CECILIA 2050

OPTIMAL EU CLIMATE POLICY

Choosing Efficient Combinations of Policy Instruments for Low-carbon development and Innovation to Achieve Europe's 2050 climate targets

Understanding the Impacts & Limitations of the Current EU Climate Policy instrument Mix

Synthesis & Conclusion

Funded by the European Union

SEVENTH FRAMEWORK PROGRAMME

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LIST OF ABBREVIATIONS

BCA	Border Carbon Adjustment
CAP	Common Agricultural Policy
CCPT	Carbon Cost Pass Through
EITE	Emissions Intensive Trade Exposed
EPBD	Energy Performance of Buildings Directive
EPC	Energy Performance Certificate
ETD	Energy Taxation Directive
EUA	European Union Allowances
EU ETS	European Union Emissions Trading System
GHG	Greenhouse Gas
ILUC	Indirect Land Use Change
LCPD	Large Combustion Plant Directive
RED	Renewable Energy Directive
RES-E	Renewable Electricity
STPM	Soft Transport Policy Measures
TCO	Total Cost of Ownership

Executive summary

The existing climate policy mix is uneven, lightly co-ordinated and sometimes difficult to define. There is a deep divide between sectors and Member States concerning the number of instruments in place to tackle emissions, instrument design and implementation, and the level of ambition. The power and industry sectors experience the most coherent policy landscape, with the EU ETS producing a single, EU wide carbon price. The promotion of renewable electricity under the Renewable Energy Directive is considered in EU ETS cap-setting exercises to prevent negative overlap, which has likely been achieved so far. However, the individual implementation of instruments to promote electricity from renewable sources varies significantly between Member States. The Energy Taxation Directive places uneven minimum taxation requirements on different energy carriers and sectors, and permits substantial variation between Member States (particularly in transport). Several instruments act on passenger cars, whilst transport modes such as international aviation and shipping are largely untouched. No explicit climate policy exists at EU level for the agriculture sector. Instead, provisions in the Nitrates Directive and Common Agricultural Policy, both introduced for non-climate purposes, are likely to have had the most impact on abatement.

Despite this, the climate policy mix in the EU has been broadly effective in producing GHG abatement. Meyer & Meyer (2013) calculate that the presence of the EU ETS, instruments to promote renewable electricity and environmental tax reforms reduced CO_2 emissions by up to 12-13% below the counterfactual in some Member States in 2008 (but with significant variation). This value is likely to increase with the consideration of the impact of flanking instruments.

Whilst economic instruments have been important, they are not the only climate policy instruments to have produce abatement. The economic instruments currently in place, whilst effective to different extents, are not exploiting their full potential as a result of design flaws, imperfect implementation, and negative interaction with other climate and non-climate policy instruments. Meyer & Meyer (2013) calculate that the EU ETS produced CO_2 abatement of 1-3% against the counterfactual in 2008 across the EU (in line with other estimates in the literature), delivered principally through fuel switching in the power sector. Although, the level of abatement induced has varied significantly over time. Instruments for the promotion of renewable electricity provided the largest contribution to policy-induced abatement at an average rate of 3.2%-3.9% across Member States against the counterfactual in 2008. The use of fuel taxes appears to be effective in influencing road travel demand, but not significant in driving demand for more efficient vehicles. However, incentives for both reduced demand and for more efficient vehicles by any road transport pricing instruments (including registration and circulation taxes, and often other road pricing mechanisms) are restricted by company car taxation arrangements. However, Regulation 443/2009, which sets binding CO_2 performance standards for new passenger cars, has been effective in increasing the rate CO_2 intensity reductions, with the 2015 target of 130gCO_2/km likely to be achieved ahead of time.

There is no evidence to suggest that 'carbon leakage' has occurred. Whilst much of the *ex-ante* analysis predicted significant rates of carbon leakage, the *ex-post* evidence suggests that no loss of competitiveness leading to carbon leakage has occurred amongst the Energy Intensive Trade Exposes (EITE) sectors as a result of the EU climate policy.

From a broad perspective, the climate policy mix may have produced net economic benefits to the EU. Meyer & Meyer (2013) conclude that the introduction of the EU ETS, renewable electricity support mechanisms and environmental tax reforms overall did not reduce GDP in the EU, and likely had a positive impact. Employment is also likely to be higher than the counterfactual in most Member States, with the exception of some of the smaller transition economies. The EU ETS, taken individually, is likely to have reduced GDP and employment by an average of 0.5% and 0.34% respectively across Member States in 2008, whilst investment in renewable electricity is estimated to have increased both GDP and employment by an average of 0.32% and 0.09% respectively across Member States in 2008. Although this average value is lower than the average negative impact from the EU ETS, the net benefits have been the most substantial in the larger European economies, producing a weighted total net benefit for the EU.

Instrument mix 'Optimality' is difficult to achieve, but improvements are possible. Under the concept of optimality developed under the CECILIA2050 project and employed in this report, which examines environmental effectiveness, economic efficiency (static and dynamic) and feasibility, it is clear that the existing climate policy mix is sub optimal. However, it must be made clear that this concept must be considered a theoretical point of reference. Policies and policy mixes in real-world application are always faced with trade-offs and compromises between each of these components. Based on the research summarised in this report, and the reports underlying it, the lessons learned can be used to enable improvements to the existing policy mix to be investigated and pursued.

1 Introduction

This report synthesises the outputs from the series of sector-specific and cross-sectoral studies produced under Work Package 2 – 'Understanding the Impacts and Limitations of the Current Instrument Mix in Detail' of the CECILIA2050 project. The titles of the reports summarising these studies are in Table 1, and each publication may be found on the CECILIA2050 website[1].

Table 1 - 'Understanding the Impacts and Limitations of the Current Instrument Mix in Detail' - Individual Report Titles

Authors	Title
Agnolucci & Drummond (2014)	The Effect of Key EU Climate Policies on the EU **Power Sector**: An Analysis of the EU ETS, Renewable Electricity and Renewable Energy Directives
Branger & Quirion (2013)	Understanding the Impacts and Limitations of the Current Instrument Mix in Detail: **Industrial Sector**
Maca *et al* (2013)	Climate Policies and the **Transport Sector**: Analysis of Policy Instruments, their Interactions, Barriers and Constraints, and Resulting Effects on Consumer Behaviour
Kuik & Kalfagianni (2013)	**Food and Agriculture**: The Current Policy Mix
Nauleau, Branger & Quirion (2014)	Abating CO_2 Emissions in the **Building Sector**: The Role of Carbon Pricing and Regulations
Zverinova, Scasny & Kysela (2014)	What Influences **Public Acceptance** of the Current Policies to Reduce GHG Emissions?
Meyer & Meyer (2013)	Impact of the Current Economic Instruments on **Economic Activity**: Understanding the Existing Climate Policy Mix
Mazzanti & Antonioli (2013)	Inducing Greenhouse Gas Abating **Innovations** Through Policy Packages: Ex-Post Assessments from EU Sectors
Kuik, Branger & Quirion (2013)	**International Competitiveness and Markets**
Mehling *et al* (2013)	The Role of **Law and Institutions** in Shaping European Climate Policy: Institutional and Legal Implications of the Current Climate Policy Instrument Mix

This report is structured using two framing tools developed and applied by the CECILA2050 project. The primary structure follows the three aspects of the CECILIA2050 concept of policy mix 'optimality'; environmental effectiveness, economic efficiency and feasibility[2]. Under each of these headings the discussion is delineated by climate policy instrument 'type', namely 'Carbon Pricing', 'Energy Efficiency and Energy Consumption', 'Promotion of Renewable Energy' and 'Non-CO_2 GHGs', based on the objective of the instrument or instrument mix under discussion. A discussion of the specific interactions present between climate policy instruments is also provided under each aspect of optimality. Conclusions are then drawn in the final section. It must be noted here that interactions between policy

[1] www.cecilia2050.eu/publications

[2] Refer to Görlach (2013) for further information on these aspects and the overall concept.

instruments are often complex and multi-faceted. As such, the structure provided in this report serves to highlight some interactions, but to make others less clear. Such conflict is inevitable, however, and is minimised as much as possible. Necessarily, not all policy instruments that may be considered ‘climate’ policy instruments are discussed, only those which are of highest importance. As this report mainly focuses on ex-post assessment, some instruments, such as the Energy Efficiency Directive, are excluded due to their recent introduction. Mention of the ‘EU’ or ‘Europe’ refers to the EU28 Member States, unless otherwise stated.

2 Environmental Effectiveness of the EU’s Climate Policy Instrument Mix

Total GHG emissions in the EU27 decreased by around 18.5% between 1990 and 2011 (~1,024MtCO_2e), with CO_2 decreasing by around 11% (~450MtCO_2) (European Environment Agency, 2014). Meyer & Meyer (2013) used the global economic environmental model GINFORS (discussed further in the next section), to estimate to what extent the three main types of climate policy instruments used in the EU – the EU ETS (EU Emission Trading System), support schemes for renewable energy and environmental taxes have contributed to this trend between 1995 and 2008. They conclude that CO_2 emissions in 2008 would have been up to 12-13% higher than observed in some Member States (but with significant variation across the EU).

Carbon Pricing

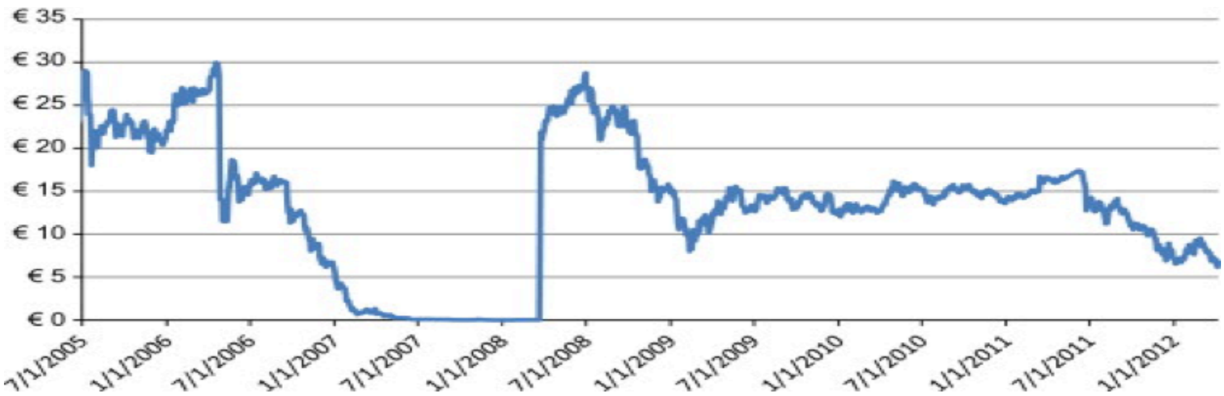
The EU ETS is the cornerstone of EU climate policy, and since 2005, the primary instrument through which around 50% of Europe’s CO_2 emissions are priced[3], including those from power (and parallel heat) generation, and most heavy industry[4]. Meyer & Meyer (2013) calculate that in 2008, CO_2 emissions in the EU would be 1-3% higher than if the EU ETS was not in place (with significant geographical variation, discussed below) – a range comparable to that found in the literature for Phase 1 (2005-2008), as summarised by Laing *et al* (2013). The EU ETS is the primary instrument employed to reduce CO_2 emissions from the power sector. In turn, the power sector accounts for around half of emissions capped under the EU ETS, and is key in achieving abatement under the scheme. CO_2 emissions from power and heat generation decreased by 15.6% between 1990 and 2011, and account for around 27% of GHG emissions in the EU (European Environment Agency, 2014). Agnolucci & Drummond

[3] For more information on the specific policy instruments discussed in this report, refer to Drummond (2013).

[4] From 2012, domestic and international aviation is also covered by the EU ETS (using aviation-specific allowances). However due to international pressure, flights that originate or terminate outside the EU are currently in effect exempt, in anticipation of an ICAO global mechanism to be agreed by 2016 (Maca *et al*, 2013). As this mechanism was only introduced in 2012, and as EU-only aviation emissions are under 1% of EU total GHG emissions (and as 85% of aviation allowances are currently grandfathered, producing little abatement incentive) (Drummond, 2013), the impact of the EU ETS on aviation will not be discussed further.

(2014) conclude, based on an extensive literature review, that abatement in the power sector attributable to the EU ETS in Phase 1 peaked at the beginning of the scheme in 2005 (around 2.3% counterfactual power sector emissions), before rapidly decreasing in 2006 (1.3% counterfactual power sector emissions) and largely ceasing in 2007[5]. Such a trend reflects the evolution of the European Union Allowance (EUA) spot price over this time, as reflected in Figure 1.

Figure 1 – Trend in EUA Spot Price (Source: Venmans, 2012)



At the beginning of the EU ETS the price remained relatively stable, peaking at €29.20/tCO_2 in April 2006. In the same month, when the release of verified emissions data confirmed a likely oversupply of allowances for the Phase, the price roughly halved. This trend continued, and as the banking of allowances into Phase 2 was not permitted, the price crashed to nearly zero by the end of the Phase. The price rebounded at the beginning of Phase 2 with the oversupply issue removed (peaking at €27/tCO_2 by July 2008), before dipping to around €10 in February 2009 at the onset of the financial crisis, which reduced electricity demand (and consequential emissions) (Agnolucci & Drummond, 2014). The price recovered and remained stable at around €15/tCO_2 between April 2010 and April 2011, before reducing to around €7/tCO_2 in April 2012, in response to the European sovereign debt crisis (Venmans, 2012). The price subsequently dropped to around €5/tCO_2 in response to expected allowance oversupply of allowances until 2020 as a result of the financial crisis, and has remained approximately at this level into Phase 3 (with the ability to bank allowances preventing a repeat of a zero price) (Agnolucci & Drummond, 2014).

Few studies have yet attempted a rigorous estimation of EU ETS-attributable abatement in Phase 2 (2008-2012), at either cumulative or sectoral levels – in part due to a delay in data availability. As Meyer & Meyer (2013) estimate for the EU ETS as a whole, Agnolucci & Drummond (2013) conclude that abatement in the power sector is geographically varied, with the UK (followed by Germany) experiencing the largest ETS-attributable abatement[6], with negligible abatement in many others (such as Sweden, France, Poland and the Czech

[5] Percentage values calculated by Agnolucci & Drummond (2014) based on primary data from Delarue *et al* (2008).

