emissions may be challenging given the other policies implemented at the same period at the UK and European level. The UK and other countries in the European Union were subject to the same policies enacted at the EU level, in particular the EU carbon market, air quality regulations, and the 2020 strategy setting targets for emission reductions and the deployment or renewable energy. It means that I can differentiate out the effect of these policies by using other European countries

[11] I leave aside the *network efficiency* channel, which is a technical component stable over time and unlikely to be influenced by carbon pricing. L can be estimated with Eurostat data as gross production Q_{CO_2e} plus net imports $(I-X)$ minus net consumption Y_n. For all European countries, L is constant over time in proportion of total gross production, at about 18%.

as a counterfactual. UK-specific policies enacted at the same time as the CPS cannot be differentiated out this way. I will make different assumptions on the effects of these policies to bound their effects at the period considered. There are four important policies to consider. These policies are described in more detail in appendix A.2, and below I summarize how each of them may have impacted UK emissions.

At the European level, the LCP directive (LCPD) is an air quality directive enacted in 2001, which entered into force in 2008. It imposes emission limit values for local air pollutants to all combustion plants with a rated capacity above 50MWth. Regulated plants had to respect the emission standards by 2008, or they could choose to opt out from the directive. Opt-out plants were exempted from the emission standards, but could not operate for more than 20,000 hours between 1 January 2008 and 31 December 2015 (European Commission, 2001), and had to shut down once they had run for 20,000 hours or in 2015 (whichever came first). Plants had to decide by 2004 whether they wanted to opt-out or not[12]. The UK had the highest share of opt-out capacity per capita in 2004, followed closely by Slovakia and Finland[13]. LCPD-induced plant closures could explain part of the decrease in emission seen in the UK compared to the average EU country. To avoid confounding the impact of the CPS and that of these two air quality directives, I will control for the emissions coming from LCP opt-out plants in my estimation strategy. The LCPD was replaced by the IED directive in 2016. The IED Directive had a similar opt-out option, upon which plants had to decide by 2013. Given that the CPS had already been announced at that time, I consider the decision to opt out from the IED directive endogenous to the CPS. The UK has two IED opt-out plants, which have limited operating hours between 1 January 2016 and 31 December 2023 and have to shut down in 2023 at the latest.

At the UK level, three specific policies part of the Electricity Market Reform may have contributed to the decrease in emissions after 2013. First, the UK government subsidised

[12]The decision to opt-out is made for each generating units, and some combustion plants only opted out some but not all of their generating units

[13]Own calculation based on EEA website: https://www.eea.europa.eu/data-and-maps/data/large-combustion-plants-lcp-opted-out-under-article-4-4-of-directive-2001-80-ec-4

the conversion of coal-fired power plants to biomass starting from 2012, and two plants representing 15% of UK emissions in 2012 benefited from a Contract for Difference for the biomass conversion of part (for Drax plant) or all of their units (for Lynemouth plant). Whether the subsidy for biomass conversion was decided to facilitate the conversion of coal-fired plants facing the CPS, or whether it was independent from the CPS is ambiguous. In section 4.2, I develop a strategy to exclude the emission reduction induced by biomass conversion from my estimation.

Second, the Contracts for Difference system introduced in 2014 and its 2012 predecessor, the FID Enabling for Renewables, could have impacted the fuel mix by increasing the share of renewable energy in UK electricity production over the 2013-2017 period (outside the specific case of biomass generated by former coal-fired plants). However, available data on the projects being awarded a CfD in 2014, 2015 or 2017 reveal that only few of them were operational over the 2013-2017 period. Given the date at which the projects became operational, the amount of clean electricity generated between 2013 and 2017 that can be imputed to the CfD projects represents only 0.4% of electricity generated with renewable sources (including waste and biofuels) and 0.1% of total electricity generated in the UK over the 2013-2017 period. If this electricity had been produced by coal-fired plants, the associated CO_2e emissions would have been 1.4 MtCO_2e (see Appendix A.2 for more details on the calculations).

Third, the capacity market introduces payments for electricity generators being awarded a capacity contract, in exchange for providing generation capacity at a pre-determined period of time. Since most of the capacity secured is for after 2018, this policy could only reduce UK emissions over the 2013-2017 period if two conditions are met: first, if the prospective capacity payment incentivised new capacity to be rolled out ahead of the capacity delivery year; second, if this new capacity has a lower emission intensity than existing plants. Using available public data on plants being awarded a capacity contract and new-build conventional plants in the UK between 2014 and 2017, I estimate that at most 2,590 GWh of electricity were generated over the 2013-2017 period from plants meeting the following conditions: 1)being awarded a capacity contract between 2014 and 2017 2)starting generation after 2014 and 3)not having been planned before 2014 based on available evidence[14]. The plants meeting these conditions are all fired with municipal solid waste and their generation over the 2013-2017 period represents only 0.6% of electricity generated with renewable sources and 0.2% of total electricity generated in the UK over the 2013-2017 period. If this electricity had been produced by coal-fired plants, the associated CO_2e emissions would have been 2.3 MtCO_2e (see Appendix A.2 for more details on the calculations). Overall, the Contract for Differences and the capacity market are unlikely to have triggered important changes in power sector emissions over the 2013-2017 period (except for the impact of CfDs on the biomass conversion of coal-fired plants, considered separately).

3 Empirical strategy

3.1 The synthetic control method

To estimate the impact of the Carbon Price Support from other factors, I use the synthetic control method (SCM) exposed in Abadie and Gardeazabal (2003) and Abadie et al. (2010, 2015). This method consists in building a counterfactual UK power sector by applying appropriate weights to the set of other European countries' power sectors. Providing that the obtained "synthetic" UK accurately reflects what the UK power sector would have looked like without the CPS, this method allows to estimate the causal impact of the CPS on per capita power sector emissions, and more generally on absolute abatement. The SCM method is particularly appropriate in the context of the CPS since the "treatment" applies to one country only, and within the country it affects almost all power installations, without time variation in treatment. Within the UK, there is then no obvious group of installations that

[14]The largest new gas-fired plant which opened in 2016 and won several capacity contracts, Carrington power station, started being constructed in 2009 https://en.wikipedia.org/wiki/Carrington_Power_Station

could serve as counterfactual for how treated power plants would have evolved absent the policy.

Using the notation from the Neyman-Rubin Causal Model (Rubin, 1974), the challenge is to estimate $\beta_{\mathrm{UK}t}$ when t≥2013, defined as:

$$\beta_{\mathrm{UK}t} = Y^1_{\mathrm{UK}t} - Y^0_{\mathrm{UK}t} = Y_{\mathrm{UK}t} - Y^0_{\mathrm{UK}t} \tag{3}$$

$Y^1_{\mathrm{UK}t}$ designates, at each period, UK per capita power sector emissions in the presence of the CPS policy. $Y^0_{\mathrm{UK}t}$ designates, at each period, UK per capita power sector emissions in the absence of the policy. $\beta_{\mathrm{UK}t}$ designates the difference between the two. $Y_{\mathrm{UK}t}$ designates the observed outcome. The challenge to estimate $\beta_{\mathrm{UK}t}$, or "fundamental problem of causal inference" (Rubin, 1974), comes from the fact that $Y^1_{\mathrm{UK}t}$ is observed when t≥2013 but $Y^0_{\mathrm{UK}t}$ is not.

Let us assume after Abadie et al. (2010) that the outcome in the absence of intervention Y^0_{ct} can be modelled as the following linear factor model, for each country c and period t:

$$Y^0_{ct} = \delta_t + Z_c\alpha_t + f'_t\lambda_c + \epsilon_{ct} \tag{4}$$

δ_t is a time fixed effect, Z_c is a vector of observed exogenous country characteristics, α_t is a vector of unknown parameters, f_t is a vector of unobserved common factors (and f'_t its transpose), λ_c is a vector of unobserved country-specific effects or factor loadings, and ϵ_{ct} is an error term with mean 0 (typically capturing transitory shocks at the country level).

Such a model is more flexible than the typical difference-in-difference equation because time effects and individual (country), time-invariant effects are allowed to interact. It is assumed that there is no permanent additive difference between the treated and control units (Doudchenko and Imbens, 2016). Abadie et al show that under this specification, it is possible to use a function of outcomes observed post-treatment in other countries as an estimator of $\beta_{\mathrm{UK}t}$:

$$\hat{\beta}_{\mathrm{UK}t} = Y_{\mathrm{UK}t} - \sum_{j=1}^{J} w_j^* Y_{jt} \tag{5}$$

Where $\sum_{j=1}^{J} w_j^* Y_{jt}$ is a weighted combination of the outcome for J countries having not implemented the policy, and the vector $W^* = (w_1^* ... w_J^*)'$ should satisfy the following conditions:

$$\begin{cases} w_j^* & \geq & 0 \; \forall j = 1..J \\ \sum_{j=1}^{J} w_j^* & = 1 & \\ \overline{Y}_{\mathrm{UK}}^{K} & = & \sum_{j=1}^{J} w^* \overline{Y}_j^K \\ Z_{\mathrm{UK}} & = & \sum_{j=1}^{J} w^* Z_j \end{cases}$$

With $\overline{Y}_{\mathrm{UK}}^{K}$ a linear combination of pre-intervention outcomes in the UK and $\overline{Y}_j^K$ a linear combination of pre-intervention outcomes for country j (The linear combination is defined by the vector $K = (k_1, ..., k_{T_0})'$. For example, it can be the simple mean of pre-intervention outcomes $\overline{Y}_j^K = 1/T_0 \sum_{t=1}^{T_0} Y_j$). Abadie et al also show that the estimator gets closer to the true parameter $\beta_{\mathrm{UK}t}$ when the number of pre-treatment periods is high compared to the scale of transitory shocks affecting countries.

In practice, to find the appropriate W vector I rely on an algorithm created by Abadie et al. The algorithm minimizes the distance between a vector of pre-intervention characteristics (also called predictors) in the treated country, X_{UK} (with dimensions $K \times 1$) and a weighted matrix of pre-intervention characteristics in the non-treated countries, $X_0 W$ (with dimensions $K \times K$). Pre-intervention characteristics are of two types: 1) the linear combinations of pre-intervention outcomes $\overline{Y}_j^K$, and 2) the country characteristics Z_j not affected by the intervention. To obtain the W vector, the programme starts with a positive and semi-definite matrix V that defines a dot product. The distance between X_{UK} and $X_0 W$ can then be written as

$$X_{\mathrm{UK}} - X_0W = \sqrt{(X_{\mathrm{UK}} - X_0W)'V(X_{\mathrm{UK}} - X_0W)} \tag{6}$$

The goal is to find the vector $W^*(V)$ that minimizes this distance. Such minimization comes down to finding the right V matrix, which can be shown to be equivalent to a diagonal matrix assigning weights to linear combination of characteristics in X_{UK} and X_0W. Like Abadie and Gardeazabal (2003), I choose the V minimizing the mean squared prediction error (MSPE)[15] of the outcome variable in the pre-treatment periods. Formally, let Y_{UK} be the (8×1) vector of pre-2013 power sector emissions from 2005 to 2012 for the UK and Y_j be the $(8 \times J)$ matrix of pre-2013 power sector emissions for the J other European countries. Then V^* is chosen among the set V of all non-negative diagonal $(K \times K)$ matrices, such that:

$$V^* = argmin(Y_{\mathrm{UK}} - Y_jW^*(V))'(Y_{UK} - Y_jW^*(V)) \tag{7}$$

The ability to build a good synthetic control can be assessed with at least two criteria: first, pre-intervention characteristics of the treated unit should be close to those of the synthetic unit. This depends on how well these characteristics predict the outcome and can be assessed by comparing pre-intervention characteristics for the treated and synthetic country. Second, the pre-intervention outcomes of the synthetic unit should be close to the pre-intervention outcomes of the treated unit. This can be checked graphically or by computing the MSPE. Compared to the difference-in-difference method, the number of pre-treatment periods should be large to limit the size of the bias, and relatively larger than transitory shocks affecting the countries (Abadie et al., 2010). As explained below, my main outcome variable is only available from 2005, which implies that my pre-treatment period has only eight years for the main specification. This is rather low compared to other published papers using the synthetic control method. I apply the same method on less

[15] The MSPE gives the average of the squared difference between the treated unit's and the synthetic control's pre-intervention outcomes.

precise aggregate data available since 1990 in appendix A.11 to assess whether the results change. The countries entering the synthetic UK are not the same, but the estimate of the impact is very close to the original one.

3.2 The Data

The empirical strategy relies on a comparison between the UK and other European countries and requires assembling a dataset at the country level. I combine installation-level data on carbon emissions aggregated at the country-level with country-level data obtained from different sources. The installation-level emission data come from the European Union Transaction Log (hereafter EUTL), the official register of the EU ETS, managed by the EU Commission. The EUTL checks, records and authorises all transactions taking place between participants in the EU ETS. Every year since 2005, the start date of the EU ETS, participants have had to report their CO_2e emissions and surrender enough emission allowances to cover their emissions. Reported emissions are verified by an accredited verifier.

Given that not all ETS participants are power installations[16], one crucial step is to identify power installations in the EUTL data. The main activity of each installation is publicly available, but there is no specific activity category for power installations. I rely on data provided by the UK-based think-tank Ember (formerly Sandbag) and a one-off file with more precise activity codes circulated by the EU Commission to identify power installations. Appendix A.3 describes the specific steps followed. I identify a total of 4,938 power plants, including 302 in total for the UK, with an average of 190 active power plants per year in the UK over the 2005-2012 period and 189 active power plants per year in the other twenty EU countries[17], over the same period (See Appendix A.4 for summary statistics).

Almost all the UK power plants subject to the CPS are included in the data, except

[16]The ETS covers combustion installations with a rated capacity above 20 MWth, including power installations, and energy-intensive industries

[17]I exclude the countries which joined the ETS after 2005 (Romania, Bulgaria, Croatia), those which are not part of the European Union for the entire period considered (Slovenia, Norway, Liechtenstein and Iceland), and the three countries having less than ten power plants subject to the EU ETS: Luxembourg (only nine power plants), Cyprus (only three) and Malta (only two)

those with a rated thermal input between 2 and 20 MWth, not covered by the EU ETS. These small plants logically represent a very small share of total emissions. Two categories of UK plants present in the data are not subject to the CPS: power installations located in Northern Ireland, which represent a small share of UK power sector emissions (2.4% in 2012); and standby generators, also representing a small share of emissions[18]. I aggregate plant-level emissions at the country level, separately for power and non-power plants[19]. I obtain emission data for a panel of 21 European countries for the 2005-2017 period.

I add to this panel a set of annual country-level variables which are used in the descriptive analysis (see section A.1) or/and in the empirical strategy: country population, installed capacity and power generation by source, electricity consumption, electricity imports and exports, coal and gas prices, availability of lignite resources (a particularly polluting type of coal only used domestically), average age of the coal-fired plants. Most of these data come from Eurostat. See appendix A.5 for details on each variable's source. Table A.3 shows summary statistics by country for the main variables considered.

3.3 Choice of predictors

Keeping the notation used in section 3.1, the set of predictors X_0 used to build the synthetic UK should be variables predicting country-level per capita power sector emissions, and which values are not affected by the CPS. Choosing characteristics' values for the pre-treatment period ensures that the values are not affected by the CPS[20]. The pre-intervention predictors chosen here are common drivers of emissions identified in the literature (Ellerman and McGuinness, 2008; Van den Bergh and Delarue, 2015; Lee and Melstrom, 2018). Appendix A.5 gives details on how each predictor variable is constructed.

[18] Such generators are likely to be found in hospitals. In 2012, the six ETS power installations from the UK belonging to hospitals represent only 0.05% of UK power sector emissions

[19] Non-power plants are only used in figure 2b, to verify that the UK decrease in emissions only occurs in the power sector. Based on my categorization of power and non-power installations, there is a total of 9,127 non-power plants covered by the ETS, with an average of 618 active plants per year in the UK over the 2005-2012 period and 259 active plants per year in the other twenty EU countries.

[20] In theory, post-treatment values can also be included if the predictors are not affected by the treatment(Abadie et al., 2010).

In countries like the UK relying both on coal- and gas-fired power plants for electricity generation, fuel switching has been identified as an important determinant of emissions variation. Fuel switching is influenced by the coal-to-gas price ratio (Ellerman and McGuinness, 2008), which is directly impacted by the CPS since the tax rate for coal is higher than for gas. I use country-level data on coal and gas prices to build a country-specific time-varying variable of coal-to-gas price ratio.

The coal price data are derived from trade statistics and do not take into account domestic coal resources. In particular, it does not take into account the availability of lignite, a low-quality type of coal with a very high emission intensity, used almost exclusively for power generation and mostly consumed domestically (Berghmans and Alberola, 2013). To account for the large differences in lignite resources across European countries - and its use for electricity generation -, I add a time-invariant predictor defined as a binary variable identifying the countries with large lignite resources: Germany, Poland, Hungary, Greece, and Czech Republic[21]. Since the UK value is 0, the lignite variable constraints the programme to find a synthetic UK with as few countries with lignite reserves as possible.

Power sector emissions also depend on how much electricity demand needs to be covered by CO_2-emitting power plants. Residual load measures this amount of electricity demand that needs to be covered by fossil fuels and biomass once generation from so-called "must-run" power generators (nuclear power plants) and those that generate with almost no marginal cost (solar, wind and hydro) is withdrawn. I build a country-level time-varying variable of per capita residual load by taking the difference between electricity consumption and the generation from renewables and nuclear power plants, and dividing it by total population.

To account for the impact of the European air quality regulation mentioned in section 2.3 and in appendix A.2, I add one predictor measuring for each country the amount of emissions coming from installations that opted out from the LCP directive in 2004 and are expected

[21]The lack of data on lignite reserves covering all Europe constraints me to build a binary rather than a continuous variable (such as the amount of proven reserves by country)

to shut down in 2015. To build this variable, I first identify the name and location of plants opted-out from the LCP directive based on the LCP data available on the European Environmental Agency's website. I then manually identify these plants in the EUTL installation-level emission data[22] to know how much CO_2 these plants emit each year. For each country and each year, I calculate the sum of power sector emissions coming from LCP opt-out plants. I divide each sum by the country population to obtain a variable of per capita LCP opt-out emissions. The opt-out decision had to be made before the CPS was introduced. Thus the share of emissions coming from opt-out plants *before* the announcement of the CPS cannot be affected by the CPS. Using this predictor ensures that the synthetic UK will have about the same quantity of emissions from plants "at risk of closure" by 2015 as the UK. I take as a predictor the value of per capita LCP opt-out emissions in 2009, shortly before the announcement of the introduction of the CPS in 2011.

The two last predictors I use are two lagged outcomes, which is standard in the SCM literature. I take per capita power sector emissions in 2005 and 2012, the first and last year of the pre-treatment period.

For the optimization, the residual load predictor is averaged for the period 2005-2012 and the coal-to-gas price ratio is averaged for the period 2007-2012 to ensure data consistency over time (see appendix A.5 for more details). The remaining predictors are taken for one period only (lagged outcome, per capita opt-out emissions) or are time-invariant (lignite dummy) so they do not need to be averaged.

In a sensitivity analysis presented in appendix A.9, I run the SCM with alternative sets of predictors. The magnitude of the results is unchanged when the installed capacity from plants using combustible fuels, the pre-treatment trend in renewables' installed capacity, the number of heating degree days or the average age of coal-fired plants are included in the set of predictors (although some alternative sets of predictors satisfy less well the requirements of the synthetic control method).

[22]The LCP data use a different installation identifier from the EUTL identifier

3.4 Choice of the donor pool

The "donor pool" designates the set of countries not affected by the CPS that will potentially enter the composition of the synthetic UK. The starting pool of countries consists of the twenty European countries part of the European Union (EU), other than the UK, described in the data section. Restricting the donor pool to EU countries rather than including other OECD countries has several advantages and one drawback. The main advantage is that over the period considered, the UK and other EU countries are subject to the same EU-level policies (in particular the EU ETS and the LCP directive, but also other energy policies). European countries are also likely to be affected in a similar way by global shocks on the energy market, such as the 2011 US shale gas revolution. One drawback is that such geographic proximity and sectoral integration makes spillovers between treated and synthetic unit more likely.

Starting with this initial pool of twenty countries, it is important to discard the countries that are likely to be poor counterfactuals (Abadie et al., 2010). This can include three types of countries: First, countries that suffered idiosyncratic shocks to the outcome of interest, either by directly introducing a policy targeting the power sector or via a more generic exogenous shock likely to affect the electricity sector. Second, countries that are likely to be directly affected by the CPS. Third, countries with very different characteristics compared to the UK, which may cause severe interpolation biases.

