4

Climate Policy Designs

Contexts, Choices, Settings and Sequences

4.1 Introduction

Chapter 3 described the design space in which EU climate policies have emerged. It revealed an ongoing game of design that has simultaneously worked across and involved many different actors, governance levels and policy elements. One of the most significant policy design dynamics has been the one connecting programmatic goals and specific policy designs. In general, longer-term goals set at EU level to match international-level processes centred on the UNFCCC have been gradually back-filled with policy programmes and policy instruments.

This chapter examines the instrument-level dynamics and their initial feedback effects in more detail. Sections 4.2–4.4 provide more detail on the three policy areas that were originally introduced in Chapter 1. For each instrument, we briefly outline the relevant sector's emissions before the policy design process commenced, and introduce the main designers, target groups and interest groups. Then, the pre-existing policy design space is summarised and the most salient features of each instrument are described and an initial preview is given of the most significant post-adoption feedback effects. Over time, these effects have changed actor dynamics, leading to new policy changes and, eventually, long instrument sequences. These sequences, which are summarised in the concluding section, span many decades and will be discussed in much greater detail in subsequent chapters.

4.2 The Governance of Biofuels: An Extended Regulatory Sequence?

Emission Patterns

Biofuels are a type of bioenergy[1] resource that are derived from organic matter (International Energy Agency [IEA], 2011: 5; Bouthillier *et al.*, 2016). The 2003 Biofuels Directive – the EU's first main foray into biofuel-related policy design – defines them as 'liquid or gaseous fuel for transport produced from

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biomass' (OJ L123, 17.5.2003: 44). Unlike fossil fuels, biofuels are, at least in theory, a fully renewable resource because they derive from plant material (Environmental Audit Committee, 2008). They are also a central element of the EU's wider renewable energy ambitions. The economic potential of biofuels has been recognised for almost as long as cars have been produced: early versions of the diesel engine could run on a biofuel derived from peanuts (Knothe, 2001: 1104; International Energy Agency, 2011: 10). Since then, the economic fortunes of the industry have fluctuated, often in line with international energy prices. As an influential OECD review has noted, the high cost of biofuels relative to fossil fuels has repeatedly limited their uptake (Doornbosch and Steenblik, 2007: 11). Thus, biofuels were regarded as a technologically viable option until the 1940s, when falling oil prices rendered them uncompetitive (IEA, 2011: 10). They were actively re-promoted after the 1973 oil crisis when fossil fuel prices soared (COM (2001) 547: 5), and in the 2000s when they rose again.

However, in spite of these cycles, global biofuel production has nonetheless grown spectacularly by 525 per cent between 2000 and 2010, from 16 billion litres to 100 billion litres (IEA, 2011: 10). This growth was facilitated by a 'frenzy' of new policies, mostly adopted at a national level (Ackrill and Kay, 2014: 3). Nonetheless, the global uptake of biofuels slowed considerably after 2010 as fossil fuel prices dropped (IEA, 2019). In 2017 its share of the global transport fuel market still stood at only 4 per cent, up marginally from 3 per cent in 2010 (IEA, 2011: 1), leaving the biofuel industry as a relatively niche player in what is a huge, highly globalised market in transport and other fuels.

There are many different types of biofuel, but two in particular have repeatedly attracted the attention of policy designers: bioethanol and biodiesel, respectively accounting for 72 per cent and 27 per cent of global biofuel production in 2017 (IEA, 2019; see also Ackrill and Kay, 2014: 5). *Bioethanol* is a type of alcohol derived from corn, barley, wheat and sugarcane – i.e. food and fodder crops. It can be blended to produce a drop-in alternative to conventional petrol (Ackrill and Kay, 2014: 5). By contrast, *biodiesel* is an oil-based fuel derived from the fats and oils in rapeseed, palms, soya and even animal meat. It can be blended to provide a drop-in alternative to diesel fuel.