[6] See Agnolucci & Drummond (2014), Table 4. All values proportional reductions on estimated counterfactual emissions. UK = 7.5-12% (2005), 3.7-12% (2006), 0.4% (2007) and Germany = 2.1-8.3% (2005), 1.2% (2006), 0% (2007).

Republic). This is a result of the process by which ETS-induced abatement has been almost entirely achieved in the power sector, namely 'fuel switching' - primarily from coal generation to natural gas generation[7] - which occurs when the relative marginal cost of generation of generally cheaper coal and more costly (but less CO_2-intensive) natural gas switch places in the merit-order in the presence of a (relatively significant) carbon price. However, the structure and profile of the power sector and power market in each Member State determines to what extent this effect may occur. There are two primary influences. Firstly, the largest potential for fuel switching between coal and gas rests in those Member States with a high proportion of both types of capacity to switch from and to. Such is the case in the UK and Germany. In addition, the majority of abatement has thus far occurred at times of low electricity demand, such as in the summer, over weekends and overnight. At such times there is idle capacity in the system to allow switching (under the right influences), whereas during times of peaking demand relatively little capacity of any type is left unutilised, reducing the ability to chose between generation capacities, regardless of the carbon price. Member States with a high proportion of coal capacity but not gas (such as Poland or the Czech Republic), or vice versa, have experienced little fuel-switching[8], along with Member States with a high proportion of low-carbon capacity such as nuclear (e.g. France) or renewables (particularly hydropower, e.g. Sweden), for which the marginal cost of generation is very low or zero (and therefore first in the merit order), and upon which the carbon price has no influence (Agnolucci & Drummond, 2014).

Secondly, the Carbon Cost Pass Through (CCPT) rate, defined as the change in wholesale power price resulting from a change in CO_2 price, varies over time and between Member States, determining the extent to which the headline carbon price generated by the EU ETS is factored into dispatch decisions (Agnolucci & Drummond, 2014). This will be discussed further in the next section.

The abatement achieved via fuel switching between different types of existing capacity is immediately reversible when the incentive to maintain the switch is reduced or removed (as likely happened in 2007, with the carbon price crash), and not indicative of long-term systemic change in the power system. Between 2000 and 2010, EU coal capacity slowly declined, whilst gas capacity nearly doubled. However, the literature concludes that the EU ETS was an insignificant influence on this trend, as predictability of opportunities and liabilities is key to making efficient investment decisions (Egenhofer, 2007), and despite the certainty of the emissions cap to 2020 the lack of a high, stable price experienced so far combined with significant uncertainty regarding future prices and Phase design (relevant to such long-term investment), prevents such reliable foresight. Pahle *et* al (2011) even suggest that the design of the first two Phases, in which permits were almost entirely grandfathered

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[7] Delarue *et al* (2008) calculate 90% of Phase 1 power sector abatement due to fuel switching, with remainder the result of electricity trading between Member States, discussed later in this report.

[8] Although Schumacher *et al* (2012) found that relatively significant switching from lignite to slightly less CO_2-intensive hard coal is likely to have occurred in the Czech Republic in 2005. Convery *et al* (2008) found a similar effect in Germany in Phase 1 (alongside the broader coal-to-gas trend).

across Member States, may have incentivised investment in higher-carbon fuels (e.g. coal), as windfall profits would be higher (discussed in the following section). However, there is little evidence in the literature to suggest this perverse incentive made a material impact on investments in new capacity (Agnolucci & Drummond, 2014).

Relative fuel prices (particularly gas and coal), and changes over time were a key determinant behind the rapid expansion of gas capacity between 2000 and 2010. Relative fuel prices determine the marginal cost of generation, and therefore the merit order for generation (and subsequent CO_2 emissions), including the emergence of 'switching points' – where the difference between marginal costs of generation are small enough to allow the EU ETS price to induce fuel switching. As such, underlying (relative) fuel prices are a principal factor in determining the electricity generation profile across the EU. However other factors such as domestic fuel availability (e.g. North Sea gas in the UK), the shorter lead-time gas plant exhibit over coal, gas plant efficiency improvements (via CCGT) and energy market liberalisation in the EU also play influential roles (Agnolucci & Drummond, 2014).

The literature is also in consensus that the EU ETS has not induced investment in renewable electricity capacity in the EU (Agnolucci & Drummond, 2014), although New Energy Finance (2009) find evidence that the presence of biomass co-firing capacity is 'clearly' positively impacted by the EU ETS in Phase 1.

The EU ETS is also the primary policy instrument in place to reduce GHG emissions in the EU's heavy industry. In a number of Member States the sector is also notionally subject to other instruments such as carbon and energy taxes or regulatory measures (including the Industrial Emissions Directive), although in practice, in most instances, exemptions for EU ETS sectors are in place. Subsidies for GHG abatement in the industry sector are also present in a number of Member States, however they are constrained by state aid rules, and represent a very limit implicit CO_2 price (Branger & Quirion, 2013).

Industry accounted for around 22.6% of total EU GHG emissions in 2011, with CO_2 emissions from the sector decreasing by around 4% between 2005 and 2012[9] (28.2% from 1990) (European Environment Agency, 2014), although few studies have produced robust estimates to which extent this was driven by the EU ETS (Martin *et al*, 2012). Studies that have attempted such a sector-specific calculation have clear shortcomings and tend to produce contradictory results, as discussed by Branger & Quirion (2013), who conduct an original study to examine the extent to which the EU ETS has produced CO_2 abatement in the cement sector (the most carbon intensive of all global industries (Grubb, 2014)), using best available data and examining trends to 2011, beyond that of most previous literature.

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[9] CO_2 emissions for the industry sector actually increased across EU ETS Phase 1 (by around 10% between 2005 and 2007), before decreasing sharply over Phase 2 (over 18% between 2008 and 2012) (European Environment Agency, 2014).

CO_2 emissions from the production of cement[10] in the EU decreased by around 7.5% between 2005 and 2012[11]. Much of this reduction (delivered mostly within Phase 2) coincides with a significant reduction in cement production (260Mt in 2007 to 186Mt in 2010), responding to declining demand as a result of the financial crisis (Branger & Quirion, 2013). On a CO_2 intensity basis (tCO_2 per tonne of cement produced), whilst a steady decrease was experienced between 1990 and 2000 (0.7% per annum), rising to 1% per annum between 2000 and 2005, an average annual decrease of just 0.33% was experienced between 2005 and 2011. Taking the trend for 2005 to 2010 alone, CO_2 intensity of cement production actually increased (emissions between 2010 and 2011 appear to decrease by around 2%, however this is largely due to a change in reporting requirements[12]). A key driver of CO_2 intensity – energy intensity – decreased by around 8% between 1990 and 2000, but only 1% between 2000 and 2005, and has actually experienced an increase to 2011. Branger & Quirion (2013) conclude that the EU ETS has therefore not reduced the energy intensity of cement (clinker) production, and coupled with a lack of change in the fuel mix, is unlikely to have impacted carbon intensity and therefore total emissions from cement production, through this route. This may be due to an insufficient carbon price, as Climate Strategies (2014) suggest that even a EUA price of €20 (a spot price achieved only rarely and for short periods, as illustrated in Figure 1) is unlikely to justify a shift to more efficient kilns. However, another driver may be a linked to free allowance allocation in the first two Phases. In the first two phases, allowances were allocated based on historical emissions in a base period. Firms may also have anticipated this to continue into the second phase, creating a perverse incentive against abatement (Branger & Quirion, 2013). Phase 3, which began in 2013, introduced new rules to determine allocations based on an emission 'benchmark' calculated as the average emissions of the top 10% performing installations in the EU producing a given product. In general, installations that meet these benchmarks will receive 100% free EUA allocations, with those not meeting the benchmark receiving proportionally less (Drummond, 2013). Whilst this reduces previous abatement disincentives, rules surrounding the closure of an installation retains distortions. If an installation produces less than 50% historic rates, it loses 50% of its allowances, and 75% of its allowances if operation is less than 25%. This produces an incentive for a firm operating multiple installations to operate all at lower rates, but above 50%, rather than closing the least-efficient plants and increasing production from the most efficient, reducing overall emissions (Branger & Quirion, 2013; Climate Strategies, 2014).

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[10] Specifically the production of clinker; an intermediate product that constitutes about 75% of cement in mass, and is used for no other purpose (Branger & Quirion, 2013)

[11] As with the Industry sector as a whole, CO_2 emissions from cement production (clinker) increased in Phase 1 (by over 14% between 2005 and 2007), before decreasing sharply by the end of Phase 2 (around 26% between 2008 and 2012). Cement production accounted for around 8% total EU ETS emissions in 2011 (around 22% of non-power sector EU ETS emissions) (European Environment Agency, 2014).

[12] Prior to 2011, emissions from the combustion of waste with both a biomass and fossil fuel component (e.g. used tires, which contains both latex and oil) were reported in their entirety. As of 2011, only emissions from the fossil fuel component of such items must be reported, producing an artificial reduction in reported emissions (Branger & Quirion, 2013).

Although, the proportion of clinker in cement decreased by an average 0.55% per annum between 2005 and 2011, a significant increase from the 0.24% achieved between 2000 and 2005, itself a significant increase from the 0.09% experienced between 1990 and 2000. Branger & Quirion (2013) suggest that this may indeed have been the result of the EU ETS, reducing cement-related emissions by around 2% from the counterfactual if the 2000-2005 trend had continued. However, the impact of the significant increase in the price of steam coal (and petcoke) since 2000 (doubling between 2003 and 2010 (Cembureau, 2011), making clinker substitutes more economically attractive, cannot be excluded as a primary cause of this trend.

Meyer & Meyer (2013) attempt to estimate the impact of Environmental Tax Reform (ETR) across Member States that have implemented such reforms (Czech Republic, Denmark, Estonia, Finland, Germany, Netherlands, Sweden and the UK) for the consumption of coal, oil, natural gas, mineral oils and electricity by creating a counterfactual scenario in which rates active in 1998 remain static until the end of the assessment horizon (2008). They estimate that by 2008, CO_2 emissions in these countries were on average around 0.8% lower than if 1998 tax rates on these products had remained, delivered by a rise in energy productivity. These ETRs took place within the context of the Energy Taxation Directive (ETD) (2003/96/EC), although they are not a consequence of it. The ETD is the second EU-wide instrument to place an (implicit) price on CO_2 emissions, and came into effect on 1st January 2004. The ETD places minimum tax rates on the consumption of gas oil, heavy fuel oil, natural gas, coal, coke, gasoline, diesel and electricity. Rates vary by energy product and end user (commercial and non-commercial), with energy products used to generate electricity exempt from taxation. Energy products consumed in energy Intensive industry may also receive a reduction or exemption if such industries are subject to other instruments (or voluntary agreements). A significant number of Member States make use of this exemption, for a range of sectors (Drummond, 2013). Rates are defined in terms of physical units (litre, KWh, etc.), with implicit rates per gigajoule (GJ) varying from €0.1/GJ for coal to €10/GJ for gasoline, and implicit CO_2 prices varying from €1.1/tCO_2 to €151/tCO_2 for the same products, respectively (European Commission, 2011 and OECD, 2013).

Fuels consumed primarily in the (road) transport sector are taxed most heavily under the ETD, with minimum rates for gasoline and diesel set at €359/1000l and €330/1000l, respectively (Maca *et al*, 2013)[13], with significant variation across Member States, as illustrated in Figure 2, below[14].

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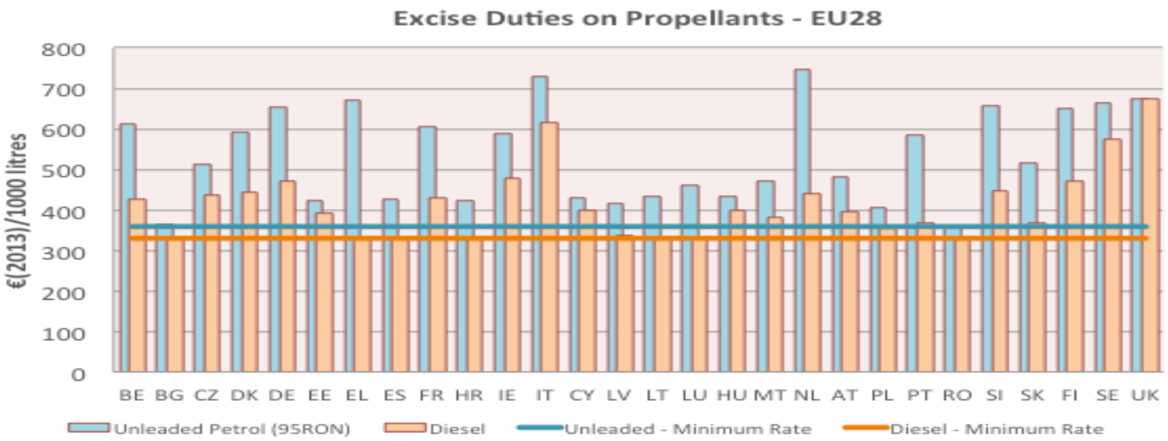
[13] With implicit CO_2 costs of €151/tCO_2 and €126/tCO_2, respectively (OECD, 2013a)

[14] Whilst the ETD excludes the aviation sector from its provisions, electricity and diesel used in rail transport is covered, although exemptions are possible for both fuels for rail transport. 10 Member States currently set reduced rates or exemptions for electricity use, and 14 for diesel. Malta and Cyprus are excluded, as no railways are in operation. Inland water navigation in also subject to exemptions in most Member States (Maca *et al*, 2013).