By 2017, no other European country had adopted a carbon tax or carbon price floor in addition to the ETS price in the power sector (Metcalf and Stock, 2020).[23] The biggest change in other European countries' power sectors is the case of Germany, which unexpectedly decided to phase out nuclear energy following the 2011 Fukushima nuclear accident. I therefore exclude Germany from the donor pool. Since the European debt crisis affected the Greek economic environment very heavily over the period, I also exclude Greece. However,

[23] France and the Netherlands discussed introducing a carbon price floor as well (Newbery et al., 2018), only the Netherlands have passed a concrete law in August 2018, and the Dutch CPF was scheduled to start in 2020.

including them in the donor pool does not change the results, as shown in appendix A.10.

Regarding the second type of countries, there can be a tension between discarding from the donor pool countries whose outcomes are affected by the treated unit, and finding countries sufficiently comparable to the treated unit (Abadie, 2021). I do not exclude any country based on the risk of spillover, but I discuss this risk and estimate the amount of potential spillovers in section 4.4.

Finally, to avoid having countries too different from the UK, I eliminate Estonia, a country which high emissions per capita are due to the unusual use of oil shale for power generation, a high-emitting input fuel. I also exclude the two other Baltic countries, Latvia and Lithuania, which unlike the UK do not use coal for power generation (see Figure A.3). Since coal-to-gas fuel switching is expected to be an important driver of decarbonisation, it is relevant to restrict the analysis to countries who can experience it.

In the end, the donor pool includes 15 EU countries. Appendix A.10 shows that changing the composition of the donor pool does change the composition of the synthetic UK and the estimates, but not their order of magnitude. To ensure that building a convex combination of countries (having positive weights) that closely reproduce the UK's values for predictors and emissions is possible, there needs to be common support between the distribution of the predictors in the donor pool and in the UK. I check that this is the case for all variables (See the histograms in appendix A.6).

4 Results

4.1 Upper Bound

I start by applying the SCM method using the emission outcome variable, donor pool and predictors exposed in the previous section. Figure 3a shows that the obtained synthetic UK (dashed line) reproduces well the trajectory of UK per capita power sector emissions (continuous line) before 2013, with a Mean Squared Prediction Error (MSPE) of 0.01. Compared to the average per capita power sector emissions for the donor pool (dotted line), the synthetic UK has a relatively close trajectory but higher per capita emissions. Table 2 shows the weights received by each country in the synthetic UK. The synthetic UK is made of five countries: Ireland (49.2%), Slovakia (25.6%), the Netherlands (13.7%), Finland (5.8%), Czech Republic (5.7%). The remaining potential control countries receive a weight of 0. The large weight of Ireland is not surprising: the two countries have close institutions and energy markets, and like the UK, Ireland has a substantial portfolio of coal- and gas-fired power plants. The Netherlands and Slovakia also have a potential for coal-to-gas fuel switching (see figure A.3a). The Netherlands has a residual load per capita close to the UK, and Slovakia and Finland have, like the UK, a substantial amount of LCP opt-out emissions.

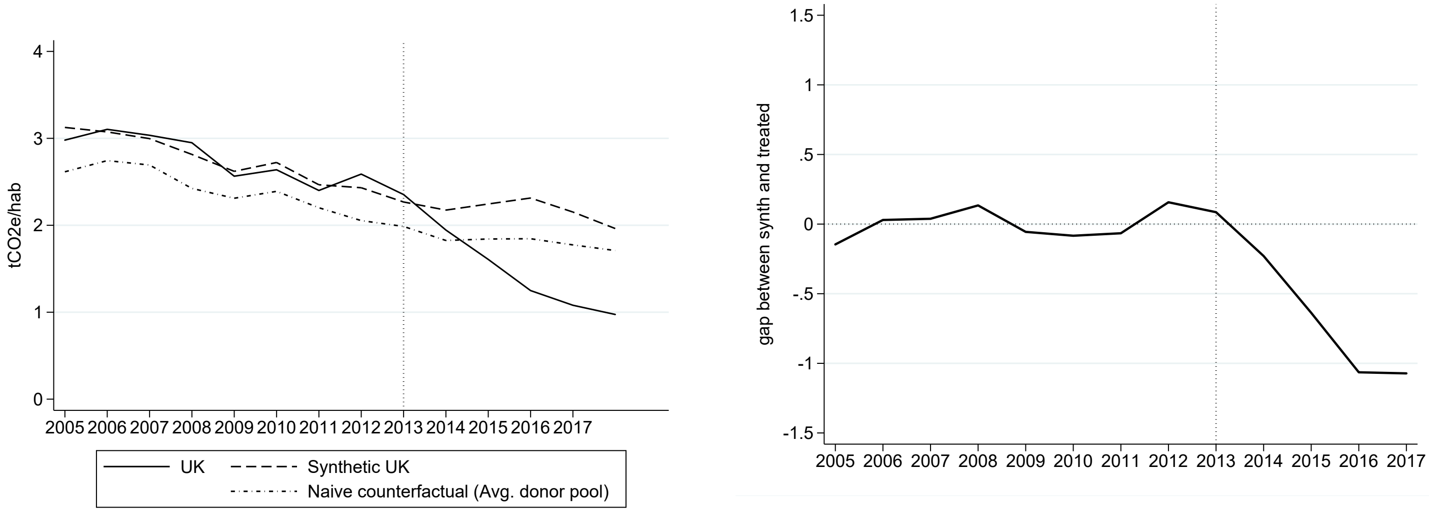
The good pre-treatment fit between the UK and Synthetic UK suggests that after 2013, the synthetic UK accurately replicates the evolution of per capita emissions in the UK power sector absent the CPS (assuming no other UK-specific confounder). The fit is less good in 2012, where UK emissions peak. It is also the year where the share of coal in the UK fuel input mix is the highest, which can be partly explained by the low coal-to-gas price ratio that year (lowest point since 2007) (BEIS, 2018). If power could easily be stored, the 2012 peak could also be interpreted as an anticipation effect of the CPS, which was announced in 2011. Coal-fired plants would then have an interest to use their coal before it becomes taxed in 2012, store the electricity, and sell it over subsequent years. But electricity cannot be stored, and production has to match demand at every point in time. The generation mix at each point in time depends on the merit order, that is, the ranking of plants' marginal costs. Anticipation can only materialize if some coal-fired plants alter the merit order by accepting to sell at a price lower than their marginal cost in order to get rid of their coal reserves. Power plants scheduled to close because of the LCPD may have had an interest in adopting such behaviour, especially if they had excess coal stocks that they wanted to get rid of before being taxed[24].

[24]Anecdotally, official data on annual coal consumption and stocks by electricity generators indicate that coal stocks as a share of stocks and consumption are lower in 2012 compared to previous periods (20% vs 27%

Table 3 shows the average value of each predictor for the UK, synthetic UK, and average of the donor pool. The values of the predictors for the synthetic UK are close to the values for the actual UK - indicating that the synthetic UK is a relatively good counterfactual to the UK. The balance in predictors' values between the UK and synthetic UK is better than between the UK and average of the donor pool for all predictors, further justifying the use of the SCM method.

Figure 3b shows the emission gap between the UK and synthetic UK at each period. The gap between the UK and synthetic UK widens significantly between 2014 and 2016, while UK emissions are slightly higher than synthetic UK emissions in 2013. This evolution is consistent with the timing of the introduction of the CPS (April rather than January 2013) and with the strong increase in the CPS rate between 2013 and 2015, and the CPS freeze in 2015/2016. The corresponding annual abatement for each year $t \in [2013, 2017]$ can be calculated by multiplying the annual gap in per capita emissions by the UK annual total population. On an average year, emissions decrease by 26 percent, with an associated semi-elasticity of -1.65% of emissions per Euro of the tax on average. Adding up all annual abatements gives a total cumulative abatement of 191 million tCO_2e ($MtCO_2e$) over the 2013-2017 period. Abatement is the strongest in 2017, where emissions are lower by 50% in the UK than in the synthetic UK.

on average over 2005-2012), although the difference is not large (BEIS (Department for Business, Energy & Industrial Strategy), 2019).



(a) Absolute per capita emissions

(b) Emission gap between treated and synthetic

Figure 3: UK and synthetic UK per capita emissions

Notes: For each period, the variable of per capita emissions corresponds to the sum of CO_2e verified emissions from power installations subject to the EU ETS, divided by the average country population that year. The vertical line is set in 2013, date where the CPS is introduced. The synthetic UK is made of five countries: Ireland (49.2%), Slovakia (25.6%), the Netherlands (13.7%), Finland(5.8%), Czech Republic (5.7%).

Table 2: Country weights in Synthetic UK

Country	**Weight**	**Country**	**Weight**
Austria	0	Ireland	0.492
Belgium	0	Italy	0
Czech Republic	0.057	Netherlands	0.137
Denmark	0	Poland	0
Spain	0	Portugal	0
Finland	0.058	Sweden	0
France	0	Slovakia	0.256
Hungary	0		

Note: All weights are between 0 and 1 because the Synthetic control method imposes positive weights summing to 1.

Table 3: Predictors' values for the UK, synthetic UK and average of the donor pool

Variable	UK	Synth. UK	Avg. Donor pool
Per capita residual load	4.29	4.30	3.37
Coal-gas price ratio	0.52	0.51	0.71
Per capita LCP opt-out emissions	0.29	0.24	0.22
Lignite dummy	0.00	0.06	0.20
Per capita emissions 2005	2.98	3.13	2.62
Per capita emissions 2012	2.59	2.43	2.05

Notes: The per capita residual load is averaged for the period 2005-12, and the coal-to-gas price ratio for the period 2007-12. LCP opt-out emissions are taken in 2009, the lignite dummy is time-invariant. Outcome lags are taken in 2005 and 2012.

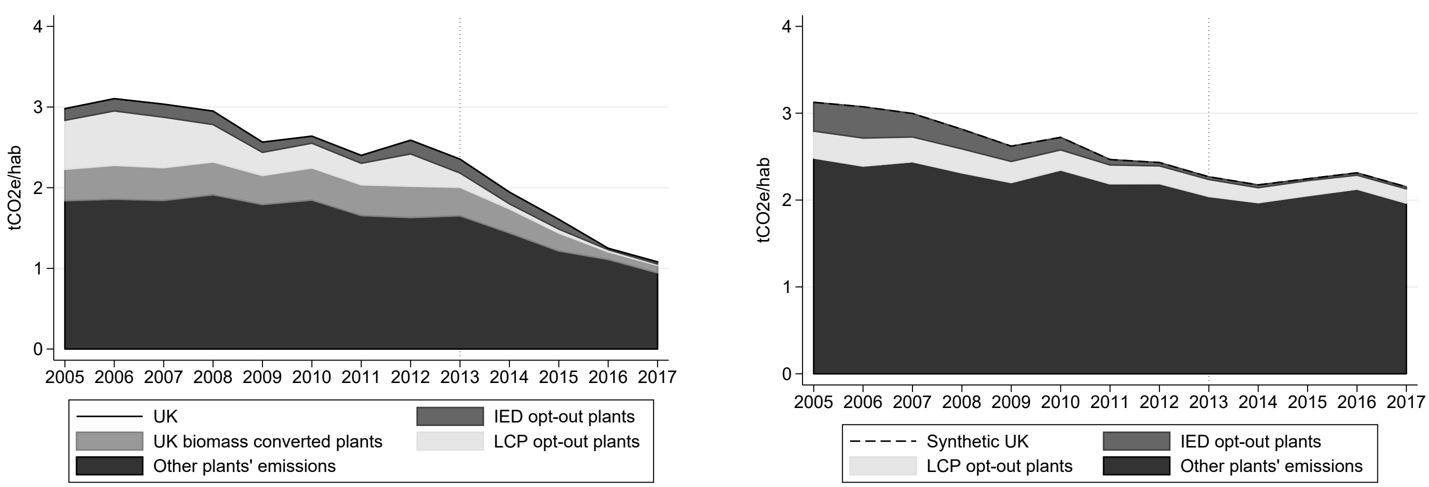
4.2 Lower bound

Potential confounders and emission decomposition: In the main result, the difference in emissions between the UK and synthetic UK after 2013 is assumed to be caused by the Carbon price support only. As mentioned in section 2.2, UK-based policies and European policies affecting the UK differently from other countries may have further contributed to the observed decrease in emissions in the UK.

Regarding European policies, the predictor of LCP opt-out emissions should guarantee that the UK and synthetic UK have approximately the same amount of emissions coming from plants facing a high risk of closure. Given the close values of the LCP opt-out predictor, any difference in the evolution of emissions from opt-out plants between the UK and synthetic UK is assumed to be caused by the CPS. For example, the CPS may affect the way in which the remaining operating hours of each opted-out plant have been spread over the 2005-2015 period; it may also influence the willingness of opt-out plants to lobby governments to remain open in spite of running out of operating hours. The decision to opt out from the IED directive occurred after the announcement of the CPS, so any difference observed in opt-out behaviour between UK and non-UK plants could be a consequence of the CPS.

In contrast, UK-specific policies happening at the same time as the CPS cannot be controlled for in the SCM framework. In Appendix A.2, I estimate that both the Contracts for Differences and the capacity market likely had a limited impact on the fuel mix over the period considered, with estimated emissions reductions of at most 1.4 $MtCO_e$ for the CfD, and at most 2.3 $MtCO_e$ for the capacity market. The situation is different for the biomass conversion policy: the largest UK coal-fired plant, Drax, converted half of its production units from coal to biomass between 2013 and 2016, and the smaller station Lynemouth stopped using coal in December 2015 to prepare for biomass conversion.

To assess the role of the air quality directives and of the biomass conversion, I take advantage of having plant-level emission data. I decompose emissions in four categories for the UK and synthetic UK: emissions coming from LCP opt-out installations (light grey); emissions from IED opt-out installations (dark grey); only for the UK, emissions from installations having benefited from subsidies to convert to biomass (medium grey); and remaining emissions coming from other installations (black). Figure 4 shows the results of this emission decomposition for the UK and Synthetic UK.



(a) UK

(b) Synthetic UK

Figure 4: Per capita CO_2e emissions by source, UK and synthetic UK

Notes: The synthetic UK is made of five countries: Ireland (49.2%), Slovakia (25.6%), the Netherlands (13.7%), Finland(5.8%), Czech Republic (5.7%).

LCP opt-out emissions are higher in the UK than in the Synthetic UK before 2008, but they are close between 2009 and 2012, before the CPS is introduced. After 2012, LCP opt-out emissions decrease sharply in the UK while they remain relatively constant in the synthetic UK. The evolution of the Synthetic UK opt-out emissions may seem surprising: opt-out plants are expected to shut down by 2015 at the latest and we should have zero emissions from opt-out plants in 2016 and 2017, both in the UK and synthetic UK. Singhal (2019) confirms that 60% of opt-out plants actually continued to operate after 2015. The difference in the trajectory of UK and non-UK LCP opt-out plants suggests that the CPS intensified UK plants' response to the LCP opt-out option and accelerated their closure[25]. The figure also confirms that the UK emission peak in 2012 mainly comes from LCP opt-out plants, which had an interest in using their polluting inputs before the introduction of the tax. IED opt-out emissions are relatively low in both the UK and Synthetic UK just before the opt-out decision.

Finally, the emissions from UK plants converted to biomass represent a substantial share of UK emissions and decrease after 2013, in particular between 2015 and 2016. Drax power plant - which represents more than 90% of the emissions converted to biomass - had only half of its six units converted to biomass, so the decrease in emissions after 2013 must be partly explained by the impact of the CPS on the non-converted units. Furthermore, the introduction of the CPS may have influenced the willingness of UK plants to convert to biomass. The estimate above implicitly assumes that this was the case.

To avoid biomass conversion confounding the impact of the CPS on emissions, below I estimate counterfactual CO_2e emissions for the biomass converted plants if they had not converted to biomass. In appendix A.8, I run a second test where I remove from the UK emissions variable all the emissions coming from biomass converted plants and generate a new synthetic UK based on this modified emission variable.

[25] Such an interpretation would also confirm a Guardian journalist's statement that "[UK coal-fired] Plants have closed in recent years as EU pollution standards started to bite, but it was increases in the UK carbon tax that sealed their fate" (Vaughan, 2018).

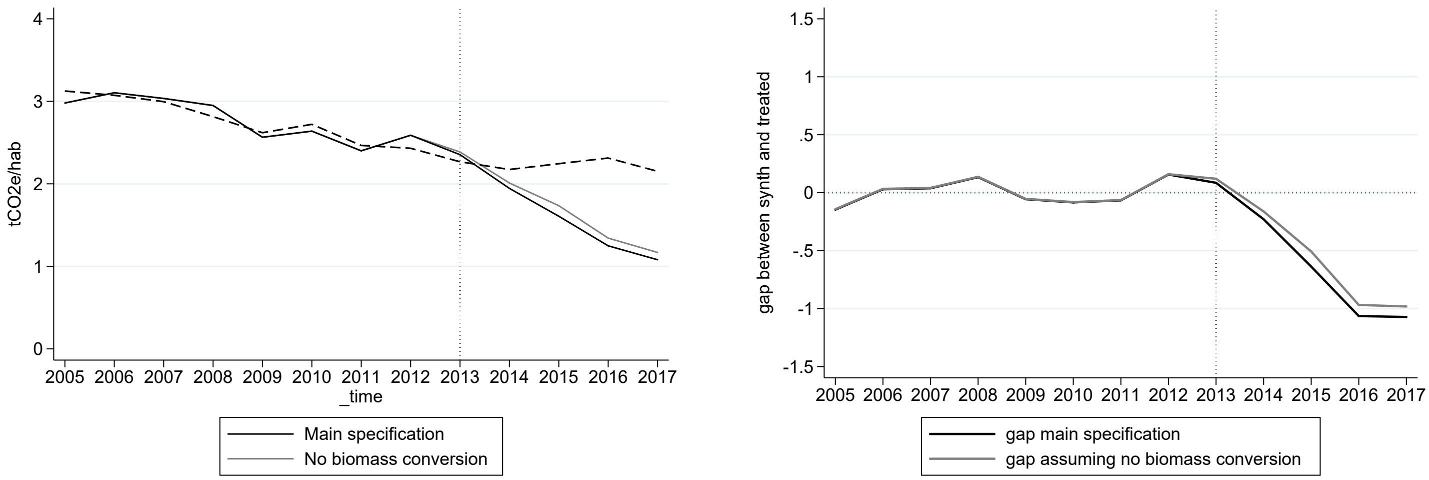
Lower bound: Counterfactual emissions of plants converted to biomass if they had not converted: Appendix A.7 provides details on the imputation of emissions in the absence of biomass conversion. I summarize below the method used for each plant. Estimating what emissions from Drax plant would have looked like in the absence of biomass conversion is relatively easy because I observe the evolution of emissions for its coal units which did not convert. First, I combine data on the monthly generation of the three Drax coal units over the 2009-2016 period, combined with their average emission intensity (kindly provided by Mirjam Kosch and Jan Abrell) to estimate the amount of CO_2 emissions coming from Drax coal units. Second, I subtract the emissions coming from the coal units from the total emissions reported for Drax in the EUTL data to estimate the emissions coming from the three units converted to biomass between 2013 and 2016. In 2016, the estimated CO_2 emissions for these units are close to zero, which makes sense given that the three units run entirely on biomass in 2016. I can then assume that their emissions are also zero in 2017, which means that all the emissions reported for Drax in the EUTL in 2017 come from the three coal units. Third, I assume that absent the biomass conversion, biomass converted units would have had a similar emission trend to the three coal units. Concretely, I start with their estimated CO_2 emissions value for 2012, and I apply the same annual percent change as the annual percent change for the three coal units. I obtain "counterfactual" emissions for the three units converted to biomass. Finally, I add these "counterfactual" emissions to the actual emissions of the the three coal units and I obtain counterfactual emissions for Drax in the absence of the biomass conversion policy.

For Lynemouth plant, I cannot use the same method because the whole plants started its conversion in December 2015[26]. I make a cruder assumption and consider that absent the conversion to biomass, emissions in 2016 and 2017 would have been the same as in 2015. For the two plants, the estimated counterfactual emissions absent the biomass conversion policy only differ from the actual emissions for the 2013-2017 period. For the 2005-2012 period,

[26] https://www.power-technology.com/projects/lynemouth-biomass-power-station-northumberland/

counterfactual emissions are equal to actual emissions.

Finally, I generate a modified outcome variable for the UK, which include Drax and Lynemouth's counterfactual emissions instead of their actual emissions. UK pre-treatment emissions remain the same as before with this modified variable, such that the synthetic UK obtained in the previous section is still an appropriate comparison unit for the UK. Figure 5a shows the UK emission trajectory with this modified outcome variable (dark grey), overlaid to the actual UK and synthetic UK emission trajectories (in black). After "removing" the effect of biomass conversion, emissions are logically higher after 2013 for the modified UK emission variable. The gap between UK and synthetic UK is then smaller (Figure 5b). On an average year, emissions decrease by 22.5 percent, with an associated semi-elasticity of -1.41% per Euro of the tax. The total cumulative abatement is 164 million of tCO_2e. Withdrawing the upper bound estimate for the effect of the capacity market and the CfD ($\approx$ 4.7 $MtCO_2e$ in total), I obtain a cumulative lower bound abatement of around 159 $MtCO_2e$. The estimated abatement is lower in appendix A.8, where the emissions from the biomass converted plants are taken out of UK emissions; the difference between the two abatement results corresponds roughly to the decrease in Drax and Lynemouth counterfactual emissions (in the absence of biomass conversion) between the pre- and post-treatment period.