Over time, different generations of biofuel have been developed (IEA, 2011: 8). *First-generation biofuels* are produced using food crops. When combusted, they produce fewer pollutants than fossil fuels, but they also suffer from a number of well-known drawbacks. Production processes are relatively inefficient, typically requiring significant inputs of energy that is often, paradoxically, derived from fossil fuel (Charles *et al.*, 2007: 5738). In addition, first-generation biofuels are typically derived from feed stocks that require high-quality land for cultivation. As a result, there have been repeated claims that they undermine food security and

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encourage land-use intensification (through requiring chemical fertilisers and pest controls). *Second-generation biofuels* are derived from specially grown crops such as switch grass, the non-edible parts of food crops (husks, shells and cobs) and waste materials such as straw and cooking oil. Second-generation production processes tend to generate fewer greenhouse gases than those for first-generation biofuels (Charles *et al.*, 2007: 5738). However, they still require significant quantities of land for cultivation. Finally, *third-generation biofuels* are derived from algae (Ackrill and Kay, 2014: 9), do not compete with other potential land uses and have a lower greenhouse gas footprint, although they do require inputs (typically, sunlight, water and nutrients) that may not be available in all locations (IEA, 2011: 14).[2]

First-generation biofuels still account for the majority of current global production and consumption. By contrast, many second-generation and third-generation biofuels are still in the research and development stages. However, their market share has been growing significantly. In 2018, second-generation biofuels (both bioethanols and biodiesels) together accounted for 23 per cent of total biofuel usage in the EU (United States Department of Agriculture, 2018: 3–4). However, in order to fully appreciate the multifaceted challenge of developing durable policy designs, one needs to understand the distinction between different generations of the two main types of biofuel: bioethanol and biodiesel. Thus the most common first-generation *bioethanols* are derived from sugar and starch-based crops grown on high-quality agricultural land (Demirbas, 2009: s108; IEA, 2011: 8). The most common second-generation types derive from non-food crops such as perennial grasses (Sharman and Holmes, 2010: 310). However, only recently have these been produced at a sufficiently large scale to make them economically viable (IEA, 2011: 13). The most common first-generation *biodiesel* supplies derive from a variety of feedstocks including rapeseed in the EU and palm oil in tropical countries (Ackrill and Kay, 2014: 6). Second-generation biodiesels mostly derive from vegetable oils (Di Lucia and Nilsson, 2007: 534) including recycled cooking fats and oils (Ackrill and Kay, 2014: 8).

The main challenge facing policy designers is how best to incentivise the right type of biofuel use, which in practice entails matching production capacities with current and future levels of consumer demand. One influential OECD review identified a ‘huge array’ of policy design options for achieving these goals (Doornbosch and Steenblik, 2007: 24). However, according to the UN-affiliated High Level Panel of Experts on Food Security and Nutrition (HLPE), existing national-level interventions have tended to fall into two main categories: *demand-focused* and *production-focused* (HLPE, 2013: 11). Demand-focused policies have sought to create new markets for biofuels. They include tax exemptions for producers, so-called blending mandates that stipulate what percentage of fuel in a particular sector should be sourced from biofuel, and subsidies to nurture demand (e.g. encouraging private owners and fleet operators to convert to biofuels). By contrast, production-focused policies have targeted fuel producers and dealers by offering subsidies to compensate for the additional cost of producing biofuels (and especially the more advanced types) as compared to petroleum fuels, or alternatively establishing border taxes to give domestic producers a competitive advantage.

Why have so many policy designers sought to manipulate the demand for and supply of biofuel? The answer to this question is not straightforward – in fact, the perceived benefits of switching have changed quite significantly over time and within particular parts of the world (Demirbas, 2009: s109; Ackrill and Kay, 2014: 216). Three benefits in particular have been regularly cited by advocates: superior environmental performance (including lower greenhouse gas emissions); greater energy security through reduced reliance on imported fossil fuels; and economic benefits, particularly in rural areas that are reliant on agriculture (IEA, 2011: 7; Ackrill and Kay, 2014: 11). It is worth remembering that biofuel production first took off in Brazil and the USA in the 1970s, primarily to address energy security. In Europe, governments were much slower to promote production and use; when they did, decarbonisation, energy security and rural employment were the most regularly cited rationales (Palmer, 2014: 337–338), particularly after the early 2000s (Ackrill and Kay, 2014: 70).