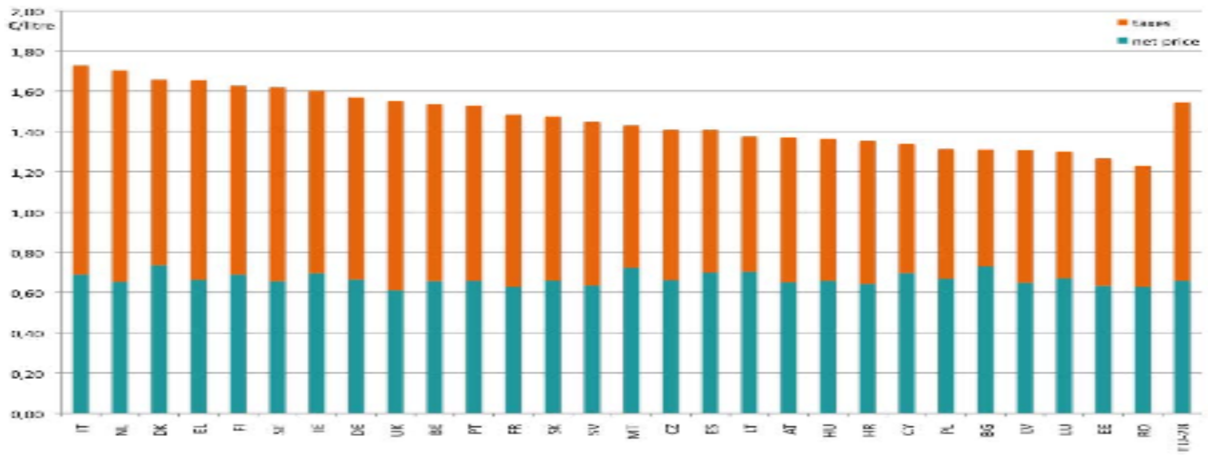
Figure 2 - Excise Rates on Petrol and Diesel in EU28 (Source: DG TAXUD, via Maca *et al*, 2013)



CO_2 emissions from transport increased steadily between 1990 and 2007 (at an average of 1.7% per annum), before declining to 2011 at an average of 1.5% per annum (producing an overall 19% increase between 1990 and 2011). Transport-related CO_2 emissions accounted for 20% of EU GHG emissions in 2011, with road transport accounting for around 94% of this (European Environment Agency, 2014). For passenger cars, Nijland *et al* (2012) conclude that increasing fuel taxes effectively reduce passenger kilometre demand at a rate of 260km annually per capita for every 10 Eurocent increase in fuel price (although fuel taxes appear to ineffective in influencing initial vehicle purchase decisions). The results of that study, compared with the level of taxation applied to transport fuels across the EU (as illustrated in Figure 3 below), suggests that fuel taxes have a significant impact on (road) transport demand and emissions.

However, by mandating only minimum taxation levels across the EU, the design of the ETD may reduce the effectiveness of fuel taxation. Member States are free to levy any rate above this level they see fit, producing considerable price differences across the EU (see Figure 2). Figure 3, along with illustrating the high proportion of taxation as a component of total price, indicates that differences in fuel price between Member States are also largely a result of taxation, rather than underlying prices.

Figure 3 - Fuel Price (Super 95) across Member States (October 2013) (Source: DG Energy Oil Bulletin, via Maca *et al*, 2013)



This leads to the practice of arbitrage, or 'fuel tourism', by which vehicle owners take advantage of price differentials by purchasing fuel where it is cheapest, if the differential is sufficient to offset the additional cost of doing so (in terms of time and fuel to cross Member State borders) (Maca *et al*, 2013), increasing travel demand and therefore emissions, amongst other issues (discussed in the following section). Countries such as Luxembourg, Austria and Switzerland are major European thoroughfares and have significantly lower fuel prices than neighbouring countries, and experience a substantial amount of fuel tourism[15]. Substantial differentials in petrol prices are also observed at the Italian-Austrian, Italian-Slovenian, Greek-Bulgarian, German-Luxembourgish and German-Polish borders, with differences ranging up to 35 Eurocents per litre. For diesel the differences are markedly lower, with only two in which price differentials exceed that of petrol. However, the literature on the incidence and impacts of fuel tourism is too isolated to provide a complete picture of the phenomenon. From the available literature, it appears that the issue is most prevalent in small, central EU Member States located on major thoroughfares, particularly those with a high population density in the border region of those countries with which a sufficient price differential exists. In these countries, the private vehicles that engage in fuel tourism appear to be international commuters and those living within 10-15km of such a border (Maca *et al*, 2013). Whilst taking advantage of price differentials is a highly plausible strategy for freight transport (as fuel costs represent 30% of all costs in the haulage industry in Europe (VDA, 2013)), widening the geographic potential for fuel tourism, the evidence in this sub-sector is even sparser. However, the opportunity cost of travelling longer routes for cheaper fuel is high in commercial transport, limiting the extent to which it might occur (Maca *et al*, 2013). As such, whilst arbitrage may be a significant issue in such regions the impact at the EU level is likely to be small, and is likely to have a negligible impact on the effectiveness of fuel taxation as an instrument in the EU.

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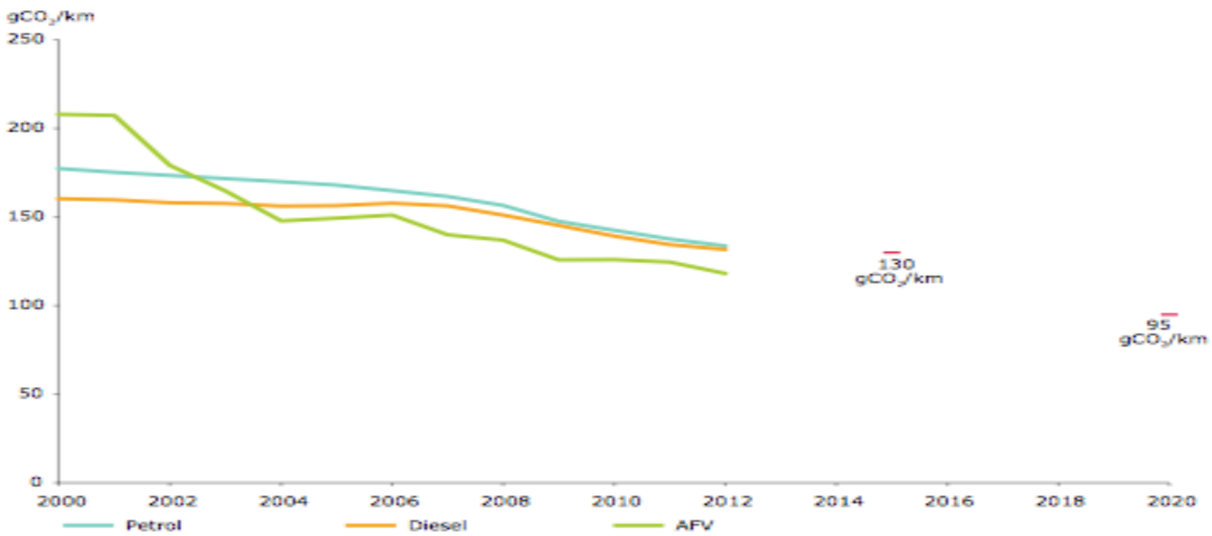
[15] For example, the Austrian Energy Agency (2012) concludes that up to 30% of all transport fuel sold in Austria is a result of fuel tourism, principally from Germany., Bleijenberg (1994) concluded that in the early 1990s, customers not resident in the country consumed two thirds of transport fuel sold in Luxembourg.

Although a minimum rate is applicable to domestic electricity use, energy products used for domestic space and water heating (which accounts for over three-quarters of all domestic energy consumption), may be exempt (including electricity for heating purposes), along with energy consumption in the agriculture sector, with many Member States exploiting this provision.

Energy Efficiency and Energy Consumption

Numerous instruments focus on improving energy efficiency and reducing energy consumption across the EU, with a clear emphasis on end-use sectors. The transport sector is subject to a variety of such instruments (both at the EU and Member State level), the most prominent of which is Regulation 443/2009, which sets binding emission performance standards for new passenger cars[16]. The regulation requires a fleet-average CO_2 intensity of 130gCO_2/km to be reached by 2015 for new cars (to be phased in from 2012), with a long-term target of 95gCO_2/km by 2020 (as approved in June 2013 by the European Parliament, Council and Commission).

Figure 4 - Trends in New Passenger Car CO_2 Intensity (Source: European Environment Agency, 2013)



As Figure 4 illustrates, between 2000 and 2007 the average CO_2 intensity decreased from 172gCO_2/km to 159gCO_2/km (average 1.2% reduction per year). However, since 2007 the rate of reduction increased markedly. By 2011 the average CO_2 intensity of new cars reduced to 136gCO_2/km (3.8% average annual reduction), suggesting the 2015 target will be achieved ahead of time (Maca *et al*, 2013). Maca *et al* (2013) attribute this step change to the requirements of the regulation (which superseded voluntary regulations), however the authors also state the effectiveness of the regulation is distorted by the weight-based nature

[16] A similar regulation was introduced in 2011 for light commercial vehicles (N1) (Regulation 510/2011), setting a 2017 target of 175gCO_2/km (to be phased in from 2014), and a longer-term target of 147gCO_2/km for 2020. Proposals for regulations to tackle Heavy Goods Vehicles (HGV) emissions will be announced are expected in 2015. As policy measures to address CO_2 these vehicle types are not yet operational, they are beyond the scope of this report.

of the specific emissions calculation[17], which penalises manufactures that reduce a vehicle's weight by then being subject to stricter CO_2 standards. Nijland *et al* (2012) found that the average weight of new vehicles had increased 10% between 2001 and 2011[18], reducing fuel efficiency of new diesel and gasoline vehicles by 6% and 2%, respectively, although when corrected for fuel type, weight, engine capacity and power, average CO_2 intensity decreased by around 23%, as observed in Figure 4. However, Mock *et al* (2013) imply that such a reduction in CO_2 intensity is optimistic, by concluding that the gap between vehicle 'type-approval' and real-world fuel efficiency (and by extension CO_2 efficiency) has increased from around 10% in 2011, to about 25% in 2011. The authors suggest that rather than a substantial change in driving behaviour, this increasing gap is likely due to a combination of factors including the increasing use of technologies with higher efficiency in type approval than actual application and the use of permitted variance in type approval and changes in external factors (e.g. increased use of air-conditioning) (Maca *et al*, 2013).

The extent to which the reduction of CO_2 intensity of new cars (regardless of the extent to which this has been achieved in practice) has lead to the abatement of (road) transport CO_2 emissions is dependent on the degree to which these new vehicles have penetrated the car fleet, and the division between vehicle type (petrol, diesel and alternative fuel vehicles (AFVs)). In 2000, petrol cars comprised the majority of new registrations at 68.9%, with diesel at 31%. By 2012 this position had reversed, with diesel holding a 54.9% share and petrol just 42.9% (with AFVs increasing from 0.1% to 2.2%) (European Environment Agency, 2013). Ajanovic (2011) concludes that this 'dieselisation' is a primarily a result of the improved fuel economy in diesel cars (20-30% compared to gasoline), and lower diesel prices compared to gasoline combining to produce a service demand around 30% cheaper per kilometre driven than gasoline cars. However, the author also finds that other related policy instruments (such as the ETD, and those discussed below) are likely to have had some impact. This change in proportional sales is only meaningful in the context of overall sales. New registrations increased between 2001 and 2007 (from around 12.8 million to 15.7 million per year), before rapidly deceasing by 2010 (13.5 million) – possibly a result of the financial crisis reducing demand (new registrations has continued to decrease, reaching around 12.1 million in 2012) (European Environment Agency, 2013).

Regardless of type, the penetration of new cars into the fleet is highly varied between Member States. Whilst the average age of the car fleet in Europe is 8 years, there is significant variation between Member States. In Germany, Italy and the UK, cars under five years old comprised more than a third of the total car fleet in 2011, whilst in Poland this value was just 11% (with over 70% of the fleet over ten years old) (Maca *et al*, 2013). The

[17] The fleet-average (or limit value curve) is set so that manufacturers that produce vehicles with specific emissions above the curve can return to the average by producing vehicles below it. The manufacturer's specific emission target is set using the vehicle's mass: *Specific CO_2 emissions = 130+0.0457*(M-M0)*, where '*M*' is the vehicle mass, and '*M0*' is set at 1372kg for the 2012-2015 period (Maca *et al*, 2013).

[18] Data from ICCT (2012) indicates a slowly increasing trend to 2007, before leveling and a slight trend reversal, after which a rapid increase occurs from 2009 to 2011, potential suggesting an incentive to actively increase vehicle weight.

EU15 still accounts for around 94% of all new registrations (European Environment Agency, 2013).

Regardless of the level of abatement achieved by the CO_2 intensity regulations, their presence in combination with other instruments may reduce overall effectiveness of the instrument mix. For example, the parallel use of other instruments simply allows manufacturers to reduce efficiency improvements on conventional vehicles, as progress towards the CO_2 intensity targets is made via indirect means (using the current 'average fleet intensity' approach) (Maca *et al*, 2013).

The structure and application of vehicle registration and ownership (circulation) taxes are examples of other instruments. Ten Member States use vehicle-specific CO_2 emissions as a main parameter in registration tax calculation. Austria and France, for example, both employ 'feebate' systems. The evidence on the effectiveness of CO_2-related tax structures in reducing road transport emissions (primarily through influencing vehicle purchase choice) is generally positive, but the impact appears relatively small (Maca *et al*, 2013). For example, Kok (2011) concludes that such an approach in the Netherlands reduced the average CO_2 intensity of new cars sold from 142gCO_2/km in absence of such a tax structure, to 136gCO_2/km. As only ten Member States currently levy tax in terms of CO_2 intensity, it is unlikely significant abatement has been achieved. Econometric modelling by Ryan *et al* (2009) concurs, stating that registration taxes (of any structure) do not impact emissions above that induced by other instruments. In addition, Anderson *et al* (2011) find that by having different taxation systems for cars and light trucks based on fuel or CO_2 intensity (meaning inefficient cars are heavily taxed whilst efficiency light trucks are subsidised), a perverse incentive exists to redesign large cars as light trucks, reducing effectiveness of such instruments.