(a) Absolute per capita emissions

(b) Gap between treated and synthetic

Figure 5: SCM with counterfactual UK emissions w/o biomass conversion

Notes: The synthetic UK is made of five countries: Ireland (49.2%), Slovakia (25.6%), the Netherlands (13.7%), Finland(5.8%), Czech Republic (5.7%).

4.3 Inference

In the SCM method, inference can be derived from a set of placebo tests, consisting in applying the SCM method to untreated units or fake treatment dates (Abadie et al., 2010).[27]. I show that the results are likely driven by the causal impact of the Carbon Price Support by measuring (1) how likely it is to find an effect of the same magnitude as what I find when I apply the method before 2013 ("in time" placebo test); (2) how likely it is that the result is driven by the peculiar behaviour of one country in the donor pool ("leave-one-out test"); (3) how likely it is to find an effect the magnitude from what I find when I apply the method to other countries ("in space" placebo test or permutation test). The in-time and leave-one-out test are run for the actual UK emissions used for the upper bound estimation from section 4.1. Results would be the same for the lower bound, given that the composition of the synthetic UK and the trajectory of emissions in the 2005-2012 period are the same. For the permutation test, I show the results both based on the upper bound and on the lower bound estimation of abatement.

[27]Having only one treated unit is not sufficient to build confidence intervals as done in Gobillon and Magnac (2016) and Isaksen (2020).

"In-time" placebo One way to check that the results observed are indeed caused by the CPS policy is to assume that a similar policy was implemented at another date prior to 2013, apply the same method to generate a synthetic UK, and check that the UK and synthetic UK have similar per capita emissions before and after this artificial intervention date. Figure 6 shows the UK and synthetic UK obtained when treatment is assumed to occur in 2010 rather than 2013. The synthetic UK closely resembles the UK emission trajectory before 2010, and there is no significant gap between treated and synthetic UK in 2011 and 2012.

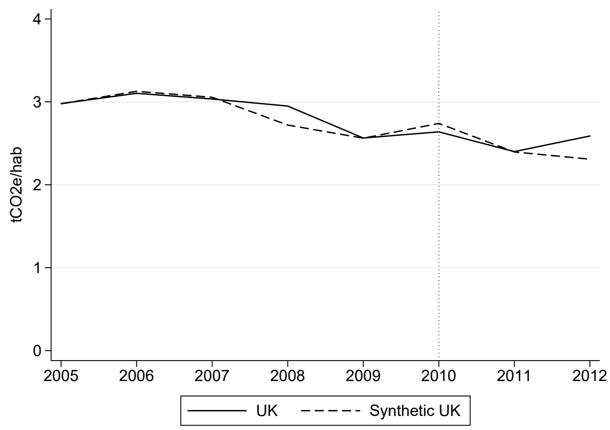
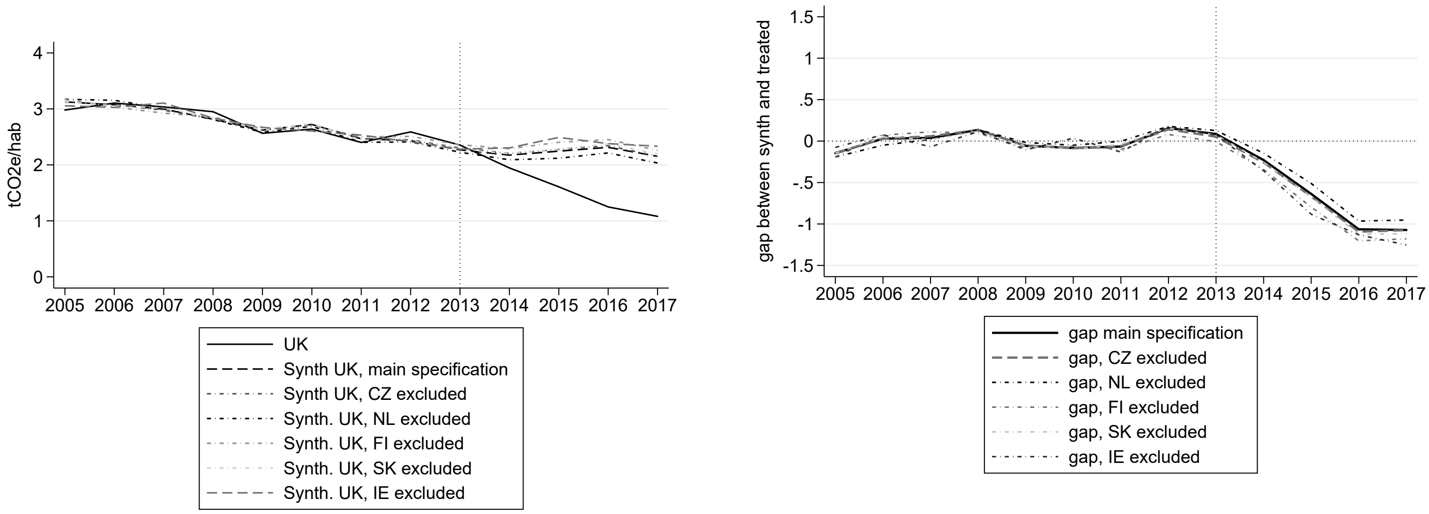


Figure 6: Gap between treated and synthetic UK, CPS assumed to start in 2010

Notes: Predictors are averaged for the period 2005-09, except for the coal-gas price ratio averaged for the period 2007-10. The lagged outcome is taken in 2009 instead of 2012. The Synthetic UK is made of seven countries with a weight above 1%: Ireland (45.8%), Slovakia (23.1%), Finland (15.3%), Czech Republic (3.3%), The Netherlands (3.1%), Sweden (2.3%) and Denmark (1.1%).

Leave-one-out test Another common test recommended in Abadie et al. (2010) is the leave-one-out test, which consists in running the synthetic control method again after iteratively removing each country receiving a positive weight in the synthetic UK. If the results change significantly when a country is removed, it means that the estimated effect may have been caused by the evolution of emissions for that country, rather than by the change in UK emissions. Figure 7 shows that the results change very little across the alternative donor pools. This test suggests that my estimate of abatement is not driven by the presence of a specific country in the donor pool.



(a) Absolute per capita emissions

(b) Emission gap between treated and synthetic

Figure 7: Leave-one-out test

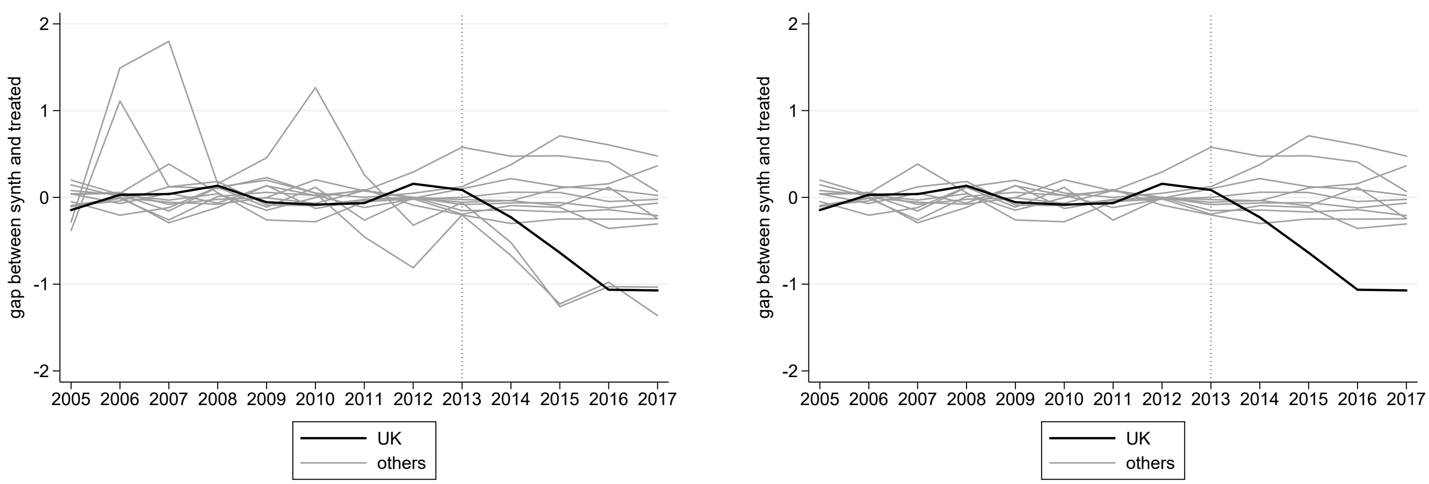
Notes: Synthetic UK for the main specification: five countries: Ireland (49.2%), Slovakia (25.6%), the Netherlands (13.7%), Finland(5.8%), Czech Republic (5.7%). Specification without Ireland: four countries: the Netherlands (45.9%), Spain (38%), Denmark (8.6%), and Czech Republic (7.5%). Specification without Slovakia: five countries: Ireland (52.5%), France (18.4%), the Netherlands (17.5%), Czech Republic (6.2%) and Finland (5.4%). Specification without the Netherlands: four countries: Ireland (58.0%), Slovakia (26.2%), Finland (10.2%), and Poland (5.7%). Specification without Finland: thirteen countries: Ireland (47.9%), Slovakia (20.5%), the Netherlands (18.8%), Poland (11.1%), all other countries have a weight below 1%. Specification without Czech Republic: five countries: Ireland (50.9%), Slovakia (22.3%), the Netherlands (14.2%), Poland (6.6%), and Finland (5.9%).

Permutation test The permutation test consists in building a synthetic counterfactual for each country of the donor pool. Then, the gap between each country and its synthetic counterpart is compared with the gap obtained for the UK in the main results section. If for many countries the gap is as large as for the UK, it means that the gap obtained for the UK could have happened just by chance, rather than as a result of the introduction of the CPS. Figure 9a shows the gap between the treated and synthetic country for the UK and all the other countries in the donor pool. For Czech Republic and France, having respectively the highest and lowest per capita emissions, and for Italy, it is impossible to find a convex combination of countries replicating the pre-2013 emissions. So these countries are not included. For Denmark and Finland, the pre-2013 fit is poor, with a pre-treatment MSPE more than 10 times greater than the UK.[28] Comparing the UK emission gap with

[28] As mentioned in appendix A.1, Denmark and Finland have a high variability in emissions, likely explained by the large inter-annual variations in renewable sources available for electricity generation.

these countries is not very meaningful since the conditions for a good synthetic control are not met. Hence Figure 9b drops these two countries, as advised in Abadie et al. (2010). The UK clearly stands out as having the largest decrease in per capita emissions after 2013.

To illustrate the difference in the magnitude of pre- and post-2013 emission gap between the UK and the other permutations, one can also compute the ratio of post to pre-MSPE for all countries (Abadie et al., 2010). We should observe an unusually high ratio for the UK. Figure 9a shows that the UK ratio is indeed the largest. Figure 9b shows that this is also the case with the modified UK emission value from the lower bound, although the ratio is lower. We can calculate the estimated probability to observe an effect as large as the one observed for the UK under a random permutation of the intervention on the data, by dividing the number of countries having a higher ratio than the UK by the total number of countries (Abadie et al., 2010). Here the UK has the highest ratio amongst the 13 countries, so the associated probability is $1/13 = 7.7\%$, the lowest possible probability with this sample size.

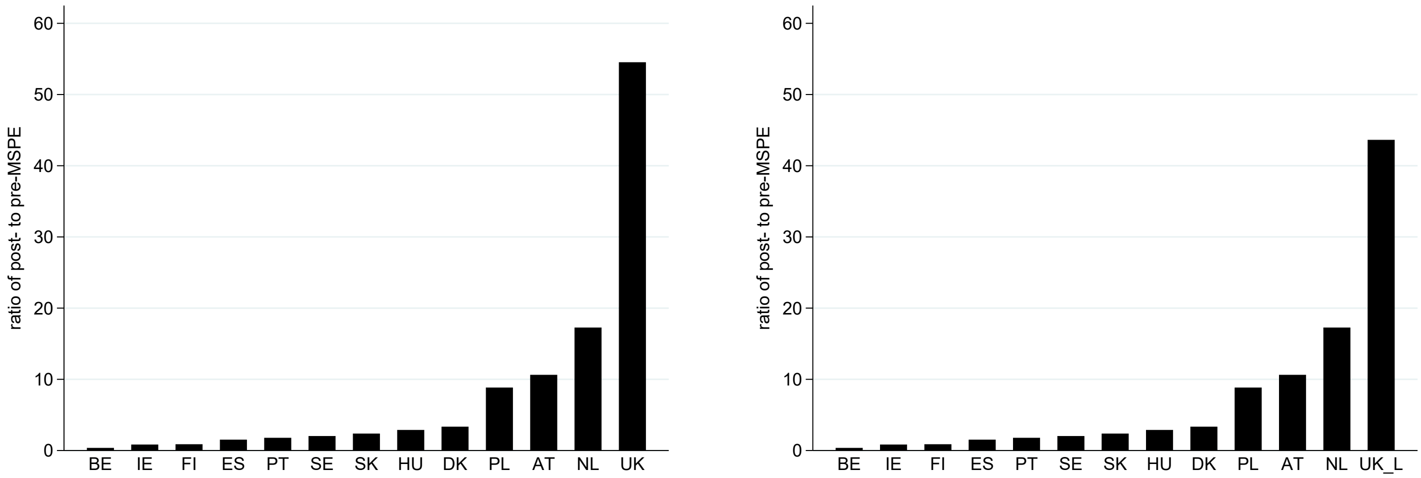


(a) All countries

(b) W/o countries with a large MSPE

Figure 8: Permutation test

Notes: In both figures, Czech Republic, France and Italy are not included: for these countries it is impossible to find a convex combination of countries replicating pre-2013 emissions. On figure b, the two countries with an MSPE 10 times higher than the UK, Denmark and Finland, are not included.



(a) Upper bound

(b) Lower bound

Figure 9: Ratio of post to pre-MSPE

Notes: Czech Republic, France and Italy are not included: for these countries it is impossible to find a convex combination of countries replicating pre-2013 emissions.

4.4 Risk of spillovers

For the synthetic control method to identify the causal impact of the intervention, candidate units for the synthetic control group should not be affected by the intervention. As an overlapping policy to an existing carbon market, The CPS could spill over to other European countries' power sectors via two channels highlighted by Perino et al. (2019): internal leakage, that is, an increase in UK net electricity imports from other European countries; or a waterbed effect, that is, an increase in emissions from European power plants not subject to the CPS, due to the negative effect of the CPS on ETS permit prices under a fixed emission cap. Quantifying the magnitude of these two effects for the EU carbon market as a whole goes beyond the scope of this paper, which focuses on the impact of the CPS on UK emissions. What I need to assess is the risk of spillovers to countries entering the Synthetic UK, given that they serve as a counterfactual for the evolution of UK emissions in the absence of a CPS.

I first estimate the amount of emissions from countries in the Synthetic UK potentially caused by import spillovers. This amount is naturally bounded by the limited interconnection capacity of the UK with the rest of Europe. I then estimate the amount of emissions in the

Synthetic UK potentially caused by a waterbed effect. The two effects combined represent 11% of the estimated abatement of the lower bound.

Risk of spillover via increased electricity imports UK net electricity imports per capita are generally low compared to other European countries (see Figure A.2b), representing 2% of gross electricity consumption in the 2005-2012 period. However, net imports increased to 5% of gross electricity consumption in the 2013-2017 period. If this increase was caused by the CPS, it could threaten the identification strategy because two of the UK trading partners, Ireland and the Netherlands, have a combined weight of 63% in the synthetic UK. The increase in UK net imports would then increase the synthetic UK's emissions *as a result of the CPS* and contaminate the counterfactual. The question is how large in magnitude this contamination is, relative to the estimated abatement. I calculate the maximum amount of Synthetic UK emissions that may have been directly caused by the CPS, considering that the increase in UK electricity imports from France, the Netherlands and Ireland after 2012 is entirely caused by the CPS[29]. I estimate the emissions associated with these exports for Ireland and the Netherlands (the countries entering the Synthetic UK). In Appendix A.12, I run another test where I exclude all interconnected countries from the donor pool to assess whether the presence of Ireland and the Netherlands in the Synthetic UK drives up the results. The estimated abatement is 14% lower without interconnected countries, but the balance in predictors' characteristics is also less good.

First, I calculate the excess electricity generation in the Netherlands and in Ireland which can be imputed to CPS-induced exports to the UK: to do so, I simply calculate, for every post-treatment year, the difference between electricity exports to the UK that year and average electricity exports to the UK in the pre-treatment period. I use electricity

[29] Guo and Newbery (2020) estimate that 0.9% of the CO_2 emission reduction taking place in the UK between 2015 and 2018 was undone by the increase in electricity imports from France and The Netherlands. I cannot use this estimate because the time period is different, the estimated UK emission reduction is based on a different method - a dispatch model of the 2015 GB power system, see (Kong Chyong et al., 2020) -, and the paper only considers France and the Netherlands (two markets coupled with the UK in 2014, which enabled cross-border electricity trading to take place as soon as market prices were different across the two sides of the interconnection) but not Ireland, which represents half of the Synthetic UK.

trade statistics from Ofgem, which give quarterly trade flow for each interconnector with the UK.[30] I estimate that on an average year between 2013 and 2017, the Netherlands produced an excess of 2,965 GWh, and Ireland produced an excess of 382 GWh, compared to the pre-treatment period.

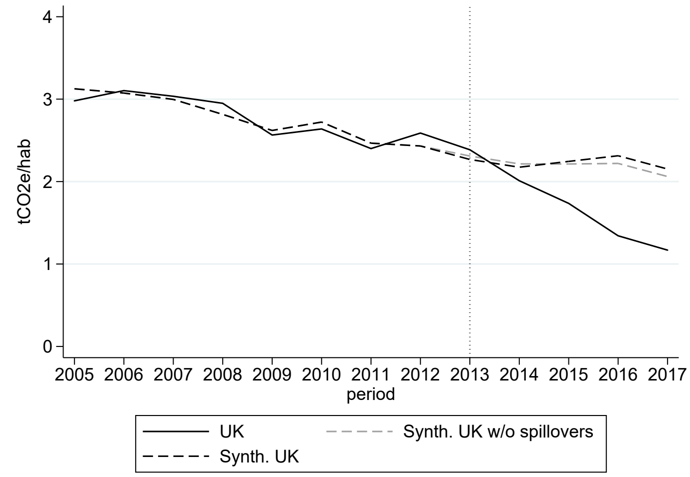
Second, I calculate the emissions associated with this electricity generation. The emission intensity of this displaced generation depends on which technology is used for marginal generation. According to Guo and Newbery (2020), gas is the marginal fuel most of the time in the Netherlands. Furthermore, the emission intensity of fuel displaced by renewable energy in Ireland in 2012 was estimated to 0.43 tCO_2e/MWh (Sustainable Energy Authority of Ireland, 2014), which is close to the emission intensity of gas in the UK. Assuming a marginal intensity of 0.43 tCO_2e/MWh both in the Netherlands and in Ireland, the excess emissions caused by the CPS are 6.4 $MtCO_2e$ over the 2013-2017 period in the Netherlands, and 0.8 MtCO2e in Ireland (exporting less to the UK than the Netherlands).[31]. Third, I remove these excess emissions from Dutch and Irish emission data over the 2013-2017 period. I assess how the emission trajectory of the Synthetic UK changes when these excess emissions are removed.[32]

Figure 10a shows how the trajectory of the Synthetic UK emission changes after removing these "leaked" emissions from Ireland and the Netherlands. Net imports from both the Netherlands and Ireland are higher than the pre-treatment average in 2015, 2016 and

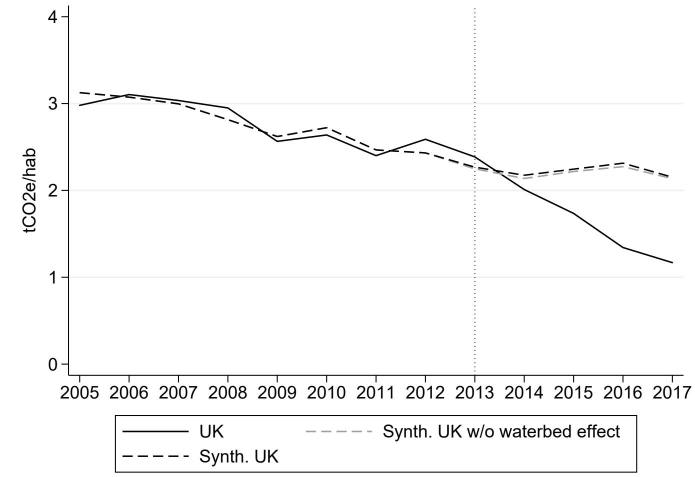
[30] Since the Netherlands-UK interconnector entered into service in April 2011 only, I average trade flows between the second quarter of 2011 and the fourth quarter of 2012 to get average trade flow pre-treatment (I include the first quarter of 2013 in the post-treatment period to be consistent with the rest of the analysis). For Ireland-UK trade, I consider separately the interconnectors between Ireland and Northern Ireland, open for the entire pre-treatment period and for which I average trade flows for the 2005-2012 period, and the interconnector between Ireland and Wales (East-West interconnector), which only opened in September 2012 and for which I only consider the trade flow of the last quarter of 2012.