The emergence of the Biofuels Directive in 2003 should be seen in the context of these broader technological, policy and scientific developments. In fact, many of them are explicitly cited in its opening recitals. The full story of the Directive's design is recounted in Chapter 5. It reveals that decarbonisation was one of the main rationales cited by advocates. As Palmer (2014: 337) explains, biofuels are, at least in theory, carbon-neutral: on combustion they release into the atmosphere the carbon which was originally sequestered during their growth. In a Communication published alongside the formal proposal for the Directive in 2001, the Commission boldly asserted that they 'offer an *ideal alternative* since, when based on EU grown crops, they are practically 100% indigenous and CO_2 neutral' (COM (2001) 547: 5, emphasis added). To be fair, it did qualify this headline statement with a discussion of the high cost of some production techniques. It also noted that 'up to half, or more than half, of the CO_2 benefit is offset in the production process for biodiesel and bio-ethanol respectively' (COM (2001) 547: 5). But it remains the case that, around this time, hopes in some parts of the Commission were high that biofuels would simultaneously solve many long-standing policy problems in the EU.

In practice, the emissions saved by switching to biofuels vary significantly (Ackrill and Kay, 2014: 24), depending on the precise type and generation of the biofuel used, and how it is produced, including how the land in which the feedstock was cultivated was previously used (Howes, 2010: 140). Biofuel production can lead to increased emissions from direct land use change, when the feedstocks displace food crops (Royal Academy of Engineering, 2017: 8; COM (2010) 811: 3). If, on the other hand, the feedstocks are grown on carbon-rich land which had not previously been farmed, such as forests, the resulting effects are categorised as indirect – hence the term 'indirect land use change' (ILUC) emissions. It has proven very challenging to quantify the scale of ILUC emissions from biofuel (Giljam, 2016: 102). As long ago as the late 2000s, scientists began to suggest that when the direct and indirect effects are fully accounted for, the total emissions arising from production may even exceed those associated with some fossil fuels (Environmental Audit Committee, 2008: 6; Ackrill and Kay, 2014: 24), although again the exact amount varies from one fuel type to another. Indeed, far from becoming more widely accepted as production has grown, the case for converting to biofuels to decarbonise society has become more contested as new information has emerged and circulated amongst actors.

As the biofuel industry has expanded in Europe, an increasingly complex set of feedstocks has been used, many imported from outside the EU. In 2015, the European Environment Agency (EEA) concluded that when the direct and the indirect effects arising from these extended supply chains are taken into account, the net environmental benefits of biofuels are subject to 'considerable uncertainty' (EEA, 2015a: 58). These uncertainties have stimulated – and in turn been greatly compounded by – a succession of political conflicts between actors promoting or opposing the use of different types (and indeed particular generations) of biofuel, many employing scientific information in a 'partial and tendentious fashion' (Ackrill and Kay, 2014: 217).

Target Groups and Other Interest Groups

The key policy actors related to EU biofuels policy have included the fuel producers, the vehicle producers, environmental groups, some (but not all) Member States and the EU institutions. Between 2000 and 2010, global biofuel production expanded massively from 16 billion litres per year to over 100 billion per year (IEA, 2011: 11). Growth then slowed, but by 2017 production had nevertheless increased to 143 billion litres annually (IEA, 2019). As noted above, production initially centred on the USA and Brazil; by 2016, these two countries still hosted 70 per cent of global production (IEA, 2017: 103). But gradually production has also taken off in some – but not all – Member States (Ackrill and Kay, 2014: 4). Our main point, however, is that production and consumption have rarely been uniformly distributed across time and space: certain countries and regions have actively promoted certain biofuel types for distinct reasons and using different mixes of policy instruments. These differences have led to countries trading in both fuels and feedstocks.

Globally, bioethanol is still produced at much higher volumes than biodiesel (IEA, 2017: 53). Global production grew much quicker between 1980 and 2007 – again, principally in the USA and Brazil. These two countries were anxious to secure new uses for existing agricultural production after the 1973 oil crisis (HLPE, 2013). By contrast, biodiesel production has grown more slowly, but has traditionally been the dominant biofuel in Europe (Demirbas, 2009: s110), strongly supported by biofuel producers associated with the agricultural sector, including farmers (Skogstad, 2017: 30, 34–35). The European car producers have generally been supportive of biofuels (see below), having staked their future profitability on 'dieselising' their car fleets to stay ahead of stricter greenhouse gas emission targets (HLPE, 2013: 12).