Company car taxation in four Member States is also determined at least in part by CO_2 intensity or fuel consumption. In the UK for example, the private use of a company (purchased and owned) car is taxed as a benefit in kind under personal income tax, with rates ranging from 5% of the purchase price (for CO_2 intensity up to 75gCO_2/km) up to a maximum of 35% (for CO_2 intensity of 225gCO_2/km and over). Although such a structure appears to have had a material impact (average CO_2 intensity of the company car fleet in the UK is lower, and decreased quicker than those in private ownership) (Veitch & Underdown, 2007), company car taxation arrangements across the EU present issues with incentives and interactions with other instruments. Whilst all Member States require an employee to declare company car use as an in kind benefit, the calculation to determine the proportion of the vehicle's catalogue price to be levied as taxable income varies (calculations may consider distribution of private and business use, age of the car, CO_2 intensity, etc.), although the range usually falls between 10% and 30% within and between Member States. It is also common practice for the employer to absorb the cost of the fuel (via fuel cards, for example), rendering the use of a company car in place of private vehicle ownership financially attractive to employees. The employer also benefits, primarily through the deductibility of the VAT paid for vehicle and fuel purchase, and maintenance and repair costs. A company car as an in kind benefit is also not liable for social security contributions (from either employer or employee), as opposed to the monetary salary component (Maca *et al*, 2013). The fiscal attractiveness of company cars to both parties explains their high share in the car fleet – in eighteen Member States in 2008, company cars accounted for around half (6 million) of all new car registrations (Copenhagen Economics, 2010). However, the structure of these arrangements is in conflict with the instruments discussed above, and below. The effect of registration and ownership taxation, even when graded by CO_2 intensity, is dulled or even reversed (to increase costs and therefore increase VAT deductibility, etc.), leading to company cars accounting for 70% of all new emission intensive cars in 2008 (Copenhagen Economics, 2010). In Germany, over 85% of high end cars are sold to companies, with some luxury car models registered exclusively as company cars (Federal Motor Transport Authority, 2013. This is exacerbated by the company (often) absorbing the cost of fuel, and thereby reducing the incentives for fuel (and CO_2) efficient vehicles and reduced consumption by the driver, along with the effect of fuel tax regimes. Graus & Worrell (2008) estimate that this phenomenon leads to a net increase in fuel consumption of 1-7% in the Netherlands, where the 11% share of company cars in the total car fleet consumes 21% of the fuel. Increased fuel consumption also lessens the abatement effect of biofuel-promoting instruments. Companies also absorb the cost of congestion charging or other road pricing, again reducing or removing the incentive for the driver to alter their behaviour.

Aside from the mechanisms above, which attempt to both 'push' and 'pull' the market towards cars with a lower CO_2 intensity, a number of Member States have implemented instruments to specifically increase the penetration of plug-in hybrid (PHEV) and electric vehicles (EVs), including registration and circulation tax exemption, additional preferential treatment under the bonus-malus system in France and direct subsidies, in order to reduce the difference in total cost of ownership (TCO) between PHEVs, EVs and conventional vehicles (the former two classifications have lower lifetime running costs but high capital costs, whilst the reverse is true for conventional vehicles – however the TCO for conventional vehicles remains much lower). Maca *et al* (2013) conclude that immediate financial incentives such a the French bonus-malus system and direct subsidies appear to be the most effective in reducing this differential and encouraging take-up of these vehicles, as this overcomes the high discount rates consumers apply to future savings in operational costs, whilst measures which accrue savings over the lifetime of the vehicle (such as reduced ownership taxes), are largely ineffective. However, the results vary based on the specific profile of the vehicle in question, and the usage.

Another instrument which attempts to 'pull' the market towards higher efficiency is Directive 1999/94/EC on consumer information on fuel economy and CO_2 emissions. This Directive was adopted in March 1999 to mandate that such information be displayed via vehicle labelling and in all means of marketing material[19]. Member States use a variety of different approaches to labelling to implement this Directive, although several employ a system that

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[19] From November 2012, Regulation 1222/2009 also made the labeling of tyres mandatory for cars, LGVs and HGVs, with information of fuel efficiency or rolling resistance, wet-weather adhesion and noise levels. However, as this is a relatively new instrument, it is excluded from further analysis of impacts.

mimics that for energy-related products (discussed below). Again, there is mixed evidence regarding labelling effectiveness in influencing purchase decisions. Whilst a survey by Deutsche Energie Agentur reported that labels were an important or very important factor in purchase decisions for around 63% of respondents (VDA, 2013), Codagnone *et al* (2013) found that respondents are only moderately familiar with the labels used, and that there is a gap between attitudes and actual behaviour (e.g. consumers first select vehicle class, then only consider environmental concerns when selecting a model) (Maca *et al*, 2013). The authors also conclude that by the time of the study, the impact of vehicle labelling had been negligible – a position supported by AEA (2011), and by Gartner (2005), some years earlier[20].

Burguillo-Cuesta *et al* (2011) find that whilst economic incentives (both policy and non-policy induced), regulations and informational instruments exert different levels of influence over purchase decisions and transport demand, psychological aspects such as habit, play a role. 'Soft Transport Policy Measures' (STPM) try to tackle such issues, and comprise a wide range of initiatives that attempt to encourage behaviour change by directly influencing an individual's decision making by altering their perception of the objective environment by altering their judgement of the consequences associated with different travel alternatives, and by motivating and empowering them to switch to alternative travel options (Bamberg *et al*, 2011). Several narrative reviews have concluded that STPMs have been effective in reducing car mileage and CO_2 abatement (Maca *et al*, 2013). The provision of travel planning services is one of the most widely implemented STPM, including the EU. The UK's Department for Transport (2004) report a reduction in car use of at least 2% in Breisgau-Hochschwarzwald (Germany) through the use of personalised travel planning, whist Transport for London (2004) report a 5-11% reduction in car use during the pilot for it's *TravelOption* software. Based on a review of thirty-two personalised travel planning programs in Sweden, Friman *et* al (2013) conclude that in seven, a reduction in car use of 22% was observed, whilst one initiative increased public transport use by 93% (with another producing just a 2% increase). The long-term impacts of such initiatives has been debated, although Taylor& Ampt (2003) and Richter *et al* (2009) conclude that longer-term studies are required to draw sound conclusions. Regardless, due to the small number of such initiatives and their generally city-specific scope, their impact on overall transport emissions is likely to be negligible.

CO_2 emissions from buildings decreased by 23% between 1990 and 2011, but remain at around 10% of total EU GHG emissions, with the residential sector accounting for 83-85% of total building emissions (European Environment Agency, 2014b). The Energy Performance of Buildings Directive (2002/91/EC, recast by 2010/31/EC) (EPBD) was the first instrument to take a holistic approach to encourage energy efficiency in the European buildings sector, and lays down requirements regarding six aspects of energy consumption in buildings. The most prominent of these are minimum energy performance requirements for new buildings, or for

[20] Refer to Maca *et al* (2013) for a more detailed discussion on the effectiveness of specific labeling types (e.g. relative vs. absolute).

existing buildings undergoing major renovation. Member States may set their own standards (using a standardised methodology based on cost-efficiency), and values may differ by category of building (e.g. residential, office, hospital). Linked to this is the second aspect – the requirement for all new buildings must be classified as 'Nearly Zero-Energy Buildings' (NZEBs) by 31st December 2020 (with all new buildings owned and occupied by public authorities achieving this by 2018). Member States may also interpret this requirement according to national circumstances. The third aspect is the provision of Energy Performance Certificates (EPCs), containing information on the energy performance on the building, along with reference values (such as minimum requirements), which must be issued and supplied to the new owner or tenant when a building changes ownership or occupancy. The remaining three aspects are a unified methodology for calculating building energy performance, inspection requirements for large heating and cooling systems and the use of independent control systems for the issuance of EPCs and heating and cooling system inspections (Drummond, 2013). Whilst the impact assessment for the 2010 recast of the Directive estimates annual energy savings of 60-80Mtoe and CO_2 savings of 160-210MtCO_2 by 2020 (equivalent to 5-6% of projected final energy demand and 4-5% of projected CO_2 emissions in 2020) (European Commission, 2008), there appears to be a gap in the literature regarding the effectiveness of the EPDB (both original and recast) and its provisions thus far (Drummond, 2013).

The Ecodesign Directive (2005/32/EC, recast by 2009/125/EC) aims at integrating environmental aspects into the design of energy-related products (energy-using products only, pre-recast), with the aim of improving the environmental performance of the product throughout its lifecycle (and to ensure effective functioning of the internal market by ensuring varied levels of product environmental performance does not produce a barrier to inter-EU trade). Thus far, twenty-one 'implementing measures' have been introduced under the Ecodesign 'framework' (plus two voluntary agreements), placing requirements on a range of products including domestic cooking appliances, water heaters, air conditioners, lighting products and refrigeration equipment, with requirements often focussing on reducing the energy intensity of product operation. Ex-ante assessments of the first twelve implementing measures project savings of 385TWg per annum by 2020, approximately equivalent to 14% of residential energy consumption in the EU in 2009. Whilst it is clear that most products covered by implementation measures are becoming more energy efficient, and that this Directive is at least in part driving this trend (particularly in the domestic and tertiary lighting sector), the data is unavailable to determine the extent of the contribution and progress towards the ex-ante assessments (CSES, 2012). The Energy Labelling Directive (92/75/EC, recast by 2010/30/EU) supports the Ecodesign Directive by requiring energy-related products subject to a delegated act under the Directive to supply a label and a fiche (table of information) containing information on the energy consumption (electric and other), and other resources, where relevant. Fourteen product types are currently covered by a delegated act (mostly those also subject to Ecodesign directive implementing directives). There a few studies examining the effectiveness of the Energy Labelling Directive, and the extent to which it has driven the observed increases in product efficiency. Waide & Watson (2013) found that almost half of respondents to their survey in a multi-country study considered energy efficiency to be a key aspect in informing their purchases, with energy labels informing these decisions, however some labels produced confusion, particularly with the introduction of 'A+' to A+++' energy efficiency classifications on labels for some products after the 2010 recast, which was also found to induce perceptions of diminishing returns (Heinzle & Wustenhagen, 2010; Mills & Schleich, 2010). As such, although it is not clear to what extent energy labelling has contributed to increasing product efficiency, the impact is likely weakened since the Directive recast (Drummond, 2013).

CO_2 emissions in agriculture in the EU decreased by around 18.7% between 1990 and 2011, and account for under 2% of EU GHG emissions (with non-CO_2 GHGs in agriculture much more significant, as discussed later in this section) (European Environment Agency, 2014b). CO_2 is emitted through fossil fuel combustion, but also through soil processes induced by agricultural activity (Kuik & Kalfagianni, 2013). Despite the long history of successfully promoting energy efficiency in greenhouse horticulture in the Netherlands, principally via negotiated agreements ('covenants'), few instruments attempt to tackle energy-related CO_2 emissions in agriculture across the EU. Similarly, few instruments exist to directly tackle CO_2 emissions from soils.

Promotion of Renewable Energy

Directive 2001/77/EC on the promotion of electricity from renewable energy sources in the internal electricity market (Renewable Electricity Directive), placed indicative targets on Member States to achieve a certain level of renewable electricity as a proportion of gross electricity consumption by 2010 (ranging from 6% in Belgium to 78.1% in Austria), to reach an average EU target of 21%. This Directive was superseded by Directive 2009/28/EC on the promotion of the use of energy from renewable sources (Renewable Energy Directive, 'RED'), and whilst maintaining the indicative targets for renewable electricity, set binding targets on Member States for renewable energy[21] in total gross final consumption by 2020 (from 10% in Malta, to 49% in Sweden), producing a target of 20% at the EU level, as laid down by the 2008 Climate and Energy Package. The RED also contains several 'enabling' provisions, such as requiring priority dispatch for renewable electricity, and the development of grid infrastructure (building on those already present in the preceding Directive). The proportion of renewable energy in gross final consumption increased steadily between 2004 and 2011, from 8.3% to 12.9%. Renewable electricity accounted for around 30% of this increase (with heating, cooling and transport sectors accounting for the remainder) (EUROSTAT, 2014).

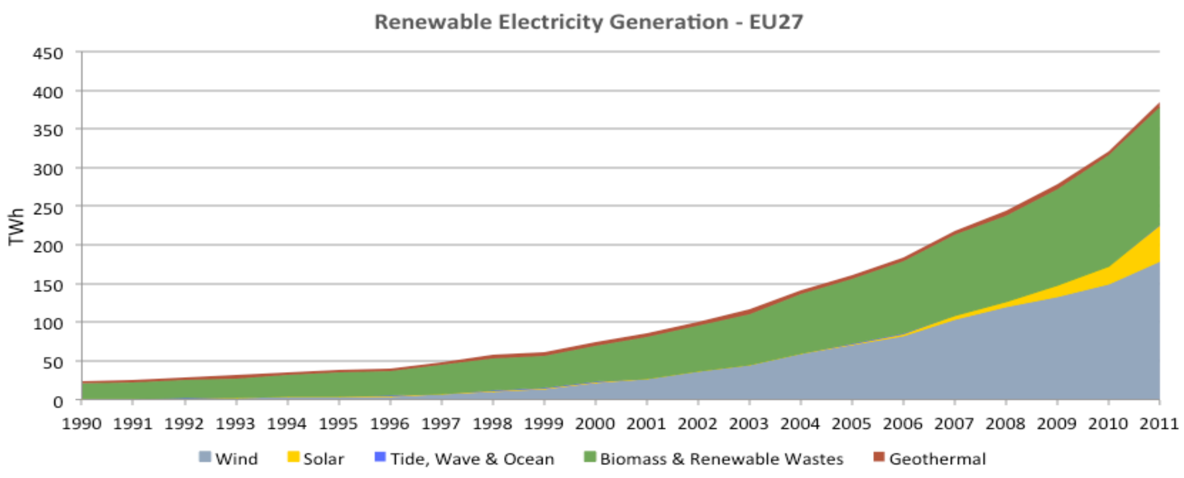
By 2010, RES-E accounted for 19.9% of EU gross electricity consumption, missing the 21% objective, as fifteen Member States failed to reach their non-binding targets (European Commission, 2013). Despite this, Figure 5 illustrates the rapid increase of RES-E generation between the late 1990s and 2011, whilst Figure 6 places this in context of total electricity

[21] Renewable energy sources (RES) considered by the Directive are: wind power (onshore and offshore), solar power (PV and solar thermal electricity), geothermal power, hydropower, wave power, tidal power, biomass and biogas (landfill and sewage gas).

generation[22]. Agnolucci & Drummond (2014) highlight that the literature is in general consensus that the rapid increase in RES-E experienced since the adoption of Directive 2001/77/EC is due entirely to targeted RES-E support mechanisms and enabling initiative, with preceding growth also due to pre-existing instruments driven by individual Member States. Each of the EU15 already had some form of support mechanism in place by 2000 (although only ten had 'major' instruments) (Kitzing *et al*, 2012).