[31] If I instead calculate emissions assuming that gas is the marginal fuel, with an emission intensity of 0.4 tCO_2e/MWh (which is the average for the UK, see (Abrell et al., 2019)), these excess emissions are 5.9 $MtCO_2e$ in the Netherlands and 0.8 $MtCO_2e$ in Ireland . If I assume that coal is the marginal fuel, with an emission intensity of 0.89 tCO_2e/MWh (which is the average for the UK, see (Abrell et al., 2019)), the excess emissions are 13.2 MtCO2e for the Netherlands and 1.7 $MtCO_2e$ for Ireland.

[32] I do not impute these excess emissions back to the UK because the goal is not to estimate the impact of the CPS net of leakage, but rather to accurately estimate the impact of the CPS on UK emissions by making sure that the counterfactual does not include spillover effects.



(a) UK and synthetic UK w/o imports spillovers



(b) UK and synthetic UK w/o waterbed effect

Figure 10: Spillover risk

Notes: The synthetic UK is made of five countries: Ireland (49.2%), Slovakia (25.6%), the Netherlands (13.7%), Finland(5.8%), Czech Republic (5.7%). UK emission values include estimated counterfactual emissions in the absence of biomass conversion for Lynemouth and Drax plants.

2017, such that removing the estimated "leaked" emissions reduces emissions from the Synthetic UK.[33] Overall, the gap between the UK and synthetic UK is smaller than when these spillovers are not accounted for, which is expected. The resulting cumulative abatement is smaller by 5% compared to that estimated in section 4.2.

Risk of spillover via a waterbed effect Theoretically, the waterbed effect designates the mechanism via which, under a common emission cap - in this case the cap set by the ETS carbon market -, any emissions reduction in a given country only leads to an increase in emissions elsewhere (Böhringer et al., 2008; Goulder and Stavins, 2011; IPCC et al., 2014; Perino, 2018). The waterbed effect would arise because the CPS decreases demand for emission permits from UK installations subject to a higher carbon price. On the EU ETS market as a whole, the shift to the left of the demand curve can only be compensated by a price decrease, since the supply is fixed and perfectly inelastic because of the emission cap. With cheaper permits, individual installations subject to the ETS but not to the CPS can

[33]Emissions slightly increase in the modified Synthetic UK in 2013 and 2014, because net imports from Ireland decrease at this period and are not compensated by the increase in net imports from the Netherlands given the much higher weight of Ireland than the Netherlands in the Synthetic UK.

buy more allowances and emit more. Aggregate emissions remain unchanged.

The concern for the empirical strategy is a waterbed effect affecting the power installations composing the synthetic UK. This risk exists but the magnitude of the effect is likely to be small for two reasons. First, UK power installations represent only a small share of the total ETS market (in 2012 they represent 8.8% of total emissions covered by the EU ETS), so the demand-side shock coming from UK power installations is likely to be small. To illustrate this, Figure 10b simulates a 100% waterbed effect scenario. Under this scenario, the decrease in UK power sector emissions after 2013[34] is assumed to be caused by the CPS and be compensated by an equivalent increase in emissions coming from other ETS installations. Observed power sector emissions outside the UK therefore include a waterbed effect component, compared to what non-UK emissions would have looked like in the absence of the CPS. Assuming that the waterbed effect was spread across the different sectors and countries based on their share in ETS emissions in 2012, I estimate the waterbed effect component for each country's power sector. I then estimate each country's adjusted, *lower* emission value excluding the waterbed effect component[35]. Figure 10b shows the emission trajectory for the modified UK emission variable (corresponding to the lower bound estimate), and for the synthetic UK after removing the hypothesized waterbed component. Given the low weight of UK power installations in the market's total emissions, the waterbed effect component is small once spread over all ETS countries, and the adjusted synthetic UK emissions are only slightly higher than in the main specification. The magnitude of this waterbed component is similar to that of the import spillover component estimated above: the cumulative abatement based on Figure 10b is also smaller by 5% compared to that estimated in section 4.2.

The second reason why a strong waterbed effect is unlikely is linked to the specific context of the EU ETS in the 2013-2017 period. At the time, there was a structural oversupply of al-

[34]using the modified UK emission variable, the one including emissions from biomass converted plants if they had not converted to biomass

[35]To take a concrete example: in 2014, UK emissions were lower by 25 $MtCO_2e$ than in 2013; Ireland represents 1.2% of ETS power sector emissions in 2012 (excluding the UK); power installations represent 66% of emissions in the whole ETS; the 2014 waterbed component for Ireland is estimated to be $1.2\% \times 66\% \times 25 = 0.2MtCO_2e$, which represents 1.8% of Ireland's observed power sector emissions in 2014.

lowances on the ETS, leading market participants to *bank* more allowances (Ellerman et al., 2016). In this context, if cheaper ETS permits were purchased by non-UK power installations, these permits are likely to have been banked for future use rather than transformed in contemporaneous emissions, leaving synthetic UK emissions uncontaminated.

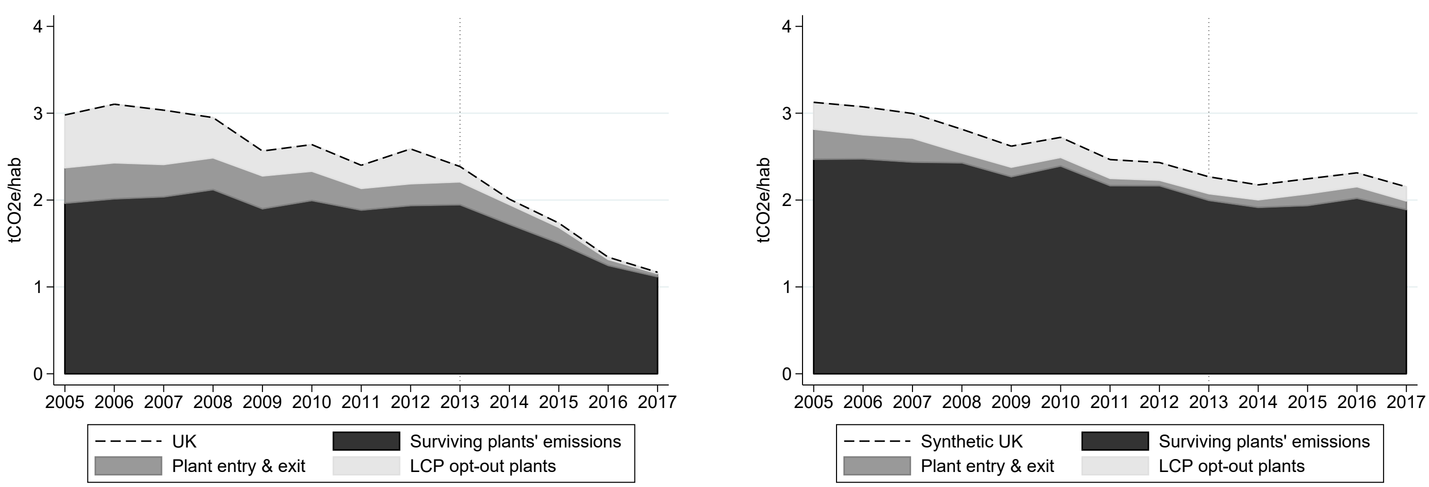
Overall, the combined effect of internal leakage and the waterbed effect likely contributed to increasing emissions from other European countries, but this increase is deemed to be small. Emissions in the Synthetic UK are thus slightly overestimated compared to what they would be in the absence of the CPS. Taking out the simulated import spillovers and waterbed components from the synthetic UK yields a gap between the UK and synthetic UK that is about 18 $MtCO_2e$ lower than if we assume spillovers to be insignificant. Accounting for these spillovers and all potential confounding factors would decrease abatement from 159 $MtCO_2$ (lower bound estimation including the potential effect of the capacity and CfD policies) to 159-18=141 $MtCO_2$. The average annual abatement would be -20.5%.

5 Discussion

5.1 Channels contributing to emission decrease

My results suggest that in the absence of the CPS, UK power sector emissions would have been higher by between 141 and 191 $MtCO_2e$. The upper bound is the SCM estimate assuming that biomass conversion was caused by the CPS, that other UK-specific policies had a negligible impact, and that spillovers were negligible. The lower bound estimate takes out from the upper bound 1)the estimated effect of biomass conversion 2)an upper bound for the CfD and capacity market 3)an upper bound for import spillovers and the waterbed effect.

Using the plant-level emission data, I estimate the relative contribution of three mechanisms contributing to emission reductions, based on the lower bound estimation from section 4.2: 1)the decrease in the emission intensity of existing plants; 2)the closure of plants having not opted out from the LCP directive; and 3)the differentiated behaviour of UK LCP opt-out plants induced by the CPS. Figure 11 shows the results from this decomposition for the UK and synthetic UK: emissions from installations present in the EUTL data every year over the 2012-2017 period are in black; emissions from installations which appear in the EUTL data or disappear from it between 2012 and 2017 (which I interpret as a plant entry in the first case and a plant exit in the second case) are in medium grey; and emissions from LCP opt-out installations are in light grey. The difference in black areas reflects the impact of the CPS at the intensive margin (excluding LCP opt-out plants), that is to say how much more or less existing plants emit as a result of the policy. The difference in medium grey areas reflects the impact of the CPS at the extensive margin (excluding LCP opt-out plants), that is to say how much more or less power plants enter and exit the market as a result of the policy; finally, the difference in black areas between the UK and synthetic UK before and after 2013 captures the impact of the CPS on the emission trajectory of LCP opt-out plants.



(a) UK

(b) Synthetic UK

Figure 11: per capita CO_2e emissions by source, UK and synthetic UK, lower bound estimation

Notes: For each period, the variable of per capita emissions corresponds to the sum of CO_2e verified emissions from power installations subject to the EU ETS except those in the UK converted to biomass, divided by the average country population that year. The synthetic UK is made of five countries: Ireland (49.2%), Slovakia (25.6%), the Netherlands (13.7%), Finland(5.8%), Czech Republic (5.7%).

Drawing on the difference-in-difference methodology, I calculate for each component the double difference between the average pre-treatment and post-treatment emissions, between the UK and synthetic UK. For the pre-treatment period, I take the average over the 2009-2012 period rather than the 2005-2012 period, because UK and synthetic UK opt-out emissions are different before 2009. These back-of-the-envelope calculations suggest that the different behaviour of LCP opt-out plants contributed roughly 52 $MtCO_2e$ over the 2013-2017 period (double difference in the light grey areas). The intensive margin excluding opt-out plants contributed roughly 45 $MtCO_2e$ over the 2013-17 period (double difference in the black areas). The extensive margin excluding opt-out plants contributed roughly 50 $MtCO_2e$ over the 2013-17 period (double difference in the medium grey areas). This extensive margin effect is the combined effect of having more emission reductions from coal-fired plants shutting down in the UK than in the synthetic UK[36]; and also fewer high-emission plants entering the market in the UK compared to the Synthetic UK[37].

5.2 Comparison of the results with existing estimates

Two other working papers, by Abrell et al. (2019) and Gugler et al. (2020), evaluate the impact of the CPS on emissions reductions, using a different data coverage and different methods. In terms of data coverage, Gugler et al. (2020) considers the 2013-2015 period and Abrell et al. (2019) consider the 2013-2016 period. Both papers also estimate emissions at the monthly or hourly level by combining hourly generation data with estimated plant-specific emission factors. Doing so, they are able to allocate emissions from January to March 2013 to the pre-treatment period, while I use annual emission data and have to allocate all the

[36]Two large coal-fired UK plants shut down in the period considered while they had neither opted-out from the LCP or IED directives: Rugeley power station closed in March 2012 and Longannet power station closed in March 2016. For Rugeley, the official reason was "a "continued fall in market prices" and increases in carbon costs" (Source: https://www.bbc.com/news/uk-england-stoke-staffordshire-35526894). For Longannet, the official reason was that it was it was ""uneconomic to continue"[..]because of the high transmission charges and carbon taxes."(Source: https://www.theguardian.com/environment/2016/mar/24/longannet-power-station-closes-coal-power-scotland).

[37]three new Dutch coal-fired plants entering the market in 2015 explain the increase in synthetic UK emissions at the extensive margin after 2014

2013 emissions to the pre-treatment period.

The annual results reported by Gugler et al. (2020) using a Regression Discontinuity in Time (RDiT) imply a cumulative abatement of 39 $MtCO_2e$ over the 2013-2015 period. If I restrict my estimates to the 2013-2015 period, I find a cumulative abatement of between 36 $MtCO_2e$ (lower bound from section 4.2) and 51 $MtCO_2e$ (upper bound from section 4.1). Their results are thus included in my estimate range and are consistent with mine.

Abrell et al. (2019) estimate counterfactual generation for treated plants using a machine learning algorithm based on a short-run equilibrium model of the electricity wholesale market, and find a cumulative abatement of 26 $MtCO_2e$ over the 2013-2016 period, by . If I restrict my estimates to the 2013-2016 period, I find a higher abatement, of between 100 $MtCO_2e$ (lower bound from section 4.2) and 120 $MtCO_2e$ (upper bound from section 4.1). Abrell et al mention that their estimate "should best be viewed as providing a lower-bound empirical estimate of the environmental effectiveness of the UK carbon tax" (Abrell et al. (2019), p41), and the difference between our results is likely explained by the difference in data coverage and methodology. The main difference is that Abrell et al. only consider large gas- and coal- fired plants running for the whole period of analysis (from 2009 to 2016), while I include all types of power plants, including those closing or opening during the period of analysis. Doing so, I can measure the impact of the CPS at the extensive margin and how the CPS interacted with the LCP directive to accelerate closure, while the scope of Abrell et al is closer to the intensive margin channel identified in section 5.1.

Based on the decomposition analysis from the previous section, I find an abatement of 14.5 $MtCO_2e$ at the intensive margin over the 2013-2016 period. This estimate is closer to the order of magnitude of Abrell et al's estimate. It is lower, which may be due to the difference in the method used, to the emission scope, as the emissions included in my "intensive margin" channels do not overlap perfectly with the emissions included in Abrell et al[38]; and to the time scope, as my estimate for 2013 includes the Jan-March 2013 period,

[38]some of the plants they consider are not in my "intensive margin" emissions, but rather in the "extensive margin" (for plants closing in 2016, such as Rugeley) or in the "lcp opt-out" channels; the other way around,

three winter months with a presumably high electricity consumption where the carbon tax was not yet in force.

6 Conclusion

I find that the carbon tax implemented in the UK power sector in 2013 resulted in a large decrease in carbon emissions coming from a mix of mechanisms: a decrease in carbon emissions of plants staying in the market, a stronger response of opt-out plants to the LCP Directive than in countries without carbon tax, and the closure of some high-emitting plants. While an advantage of the SCM method applied to aggregated plant-level data is to take into account different channels via which carbon pricing impacts emissions, it also has some limitations: first, I cannot estimate precisely the relative contributions of these different channels. Second, I cannot estimate heterogeneous treatment effects across the different treated plants.

From the point of view of its effectiveness, the CPS policy can be considered successful: the estimated abatement represents between 60% (for the lower bound estimate of 141 $MtCO_2e$ taking into account biomass conversion, the Contract for Differences, the capacity market and spillovers) and 81% (for the upper bound abatement of 191 $MtCO_2e$) of the abatement necessary to achieve the targets set for the second carbon budget. While this is not the focus of this paper, other work suggests that this abatement has been achieved at a relatively low cost (Gugler et al., 2021).

Regarding the external validity of the results, it is important to have in mind three factors that arguably enabled the tax to have such a high impact on abatement with limited carbon leakage: the relatively high potential for fuel switching from coal to gas, the UK relative isolation from other electricity markets that limited the risk of carbon leakage, and the context playing against new investments in high-emitting generation. Several countries meet these criteria and could be good candidates to replicate the UK experience. Wilson and

some plants included in my "intensive margin" are not included in Abrell et al's analysis, such as the plants from Northern Ireland, not subject to the CPS

Staffell (2018) estimate that many European countries have sufficient idle gas capacity to completely eliminate coal via fuel switching, while Russia and the United States could switch 40-50% of their coal generation and China and India only 6-12%. Avoiding carbon leakage may be more difficult for countries with strong interconnections, but a solution could be to implement a carbon price floor at the regional level, as suggested in Newbery et al. (2019) for North-Western Europe. Finally, my results suggest that the interaction between increasingly stringent regulations of industrial emissions and a carbon price accelerated several plant closures. This context may also have encouraged companies operating multiple power plants to view the CPS favourably, as an instrument providing a clear price signal and making the case for coal even worse as it was before. An evidence for this is that UK power companies supported the Carbon Price Floor (Hirst, 2018). Again, such a context is not unlikely to be found in other countries, given the global trend towards more stringent air quality regulations.

References

Abadie, A. (2021). Using Synthetic Controls: Feasibility, Data Requirements, and Methodological Aspects. *Journal of Economic Literature*.

Abadie, A., A. Diamond, and J. Hainmueller (2010, June). Synthetic Control Methods for Comparative Case Studies: Estimating the Effect of California's Tobacco Control Program. *Journal of the American Statistical Association 105*(490), 493–505.

Abadie, A., A. Diamond, and J. Hainmueller (2015). Comparative Politics and the Synthetic Control Method. *American Journal of Political Science 59*(2), 495–510.

Abadie, A. and J. Gardeazabal (2003, March). The Economic Costs of Conflict: A Case Study of the Basque Country. *American Economic Review 93*(1), 113–132.

Abrell, J., M. Kosch, and S. Rausch (2019). How Effective was the UK Carbon Tax?—A Machine Learning Approach to Policy Evaluation. *Unpublished Working Paper*.

Andersson, J. J. (2019). Carbon Taxes and CO2 Emissions: Sweden as a Case Study. *American Economic Journal: Economic Policy*.

Ares, E. and J. Delebarre (2016). The Carbon Price Floor. Technical report, House of Commons Library.

BEIS (Department for Business, Energy & Industrial Strategy) (2019). UK Coal statistics.

Berghmans, N. and E. Alberola (2013). The Power Sector in Phase 2 of the EU ETS: Fewer CO2 Emissions but just as much Coal. *Climate Report. paris CDC Climate research* (42).

Böhringer, C., H. Koschel, and U. Moslener (2008). Efficiency losses from overlapping regulation of EU carbon emissions. *Journal of Regulatory Economics 33*(3), 299–317.

Brown, G. (2017). British power generation achieves first ever coal-free day. *The Guardian*.

Chyong, C.-K., B. Guo, and D. Newbery (2019, January). The impact of a Carbon Tax on the CO2 emissions reduction of wind. Technical Report 1904, Faculty of Economics, University of Cambridge.

Colmer, J., R. Martin, M. Muûls, and U. J. Wagner (2020, November). Does Pricing Carbon Mitigate Climate Change? Firm-Level Evidence From the European Union Emissions

Trading Scheme. Technical Report crctr224_2020_232, University of Bonn and University of Mannheim, Germany. Publication Title: CRC TR 224 Discussion Paper Series.

Cropper, M. L., R. D. Morgenstern, and N. Rivers (2018). Policy Brief—Facilitating Retrospective Analysis of Environmental Regulations. *Review of Environmental Economics and Policy 12*(2), 359–370. Publisher: Association of Environmental and Resource Economists.

Doudchenko, N. and G. W. Imbens (2016). Balancing, regression, difference-in-differences and synthetic control methods: A synthesis.

Ellerman, A. D., C. Marcantonini, and A. Zaklan (2016). The European Union Emissions Trading System: Ten Years and Counting. *Review of Environmental Economics and Policy 10*(1), 89–107.

Ellerman, A. D. and M. McGuinness (2008). CO2 Abatement in the UK Power Sector: Evidence from the EU ETS Trial Period. Working Paper.