In the early 2000s, when the Biofuels Directive was being formulated within the Commission, biodiesel production in the EU stood at around 2.3 billion litres – i.e. around four times the total production of bioethanol (Demirbas, 2009: s109). However, only six Member States produced significant quantities (COM (2001) 547: 19), amongst which three were dominant (France, Italy and Germany), although there were also significant production facilities in Sweden, Spain and Austria. The remaining Member States produced very little or even no biofuel (COM (2001) 547, 19). In 2001, net consumption accounted for less than 0.5 per cent of overall fossil fuel consumption in the EU (COM (2001) 547: 6). In effect, policy designers were starting from a very low base.

When the European Commission began to seriously engage in policy design in the late 1990s, the biodiesel producers were better organised than the bioethanol producers. The key industry associations dated back to the late 1990s and included the European Biodiesel Board (EBB) (1997) and European Vegetable Oil and Proteinmeal Industry (FEDIOL) (1957). Bioethanol producers were initially represented by the European Union of Alcohol Producers. Formed in 1993 to represent producers of industrial alcohol as well as biofuels, it was renamed the European Union of Ethanol Producers in 2004. In 2005, the European Bioethanol Fuel Association (eBIO) was formed to specifically represent bioethanol producers. In 2010, the European Union of Alcohol Producers and eBIO merged to form the European Renewable Ethanol Association (ePure). As biofuel production expanded, other industry groups emerged claiming either to represent the interests of the whole industry (e.g. the European Biofuels Technology Platform (EBTP), established in 2006) or particular users and producers (e.g. the Leaders of Sustainable Biofuels, established in 2010 to promote the use of advanced biofuels in the aviation sector). Associations representing feedstock industries (e.g. COPA/COGECA – a very powerful interest group representing European farmers) and competing fuel types (e.g. EUROPIA for the oil and gas industries) also became more involved.

The involvement of environmental NGOs expanded from a small number (the European Environment Bureau [EEB], Transport and Environment [T&E] and the World Wildlife Federation [WWF]) to include most environmental NGOs campaigning on climate change-related themes (Skogstad, 2017: 30). As public awareness of the associated environmental and social impacts of driving up biofuel use grew, big international development charities such as Oxfam and Action Aid began to take a more active interest (Skogstad, 2017: 35).

The Design Space

The first policies to promote biofuels were adopted in Brazil and the USA and generally took the form of subsidies. By 2011, over 50 countries had become involved, adopting a much wider array of policy instruments, each having a distinctive national twist (HLPE, 2013: 11–12). Prior to the EU becoming more involved, those Member States that had biofuel-focused policies incorporated a mixture of production-focused and demand-focused instruments. Production subsidies (both direct and indirect, part or wholly paid through the EU's Common Agricultural Policy) were an important part of pre-existing national policy mixes. By 2006, the International Institute for Sustainable Development (2008) estimated that the EU and individual Member States were subsidising biofuels to the tune of around €3.7 billion per year[3] (see also COM (2001) 547: 17). But according to the Commission, these policy approaches were nonetheless still too weak and too diffuse to facilitate deep decarbonisation in the EU transport sector. Therefore, it began to explore ways to institute stronger and more harmonised policy support at EU level.

In theory, there were many ways in which policy designers at EU level could have approached the design challenge. Given the design space in which the Commission was working (see above and Chapter 3), it was entirely understandable that the Commission opted to employ regulation to set a blending mandate to drive new sources of production, although it also sought to shape demand. The objective of the 2003 Biofuels Directive was thus to promote the 'use of biofuels or other renewable fuels for transport' in each Member State 'with a view to contributing to objectives such as meeting climate change commitments, environmentally friendly security of supply and promoting renewable energy sources' (OJ L123, 17.5.2003: 44). Article 3 mentioned several renewable fuels, but the only ones then in existence were biofuels. Article 2 thus defined a number of different types of biofuel including bioethanol and biodiesel.