Meyer & Meyer (2013) ran two scenarios to estimate the impact of RES-E support mechanisms (and by broad extension, the Renewable Electricity/Energy Directives). The first assumes that investment in renewable capacity displaces investment into conventional energy sources, whilst the second assumes that renewable capacity is in addition to investment in conventional carriers. The authors suggest that the true situation is somewhere in between these binary scenarios, the latter is the more likely one. Both scenarios assume that no investment in additional RES-E would have taken place from 1998 onwards in the absence of targeted support mechanisms, For 2008, the authors estimate average CO_2 reductions of 3.9% and 3.2% for each scenario in Member States, respectively, with the difference due to increased demand for generation equipment (and thus supply chain emissions), in the latter scenario. Both scenarios exhibit a significant range between Member States (although of slightly different magnitudes), dependent on the level of observed RES-E deployment. Member States with particularly high deployment such as Germany, Portugal and Spain have experienced the highest abatement (7.88%, 6.91% and 5.03%, respectively), whilst the opposite holds for those with low RES-E deployment such as Poland and France (0.72% and 0.57%, respectively)[23].

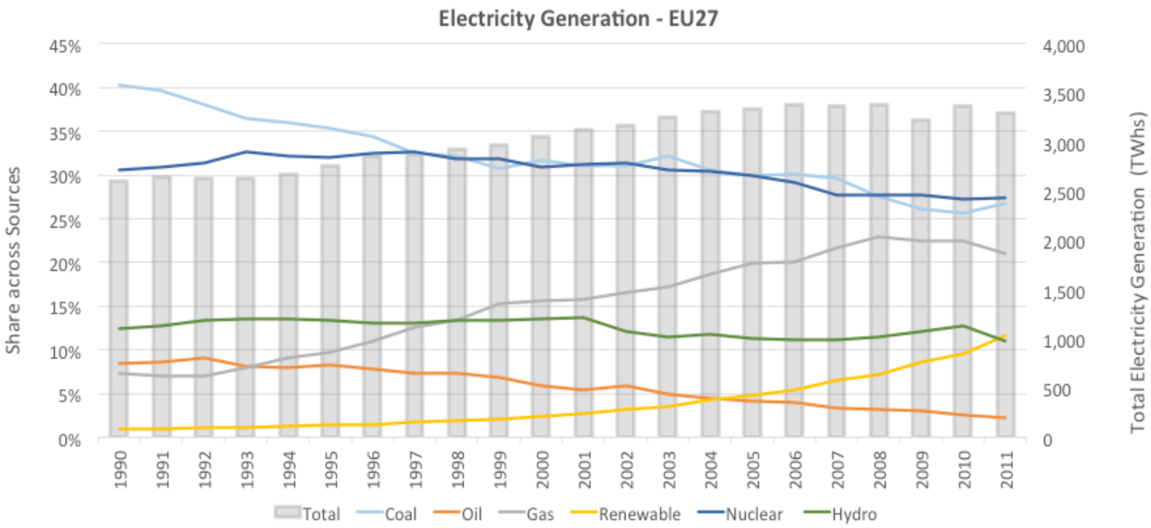
Figure 5 - Renewable Electricity Generation Trends - EU27 (Source: EUROSTAT)



[22] As Croatia joined the EU in July 2013, after the end of the trend presented, it is not included in the analysis.

[23] Results presented are for the scenario with no investment displaced from conventional carriers, as this is more representative of reality (Meyer & Meyer, 2013).

Figure 6 - Gross Electricity Generation, in Total and by Source - EU27 (Source: Agnolucci & Drummond, 2014)



Feed-in Tariffs (FiTs), in which guaranteed rates are paid to generators of renewable electricity per unit generated and exported to the grid (often for 10-20 years), are the most common form of primary RES-E support mechanism in the EU (in use in fifteen Member States - often in combination with direct subsidies) and are generally considered the most effective support mechanism as they provide long-term financial certainty upon which investment decisions may be taken. 'Green certificates', by which governments impose an obligation on consumers, suppliers or generators to source or generate a certain proportion of their electricity from renewables (as used in six Member States) are considered less effective, as they inherently present a higher risk to investors (Butler & Neuhoff, 2008). Whilst Ragwitz *et al* (2012) conclude that around 93% of onshore wind capacity, and nearly all solar photovoltaic capacity installed by the end of 2010 across the EU was initiated by the presence of FiTs, the specific design of each national scheme is a key feature in determining the distribution of deployment between Member States. Germany and Spain, leaders in RES-E deployment in the EU[24], are both considered as having 'best-practice' FiT design[25]. However, regardless of the instrument employed, both the European Commission (2009) and del Rio & Tarancon (2012) highlight instrument stability (e.g. tariff level, budget, access criteria) as a critical factor in effectiveness.

It is often argued that the presence of instruments specifically to promote the penetration of RES-E reduces the abatement effort required under the EU ETS (producing economic inefficiencies, discussed in the following section) (Frondel *et al*, 2010). However, the abatement expected from increasing RES-E penetration (in the context of indicative targets set under the Renewable Electricity Directive, and subsumed by its successor) was considered in the various cap-setting exercises for the EU ETS. As such, only overachievement of these

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[24] In 2010, Germany accounted for 35% and 66% of installed wind and solar PV capacity in the EU respectively, whilst Spain held 2% of wind and 20% of solar PV capacity (Agnolucci & Drummond, 2014).

[25] Exemplar components of 'best-practice' design include stepped tariffs (appropriate to the local resource), and tariff degression, in which payments are reduced over time reflecting falling costs (Ragwitz *et al*, 2012)

indicative targets would induce such an effect. As fifteen Member States failed to meet their 2010 targets (European Commission, 2013), this is unlikely that this effect has yet occurred (Agnolucci & Drummond, 2014).

The production of biogas, which may be produced from a number of feedstocks including agricultural waste such as manure and crop by-products (along with landfill waste and sewage sludge), and often used in CHP installations to produce both heat an electricity[26], has rapidly increased in the agricultural sector since 2004, moving from around 400MW capacity to nearly 4,500MW by 2011. Twenty-two Member States currently provide support for such production, although 78% of capacity exists in Germany and Austria alone (Kuik & Kalfagianni, 2013).

The RED, as part of the overarching 2020 target for 20% renewable energy in gross final consumption and constituent Member State targets, also contains a sub-target for 10% of energy in transport to be sources from renewables by 2020, applicable to all Member States. Prior to the adoption of the RED, Directive 2003/30/EC on the promotion of the use of biofuels and other renewable fuels for transport (the 'Biofuels Directive') required that Member States adopt measures (and indicative national targets) to achieve a 5.75% share of biofuels in transport by 2010. Due to the likelihood of this target being missed, along with increasing concerns regarding the environmental and social impacts of biofuel production, this was superseded by the RED target, which considers all renewable sources (and implemented biofuel sustainability criteria). However, the vast majority of Member States have chosen to comply with this new target by continuing the promotion of biofuels, often doing so by obliging suppliers of motor fuels to ensure a specific proportion of their sales are biofuels (largely met through blending biofuel with diesel and petrol). The provisions of the Biofuel Directive and subsequent RED have increased the share of renewable energy in transport in the EU (almost entirely biofuel) from 1% in 2004, to 4.8% in 2010 (twenty-two Member States failed to meet their indicative biofuel targets, missing the 5.75% target), to 5.1% in 2012. Increasing renewable transport accounted for around 15% of the increase in renewable energy in gross final consumption between 2004 and 2011 (EUROSTAT, 2014). However, lifecycle emissions from biofuel production (including those from indirect land use change (ILUC)) mean that the magnitude of abatement is likely to be small. The biofuel sustainability criteria introduced by the RED in 2009 stipulates that biofuels must produce a minimum GHG saving of 35% against conventional fuels. If this were achieved, a 5.1% biofuels share would have, *ceteris paribus*, produced emission savings of around 1.8% against the counterfactual.

The use of renewables for space and water heating and cooling in buildings and for industrial and other commercial purposes in the EU28 increased from 9.9% to 15.6% between 2004 and 2012 (EUROSTAT, 2014), accounting for the remaining 55% of the increase in renewables in final energy consumption between these years.

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[26] Although it can be used to produce one of the other, and may be upgraded to biomethane to be injected into the national gas grid or as a transport propellant (Kuik & Kalfagianni, 2013).

Non-CO_2 GHGs

Non-CO_2 emissions accounted for around 12% of total EU GHG emissions in 2011, mostly from methane (CH_4) and nitrous oxide (N_2O). The majority of such emissions in the EU (and globally) stem from the agriculture sector (excluding LULUCF), in which around 85% of total GHG emissions in 2011 (in CO_2e terms) were non-CO_2 (European Environment Agency, 2014). Agriculture emissions in the EU reduced by around 22.5% between 1990 and 2011 (European Environment Agency, 2014). This was achieved through a 23.5% reduction in both CH_4 and N_2O (CH_4 emissions are mainly produced from the rearing of livestock and manure handling, whilst N_2O is emitted through the application of fertiliser and manure to soils (Kuik & Kalfagianni, 2013)), and an approximate 18.7% reduction in CO_2 (European Environment Agency, 2014). The largest proportional reductions were achieved in Central and Eastern European Member States (up to around 64% in Latvia (Kuik & Kalfagianni, 2013).

Policies and instruments for the mitigation of non-CO_2 GHGs are relatively new in the EU, and whilst some specific instruments do exist (such as the UK's Greenhouse Gas Action Plan (GHGAP)[27]), they are largely very recent, focus on information dissemination and R&D efforts (and therefore are without significant ambition in the short-term), and are implemented on a voluntary basis (Kuik & Kalfagianni, 2013). As such, as with agricultural CO_2 emissions, the reduction in CH_4 and N_2O emissions over time is driven instead by non-climate related policy instruments, and non-policy drivers. The decrease in CH_4 emissions can be attributed to significant decreases in cattle numbers that followed increases in animal productivity (milk and meat), and related improvements in the efficiency of feed use. Reductions in N_2O emissions are largely a result of the reduced use of mineral and organic fertilisers following productivity increases and declines in cattle herds, but also due to the implementation of provisions in the Nitrates Directive (91/676/EEC) (also a partial driver behind CO_2 emissions from soil processes), which now forms part of the Water Framework Directive (2000/60/EC) (Fernagut *et al*, 2011).

The Common Agricultural Policy (CAP) is also likely to have been a driver for abatement, to some extent. Reforms initiated in 1992 (MacSharry Reform) introduced several changes to the CAP, including an increased focus on environmental protection though specific initiatives aimed at encouraging agri-environment programmes, afforestation and crop diversification. These initiatives were expanded by Agenda 2000 (and further strengthened in 2003), which divided the CAP into two 'Pillars'; support to farmer's incomes via direct payments and market measures, and secondly, support for rural development programmes co-financed from the European Agricultural Fund for Rural Development. Agricultural GHG abatement in Italy, at least, is largely driven by this second pillar (Kuik & Kalfagianni, 2013).

[27] Refer to Kuik & Kalfagianni (2013) for further information on the GHGAP.

3 Economic Impacts & Efficiency of the EU's Climate Policy Instrument Mix

This section assesses the economic efficiency (static and dynamic), of the existing climate policy instrument mix, along with broader economic impacts. GDP in the EU28 grew by 85% between 1995 and 2013, growing at an average rate of 4.9% between 1995 and 2007, before rapidly reducing to 0.6% in 2008 and contracting by 5.8% in 2009 at the onset of the financial crisis, before slowly recovering to an average growth rate of 2.6 between 2010 and 2013 (EUROSTAT, 2014). In their assessment, Meyer & Meyer (2013) conclude that the presence EU ETS, renewable electricity promotion and ETRs likely increased both GDP growth (but not reduced) and employment against the counterfactual in most Member States (with the exception of some smaller transition countries).

Carbon Pricing

As a cap-and-trade instrument covering over 40% of the EU's GHG emissions (50% of CO_2), theoretically allows abatement to be achieved where it is cheapest within the obligated industries across the EU28 (plus Norway, Iceland and Lichtenstein), producing an equalised marginal carbon price. This affords EU ETS a relatively high static efficiency (Drummond, 2013). As with the production of any good, the costs of production are expected to pass through to the prices paid by the final consumer. Such logic stands for the generation of electricity and the EU ETS price (Agnolucci & Drummond, 2014). The wholesale power spot market[28] is set by the marginal costs of the marginal generator, which under perfect competition and with a low price elasticity of demand should include 100% carbon cost pass-through (CCPT) – defined as a change in the wholesale price resulting from a change in the CO_2 price, as firms have little ability to absorb the additional cost (Guilli & Chernyavs'ka, 2013). However under real-life circumstances, which hold imperfect competition and other distortions, CCPT rates may be higher or lower than 100% (including negative), depending of whether the demand curve is linear or isoelastic, as firms maximise profits either by increasing the CCPT rate (and absorbing reduced demand) or reducing the CCPT Rate (and absorb reduced margins) (Agnolucci & Drummond, 2014), or they pursue a strategy other than simple profit maximisation, such as a longer-term profit target (consisting of constant profit whilst minimising price volatility, possibly in order to reduce the risk of regulatory intervention). In addition, CCPT rates vary by time (peak and off-peak) and by Member State, depending on the market structure of the power system[29]. At the same CCPT profile (and with generally parallel market structures), the total average additional cost to wholesale power prices as a result of the EU ETS would be expected to vary across Member States in proportion to the average CO_2 Intensity of generation (for example, higher costs would be expected in Poland than in France). Agnolucci & Drummond (2014) attempt to calculate this

[28] Although only a relatively small proportion of electricity is traded on the day-ahead spot market (25% in Germany (Pietz, 2009), these spot prices inform the pricing of there electricity contracts (Tveten *et al*, 2013).

[29] For further analysis and discussion surrounding CCPT rates, please refer to Agnolucci & Drummond (2014).

additional cost in Germany and Spain, with total average additional costs peaking in 2005 (at an additional €12.77/MWh and €9.01/MWh for Germany and Spain respectively, representing a 34% and 20% increase on the counterfactual wholesale price, respectively), with lows in 2007 (at an additional €0.28/MWh and €0.17/MWh for Germany and Spain respectively, representing a 0.6% and 0.4% increase on the counterfactual wholesale price, respectively). However, these values are crude estimates and must be used with care[30]. As the majority of EUAs allocated under the first two Phases of the EU ETS were grandfathered, any positive CCPT rate generates 'windfall' profits for generators, as confirmed by the literature. For example, Venmans (2012) concludes that windfall profits in the electricity sector in Phase 1 are likely to have been €19-25 billion per year. In addition, Koch & Bassen (2012), Oberndorfer (2009) and Veith *et al* (2009) analyse the share price of listed generation companies, and found that investors understood that the EU ETS had thus far increase revenues for the European power sector through free allowance allocation (Agnolucci & Drummond, 2014). However, from Phase 3, allocations to the power sector are fully auctioned (with transition arrangements for some Member States).