European Commission (2001). Large Combustion Plants Directive (2001/80/EC).

Evans, S. (2015, Jan). Old coal and gas plants won largest share of capacity market, final results confirm | Carbon Brief. [Online; accessed 12. Jan. 2021].

Evans, S. (2019). Analysis: UK electricity generation in 2018 falls to lowest level since 1994.

Gobillon, L. and T. Magnac (2016). Regional Policy Evaluation: Interactive Fixed Effects and Synthetic Controls. *The Review of Economics and Statistics 98*(3), 535–551.

Goulder, L. H. and R. N. Stavins (2011, May). Challenges from State-Federal Interactions in US Climate Change Policy. *American Economic Review 101*(3), 253–257.

Green, J. F. (2021). Does carbon pricing reduce emissions? A review of ex-post analyses. *Environmental Research Letters*.

Gugler, K., A. Haxhimusa, and M. Liebensteiner (2020). Carbon Pricing and Emissions: Causal Effects of Britain's Carbon Tax.

Gugler, K., A. Haxhimusa, and M. Liebensteiner (2021, March). Effectiveness of climate policies: Carbon pricing vs. subsidizing renewables. *Journal of Environmental Economics and Management 106*, 102405.

Guo, B. and D. Newbery (2020, March). The Cost of Trade Distortion: Britain's Carbon Price Support and Cross-border Electricity Trade. Technical Report 2014, Faculty of Economics, University of Cambridge. Publication Title: Cambridge Working Papers in Economics.

Guo, B., D. M. Newbery, and G. Castagneto Gissey (2019). The Impact of Unilateral Carbon Taxes on Cross-Border Electricity Trading. Technical Report 1951, Faculty of Economics, University of Cambridge.

Hirst, D. (2018, January). Carbon Price Floor (CPF) and the price support mechanism. Technical Report Number 05927, House of Commons Library.

HM Revenue & Customs (2017). Excise Notice CCS 1/6: a guide to carbon price floor. Technical report.

IPCC (2014). Climate Change 2014: Impacts, Adaptation, and Vulnerability. Technical report.

IPCC, E. Somanathan, T. Sterner, T. Sugiyama, D. Chimanikire, N. K. Dubash, J. Essandoh-Yeddu, S. Fifita, L. Goulder, A. Jaffe, X. Labandeira, S. Managi, C. Mitchell, J. P. Montero, and F. Teng (2014). National and Sub-national Policies and Institutions. In *Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.*

Isaksen, E. T. (2020, September). Have international pollution protocols made a difference? *Journal of Environmental Economics and Management 103*, 102358.

Kim, M.-K. and T. Kim (2016). Estimating impact of regional greenhouse gas initiative on coal to gas switching using synthetic control methods. *Energy Economics 59*, 328 – 335.

Kong Chyong, C., B. Guo, and D. Newbery (2020, January). The Impact of a Carbon Tax on the CO2 Emissions Reduction of Wind. *The Energy Journal 41*(1).

Lee, K. and R. T. Melstrom (2018, October). Evidence of increased electricity influx following the regional greenhouse gas initiative. *Energy Economics 76*, 127–135.

Martin, R., L. B. de Preux, and U. J. Wagner (2014, September). The impact of a carbon tax on manufacturing: Evidence from microdata. *Journal of Public Economics 117*, 1–14.

Martin, R., M. Muûls, and U. J. Wagner (2016, January). The Impact of the European Union Emissions Trading Scheme on Regulated Firms: What Is the Evidence after Ten Years? *Review of Environmental Economics and Policy 10*(1), 129–148.

Metcalf, G. E. and J. H. Stock (2020, July). The Macroeconomic Impact of Europe's Carbon Taxes. Technical Report w27488, National Bureau of Economic Research.

Newbery, D., D. Reiner, and R. Ritz (2018, June). When is a carbon price floor desirable? Technical Report 1833, Faculty of Economics, University of Cambridge.

Newbery, D. M., D. M. Reiner, and R. A. Ritz (2019, January). The Political Economy of a Carbon Price Floor for Power Generation. *The Energy Journal 40*(1).

OFGEM (2013). Electricity interconnectors.

Perino, G. (2018, April). New EU ETS Phase 4 rules temporarily puncture waterbed. *Nature Climate Change 8*, 262–264.

Perino, G., R. A. Ritz, and A. v. Benthem (2019, March). Overlapping Climate Policies. Technical Report 25643, National Bureau of Economic Research, Inc. Publication Title: NBER Working Papers.

Rivers, N. and B. Schaufele (2015, November). Salience of carbon taxes in the gasoline market. *Journal of Environmental Economics and Management 74*, 23–36.

Rocha, M., B. Hare, P. Yanguas Parra, N. Roming, U. Ural, A. Ancygier, J. Cantzler, F. Sferra, H. Li, and M. Schaeffer (2016, November). Implications of the Paris Agreement for Coal Use in the Power Sector. Technical report, Climate Analytics.

Rubin, D. B. (1974). Estimating causal effects of treatments in randomized and nonrandomized studies. *Journal of educational Psychology 66*(5), 688.

Shearer, C., N. Mathew-Shah, L. Myllyvirta, A. Yu, and T. Nace (2019). Boom and bust 2019. tracking the global coal plant pipeline. pp. 14.

Singhal, P. (2019, May). Are Emission Performance Standards Effective in Pollution Control?

Evidence from the EU's Large Combustion Plant Directive. SSRN Scholarly Paper ID 3297528, Social Science Research Network, Rochester, NY.

Staffell, I. (2017, March). Measuring the progress and impacts of decarbonising British electricity. *Energy Policy 102*, 463–475.

Sustainable Energy Authority of Ireland (2014, May). Quantifying Ireland's Fuel and CO2 Emissions Savings from Renewable Electricity in 2012.pdf. Technical report.

Van den Bergh, K. and E. Delarue (2015, May). Quantifying CO2 abatement costs in the power sector. *Energy Policy 80*, 88–97.

Vaughan, A. (2018, January). UK government spells out plan to shut down coal plants. *The Guardian*.

Wilson, I. A. G. and I. Staffell (2018, May). Rapid fuel switching from coal to natural gas through effective carbon pricing. *Nature Energy 3*(5), 365–372. Number: 5 Publisher: Nature Publishing Group.

World Bank and Ecofys (2018). State and Trends of Carbon Pricing 2018.

A Appendix

A.1 Evolution of per capita emissions, demand, trade and emission intensity in the UK and other countries

Per capita power sector emissions : Figure A.1 shows the evolution of per capita power sector emissions for the UK and twenty other European countries[39], using emission data from the European carbon market described in section 3.2. While the UK was among the top emitters before 2013, by 2017 it had joined the bulk of lower-emitting countries. The figure also shows that most countries tend to have stable emissions per capita, except for a few outliers, which emissions are shown in dashed or dotted lines[40].

The CPS may impact each of the three channels mentioned in section 2.2: demand, because the CPS increases the marginal cost of producing electricity and generators are likely to at least partially pass on this cost to consumers, as evidenced in Guo et al. (2019) and Ares and Delebarre (2016). Trade, because the CPS increases the relative cost of domestically produced electricity compared to imported electricity. Emission intensity, because the CPS can directly impact the fuel mix used for electricity generation, in the short-term and in the long-term. In the short-term, the higher tax on coal relative to gas increases the cost of coal-fired relative to gas-fired power generation. In the long-term, the CPS also makes it less profitable to run fossil fuel high-emitting plants, and might dampen investments in those plants to the benefit of low-carbon generation (Van den Bergh and Delarue, 2015). In the next paragraphs, I show how demand, trade and emission intensity evolved over time in the UK compared to other European countries.

[39] all 28 EU countries except Romania, Bulgaria, Slovenia, Croatia, Malta, Cyprus, and Luxembourg, which are not included in the empirical analysis (see section 3.2)

[40] Estonia's emissions are both high on average and with a high variance; Czech Republic has the highest average after Estonia; Greece has decreasing emissions after 2012; Finland and Denmark's emissions have a high variance, likely due to the inter-annual variation in available hydro resources in Finland, and hydro and wind resources in Denmark.

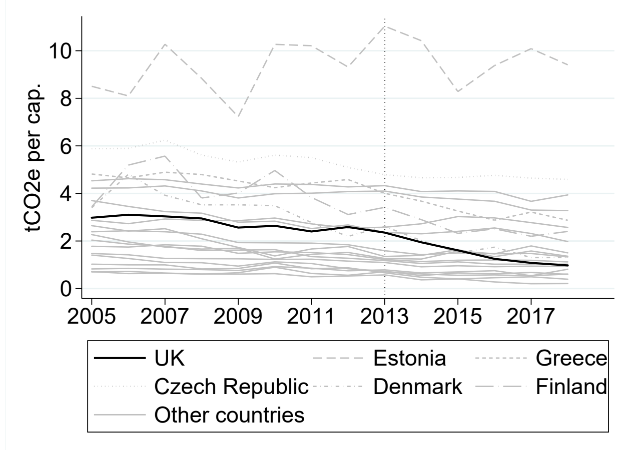


Figure A.1: Per capita power sector emissions in European countries

Notes: Emission values were obtained by aggregating plant-level emission data for ETS participants identified as power generators at the country level. Per capita emissions were obtained by dividing total emissions by the annual country population. "Other countries" include twenty European countries: Austria, Belgium, Czech Republic, Denmark, Estonia, Finland, France, Germany, Greece, Hungary, Ireland, Italy, Latvia, Lithuania, the Netherlands, Poland, Portugal, Slovakia, Spain, Sweden.

Demand, trade and intensity channels Figure A.2 shows the evolution of the three channels of demand, trade and emission intensity with the same method as Figure 2a. Demand is measured with power consumption per capita, trade is measured with net electricity imports per capita, and emission intensity is measured with emissions per gigawatt-hour of domestic power production. Demand decreases over the whole period in the UK compared to other European countries (figure A.2a). There is no obvious break in trend in 2013. This continuous decrease is consistent with the continuous improvement of energy efficiency in buildings and electric appliances in the UK since 2009 (Staffell, 2017). The lack of a visible link between the CPS and a change in demand can be linked to the finding by Chyong et al. (2019) that only about 60% of the CPS cost has been passed through to the GB day-ahead electricity market. It can also be explained by the financial compensation received by electro-intensive industries to cushion the price effect of the CPS and protect their competitiveness[41] (Hirst, 2018).

Regarding the trade channel, UK net imports per capita are low compared to other

[41]Electro-intensive industries have received a compensation of around £100 million for the period April 2013 to March 2015, and the support has been extended to 2019-2020. This compensation scheme is a specific component of a larger Energy Intensive Industries support introduced to compensate the cost increase induced by climate change policies (Hirst, 2018).

countries (figure A.2b). Being an island, the UK can only trade electricity via undersea cables and has a limited trading capacity. At the period of interest, the UK can trade electricity with three countries only: via undersea interconnectors with France, the Netherlands and Ireland from Great Britain, and via ground connections to the Republic of Ireland from Northern Ireland. Between 2010 and 2012, the UK increased its trading capacity by 50%[42], and UK net imports increased by from 2,661 GWh to 11,864 GWh. Net imports increased further between 2012 and 2015, but at a lower rate (from 11,864 GWh to 20,938 GWh), before decreasing again until 2017. Although imports started to increase before 2013, their increase after 2013 could be associated with the CPS. Using an econometric model of electricity trade, Guo and Newbery (2020) estimate that the CPS increased GB imports by 12,000 Gigawatt-hour per year between 2015 and 2018, after market coupling with France and the Netherlands in 2014. However, taken per capita and compared to other European countries, this increase in trade remains low: between the 2005-2012 and the 2013-2017 period, UK net imports per capita increase from representing 2% of gross electricity consumption to representing 5% of gross electricity consumption.

In contrast to demand and trade, the UK emission intensity of power generation decreased markedly after 2013 compared to most other European countries (Figure A.2c)[43].

The emission intensity $Q_{CO_2e}Q_{elec}$ can be further decomposed as:

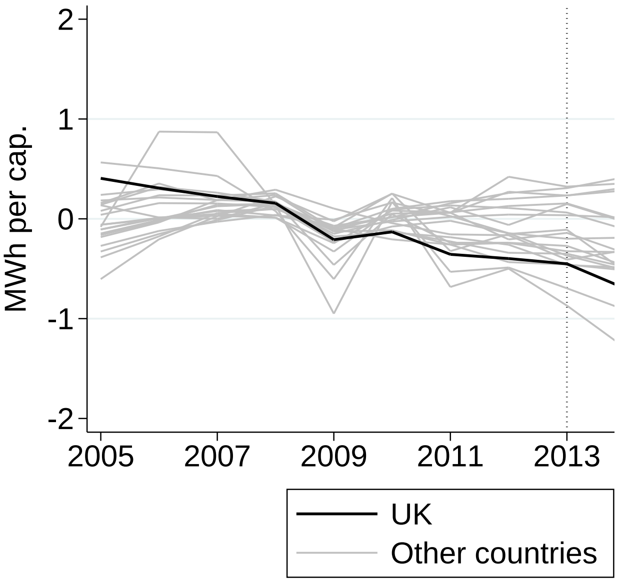
$$\frac{Q_{CO_2e}}{Q_{elec}} = \sum_i e_i q_i \tag{8}$$

Where e_i is the average emission intensity of technology i used for electricity generation and q_i represents the share of gross electricity production covered by technology i. Power generation with renewable and nuclear energy sources is emission-free[44]. What matters in

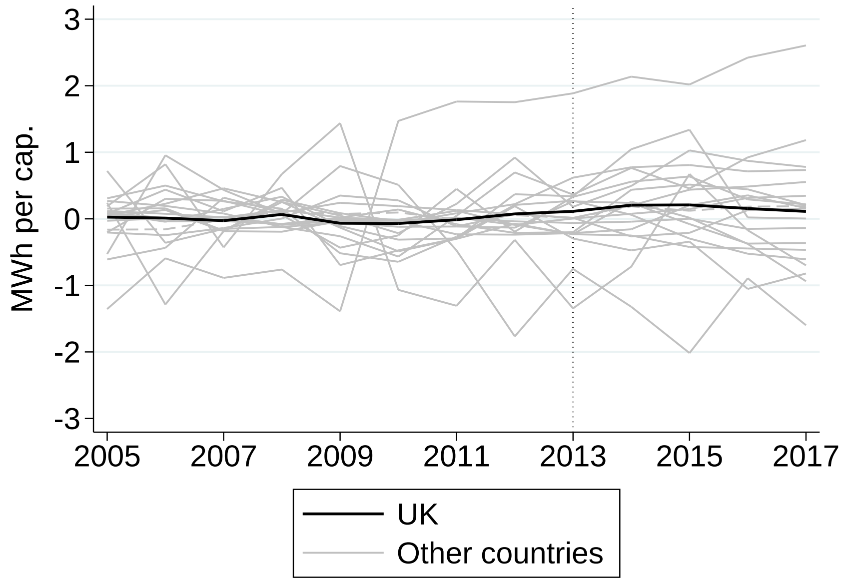
[42]GB became interconnected with the Netherlands in 2011, and in 2012 a new undersea interconnector with the Republic of Ireland was completed (OFGEM, 2013)

[43]The outlier with large variations in the emission intensity is Finland, again due to a large inter-annual variation in generation from renewables.

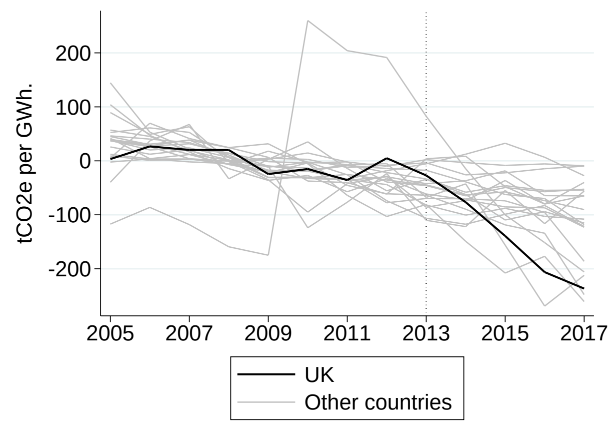
[44]these technologies embody some lie-cycle emissions, but generation itself does not emit CO_2. An exception is for plants using biomass: they do release greenhouse gases, but are not bound to pay the ETS price nor the CPS because the carbon released when solid biomass is burned is expected to be re-absorbed during tree



(a) Demand per capita



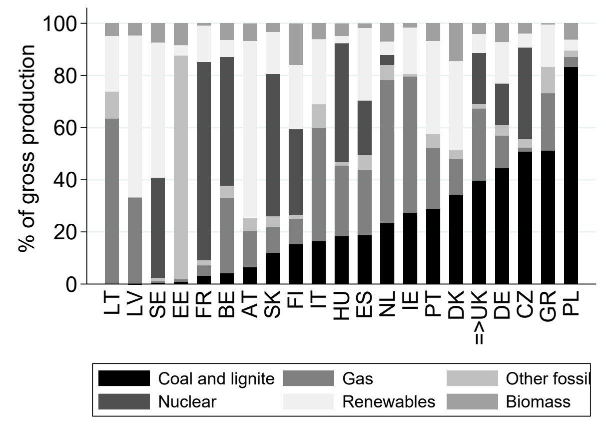
(b) Net imports per capita



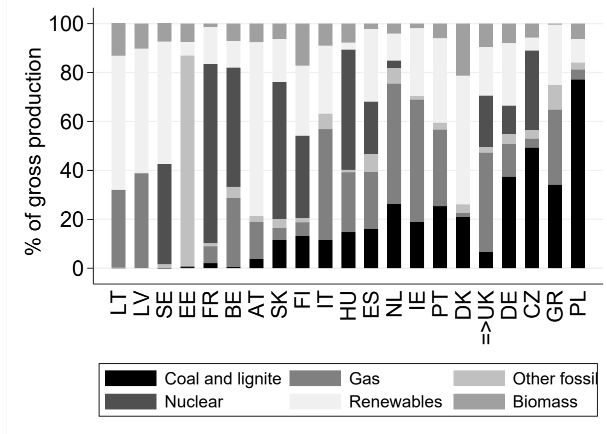
(c) Emissions per unit of electricity output

Figure A.2: Channels: evolution of electricity demand, trade and emission intensity in the UK and other European countries

Notes: The variables appearing on these two graphs were obtained by taking the difference between the original variable and the 2005-2012 average. “Other countries” include twenty European countries: Austria, Belgium, Czech Republic, Denmark, Estonia, Finland, France, Germany, Greece, Hungary, Ireland, Italy, Latvia, Lithuania, the Netherlands, Poland, Portugal, Slovakia, Spain, Sweden.



(a) 2012



(b) 2017

Figure A.3: Power sector's input fuel mix in EU countries, 2012 and 2017

Notes: EU countries are ranked by ascending order of the share of coal in electricity generation in 2012, from left to right. LT: Lithuania; LV: Latvia; SE: Sweden; EE: Estonia; FR: France; BE: Belgium; AT: Austria; SK: Slovakia; FI: Finland; IT: Italy; HU: Hungary; ES: Spain; NL: The Netherlands; IE: Ireland; PT: Portugal; DK: Denmark; UK: United Kingdom; DE: Germany; CZ: Czech Republic; GR: Greece; PL: Poland. The legend from left to right corresponds to the histogram bar colors from bottom (coal and lignite) to top (biomass). Data come from Eurostat. Renewables include production from hydro, solar, wind, and tide, wave and ocean.

this equation, therefore, is the share of fossil fuel in total electricity generation on the one hand, and the emission intensity of fossil fuel generation on the other hand. Figure A.3 shows the technologies/fuels used for power generation for each European country in 2012 and in 2017. The countries are ranked by their 2012 coal share. The UK coal share fell by 30 percentage points between the two periods, while there was only little variation in most other European countries. The decrease in the coal share was compensated by an increase in the gas share (+ 14 percentage points (pp)), in the share of non-biomass renewables (+ 9 pp) in the biomass share (+ 5 pp) and in the nuclear share (+ 2 pp).

A.2 Potential confounders

Here I give more details on policies implemented at the EU or UK level around the same time period, which may have contributed to an emission decrease

growth: see https://ec.europa.eu/clima/sites/clima/files/ets/docs/com_2018_842_final_en.pdf

European level: LCP and IED Directives Nine UK plants opted out from the LCP Directive and shut down between 2012 and 2015, and three other plants opted out partially (Source: EEA website). These 12 fully or partly opted out plants represented 11% of UK power sector emissions in 2011. The LCPD-induced plant closures could explain part of the pattern seen on figures A.2c and A.3 if the choice to opt-out and shut down occurred disproportionately more in the UK than in other European countries.

The LCPD was replaced by the IED directive in 2016. The IED Directive was enacted in 2010 and has a similar opt-out option as the LCPD. Plants had to decide by 2013 whether they wanted to opt-out or not under a limited lifetime derogation (LLD). Under the LLD opt-out option, plants are exempted from the emission standards but cannot operate for more than 17,500 hours between 1 January 2016 and 31 December 2023, and have to shut down once they have run for 17,500 hours or in 2023, whichever comes first. Two UK power plants opted out from the IED Directive. Given that plants had until 2013 to decide whether to opt-out or not and the CPS was announced in 2011, the IED opt-out decision was endogenous to the CPS.

UK level: support to biomass conversion The UK government supported the conversion of coal-fired power plants to biomass starting from 2012. This support first took the form of dedicated Renewables Obligation Certificates (ROCs). The ROCs were part of the broader Renewables Obligation scheme designed to support the deployment of large-scale renewable electricity generation; they created an obligation for electricity suppliers to source a proportion of their electricity from plants with ROCs. The ROCs were replaced by the FID Enabling for Renewables scheme in 2012 and then by the Contract for Difference scheme (CfD) introduced as part of the Electricity Market Reform in 2014. Two power plants received government support for conversion to biomass: Drax power station, representing 14% of UK power sector emissions in 2012, had already started to co-fire biomass in 2004. The company owning the station announced its intention to fully convert three of its six units to biomass in September 2012. The station benefited from Renewable Obligations Certificates for the conversion of its first unit, which was completed in 2013, and from Contracts for Differences for the conversion of its second and third units, which were completed respectively in 2014 and 2016. By the end of the period considered in this analysis, only three of the six units were converted to biomass, and the three remaining units continued to run with coal[45]. Lynemouth power station, a smaller plant, also received support under the FID Enabling for Renewables scheme. The station stopped burning coal and started the biomass conversion process in December 2015[46].

The biomass conversion of these two plants, partial but early in the case of Drax, and full but later in the case of Lynemouth, led to a decline in their carbon emissions over the 2013-2017 period, since biomass is considered a zero-emission fuel. The dates of introduction of the CPS and of the support policies for biomass conversion are close. The UK government may have decided to subsidise conversion from coal to biomass partly to reduce the economic costs associated with the CPS for coal plant owners, and to facilitate the low-carbon transition. In this case, the biomass conversion could be viewed as a direct consequence of the CPS. But the two policies may also be independent, in which case the biomass conversion would have occurred even in the absence of the CPS.

UK level: Support to renewable energy Second, The FID Enabling for Renewables and CfD programmes could have impacted the fuel mix more broadly than via its impact on the conversion of coal plants to biomass, by increasing the share of renewable energy in UK electricity production over the 2013-2017 period. I combine data on projects being awarded a Contract under the FID Enabling for Renewables or CfD programmes between 2013 and 2017[47] and the renewable energy planning database put together by the UK Department for Business, Energy and Industrial Strategy (BEIS) monitoring all UK-based renewable

[45] https://www.drax.com/about-us/our-history/

[46] https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/805441/LCP_Review_Lynemouth_DD-FP3137CG-V009-draftdecision.pdf

[47] available at: https://www.gov.uk/government/publications/contracts-for-difference/contract-for-difference

projects[48] to estimate the renewable capacity which was installed over the 2013-2017 period as a direct consequence of FID of the CfDs. Compared to the planned delivery year of projects mentioned in the CfD auction results, the actual date where a CfD project becomes operational is often delayed by several months. Given the actual date where each project becomes "operational" according to BEIS's renewable energy planning database, only 1,035 MW of renewable capacity from CfD projects is operational by the end of 2017. This represents 2.4% of the total renewable capacity installed in the UK in 2017[49]. I combine data on the date where each project becomes operational with data on running hours for each project (coming from the CfD auction results) to calculate the electricity generation associated with this installed capacity over the 2013-2017 period. I estimate that at most 1,588 GWh of electricity may have been generated by CfD projects - assuming that the theoretical number of hours given in the CfD auction results is the true number of hours. This represents just 0.4% of the total electricity generated with renewable energy sources[50] and 0.1% of total electricity generated in the UK over that period. Abrell et al. (2019) report that the average capacity-weighted emission rate of UK coal-fired plants (taken before 2013) is 0.89 ton of CO_2 per Megawatt-hour of electricity produced. Such an emission rate implies that the 1,588 GWh of electricity produced by CfD projects would have caused emissions of 1.4 MtCO_2e[51] over the 2013-2017 period if they had been generated by coal instead. This amount represents less than 1% of UK power sector emissions in 2013.

Other support policies to renewable energy exist in the UK, but the bundle of feed-in-tariffs, support to R&D for renewable energy, and regulatory instruments does not look fundamentally different from that implemented in other European countries, according to the IEA/IRENA Joint Policies and Measures Database listing the support policies implemented

[48] available at: https://www.gov.uk/government/publications/renewable-energy-planning-database-monthly-ex

[49] The denominator includes waste and biofuels, since part of the CfD projects are for power generation from waste

[50] The denominator includes waste and biofuels, since part of the CfD projects are for power generation from waste

[51] $1,589.10^3 \times 0.89 = 1,414,210$ tCO_2e

in each country since the 1970s[52]. All in all, the difference in emissions observed after 2013 in the UK does not seem driven by a renewable energy policy specific to the UK, apart from the support to biomass conversion described above.

UK level: capacity market The capacity market introduced in 2013 may impact emissions in two ways: first, before the capacity payment starts, securing a capacity contract can incentivize investing in new capacity, as auction payments can be seen as a subsidy for new power generation. Indeed, the capacity market was initially supposed to facilitate investments in new gas capacity (Evans, 2015). Second, once the capacity payment starts, the payment can keep a plant being economically profitable even with at low generation levels. The first auction took place in 2014 for capacity secured for 2018. Since my 2013-2017 period of analysis is before the auction payments start, the capacity market can only impact UK emissions via the first channel. I combine data on new-build plants being awarded a contract between 2014 and 2017 [53] and data listing all UK power plants with a capacity greater than 20MW with the year of commission or year generation began [54], to gauge if the capacity market incentivized the construction of plants having a lower-than-average emission intensity over the 2013-2017 period. Only six plants were awarded a capacity contract in 2014, 2015, 2016 or 2017 and had a date of commissioning/where generation began between 2014 and 2017. One is a large gas-fired plant (CCGT), Carrington power station and the five other are smaller waste plants. The opening of Carrington power station cannot be imputed to the capacity market because the plant started being constructed in 2009[55]. On the other hand, the five waste plants may have opened as a direct consequence of being awarded a capacity contract, and have a lower-than-average emission intensity. These five plants represent an

[52]The database can be accessed here: https://www.iea.org/policiesandmeasures/renewableenergy/.For the electricity sector only, there were more than 200 support measures in force in the 28 EU countries over the period considered. Examining each piece of legislation goes beyond the scope of this article.

[53]available at: https://www.emrdeliverybody.com/CM/Registers.aspx

[54]available at: https://www.gov.uk/government/statistics/electricity-chapter-5-digest-of-united-kingdom-energy-statistics-dukes, DUKES 5.11 file

[55]https://en.wikipedia.org/wiki/Carrington_Power_Station

installed capacity of 1,141 MW. To estimate the associated power generation, I make several assumptions: first, I assume that the year of commissioning/where generation began indicated in the list of UK power plants is when generation began, and that generation began on January 1. Second, I take as a load factor the average load factor for conventional steam plants in the UK averaged over 2013-2017[56], which is 35%. I obtain an upper bound of the low-carbon power generation imputable to the capacity market of 2,590 MWh over the 2013-2017 period. This represents 0.6% of electricity generated with renewable sources and 0.2% of total electricity generated in the UK over the 2013-2017 period. Such generation would have caused CO_2e emissions of 2.3 $MtCO_2e$[57] over the 2013-2017 period if it had been generated with coal instead of renewable waste (similar calculations as the one used to estimate the emission reduction caused by the CfD). This amount represents just 1.5% of UK power sector emissions in 2013.

A.3 Identification of power installations

The plant-level emission data released by the EUTL does not provide information on which plant is a power generator. The UK-based think-tank Ember (formerly Sandbag) provided a database with total verified emissions data for 2008-2016 supplemented with a variable identifying all power plants. This identification has been performed internally by Ember in two steps: in the first step, Ember carried out an exact matching based on a file circulated by the European Commission in 2014 containing a list of individual participants with their sectoral classification. This classification is based on NACE rev2, the Statistical classification of economic activities in the European Community, which contains two-digit *divisions*, divided into three-digit *groups*, themselves divided into four-digit *classes*. Power installations are generally found in division 35 "Electricity, gas, steam and air conditioning supply", group 35.1 "Electric power generation, transmission and distribution", class 35.11

[56] data available at: https://www.gov.uk/government/statistics/electricity-chapter-5-digest-of-united-kingdom-energy-statistics-dukes, DUKES 5.10 file)

[57] $2,590.10^3 \times 0.89 = 2,305,100 \text{ tCO}_2\text{e}$

"Production of electricity". Ember classified as a power installation all the ETS participants with class 35.11. In the second step, Ember identified other power installations which either were not classified in class 35.11 (for example because they were part of an industrial site), or opened after the file was circulated by the European Commission, based on desk-based research and manual matching. For the verified emissions variable, the data provided by Ember are the same as the raw data retrieved from the EUTL.

To retrieve the power plants status of the few plants that shut down before 2008 (and are thus absent from the Ember dataset), I use the "Accounts to Firms Matching" dataset hosted by the Florence School of Regulation (FSR)[58], listing participating installations until 2013 with their Nace rev 2 sectoral classification. I first match the FSR and Ember data to check the quality of Ember's power sector classification. Among the installations found in both the FSR and Ember data, 100% of the division 35 installations having sectoral class 35.11 (according to the FSR data) are classified as power installations by Ember. 96% of the division 35 installations with a sectoral class different from 35.11 are also classified as power installations by Ember. Installations from sectoral class 35.11 represent only 30% of the installations classified by Ember as power installations, but 80% of the carbon emissions. Installations from division 35 but with a different class from 35.11 represent 61% of Ember's power installations, but only 7% of emissions. Installations from division 35 but with a missing division or a missing class represent another 6% of Ember's power installations, and 11% of emissions. The remaining 3% of Ember's power installations, representing just 1% of emissions, have a division that is either different from 35, or missing.

To be consistent with Ember's classification, I classify the few plants only present in the FSR but not in the Ember dataset as power installation when their group is 35. This way, I identify 314 additional power plants which shut down before 2008. Finally, I identify 4 additional installations having a missing sectoral division as power installation based on their name (containing "power station" or its equivalent in one of the European languages).

[58]The dataset can be downloaded on this website: http://fsr.eui.eu/climate/ownership-links-enhanced-eutl-dataset-project/

There are no ETS installations opening in 2017, that is, present in the 2017 EUTL data but not in the 2008-2016 Ember data. After this additional matching, the power plant status is missing for only 3% of all EU ETS installations over the 2005-2017 period, with only a quarter of them having non-zero CO_2e emissions for at least one period.

A.4 Summary statistics for the country-level dataset

Table A.1: Summary statistics at the country level, average 2005-2012

	(1) UK	(2) Other countries	(3) Donor pool
Nb. ETS power installations	190	190	193
	(13)	(189)	(148)
Nb. ETS non-power installations	626	261	273
	(82)	(269)	(229)
Population	62,047,417	20,142,519	20,229,390
	(1,165,199)	(23,321,330)	(20,035,738)
CO_2e emissions from power installations (tCO_2e)	171,770,195	53,067,594	43,904,503
	(13,562,672)	(78,052,009)	(47,414,350)
Per capita power sector emissions (tCO_2e per capita)	2.78	2.80	2.43
	(0.26)	(2.10)	(1.48)
Final electricity consumption (GWh)	341,442	122,819	122,593
	(13,197)	(147,040)	(119,366)
Gross Power production (GWh)	384,010	139,907	139,013
	(13,570)	(173,760)	(143,684)
Proportion of renewables in production	0.05	0.19	0.19
	(0.0183)	(0.184)	(0.175)
Proportion of nuclear	0.18	0.21	0.24
	(0.0229)	(0.250)	(0.249)
Proportion of fossil fuel	0.776	0.596	0.574
	(0.0316)	(0.271)	(0.271)
Proportion of coal	0.327	0.226	0.235
	(0.0430)	(0.232)	(0.227)
Proportion of gas	0.398	0.230	0.235
	(0.0613)	(0.185)	(0.191)
Coal price (€/kWh)	0.0101	0.0152	0.0170
	(0.00209)	(0.0166)	(0.0188)
Gas price (€/kWh)	0.0201	0.0254	0.0258
	(0.00420)	(0.00572)	(0.00527)
Installed capacity, fossil fuels (MW)	68,414	18,838	18,522
	(2,869)	(22,481)	(18,353)
Installed capacity, wind and solar (MW)	4,732	4,480	3,363
	(3,156)	(9,815)	(5,877)
Lignite resource dummy	0.000	0.250	0.200
	(0.000)	(0.434)	(0.402)
Average age of coal-fired plants	36.04	28.57	28.86
	(1.537)	(6.698)	(7.055)
Observations	8	160	120

Notes: Standard deviations in parentheses;“Other countries” include twenty European countries: Austria, Belgium, Czech Republic, Denmark, Estonia, Finland, France, Germany, Greece, Hungary, Ireland, Italy, Latvia, Lithuania, the Netherlands, Poland, Portugal, Slovakia, Spain, Sweden; The “Donor pool” includes fifteen countries: Austria, Belgium, Czech Republic, Denmark, Finland, France, Hungary, Ireland, Italy, the Netherlands, Poland, Portugal, Slovakia, Spain, Sweden; for the UK, the values are averaged over the 2005-2012 period; for the “Other countries” and the “Donor pool”, the values are averaged over the 2005-2012 period, then averaged across countries (without population weights).

A.5 Data sources

Coal-to-gas price ratio (predictor used in the main specification): To my knowledge, there is no harmonized series of country-level coal-to-gas price ratio for the period considered. I thus build a price ratio variable combining coal trade data from Eurostat and gas wholesale price data for large industrial consumers, also from Eurostat. For coal, I use annual trade data for imported coal from Comext, the official EU trade statistics. I aggregate the price and volume data for all the subcategories of coal that may be used for coal generation and obtain average nominal unit prices for imported coal. I fill the few data gaps by applying the growth rates from the closest non-missing data source, the IEA nominal coal price index for industry. The obtained coal price series compare well with the IEA price series in the electricity generation sector, for the few countries where both data are available.

For gas, I use Eurostat data on wholesale gas prices (excluding VAT and other recoverable taxes and levies) for the second largest consumption band of industrial consumers. This band corresponds to the average consumption of large gas-fired power plants as reported in the European Environmental Agency's Large combustion plant database. I fill the few data gaps by imputing values from the third largest consumption band, or, if it is also missing, from the IEA gas price data. One drawback of this data source is that the consumption band categories and the methodology changed in 2007, which makes it difficult to build a consistent series of coal/gas price ratio before 2007. For this reason, I average the coal-to-gas price ratio predictor over the 2007-2012 period only (rather than the 2005-2012 period) when I apply the synthetic control method. I convert the obtained coal and gas price series to the same unit and combine them to build coal-to-gas price ratio for all European countries over the 2007-2016 period. The obtained price ratios compare well with the price ratios available from national statistical institutes for some countries[59].

[59]For example, in the UK the Department for Business, Energy and Industrial Strategy publishes such data each quarter

Lignite resources (predictor used in the main specification): Data on lignite resources in Europe come from the industry association Euracoal (European Association for Coal and Lignite; Source: https://euracoal.eu/info/euracoal-eu-statistics/) I create an indicator variable equal to 1 for countries with lignite resources greater than 0.5 Gt in 2012, and 0 otherwise. The variable is equal to one for Germany, Poland, Hungary, Greece, Czech Republic, and Bulgaria.

Residual load per capita (predictor used in the main specification): Residual load is defined as the difference between electrical energy available for final consumption taken from Eurostat, and generation from renewables and nuclear power plants, also derived from Eurostat. Generation from renewables is the sum of total net electricity production from the five renewable sources hydro, tide wave and ocean, solar PV, solar thermal and wind[60]. Generation from nuclear power plants is the sum of total net electricity production from nuclear power plants, including conventional plants, auto producers, and co-generation plants. This variable is then divided by the average population by country given by Eurostat.

Emissions from LCP opt-out plants in 2009 and IED opt-out plants in 2012 (predictor used in the main specification): The list of LCP and IED opt-out plants is available on the European Environmental Agency's website[61]. Since there is no common identifier between the EUTL and LCP and IED data, I manually matched the 172 LCP opt-out installations and the 140 opt-out installations under the LLD option located in the UK or in a country from the donor pool to the EUTL emission data (using information on the plant name and location). No match was found for two LCP opt-out plants, one from Finland and one from Poland, and for six IED opt-out plants, one from Poland, one from Slovakia, one from Czech Republic and one from the UK (but based on these installations' names, they are unlikely to be power installations except the installation from Poland). The

[60] geothermal, biomass and waste are not included since they are available on demand

[61] Source: `https://www.eea.europa.eu/data-and-maps/data/large-combustion-plants-lcp-opt-out-under-artic` for LCP opt-out and `https://www.eea.europa.eu/data-and-maps/data/lcp-9` for IED opt-out plants

LCP emission variable is obtained by aggregating CO_2e emissions from the LCP opt-out power plants at the country-level. The IED emission variable is obtained by aggregating CO_2e emissions from the IED opt-out power plants at the country-level.

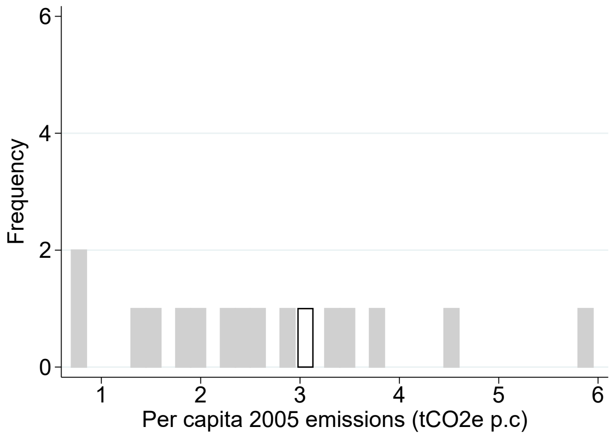
Number of heating degree days (predictor used in the sensitivity analysis): Eurostat series "cooling and heating degree days by country - annual data".

Per capita capacity for combustible fuels, gas and coal (predictor used in the sensitivity analysis): The variable is derived from Eurostat data on electricity production capacities for combustible fuels by technology and operator. I aggregate all technologies and operators to get the total installed capacity.

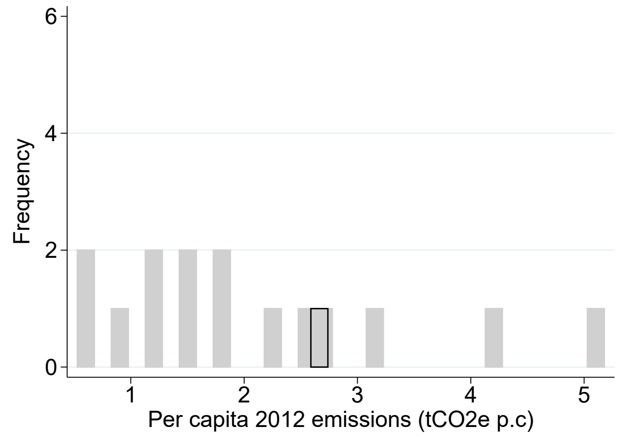
Growth in per capita renewables capacity (predictor used in the sensitivity analysis): The variable is derived from Eurostat data on net electrical maximum capacity by renewable technology. I add up capacities for wind, solar, tide wave and ocean, hydro and geothermal and calculate the average annual growth rate between 2010 and 2012. 2010 is the year where the Europe 2020 strategy was adopted (including the target of increasing the share of renewable energy in final energy consumption to 20% by 2020), and 2012 is the last year before the introduction of the CPS.

Average age of operating coal-fired plants above 30 MWth (predictor used in the sensitivity analysis): The variable is derived from the Global Energy Monitor's "Global Coal Plant Tracker" (Shearer et al., 2019), a publicly available database categorizing every known coal-fired generating unit with a rated capacity above 30 MWth. I use information on the status of the unit (operating/retired/mothballed) and its commissioning date to build a country-level variable of the coal fleet's age, defined as the average capacity-weighted age of the coal-fired power plants operating every year.