One of the primary economic concerns surrounding the EU ETS (and other climate policy instruments) is the negative effect of the additional cost burden on industrial competitiveness, leading to a loss of economic activity in the EU and in the case of the relocation of production, 'carbon leakage' ('pollution haven' effect). 'Competitiveness' may be defined at the level of the firm, sector, or nation (or supranational union, such as the EU)[31]. At the firm or sectoral level, competitiveness is defined as either the 'ability to sell' (capacity to increase market share, measured through indicators involving exports, imports and domestic sales), or 'ability to earn' (capacity to increase margins of profitability, measured using indicators concerned with profit or stock values). Only loss of competitiveness as 'ability to sell' may lead to carbon leakage. It is also useful to define two types of leakage – 'operational' leakage, by which a reduction in competitiveness of domestic production reduces the utilisation rate of existing capacity in favour of other regions, and 'investment' leakage, but which long-term changes in production capacities occur (Kuik, Branger & Quirion, 2013).

Not all sectors face the same risk of carbon leakage. Those which face the highest risk are those in which the cost of policy compliance is high (measured under the EU ETS as the carbon costs relative to the gross value added of the sector), and in which international competition is intense (trade intensity, measured by the ration between imports plus exports, and the size of the internal EU market). The industries most vulnerable to these factors are classified as Energy Intensive Trade Exposed (EITE) sectors. EITE 'at risk' sectors include cement, ceramics, coke, glass, refineries, iron and steel, and aluminium, which together account for around a third of all EU ETS emissions (two-thirds of non-electricity emissions) (Kuik, Branger & Quirion, 2013).

[30] Refer to Agnolucci & Drummond (2014) for more information.

[31] However the notion of 'national' competitiveness is controversial, and often considered meaningless (Kuik, Branger & Quirion, 2013).

Whilst much of the *ex-ante* analysis for carbon leakage predicted leakage rates of between 5% and 20% (Branger & Quirion, 2013), thus far, no evidence has been uncovered in *ex-post* studies to suggest that a loss of competitiveness or carbon leakage (operational or investment) has occurred amongst the EITE sectors as a result of the EU ETS (Kuik, Branger & Quirion, 2013). Zachman *et al* (2011) also found no evidence that the ETS affected firm's profits (and therefore 'ability to earn'). Branger & Quirion (2013), alongside examining the impact of the EU ETS on cement sector emissions, also investigate the evidence for competitiveness impacts and carbon leakage for cement and the other largest industrial CO_2 emitter in the EU - the steel sector – and confirm that a loss of competitiveness and subsequent (operational) leakage has not occurred (although, the authors highlight that there is not yet enough evidence to confirm whether or not investment leakage has occurred). There are a number of possible explanations for this difference between *ex-ante* and *ex*-post results. A primary explanation may be due to the free allocation of allowances in the first two Phases removing the incentive to abate (as discussed above for the cement sector, but applicable to all industry sectors under the EU ETS), meaning these sectors are not only (heavily) protected from any additional cost burden (which in itself, due to the persistent low average carbon price, is likely to be minor), but are actively incentivised against carbon leakage (both operational and investment) in order to maintain higher EUA allocations. The Phase 3 allocation rules are likely to have reduced such disincentives (although the closure rules maintain distorting effects), however such changes are too recent for empirical evidence to be produced. In addition, many EITE sectors hold long-term electricity supply contracts, and as such are protected from carbon price and CCPT rate volatility in electricity prices (Sijm *et al*, 2006). Other assumptions or omissions from *ex-ante* modelling may also contribute. For example, carbon price liabilities are one factor out of many other quantifiable and non-quantifiable cost and benefits that contribute to industrial competitiveness (e.g. capital abundance, labour force qualification, proximity to customers, infrastructure quality) (Monjon & Hanoteau, 2007), of which may are simply not accounted for or about which simplified assumptions regarding their future development are made. Also, such industries are paradoxically often less prone to leakage, as they are very capital intensive and less prone to relocate compared to 'footloose' industries (Ederington *et al*, 2003). Moreover, *ex-ante* projections do not (or only vaguely) consider positive aspects of environmental regulation, such as first mover advantage, climate 'spillovers', and the Porter Hypothesis[32].

Despite this, Meyer & Meyer (2013) find that the EU ETS (in 2008) is indeed likely to have slightly reduced GDP and employment in most Member States by an average of 0.5% and 0.34%, respectively (with the highest impacts felt in Estonia, Bulgaria and the Slovak Republic), due to EITE industries pricing-in the opportunity cost of the freely allocated allowances to their product increasing costs in downstream sectors.

[32] Mazzanti & Antonioli (2013) provide a literature review and full discussion of the Porter Hypothesis.

Despite high static efficiency, due to a persistent lack of a high and predictable carbon price exacerbated by the financial crisis creating long-term oversupply and the presence of free allocation of allowances in the first two Phases, the EU ETS provides low dynamic efficiency (Drummond, 2013; Mazzanti & Antonioli, 2013). However, Calel & Dechezlepretre (2012) find that obligated sectors (both power and industry) increased their low-carbon patenting by 36% as a consequence of the EU ETS, whilst Martin *et al* (2012) concluded that around 70% of obligated firms engage in 'clean process innovation' (formal or informal R&D aimed at curbing emissions and/or energy consumption), whilst 40% engage in 'clean product innovation' (formal or informal R&D aimed at developing products to help consumers reduce emissions). However, it is not clear to what extent such innovation is driven by the EU ETS, or by other policy and non-policy drivers. In the power sector, there is some evidence to suggest that the EU ETS did spur investment in Carbon Capture and Storage (CCS) R&D (process innovation) (Rogge & Hoffmann, 2010; New Energy Finance, 2009; Mazzanti & Antonioli, 2013). However, Hoffman (2007) highlights that the most significant drivers behind R&D investment in CCS are dedicated EU research programmes, which began pre-ETS. Agnolucci & Drummond (2014) conclude that any additional financing of CCS R&D induced by the EU ETS is likely to be minor, a position supported by Rogge *et al* (2011). In addition to the power sector, Mazzanti & Antonioli (2013) investigate the impact of climate policy instruments in the EU on the chemical, ceramics, coke and refinery, paper and cardboard and steel sectors, and find that the EU ETS has had little effect on technical innovation in these sectors (although evidence for 'organisational' innovation was found, discussed later in this section).

Conversely to the EU ETS, by establishing tax minima that vary extensively in implicit CO_2 price between different energy products, and permit significant variation by Member State (e.g. Figure 2) and between end uses (exacerbated by the exemption of electricity generators, significant proportions of the energy intensive industry, domestic heating and agriculture, despite relatively wide remaining sectoral coverage), the ETD is statically inefficient. Due to the incentive for arbitrage, the distribution of tax revenues between Member States is also affected, to the point where a small country prone to receiving fuel tourists (e.g. Luxembourg and Austria) may increase its revenue by lowering fuel taxes, offsetting the falling tax revenue from domestic consumers by an increase in consumption from foreign consumers. Such a pressure to maintain revenues prevents national governments' freedom to introduce welfare-maximising prices for road fuel that reflect the full external costs of transport, which may be in line with the preferences of the national electorate (Maca *et al*, 2013). The dynamic efficiency of the ETD is also low. In road transport (where the effects of the ETD are most readily felt), although a continued incentive for fuel efficiency (and therefore abatement) is present for private and commercial vehicles. Evidence is more forthcoming for induced technical energy efficiency innovation in the industry sector, such as the ratio of natural gas to electricity consumption in the steel making process. Significant 'organisational' innovation has occurred over recent years across industrial sectors, with Mazzanti & Antonioli (2013) conclude that energy taxation have had some influence on this, along with the EU ETS and RED. Such innovations include the introduction of energy consumption and emissions monitoring activities and systems (such as

Environmental Management Standards and ISO systems), and the use of renewable energy at the firm level. However, Mazzanti & Antonioli (2013) also find that the presence of numerous climate policy instruments, at both EU and Member State level, may in fact hinder innovation by generating confusion and potential perverse incentives. Once such example is the presence of several instruments acting on passenger cars, as discussed previously.

Meyer & Meyer (2013) conclude that the effect of ETRs introduced in Member States since 1998 on competitiveness (and carbon leakage) is also negligible, with a slightly net negative effect on both GDP and employment by an average of approximately -0.19% and -0.6%, respectively. However, whilst this result assumes tax neutrality by full recycling of revenue via reduced income taxes (a simplified assumption, although this did occur in many Member States when implementing reforms), the consequential reduction of labour costs is not considered, and therefore the 'double dividend' effect of reducing emissions whilst stimulating GDP growth and employment is removed. As such, the negative impact on GDP and employment is likely to be even less. Miltner & Salmons (2009) support this position by calculating that out of fifty-six cases (eight EITE sectors and seven Member States), the impact of ETRs on competitiveness was insignificant in 80% of cases, negative in only 16%, and even positive in 4%.

Energy Efficiency and Energy Consumption

As the CO_2 intensity standards for passenger cars (Regulation 443/2009) allows manufacturers to meet their obligations in the most-cost effective manner available to them, and is applicable to all Member States, it has high static efficiency. Implicit marginal carbon abatement costs of reaching the 130gCO_2/km by 2015 are estimated at €6-54/tCO_2, whilst vehicle prices have remained the same since the announcement of the Regulation in 2007 (Maca *et al*, 2013), or potentially even decreased (Varma *et al*, 2011), indicating no additional cost to the consumer (or if there was, it is offset by factors working in the opposite direction) (Drummond, 2013). Registration and circulation taxes reframed in terms of CO_2, although preferential to such taxes reframed non-CO_2 terms, have low static efficiency due to the low coverage they currently enjoy. In addition, in the first year of such reframing in Ireland from July 2008, a 33% reduction in tax revenue was experienced (Rogan *et al*, 2011)[33]. Passenger car labelling, via Directive 1999/94/EC on consumer information on fuel economy and CO_2 emissions, along with 'soft' policy measures are also statically inefficient due to their narrow scopes – however the cost of implementation of such instruments is likely to be negligible. The implicit carbon costs of capital subsidies for AFVs for a selection of fifteen OECD countries (including Denmark, Estonia, France, Germany, Spain and the UK) range from around €150/tCO_2 to €1,050/tCO_2, with an average of €420 (OECD, 2013b).

[33] A long-term driver of reductions in revenue from such a mechanism is the incentive to modify vehicles that fall close to cut-off points in 'tax notches', when a marginal change in fuel economy can create a significant change in tax treatment, marginally increasing effectiveness but reducing economic efficiency (Sallee & Slemrod, 2012).

The dynamic efficiency of CO_2 intensity regulations, car labelling, registration and circulation taxes, direct subsidies for AFVs of 'soft' policy measures, whilst all reduced in the presence of company car taxation distortions, are further reduced in combination with each other. Whilst the CO_2 intensity standards provide a binding medium-term target, encouraging dynamic efficiency with a continued incentive to innovate to meet these requirements in the cheapest manner possible (although no incentive exists to exceed the target), each of the other instruments listed aim at altering consumer preference in vehicle and mode choice and help to 'pull' the market in the direction these standards are designed to 'push', thereby helping to achieving the same via different methods. Such overlap indicates economic inefficiency, achieving the same ends at potentially higher cost.

The static efficiency of the EPBD, Energy Labelling Directive and Ecodesign Directive is low, both individually and in combination. The scope of each instrument is narrow (new and refurbished buildings, and specific energy-related products), and all focus on improving energy efficiency, rather than GHG emissions directly. As the EPBD allows Member States to set their own standards based on cost-efficiency, an equal incentive for efficiency and GHG abatement, by proxy, is not present. The Energy Labelling and Ecodesign Directives are also likely to have produced negligible costs to the economy or individual market actors (Zhou, 2013; CSES, 2012), with ex-ante estimates for the latter instrument projecting net savings to the economy of €100 billion between 2005 and 2020. The dynamic efficiency of the EPBD and Ecodesign directive is low, as no incentives are provided to exceed minimum standards. The Energy Labelling Directive conversely, by providing information on environmental performance to the customer relatively to competitors, does provide some incentive to continually improve (although this may be tempered by increasing confusion with the 'beyond-A' label rating) (Drummond, 2013). Despite this, there is evidence for innovation occurring as a result of all three instruments. Noailly & Batrakova (2010) find evidence that such instruments have induced the generation and diffusion of innovations in new materials, fluorescent lighting and condensing boilers, whilst Mazzanti & Antonioli (2013) find evidence that building regulations (under the EPBD) have been important for innovations generated in the steel sector.

Promotion of Renewable Energy

Despite the economy-wide scope, as the RED (and preceding Renewable Electricity Directive) mandates a particular method of emissions abatement (i.e. the growth of low-carbon energy), and does so through targets differentiated by Member State, which in turn use different support mechanism designs producing differentiated implicit carbon prices (demonstrated by the extremely varied support per unit of RES-E generation between Member States, illustrated in Table 3), it cannot be considered statically efficient (Drummond, 2013). The implicit carbon price also varies significantly between Member States (based on the level of support provided by support mechanisms), and between technologies. In Germany for example, Marcantonini & Ellerman (2013) find abatement costs range between €43/tCO_2 for wind energy and €537/tCO_2 for solar PV, calculated as the ratio of the net cost of such installations supported by feed-in tariffs and the attributable CO_2

abatement, as an average between 2006 and 2010. A similar range is found in the UK (OECD, 2013b). Implicit costs of biofuel promotion policies (to reach the 10% renewable transport RED sub-target) also ranges significantly. For example, in France, biofuel polices produce a cost of around €125/tCO_2 and around €230/tCO_2 in the UK (OECD, 2013b).

As RES-E technologies exhibit low marginal costs, and have guaranteed access to the grid in many Member States, renewable generation ranks first in the merit-order. As such, with increasing RES-E penetration, the supply curve shifts to the right, reducing average wholesale electricity prices (the 'merit-order effect'). The majority of studies examining this effect in the EU have focussed on Germany and Spain, with amongst the highest proportion (absolute) penetration of RES-E. Table 2 presents average estimated reductions in wholesale power prices in Germany and Spain as a result of increasing RES-E, between 2005 and 2010.

Table 2 - Estimated Average Annual Reduction of Wholesale Spot Market Prices in Germany due to RES-E Generation (Source: Agnolucci & Drummond, 2014)

Year	Average Estimated Reduction in Wholesale Prices (€/MWh) - Germany	% Reduction from Average Wholesale Spot Price - Germany	Average Estimated Reduction in Wholesale Prices (€/MWh) – Spain	% Reduction from Average Wholesale Spot Price - Spain
2005	-4.90	8%	-8.36	13%
2006	-8.73	12%	-7.07	12%
2007	-10.94	19%	-8.48	18%
2008	-12.47	14%	-7.19	10%
2009	-6.53	13%	-8.07	18%
2010	-5.51	10%	-11.15	23%

Incidences of negative wholesale spot prices have also occurred in Germany, along with some other Member States. This occurs when inflexible generation (such as nuclear or coal generation) meets low demand in the presence of significant RES-E generation. Inflexible generators submit negatively priced bids to allow them to remain generating, as the process of shutting down and restarting such plants is often more expensive (Agnolucci & Drummond, 2014). Whilst RES-E penetration depresses wholesale prices, the costs associated with operating RES-E support mechanisms are recovered via a levy added to electricity retail prices. Table 3 presents total RES-E support expenditure for eighteen Member States (both in absolute terms, and per MWh of total final electricity consumption) along with the support level paid for MWh of renewable power supplied to the grid, calculated as a weighted average across all RES-E technologies.

Table 3 - Support Mechanism Costs in Different Member States in 2010 (Source: CEER, 2013)

Member State	Weighted Average Support Level (all RES-E) (€/MWh)	Total RES-E Support Expenditure (€m)	RES-E Support Costs per unit of final electricity consumption (€/MWh)
Austria	50.91	378	6.17
Belgium	126.12	729	8.75
Czech Republic	113.37	488	8.23
Estonia	53.55	42	6.01
Finland	6.12	16	0.19
France	86.19	1,511	3.40

Germany	115.60	9,512	17.98
Hungary	101.89	247	5.81
Italy	112.17	3,427	10.38
Luxembourg	99.76	14	2.11
Netherlands	76.70	690	6.46
Norway	9.17	15	0.12
Portugal	55.84	752	15.07
Romania	55.00	37	0.90
Slovenia	49.57	36	2.97
Spain	87.98	5,371	20.61
Sweden	27.98	483	3.68
United Kingdom	126.17	1,438	4.38

A quota obligation (green certificate) system is theoretically more cost-effective than a FiT system, as the pre-defined quota should be met at least cost, since – in contrast to the FiT – there is little opportunity for producers to reap excessive profits; if the quota market functions correctly, any cost degradation for renewables should be reflected in lower levels of support. However, Butler and Neuhoff (2008) found that this is not the case when comparing the UK quota system and German FiTs, finding a lower cost per unit of RES-E generation delivered in Germany. They explain this first by suggesting that the financial certainty provided by FiTs reduces regulatory and market risk, and thus the cost of capital. A second component is the finding that stronger competition exists in Germany between wind turbine producers and constructors than in the UK – opposite to what might have been expected. As these are the stages of the value chain that contribute most to the total cost of a wind installation, increased competition here has a strong impact on total cost (Butler and Neuhoff, 2008).

Whilst the overlap between the RED and EU ETS is unlikely to have impacted the effectiveness of either instrument, as described above, it remains an economically inefficient interaction as the RED (despite an economy-wide scope) obligates the introduction of renewables in EU ETS sectors (particularly power generation), with a marginal cost of abatement that varies between Member State and technology, and is may not be as low as that achieved by the market-based approach of the EU ETS alone (Drummond, 2013). The EU ETS and RED have opposing effects on the wholesale price of electricity (EU ETS increases, whilst the RED decreases). Table 4 illustrates a basic estimate of the net impact of these two instruments in Spain and Germany, between 2005 and 2010.

Table 4 - Impact of RES-E and EU ETS on Wholesale Spot Prices in Germany and Spain (Source: Agnolucci & Drummond, 2014)

Year	Average Reduction in Wholesale Prices		Average annual EU ETS cost		Average Annual Change to Baseload Wholesale Prices	
	Germany (€/MWh)	Spain (€/MWh)	Germany (€/MWh)	Spain (€/MWh)	Germany (%)	Spain (%)
2005	-4.90	-8.36	12.77	9.01	21%	1%
2006	-8.73	-7.07	7.84	5.03	-2%	-4%
2007	-10.94	-8.48	0.28	0.17	-22%	-17%
2008	-12.47	-7.19	7.84	4.31	-7%	-4%
2009	-6.53	-8.07	6.69	3.29	0%	-11%
2010	-5.51	-11.15	6.51	2.51	2%	-19%

It is clear from Table 4 that the net influence of the EU ETS and RED can be significant and highly volatile over time, but with the RED exerting the largest influence, producing net wholesale prices below that of the counterfactual. Although these instruments act in opposing directions on the wholesale price within a Member States, at an EU level they act in concert to increase wholesale price differentials between Member States. In Member States with high average CO_2-intensity of generation EU ETS liabilities would be high, with the merit-order effect from renewables small, and vice-versa in Member States with low average CO_2-intensity. This incentivises an increase in electricity trade (with the differential-producing effects of EU ETS alone expected to have been responsible for around 10% of ETS-attributable abatement in Phase 1, according to Delarue *et al* (2008). Despite the reduction in wholesale prices, RES-E support mechanism costs still produce a significantly positive net cost. In 2010, wholesale power price reductions in Germany produced savings of around €2.8 billion, whilst support mechanisms cost around €9.5 billion - a net cost of around €6.7 billion (Agnolucci & Drummond, 2014). Although this cost is usually borne by electricity consumers, there is often inequity between consumer groups. In Germany, costs are recovered by the 'EEG (Erneuebare-Energien-Gesetz) surcharge'. Electricity intensive firms may call upon a special hardship rule that considerably reduces their payments[34], which places additional burden on non-electricity intensive firms to compensate, including households, which in addition must pay 19% VAT on the charge (Agnolucci & Drummond, 2014).

Meyer & Meyer (2013) estimate that investment in RES-E in the EU increased both GDP and employment by an average of 0.32% and 0.09% in 2008 across Member States, respectively[35] (0.55% and 0.36% respectively for Germany, and 1.12% and 0.74% respectively for Spain), due an increase in demand for equipment produced by domestic industries overcompensating for the net increase in electricity retail prices (support measure costs

[34] The largest consumers are effectively exempted. For annual consumption above 1GWh, electricity-intensive firms pay 10% of the surcharge. For consumption over 10GWh only 1% is liable, and for over 100GWh, a maximum of 0.05ct/KWh is levied.

[35] In the more likely case that investment in renewables did not reduce investment in conventional carriers. Results for the scenario in which RES-E investments replace investments in conventional carriers produce negative impacts of a similar magnitude.

minus reduction in wholesale prices). As there is no incentive to move beyond the Member State binding targets for renewable energy, dynamic efficiency is relatively low. However, Mazzanti & Antonioli (2013) conclude that RES-E support mechanisms have led to significant incremental technical product innovations, producing improved generating efficiency of existing technologies.

Non-CO_2 GHGs

As there are very few policy instruments implemented to tackle GHG emissions from the agriculture sector, and those that do exist are either very small in scope from an EU perspective, or are voluntary, there is naturally low static and dynamic efficiency, with little incentive to abate or produce innovations to abate in the future. Abatement that has occurred has been largely a consequence of other actions resulting from non-climate policy factors (including other policy instruments).

4 Feasibility Constraints & Impacts of the EU's Climate Policy Instrument Mix

Whilst the existing European climate policy instrument mix is, by virtue of its existence, feasible, many instruments will have faced, and may continue to face, challenges ranging from administrative implementation, unintended side-effects, flexibility to deal with risks and uncertainties, and political and public acceptability.

Carbon Pricing

The EU ETS, despite its current position as the 'crown jewel' of EU climate policy (Wettestad, 2005), initially had little favour as an approach to mitigation in Europe. During negotiations preceding the Kyoto Protocol, the EU had even voiced explicit opposition to market-based instruments, only relenting when the issue threatened to derail the progress due to insistence by countries such as the USA (Oberthur & Ott, 1999). This opposition was largely due to the lack of experience in using market-based instruments for environmental protection in the EU (Mehling *et al*, 2013). However, EU interest in market-based instruments was growing in Member States and at European level, with the European Environment Agency (EEA) viewing such mechanisms as a suitable means to implement the legally vested polluter-pays principle and protect public goods, alongside achieving economic and social policy objectives (European Environment Agency, 2005). This focus, however, was on a combined energy and carbon tax, which failed to secure the unanimous support required for the adoption of measures of a primarily fiscal nature (Mehling *et al*, 2013). After further unsuccessful attempts to introduce relevant legislation, the Commission proposed a less ambitious Directive, which became the ETD. Simultaneously, several factors converged to generate support for an ETS, in an 'extreme about-face' that occurred virtually 'overnight' (Hardy, 2007), leading to the eventual adoption of the Emissions Trading Directive in 2003 (2003/87/EC) At the Commission level, personnel changes, the active involvement of foreign experts in fostering better conceptual understanding of emissions trading and improved internal capacities at DG Environment improving the DG's ability to argue the merits of an ETS to counterparts in charge of competition and enterprise rendered an ETS an acceptable, if not a desired policy instrument. From a political acceptability perspective, Member States began to recognise the additional flexibility afforded by an ETS over a centralised tax. From an administrative feasibility perspective, the trading of electricity contracts had become an established business practice in the newly liberalised power sector, resulting in significant expertise and capacity within such companies. The Integrated Pollution Prevention and Control Directive (IPPC) (96/61/EC) was touted as a model for the ETS, with existing structures relating to monitoring, report, verification and enforcement providing precedence (Mehling *et al*, 2013).

The absence of unanimity requirements was arguably vital to garner sufficient momentum for the introduction of the EU ETS (Mehling *et al*, 2013), despite the rapid increase of support. The level of support or opposition to the ETS by a Member State, as with other instruments, depends on a number of factors including governance and regulatory tradition, cultural preferences, policy priorities and economic structure. The final two factors contributed to the inclusion of grandfathered permits for the first two Phases, to counter concerns of reduced industrial competitiveness. States such as Poland, in which 60% of its emissions are from EU ETS sectors (compared to the average 40%), were particularly vocal (and continue to be, securing further concessions into Phase 3). This is supported by strong industry lobbies and labour unions, which oppose ambitious climate energy regulations (Mehling *et al*, 2013), and supported by the results of recent surveys on environmental awareness, environmental protection and climate-energy issues reveal that only 16% of Poles believe that environmental pollution is a serious problem (Szewranski, 2012). As such, the high dependence on coal-based electricity generation and energy-intensive industry in Poland, along with significant domestic coal resources, and strong support from energy independence from Russia amongst government and the electorate, and the perception that environmental issues are of low importance, places action of climate change low on the list of governmental priorities. In addition, the energy sector in Poland falls within the purview of the Ministry for the Economy (although climate policy more broadly is dealt with by the Ministry for the Environment). The 'Economic Council', formed in 2010 as an informal advisory body to the Prime Minister and comprised of economists and business people, analyses all legislation and has become a real decision making centre (Mehling *et al*, 2013), entrenching purely economic interests over emissions abatement. The Polish government also cited a lack of administrative capacity as a barrier to the operation of the EU ETS (Skjaerseth & Wettestad, 2008).

Issues of feasibility continued to appear after the instrument's adoption. A number of National Allocation Plans (NAPs), in which Member States presented how EUAs would be distributed amongst obligated installations, were initially rejected by the Commission and had to be revised. In Germany, a series of legal challenges arose based on alleged violation of German constitutional basic rights. The most prominent case arose in 2005, when a cement company argued that the EU ETS violated its property rights and right of freedom to exercise a trade or profession by arguing that its right to emit CO_2 freely, as earlier guaranteed through its general operation licence, was being expropriated. However, the Federal Administrative Court did not uphold this complaint, and ruled the EU ETS compatible with the constitution. The EU ETS has also proven relatively poor at dealing with risks and uncertainties, leading to significant price volatility and from the onset of the financial crisis in late 2008, persistently low prices. The difficulty in passing the temporary 'backloading' proposal, which temporarily removes EUAs from the system to temporarily reduce oversupply and stimulate higher prices, indicates continued (and likely heightened) political opposition to the EU ETS from some quarters (and the 'consensus reflex') (Mehling *et al*, 2013).

As stated above, the ETD was introduced as a heavy compromise to initial energy and carbon price proposals. However, as the obligations are weak and the scope relatively narrow, it could no longer serve as the centrepiece of EU climate policy (Mehling *et al*, 2013). Its current configuration is highly politically feasible due to the low minima, and the several derogations and exemptions in place (and as this simply impacts tax rates, no additional burdens were placed on Member States upon its introduction). The issue of fuel tourism, whilst not caused by the ETD, is a consequence of its design, which produces the indirect issues of preventing Member States from implementing welfare-maximising taxation and difficulties in road transport emissions accounting (Maca *et al*, 2013). In 2011, the Commission proposed an amendment to the ETD, which would reframe the minimum rates in terms of the energy and carbon content of the energy carriers concerned. Thus far however, it has proven unfeasible (Drummond, 2013).

Energy Efficiency and Energy Consumption

The political acceptability of the CO_2 intensity targets for cars appears relatively high despite protestations from car manufacturers, particularly luxury brands and from Germany, where many such brands are based. The weight-based calculation was pushed through by Germany to reduce the potential burden on such manufacturers, as opposed to a single CO_2 standard or footprint-based calculation (Maca *et al*, 2013). Germany continued to push for modalities for achieving proposed 2020 and 2021 targets acceptable to these manufacturers throughout 2013 and 2014, before legislation was introduced to confirm post-2015 requirements. Administrative feasibility appears extremely high, with completeness rates for mandatory reporting parameters at 99.7% for mass and CO_2 emissions, and 99% for vehicle type, variant and version (European Environment Agency, 2013).