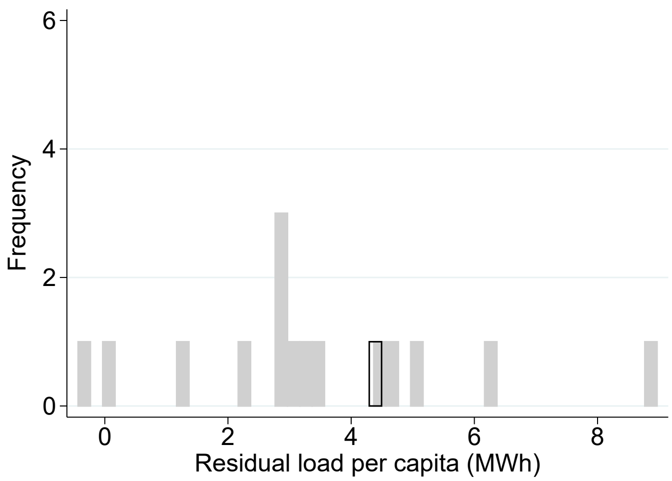
A.6 Common Support for predictors



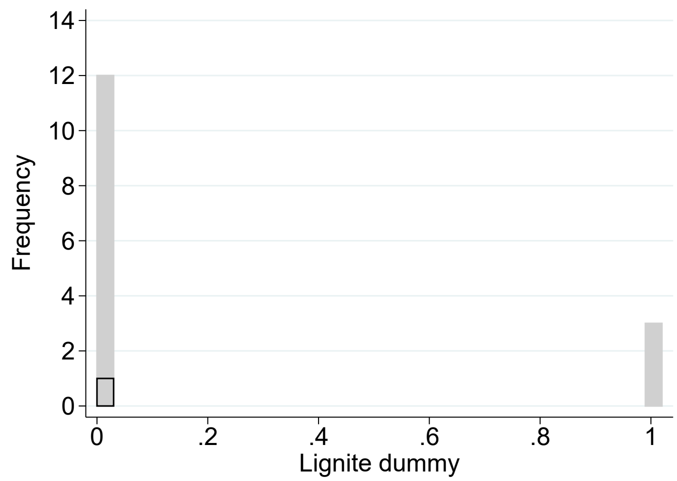
(a) Per capita CO_2e emissions (2005)



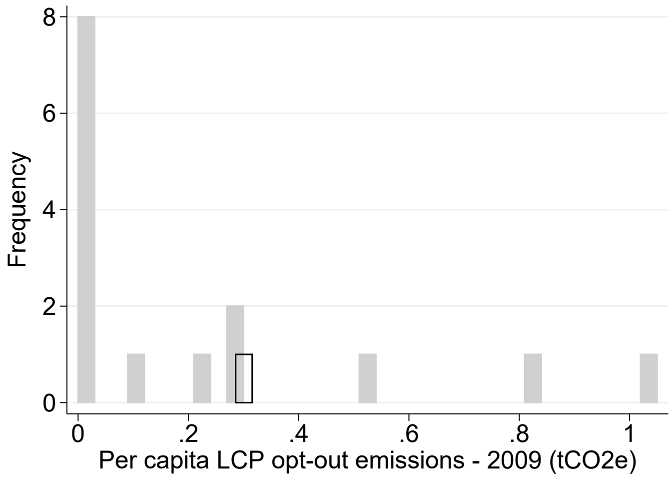
(b) Per capita CO_2e emissions (2012)



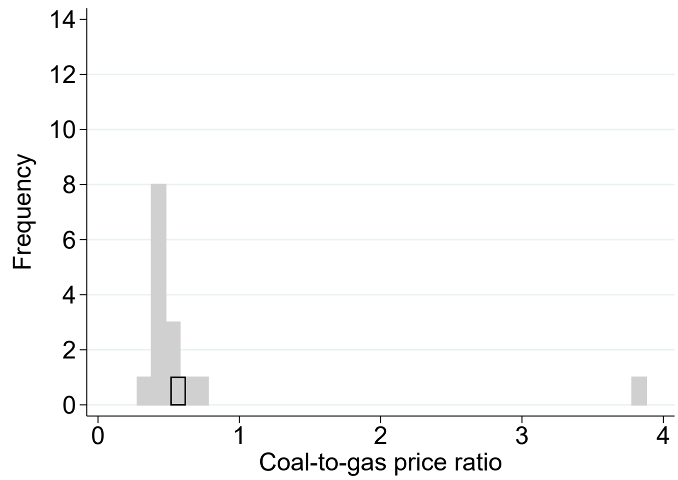
(c) Residual load per capita



(d) Lignite dummy



(e) LCP opt-out emissions per capita (2009)



(f) Coal-to-gas price ratio (2007-2012)

Figure A.4: Histograms of main predictors. UK: black with transparent fill; donor pool: grey fill.

Notes: Unless otherwise specified, all variables are averaged for the 2005-12 period

A.7 Counterfactual emissions in the absence of biomass conversion for Drax and Lynemouth plants

Table A.2: Counterfactual CO_2 emissions in the absence of biomass conversion, Drax

	(1) Total CO_2e	(2) Generation coal units (MWh)	(3) Estimated CO_2e coal units	(4) Estimated CO_2e biomass units w/o conversion	(5) Estimated CO_2e total w/o conversion (3)+(4)
2005	20,771,624	-	-	-	20,771,624
2006	22,764,847	-	-	-	22,764,847
2007	22,160,413	-	-	-	22,160,413
2008	22,299,778	-	-	-	22,299,778
2009	19,851,702	11,584,366	10,425,929	9,425,773	19,851,702
2010	22,392,487	13,537,600	12,183,840	10,208,647	22,392,487
2011	21,465,607	15,093,899	13,584,509	7,881,098	21,465,607
2012	22,694,684	14,592,305	13,133,075	9 561 609	22,694,684
2013	20,319,513	14,398,937	12,959,044	9 434 905	22,393,948
2014	16,595,193	13,364,881	12,028,393	8 757 339	20,785,732
2015	13,192,780	13,808,137	12,427,324	9 047 783	21,475,107
2016	6,261,692	7,120,958	6,408,862	4,666,008	11,074,871
2017	6,215,220	-	6,215,220*	4,525,026	10,740,246

Notes: column (1): emission data come from the EUTL. Column (2): generation data for the coal part over the 2009-2016 period come from Abrell et al. Column (3): emissions for the coal units over the 2009-2016 period were estimated by applying an average emission factor of 0.9 tCO_2/MWh, the average emission rate reported by Abrell et al for Drax plant. *Emissions for the coal units in 2017 are estimated to be the same as emissions for the entire plant as reported in column (1), since all biomass units are fully converted by then and emit zero CO_2. Column (4): emissions for the biomass units if they had not converted to biomass are estimated differently for the 2009-2012 and for the 2013-2017 period: for the 2009-2012 period, the units had not yet converted to biomass, so their emissions are simply the difference between EUTL emissions and emissions estimated for the coal units (column (1) - column (3)). For the 2013-2017 period, the units had started to convert to biomass, as reflected in the total EUTL data. Emissions if those units had not converted to biomass are estimated assuming that they would have followed the same evolution as emissions from the coal units: I multiply the estimated emissions for the coal units each year (column (3)) by the ratio of emissions from the biomass units/emissions from the coal units in 2012 (column (4)/column (3): 9,561,609/13,133,075=0.73). Column (5): total emissions in the absence of biomass conversion are simply the sum of estimated emissions for the coal units and estimated emissions for the biomass units if the had not converted to biomass (column (3) + column (4)).

Table A.3: Counterfactual CO_2 emissions in the absence of biomass conversion, Lynemouth

	(1) Total CO_2e	(2) Estimated CO_2e total w/o conversion
2005	2,685,512	2,685,512
2006	2,693,932	2,693,932
2007	2,695,748	2,695,748
2008	2,802,040	2,802,040
2009	2,543,017	2,543,017
2010	2,551,364	2,551,364
2011	2,612,450	2,612,450
2012	2,050,363	2,050,363
2013	2,284,177	2,284,177
2014	2,717,964	2,717,964
2015	1,287,305	1,287,305
2016	1,059	1,287,305 *
2017	2,421	1,287,305 *

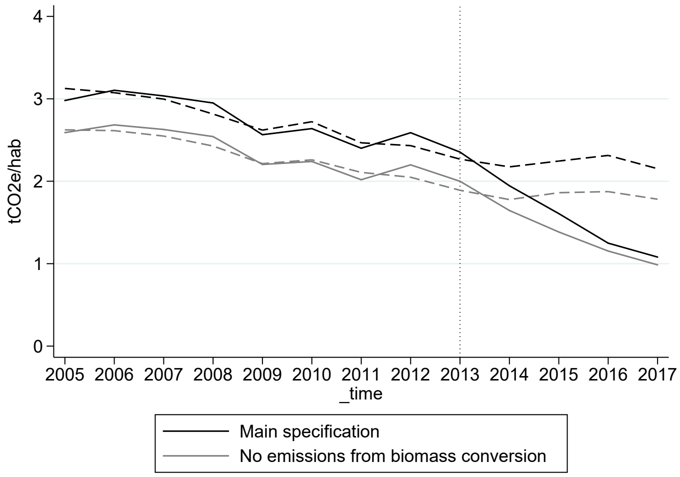
Notes: Column (1): emission data come from the EUTL. Column (2): same as column (1), except in 2016 and 2017. *It is assumed that in absence of biomass conversion, Lynemouth 2016 and 2017 emissions would have been the same as in 2015.

A.8 Lower bound removing emissions from plants converted to biomass

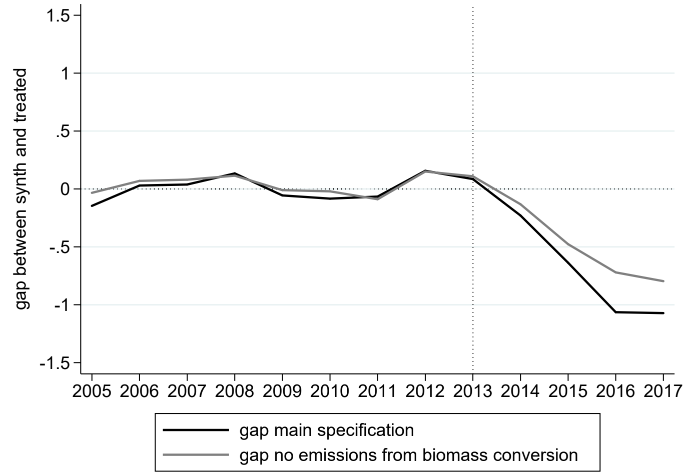
I estimate a second, more conservative lower bound of the impact, where I consider that 100% of the two plants convert to biomass and that the CPS has no separate effect. I apply the synthetic control method on a modified outcome variable, where emissions from UK plants having partly or fully converted to biomass are removed from the UK emissions for the entire period of analysis. This is conservative given that the emission decrease from Drax units not converted to biomass and the emission decrease from Lynemouth plant if it had only been subject to the CPS are not taken into account. The new per capita emission variable is 15% lower for the UK after removing emissions from these plants, while emissions from countries in the donor pool stay the same. I run the Synthetic control method again based on the modified UK emission variable. With this modified outcome variable, it becomes harder to accurately build a synthetic UK using the initial set of predictors. I therefore add a third predictor of lagged outcome for the year 2010. Figure A.5 shows the original UK and synthetic UK emission trajectories (in black) and the UK and synthetic UK emission trajectories for the modified outcome variable (dark grey).

The new synthetic UK is made of seven countries: Italy (35.9%), Ireland (23.5%), Slovakia (21.3%), the Netherlands (15%), Finland (3%), Poland (0.4%), and Denmark (0.9%). Country weights are shown in table A.4. Table A.5 shows that predictors' values are still closely aligned across the actual and synthetic UK, except for the residual load per capita - it makes sense that it is lower for the synthetic UK than for the UK, since UK emissions from fossil plants - which typically cover the residual load - have been made artificially lower than they truly are by removing the two plants converted to biomass.

The gap between the UK and synthetic UK per capita emissions is smaller than with the original outcome variable, which is expected. On an average year, emissions decrease by 18.1%. The total cumulative abatement is 132 million of tCO_2e. The difference between the total abatement from this lower bound and the one from section **??** is 164-132=32 $MtCO_2e$, which is close to the hypothesised emission decrease for Drax and Lynemouth if all their units had continued to run with coal, estimated



(a) Absolute per capita emissions



(b) Emission gap between treated and synthetic

Figure A.5: Synthetic control method excluding emissions from plants having converted to biomass

Notes: The synthetic UK is made of the following countries for each specification: Main specification: five countries: Ireland (49.2%), Slovakia (25.6%), the Netherlands (13.7%), Finland(5.8%), Czech Republic (5.7%). Lower Bound synthetic UK without emissions from plants converted to biomass: seven countries: Italy (35.9%), Ireland (23.5%), Slovakia (21.3%), the Netherlands (15%), Finland (3%), Denmark (0.9%) and Poland (0.4%).

Table A.4: Country weights in Lower Bound Synthetic UK

Country	**Weight**	**Country**	**Weight**
Austria	0	Ireland	0.235
Belgium	0	Italy	0.359
Czech Republic	0	Netherlands	0.15
Denmark	0.009	Poland	0.004
Spain	0	Portugal	0
Finland	0.03	Sweden	0
France	0	Slovakia	0.213
Hungary	0		

Note: All weights are between 0 and 1 because the Synthetic control method imposes positive weights summing to 1.

Table A.5: Predictors' values for the UK, synthetic UK and average of the donor pool, lower bound

Variable	**UK**	**Synth. UK**	**Avg. Donor pool**
Per capita residual load	4.29	4.27	3.37
Coal-gas price ratio	0.52	0.49	0.71
Per capita LCP opt-out emissions	0.29	0.29	0.22
Lignite dummy	0.00	0.004	0.20
Per capita emissions 2005	2.59	2.62	2.62
Per capita emissions 2010	2.24	2.26	2.39
Per capita emissions 2012	2.20	2.05	2.05

Notes: The per capita residual load is averaged for the period 2005-12, and the coal-to-gas price ratio for the period 2007-12. LCP opt-out emissions are taken in 2009, the lignite dummy is time-invariant. Outcome lags are taken in 2005, 2010 and 2012.

A.9 Sensitivity of the results to the choice of predictors

I test the sensitivity of the results from the upper bound to using four alternative sets of predictors to generate a synthetic UK. Note that the results would be the same if applied on the lower bound from part 4.2, since the composition of the synthetic UK is the same. In the first alternative set of predictors, I replace the per capita residual load with the annual number of heating degree days, a variable approximating the demand for energy needed for heating, likely to capture variations in peak power demand mostly covered by fossil fuels[62]. In the second alternative set of predictors, I add as predictor a measure of installed capacity for combustible fuels. This variable may influence the potential for fuel switching. Ideally, one would like to add specific variables for coal installed capacity and gas installed capacity, but these variables are not publicly available for all countries in the donor pool.

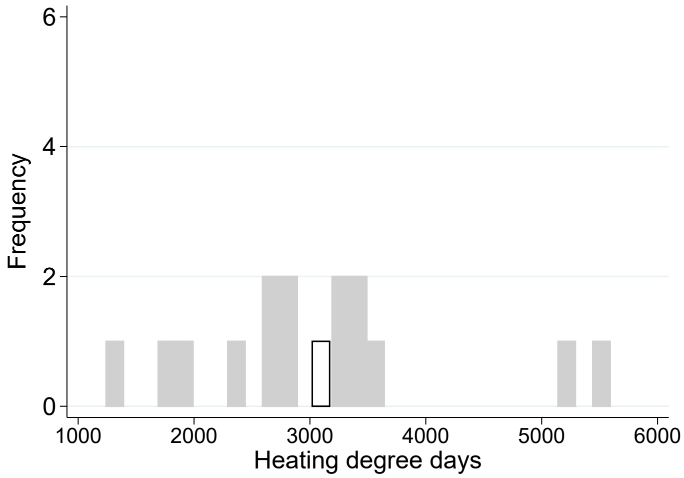
In the third alternative set of predictors, I add a predictor reflecting the growth in renewable capacity just before the introduction of the CPS. This variable can be considered as a proxy for the "business as usual" growth in renewables' capacity, which would occur absent the CPS policy. I average this variable between 2010 and 2012. 2010 is the year of implementation of the Europe 2020 strategy, which sets a target for the share of renewables in final energy consumption to be reached by 2020 for EU countries. This announcement may be followed by a growth in renewable capacity in all European countries. 2012 is the last year of pre-treatment, which makes sure that the growth in renewable capacity is not affected by the CPS.

In the last alternative set of predictors, I add a predictor reflecting the age of the fleet of coal plants for each country. Newer coal-fired plants tend to be more efficient and produce less emissions per output of electricity, so we may expect the average age of coal power plants to influence a country's emissions. Only plants with a capacity above 30 MWth are included

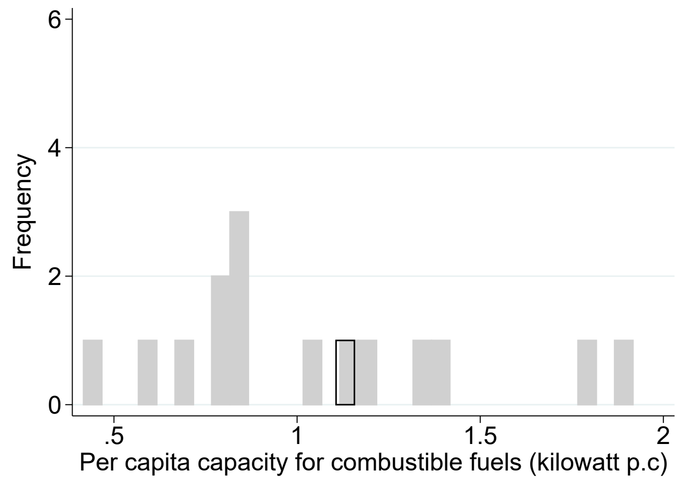
[62]In the EU, the number of heating degree days is measured as the number of days of the year where the average temperature is below a reference temperature of 15.5°C - under which energy for heating is needed - times the difference between this reference temperature and the temperature of the day. Compared with the average annual temperature used in other papers for predicting power demand, this variable better captures demand for power generation at low-temperature periods.

in the calculation of the average age of coal plants at the country level (see appendix A.5) - but we expect these plants to be responsible for most of coal-based power generation.

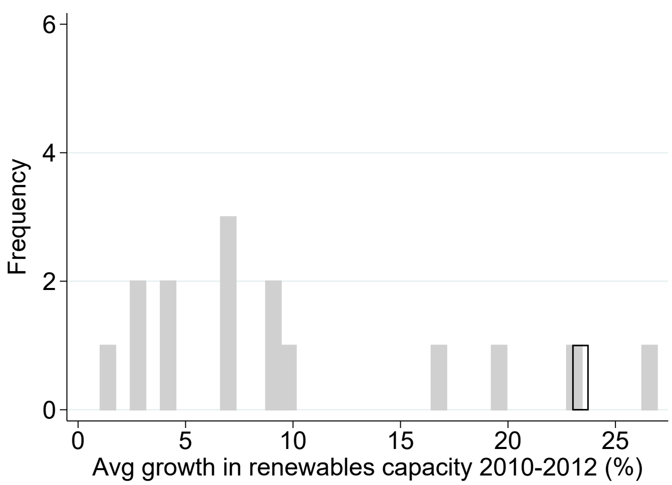
Figure A.6 shows the common support for these four predictors. For each predictor, the UK value falls within the distribution of other countries' values. Figure A.7 shows the emission paths for the different Synthetic UKs obtained with these alternative sets of predictors. The country weights are indicated below the figure for each Synthetic UK. The emission path of each new synthetic UK is relatively close to the original one. The fit is less good for the specification with the number of degree days and the one with the average age of coal-fired plants. Table A.6 shows the predictors' values for the UK, for the original Synthetic UK, and for the four alternative Synthetic UKs. The alternative predictors' values are close in the UK and in the different synthetic UKs. For the synthetic UK using the growth in renewable capacity as predictor (Alt. 3) and the synthetic UK using the average age of coal-fired plants (Alt. 4), having close values for the new predictor comes at the expense of a poorer predictor balance for some of the other predictors (lignite dummy and per capita opt-out emissions for Alt.3; lignite dummy and residual load per capita for Alt.4).



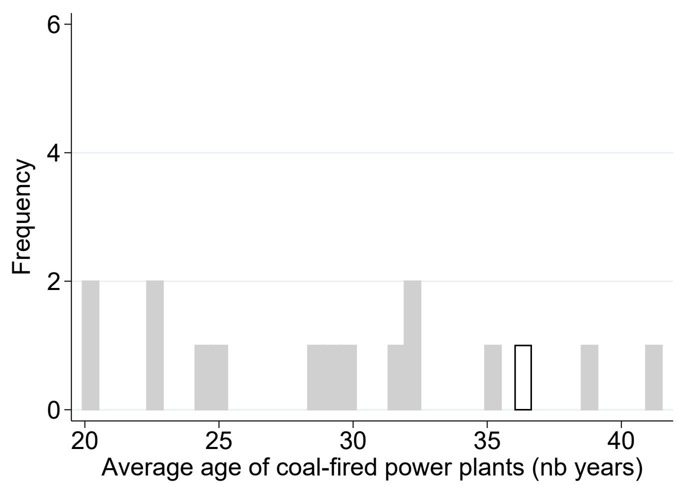
(a) Number of heating degree days



(b) Per capita capacity for combustible fuels



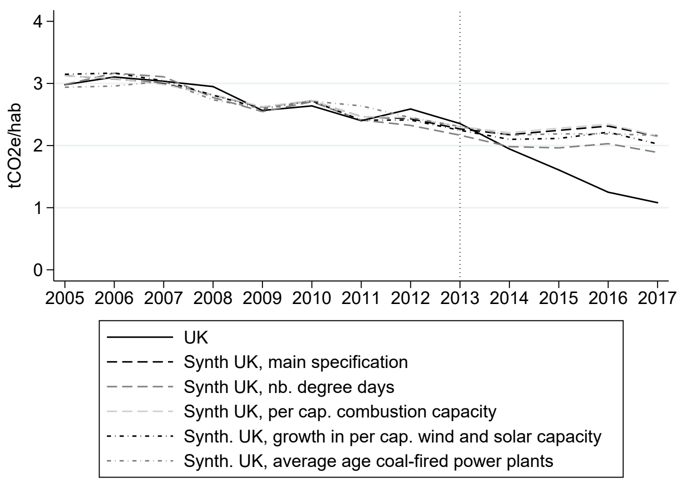
(c) Average annual growth in per capita renewables capacity (2010-2012)



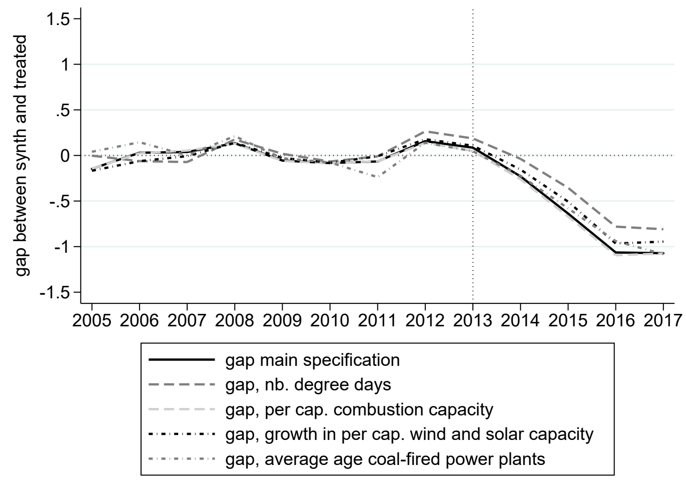
(d) Average age of operating coal-fired power plants

Figure A.6: Histograms of predictors used in the sensitivity analysis (UK: black with transparent fill; donor pool: grey fill)

Notes: Unless otherwise specified, all variables are averaged for the 2005-12 period



(a) Absolute per capita emissions



(b) Emission gap between treated and synthetic

Figure A.7: Sensitivity analysis: alternative set of predictors

Notes: The synthetic UK is composed in the following way for the different specifications: Main specification: five countries: Ireland (49.2%), Slovakia (25.6%), the Netherlands (13.7%), Finland(5.8%), Czech Republic (5.7%). Specification with the number of heating degree days: five countries: Ireland (37.9%), Italy (31.6%), Finland (15.3%), Slovakia (12.9%) and Poland (2.4%). Specification with per capita combustion capacity: five countries: Ireland (40.8.7%), Slovakia (22.4%), the Netherlands (14.1%), Poland (6.7%) and Finland (6%). Specification with the growth in per capita renewables capacity: four countries: Italy (61.5%), Poland (20.7%), Ireland (16.1%), and Denmark (1.7%). Specification with the average age of the coal-fired power plants: six countries: Slovakia (35.2%), Czech Republic (31.2%), Sweden (12%), Hungary (11.2%), Spain (7.5%) and Finland (3%).

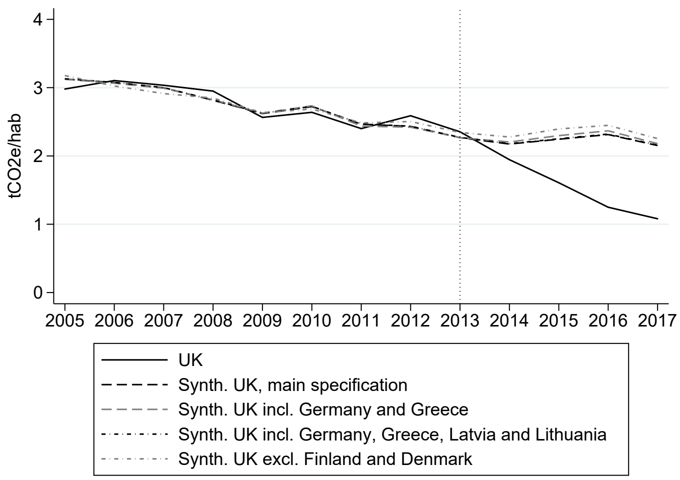
Table A.6: Predictors' values for the UK and each alternative synthetic UK

Variable	UK	Synth.UK				
		Original	Alt.1	Alt.2	Alt.3	Alt.4
Per cap. Residual load	4.29	4.30	X	4.44	4.27	2.01
Coal-gas price ratio	0.52	0.51	0.48	0.50	0.51	0.52
Per cap. LCP opt-out emissions 2009	0.29	0.24	0.29	0.29	0.36	0.31
Lignite dummy	0.00	0.06	0.024	0.07	0.21	0.42
Per cap. emissions 2005	2.98	3.13	2.98	3.12	3.06	2.94
Per cap. emissions 2012	2.59	2.43	2.32	2.45	2.49	2.45
Nb. of degree days	3020.30	X	3024.32	X	X	X
Per cap. combustible fuels capacity	1.10	X	X	1.15	X	X
Growth in per cap. renewable capacity	0.23	X	X	X	0.21	X
Avg. age of coal-fired power plants	36.04	X	X	X	X	36.07

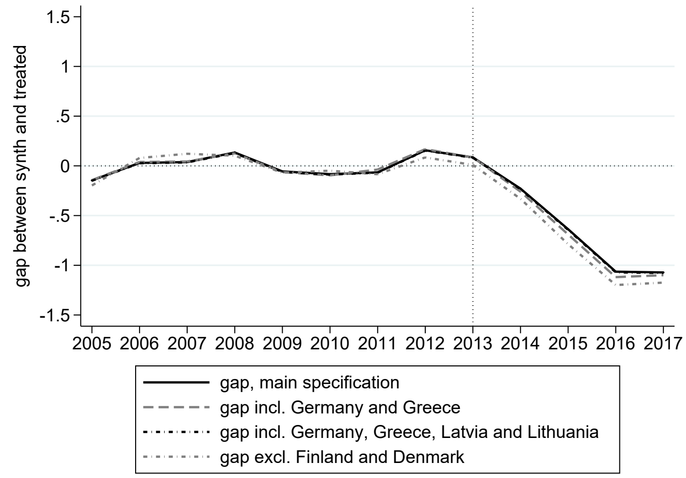
Notes: The per capita residual load, number of degree-days, per capita combustible fuels capacity and the average age of coal-fired power plants variables are averaged for the period 2005-2012. The coal-to-gas price ratio variable is averaged for the period 2007-2012. The growth in per capita renewable capacity is averaged for the period 2010-2012. LCP per capita opt-out emissions are taken in 2009, the lignite dummy is time-invariant. The outcome lags (per capita power sector emissions) are taken in 2005 and 2012.

A.10 Sensitivity of the results to the choice of the donor pool

Figure A.8 shows the sensitivity of the results from the upper bound to a different composition of the donor pool. Note that the results would be the same if applied on the lower bound from part 4.2, since the composition of the synthetic UK is the same. In a first test, I include Greece and Germany back in the donor pool, two countries that were previously excluded because they had experienced a shock in their power sector at the period of interest. In a second test, I include Germany, Greece, Latvia and Lithuania, all the countries that were previously excluded except Estonia (Adding Estonia makes it impossible to find a convex combination of countries replicating the trajectory of the UK, probably due to the too large discrepancy in emissions between Estonia and the other countries). In a third test, I exclude Denmark and Finland from the original donor pool. These two countries may be influencing the results substantially since they have large variations in per capita emissions and have a non-zero weight in the initial synthetic UK. The composition of the synthetic UK barely changes with different compositions of the donor pool. Figure A.8b shows that the emission reduction estimate obtained with the main specification is close to the estimates obtained with different compositions of the donor pool.



(a) Absolute per capita emissions



(b) Emission gap between treated and synthetic

Figure A.8: Sensitivity analysis: alternative donor pool

Notes: The synthetic UK is composed in the following way for the different specifications: Main specification: five countries: Ireland (49.2%), Slovakia (25.6%), the Netherlands (13.7%), Finland(5.8%), Czech Republic (5.7%). Specification including Greece and Germany in the donor pool: four countries: Ireland (54.2%), the Netherlands (22.5%), Slovakia (16.6%), and Finland(6.6%). Specification including Greece, Germany, Latvia, Lithuania (entire donor pool except Estonia): five countries: Ireland (49.4%), Slovakia (25.2%), the Netherlands (14.2%), Finland(5.8%), Czech Republic (5.5%). Specification excluding Finland and Denmark: four countries: Ireland (51%), Slovakia (20.6%), the Netherlands (18.1%), Poland (10.3%).

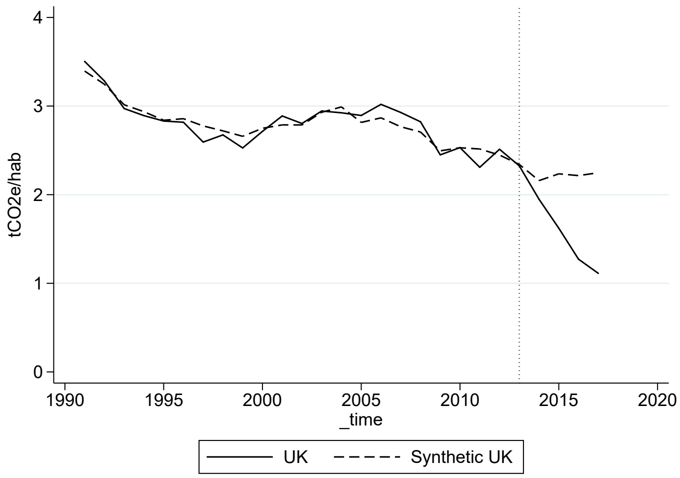
A.11 Sensitivity of the results to a longer pre-treatment period

The emission data used in the main analysis are only available from 2005, which means that only eight years of pre-treatment data can be used to generate the synthetic UK and assess the validity of the method. Although there is no rule of the thumb for the minimum number of pre-treatment periods that can be deemed "safe" to apply the synthetic control method, Abadie et al mention that "the applicability of the method requires a sizeable number of pre-intervention periods" (Abadie et al., 2015). I test whether having a relatively short time period is likely to bias my estimate by applying the synthetic control method to a similar outcome variable available for twenty-three pre-treatment years (but which does not have information at the plant level). The new outcome variable is directly available at the country level on the European Environmental Agency (EEA)'s website, and includes all greenhouse gas emissions from the public electricity and heat production sector. I divide the variable by annual country population to obtain per capita emissions, and I use it over the 1990-

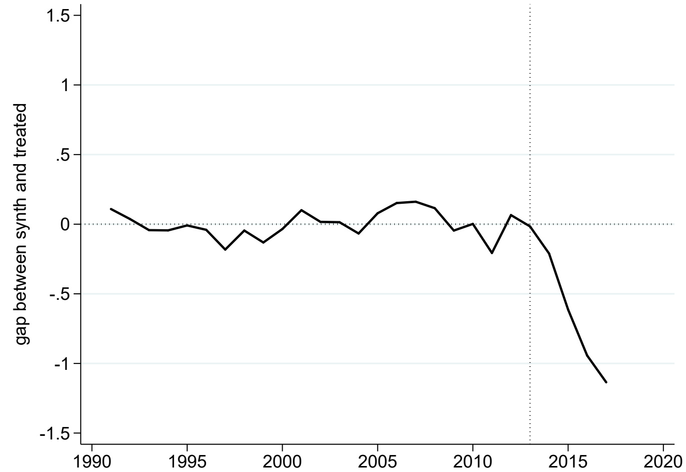
2017 period. This aggregate variable does not allow to identify individual plants and isolate confounding factors like I do in the main analysis. But it would be reassuring to find an emission reduction close to the main result when I apply the synthetic control method to this longer dataset. I keep the same predictors as in the main specification.

Figure A.9 shows the results. The composition of the synthetic UK changes compared to the main result, with the new synthetic UK made of three countries: Poland (36.0%), Italy (44.2%), and Slovakia (19.7%). It means that the combination of countries best mimicking the evolution of UK emissions over the 1990-2012 period is not the same as the combination of countries best mimicking the evolution of UK emissions over the 2005-2012 period. The estimated cumulative abatement with this longer period is the same as the estimate for the upper bound using a shorter pre-treatment period: 191 $MtCO_2e$ over the 2013-2017 period. While predictors' values were closely aligned between the UK and synthetic UK in the main result, this is no longer the case with the new synthetic UK: table A.7 shows that the predictors' values of the synthetic UK are further away from the UK than that of the average donor pool. This may indicate that averaging the predictors for the end of the pre-treatment period only is not appropriate to predict emission values from before 2005[63]. I run a permutation test similar to the one performed in section 4.3. Figure A.10 shows that the decrease in emissions seen in the UK is not found with the same magnitude in other European countries. This test suggests that using a relatively short time period in my main results should not come at the cost of a large bias in the estimation.

[63]The other way around, keeping the original weighting and synthetic UK obtained in the main specification yields an emission trajectory which does not mimic well the emission trajectory for the UK before 2005. Applying the synthetic control method before 2005 could be inherently difficult in the case of power sector emissions compared to other sectors such as transport, due to the important change in European electricity markets over the 1990s and 2000s from heavily regulated industries to more liberalized and interconnected markets subject to a single carbon market.



(a) Absolute per capita emissions

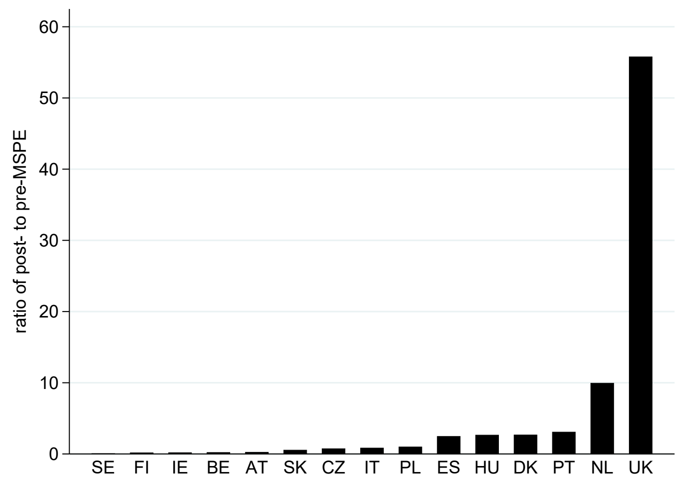


(b) Emission gap between treated and synthetic

Figure A.9: Sensitivity analysis: extended pre-treatment period with Eurostat greenhouse gas emissions by sector

Notes: The synthetic UK is made of three countries: Poland (36.1%), Italy (44.2%), Slovakia (19.7%).

(a) Permutation test with longer pre-treatment period



(b) Ratio of MSPEs with longer pre-treatment period

Figure A.10: Sensitivity analysis: extended pre-treatment period with Eurostat greenhouse gas emissions by sector, permutation test

Notes: In both figures, France is not included: for this country it is impossible to find a stable diagonal V matrix

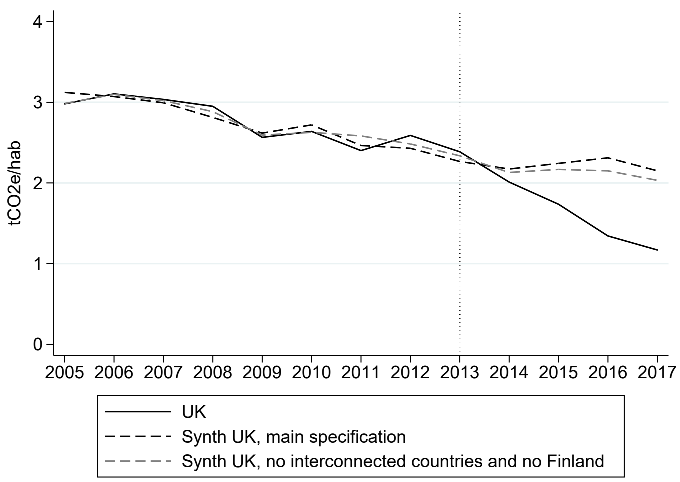
Table A.7: Predictors' values for the UK, synthetic UK and average of the donor pool, longer panel dataset

Variable	**UK**	**Synth. UK**	**Avg. Donor pool**
Per capita residual load	4.29	3.34	3.37
Coal-gas price ratio	0.52	0.44	0.71
Per cap. LCP opt-out emissions 2009	0.29	0.64	0.22
Lignite dummy	0	0.36	0.20
Per cap. emissions 1990	3.57	3.57	2.70
Per cap. emissions 1998	2.67	2.72	2.77
Per cap. emissions 2012	2.51	2.45	2.14

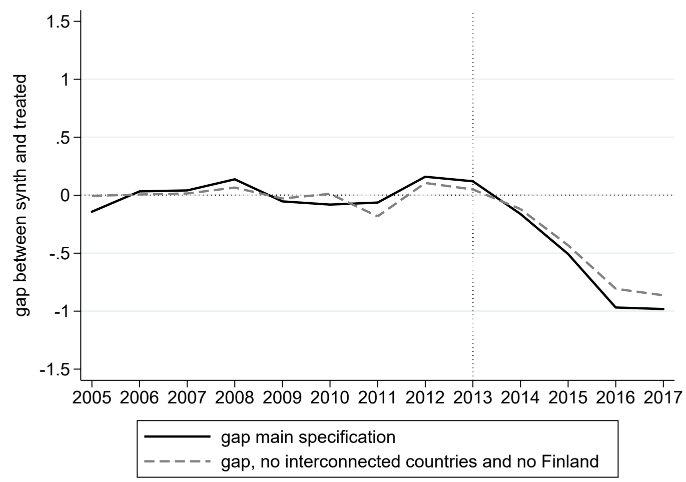
Notes: The per capita residual load is averaged for the period 2005-12, and the coal-to-gas price ratio for the period 2007-12. Per capita opt-out emissions are taken in 2009, the lignite dummy is time-invariant. Outcome lags are taken in 1990, 1998 and 2012.

A.12 Estimation of the spillovers removing all interconnected countries from the donor pool

I run the synthetic control method again, after removing the countries interconnected to Ireland from the donor pool. When I do so, the pre-treatment fit becomes quite poor due to the hight weight taken by Finland. I thus also run a specification where I also remove Finland from the donor pool. Figure A.11 shows the trajectory of these two alternative Synthetic UK compared to the trajectory of UK emissions. For the synthetic UK without the interconnected countries nor Finland, the gap in emissions is smaller and results in a cumulative abatement 22 $MtCO_2e$ lower, of 142 $MtCO_2e$, 14% less than the lower bound estimate. Predictor balance is however less good, suggesting a trade-off, also pointed by Abadie (2021), between keeping in the donor pool countries sufficiently close to the treated unit, and having countries "too close" geographically and hence subject to spillovers from the treated unit. In particular, the amount of LCP opt-out emissions is not anymore aligned with the UK value and is greater in this new synthetic UK than in the original synthetic UK, which may partly explain why the emission gap is smaller than in the main specification.



(a) UK and synthetic UK



(b) gap between UK and synthetic UK

Figure A.11: UK (incl. counterfactual emissions from biomass converted plants) and synthetic UK - no interconnected countries

Notes: The initial synthetic UK is made of five countries: Ireland (49.2%), Slovakia (25.6%), the Netherlands (13.7%), Finland(5.8%), Czech Republic (5.7%). The synthetic UK with Ireland, the Netherlands and France removed from the donor pool is made of five countries: Spain (58.9%), Finland (25.4%), Slovakia (7.6%), Czech Republic (6.5%) and Denmark (1.7%). The synthetic UK with Ireland, the Netherlands, France and Finland removed from the donor pool is made of four countries: Italy (72%), Poland (23.4%), Denmark (2.5%), and Czech republic (2.1%). UK emission values include estimated counterfactual emissions in the absence of biomass conversion for Lynemouth and Drax plants.