The Directive on CO_2 labelling for passenger cars is highly politically, as very little cost is borne and little burden is place on any actors in the market. Although in 2003, the European Court of Justice (ECJ) found against France, Italy and Germany for failing to transpose the Regulation. However, administrative feasibility also appears high, as there are increasingly low levels of reported non-compliance (Gartner, 2005). However, this may be due to a lack of regular and standardised monitoring in many Member States (Drummond, 2013). CO_2-linked registration and ownership taxation is only currently implemented at Member State level, and appears feasible on all counts in the States in which they are employed. It is unclear whether such framing of vehicle taxes could be feasibly introduced at an EU level. If new Member States were to apply this framing without EU-wide alignment, particularly in net exporters of cars, then additional emissions may ensure. Many second-hand vehicles are of high CO_2-intensity. If the domestic tax regime further reduces the attractiveness of such vehicles on the domestic second hand market, exports to other Member States are likely to increase, thereby potentially increasing emissions above the counterfactual. At the same time, the increasing build up of such vehicles may make the introduction of more ambitious policy instruments, such as increased fuel taxation, more difficult (Maca *et al*, 2013).

There is significant flexibility in the provisions of the EPBD (such as the ability for each Member State to set its own 'cost-efficient' minimum performance standards for new and refurbished buildings), which produces political acceptability. However, implementation has proven difficult in some Member States, with many holding no previous experience with energy efficiency requirements in buildings. This led to several delays and subsequent infringement cases (21 at its height) (Drummond, 2013). The Energy Labelling Directive, as with CO_2 labelling for passenger cars, places little burden upon actors, and is thus highly politically feasible. Whilst costs to government for compliance monitoring are also relatively low, however of the eight Member States that currently conduct no product testing, six cite financial (along side human resources) constraints, as a key barrier (Pahal *et al*, 2013). The flexibility of the instrument is also relatively low, as efficiency classifications cannot easily be redefined, or new classes added without causing potential confusion or a reduced impact (Drummond, 2013). The Ecodesign Directive has been proven highly feasible, although some issues have been identified. It has thus far taken between four and six years to produce an implementing measure from initiation, during which time data used in initial impact assessments may become out-dated. There is also evidence of non- compliance between 10-20%, largely due to Member States failing to dedicate the necessary resources to monitoring and enforcement (Drummond, 2013).

Promotion of Renewable Energy

The Renewable Electricity Directive, predecessor to the Renewable Energy Directive, failed to achieve its 2010 targets arguably as a result of their non-binding nature. Whilst Member States ultimately agreed on binding targets for 2020 in the RED when adopted in 2009, conflicts emerged during the legislative debate when several Member States claimed that proposed targets were unrealistically high or failed to consider historic achievements. Also, energy policy has traditionally been a sovereign matter, with the Commission reluctant to infringe on this, given the implications for economic development and energy security. However, the insertion of an explicit legal basis in the Lisbon Treaty (which entered into force in 2009), allowing for an autonomous energy policy and a broad range of measures in the energy sector (Ethricke *et al*, 2009), may reduce the potential for future legal and other conflicts in future (Mehling *et al*, 2013).

The design and implementation of specific measures (such as support mechanisms) varies by Member State, also as a result of country-specific factors. The majority of Member States employ feed-in tariffs as their primary support mechanism, however in 1998 the Commission initially concluded that feed-in tariffs violated state aid rules. Whilst this was overturned by the ECJ in 2001, legal conflict remained at the Member State level. Similarly to the legal challenges experienced in Germany in relation to EU ETS allocations, the 2005 amendment to Poland's 1997 Energy Law, which introduced a renewable electricity purchase obligation, was challenged on the basis that it interfered with the Polish constitutional right to freedom of economic activity. The Constitutional Tribunal ruled that the amendment was proportional to the objectives, and did not infringe on freedom of activity (Mehling et al, 2013). Although, such legal challenges may lead to the creation of innovative new institutions, such as the EEG Clearing House in Germany, which was established as a fast-track dispute settlement procedure under the Renewable Energy Sources Act (EEG).

Governance structures in Member States also have an impact on effective implementation. In the UK, the devolution of several competences related to climate policy to the devolved governments of Scotland, Wales and Northern Ireland arguably helping the deployment of renewables with more specifically targeted support mechanisms (such as wave and tidal in Scotland), and a 'regional centralist' approach to planning legislation more efficient than a fully centralised approach. However, this latter aspect may also prove a hindrance, as the Scottish government have used these planning powers to block the development of new nuclear installations. Regional governments are also highly important in Germany where the responsibility for administration and execution of federal legislation rests with the Lander, also allowing specific support to be tailored to local conditions, although the are several grey areas regarding specific areas of responsibility between the federal and Lander governments, such as electricity grid expansion. As such, devolution of responsibility from central government can prove both a barrier and a driver for climate policy, based on central and regional policy priorities (Mehling *et al*, 2013).

The 'enabling initiatives' contained in the RED, which include the removal of administrative barriers to deployment and guaranteed access to the grid for RES-E technologies. Such provisions assist in making the headline targets more achievable themselves, however in the second progress report on the implementation of the RED published in March 2013, the European Commission (2013) reported that progress in removing administrative barriers to renewable energy development is limited and slow, and many Member States did not even address such reforms in their 2011 progress reports. The availability of single administrative bodies for dealing with renewable energy project authorisations is rare – with only Denmark, Italy, Netherlands, Greece and Portugal taking such an approach. Authorisation and planning procedures remained a key challenge to electricity infrastructure development, however the Commission reports that there is clear evidence for most Member States making at least some progress towards reforming their electricity grid infrastructure, and rules for operation and access surrounding them, however this is happening at a relatively slow pace. Whilst this has not yet interfered particularly heavily with renewable deployment, continued slow progress could make achieving Member States' 2020 targets increasingly difficult (Drummond, 2013).

The European Commission (2013) also reports that the biofuel sustainability criteria introduced by the RED has been mostly implemented across Member States. At the end of 2013, fifteen infringement cases were active against Member States for failing to fully transpose the RED. Whilst these criteria correct some unintended consequences surrounding the lack of consideration for the indirect impacts, the European Commission (2013) also estimates that European biofuel policy contributed an additional 1%-2% to global cereal prices in 2010, and 4% to food oil crop prices. In June 2014, the proposed 'ILUC Directive', which will amend both the RED and Fuel Quality Directive, achieved political agreement in the EU Energy Council. The ILUC Directive will, if adopted, amongst other things limit the achievement of the 10% renewable target to a maximum of 7% from conventional biofuels, and a minimum 0.5% from second and third generation 'advanced' biofuels. Incentives to increase the use of renewable electricity in renewable transport would also be introduced, to compensate this amendment.

Non-CO_2 GHGs

The small scale and largely voluntary approach taken by existing climate policy instruments in the agriculture sector, along with the focus on cost-efficiency, makes such instruments highly feasible. However there are issues of feasibility preventing further action in this sector. Policies that produce additional cost to the sector are likely to be highly unpopular, as this may raise food prices and reduce competitiveness where margins are already often very small (Kuik & Kalfagianni, 2013). In addition, producing accurate measurements of emissions from the agriculture sector provides a potential administrative hurdle, which may also apply to non-agricultural non-CO_2 sources, including LULUCF activities.

5 Conclusions

Several key conclusions may be drawn from the sectoral and cross-sectoral research undertaken by the studies listed in Table 1, and summarised in this report.

The existing climate policy mix is uneven, lightly co-ordinated and sometimes difficult to define. There is a deep divide between sectors and Member States concerning the number of instruments in place to tackle emissions, instrument design and implementation, and the level of ambition. The power and industry sectors experience the most coherent policy landscape, with the EU ETS producing a single, EU wide carbon price with equalised marginal abatement costs. The promotion of renewable electricity under the Renewable Energy Directive is considered in EU ETS cap-setting exercises to prevent negative overlap, which has likely been achieved so far. However, the individual implementation of instruments to promote renewable electricity varies significantly between Member States. The Energy Taxation Directive places uneven minimum taxation requirements on different energy carriers and sectors (exempting agriculture and domestic heating, along with products used for the generation of electricity), both in terms of energy and carbon content, with the highest minimum taxation placed on gasoline and diesel for road transport. However, as the

Directive imposes only minimum rates, effective tax rates on gasoline and diesel are often much higher, producing substantial variation between Member States. Whilst other instruments, such as the 10% target for renewables by 2020 applies to all road transport (although again implemented differently across Member States), other key instruments such as CO_2 intensity and vehicle labelling regulations apply to passenger cars only (although similar regulations for heavy and light goods vehicles will be implemented in future). International aviation and shipping, both significant GHG sources, are effectively excluded from any instrument. In all sectors policy instruments with no explicit climate-related objectives impact energy consumption and other activities related to the generation of GHG emissions. This is particularly the case in the agriculture sector, in which no explicitly climate-related policy exists at the EU level. Some instruments exist at Member State level, but they are largely very recent, focus on information dissemination and R&D efforts (and therefore are without significant ambition in the short-term), and are implemented on a voluntary basis (Kuik & Kalfagianni, 2013). Instead, provisions in the Nitrates Directive and Common Agricultural Policy, both introduced for non-climate purposes, are likely to have had the most significant policy-related impact on agricultural emissions.

Despite this, the climate policy mix in the EU has been broadly effective in producing GHG abatement. Meyer & Meyer (2013) calculate that the presence of the EU ETS, instruments to promote renewable electricity and environmental tax reforms reduced CO_2 emissions by up to 12-13% below the counterfactual in some Member States in 2008 (but with significant variation). This value is likely to increase with the consideration of the impact of flanking instruments.

Whilst economic instruments have been important, they are not the only climate policy instruments to have produce abatement. The presence of phenomena such as split incentives and of factors not considered in the standard economic interpretation of a ‘rational actor’ responding to price signals means that economic instruments, whilst vital, cannot alone effectively induce abatement where it may be required in all facets of the economy and society. The economic instruments currently in place, whilst effective to different extents, are not exploiting their full potential as a result of design flaws, imperfect implementation, and negative interaction with other climate and non-climate policy instruments. Meyer & Meyer (2013) calculate that the EU ETS produced CO_2 abatement of 1-3% against the counterfactual in 2008 across the EU (in line with other estimates in the literature), delivered principally through fuel switching in the power sector. However, Agnolucci & Drummond (2014) confirm that levels of induced abatement are likely to have varied substantially over time due to the instrument design preventing adaptation to unexpected developments and external shocks (initial oversupply of allowances in Phase 1 coupled with an inability to bank, and the financial crisis reducing demand for allowances in Phase 2 and beyond). However, the central target of maintaining emissions under its remit below the given cap has thus far been achieved.

Instruments for the promotion of renewable electricity (mainly feed in tariffs, but also green certificate schemes and others), essential in enabling Member States to meet their legally binding targets for renewable energy in gross final consumption in 2020, provided the largest contribution to policy-induced abatement at an average rate of 3.2%-3.9% across Member States against the counterfactual in 2008. A significant range exits between Member States, with Germany achieving the highest estimated abatement of 7.88%. Whilst the EU ETS may have triggered only minor technological innovation in either the power or industry sectors (although organisation innovation has likely been more substantial), renewable electricity support mechanisms have led to significant incremental product innovations, particularly improved generating efficiency of existing technologies.

The use of fuel taxes appears to be effective in influencing road travel demand, but not significant in driving demand for more efficient vehicles. However, incentives for both reduced demand and for more efficient vehicles by any road transport pricing instruments (including registration and circulation taxes, and often other road pricing mechanisms) are restricted by company car taxation arrangements, in which the employer and employee reduce the tax burden from the use of a company car as an in-kind benefit, and in which the driver is often not liable for fuel costs (and thus has no incentive to reduce demand). However, Regulation 443/2009, which sets binding CO_2 performance standards for new passenger cars, has been effective in increasing the rate CO_2 intensity reductions, with the 2015 target of 130gCO_2/km likely to be achieved ahead of time. In addition, the effectiveness of such a broad-scope regulation is not (or at least, much less) affected by market distortions such as company car arrangements.

There is no evidence to suggest that 'carbon leakage' has occurred. Whilst much of the *ex-ante* analysis predicted significant rates of carbon leakage, the *ex-post* evidence suggests that no loss of competitiveness leading to carbon leakage has occurred amongst the Energy Intensive Trade Exposes (EITE) sectors. The difference between the *ex-ante* studies and *ex-post* results may be due to several reasons, including EU ETS price assumptions, the free allocation of allowances under the first two Phases of the EU ETS, the use of long-term electricity supply contracts that buffer against EU ETS price volatility, and the lack of *ex*-ante consideration of other factors such as capital abundance, labour force qualification, proximity to customers and infrastructure quality, alongside carbon and energy price liabilities.

From a broad perspective, the climate policy mix may have produced net economic benefits to the EU. Meyer & Meyer (2013) conclude that the introduction of the EU ETS, renewable electricity support mechanisms and environmental tax reforms overall did not reduce GDP in the EU, and likely had a positive impact. Employment is also likely to be higher than the counterfactual in most Member States, with the exception of some of the smaller transition economies. The EU ETS, taken individually, is likely to have reduced GDP and employment by an average of 0.5% and 0.34% respectively across Member States in 2008, due to EITE industries pricing-in the opportunity cost of freely-allocated allowances. Conversely, under the assumption that investments in conventional fuels was not displaced, investment in renewable electricity is estimated to have increased both GDP and employment by an average of 0.32% and 0.09% respectively across Member States in 2008, due to increased demand for equipment produced by domestic industries more than compensating for the net increase in electricity retail prices. Although this average value is lower than the average negative impact from the EU ETS, the net benefits have been the most substantial in the larger European economies, producing a weighted total net benefit for the EU. Despite this, distributional issues emanate from both instruments. Whilst the penetration of renewable electricity reduces wholesale electricity prices, support mechanisms costs, recovered from final consumers, have outweighed this to produce net costs. As most Member States introduce compensatory mechanisms to the EITE industries and other enterprises, the cost burden increases and falls disproportionately to residential consumers in particular. Whilst the carbon cost pass-through rate in the power sector from the EU ETS varies substantially across time and between Member States, any pass-through of the opportunity cost of the freely-allocated allowances in Phases One and Two produced windfall profits, leading to a transfer of wealth from electricity consumers to electricity generators and suppliers.

Instrument mix 'Optimality' is difficult to achieve, but improvements are possible. Under the concept of optimality developed under the CECILIA2050 project and employed in this report, which examines environmental effectiveness, economic efficiency (static and dynamic) and feasibility, it is clear that the existing climate policy mix is sub optimal. However, it must be made clear that this concept of optimality must be considered a theoretical point of reference. Policies and policy mixes in real-world application are always faced with trade-offs and compromises between each of these components. Based on the research summarised in this report, and the reports underlying it, the lessons learned can be used to enable improvements to the existing policy mix to be investigated and pursued.

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