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The Initial Policy Design

The Biofuels Directive was a form of command-and-control regulation. Thus Article 3 formally required Member States to ensure that a minimum proportion of biofuels were placed onto their national markets and 'to that effect ... set *national indicative* targets' (OJ L123, 17.5.2003: 44, emphasis added). But while the setting of these targets – which were essentially a type of blending mandate – was mandatory, their implementation was non-mandatory and they were established at national level by the Member States (i.e. the EU was simply facilitating national policy coordination). The targets were to be expressed in the form of two reference values, which were also non-mandatory: a reference value of 2 per cent (by energy content) to be achieved by the end of 2005; and a reference value of 5.75 per cent to be achieved by 31 December 2010. The policy instrument-level *durability devices* were therefore relatively weak. The Commission also proposed another, more novel *durability device*: an annual increase in blending levels by 0.75 per cent starting in 2005, to automatically raise the overall total from 2 per cent in 2003 to 5 per cent in 2009 (COM (2001) 547: 18). But this was quickly whittled away in the policy formulation process and did not appear in the final text of the Directive.

The Directive also did not say much about the precise actions that Member States should take to achieve their national biofuel targets, other than that governments should establish indicative targets to guide them (Haigh, 2009: 14–11) and regularly submit progress reports to the Commission. Significantly, the choice and calibration of specific, national-level policy instruments was left entirely to Member States (Di Lucia and Nilsson, 2007: 533). The only other *durability devices* included in the Directive related to monitoring and reporting. Thus under Article 4, Member States were required to submit annual reports to the Commission (a policy instrument-level *durability device*), in which they were supposed to explain their national indicative targets, set out the measures they had adopted to promote biofuels, and describe their impact in terms of national market shares. Each year, Member States were also required to explain and justify any difference between their national indicative target and the overall 'reference value' set at EU level. These reporting requirements were substantially stronger than those outlined in the Commission's original proposal and were the outcome of a complex trade-off between the European Parliament and the Council, in which the former insisted on stronger reporting in exchange for non-binding targets (see Chapter 5).

Crucially, Article 4 also included a flexibility clause, i.e. a policy instrument-level *flexibility device*. This obligated the Commission to publish, by 31 December 2006 'and every two years thereafter', an *ex post* evaluation report on the progress made by Member States (OJ L123, 17.5.2003: 45). Policy designers had evidently

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given some thought to this matter, because the text of the Directive included a detailed list of what the Commission should evaluate including *inter alia* the cost effectiveness of national policies, the economic and environmental costs of production, lifecycle analyses of particular biofuels, and the greenhouse gas emissions arising from each type. In hindsight, it is striking how many of these issues featured in the political controversies that gradually engulfed the sector. Finally, Article 4 concluded that 'if this report concludes that indicative targets are not likely to be achieved' the Commission should submit new proposals that 'address national targets, *including possible mandatory target*s' (Article 4 (2), OJ L123/, 17.5.2003: 46, emphasis added). In the terminology outlined in Chapter 2, this particular *flexibility clause* was a relational contract, not only designed to trigger at a precise point in time, but to be heavily biased in favour of more coercive controls at EU level.

Policy Implementation and Reform

The Commission eventually published its evaluation report in January 2007 (COM (2006) 845; see also Haigh, 2009: 14.11-11), by which point a wide variety of implementation problems had manifested themselves. In that report, the Commission confirmed what many policy designers had long suspected – the interim 2005 target had been missed and the other was 'not likely to be achieved' by 2010 (COM (2006) 845: 6). Consequently, the Commission duly recommended a new, mandatory 10 per cent biofuels target to be achieved by 2020 (COM (2006) 845: 8; see also COM (2006) 848). Moreover, it also sought to remedy a number of specific concerns that had emerged since the original Directive had entered into force. These included the direct and indirect impacts of growing crops for biofuels (e.g. the use of pesticides and fertilisers that could pollute local watercourses) as well as the mitigation potential of particular sub-types of biofuel. It claimed, however, that substituting up to 14 per cent of road fuels with biofuel would have a 'manageable' impact on agriculture (COM (2006) 845: 11; Haigh, 2009: 14.11-1), provided adequate policies were in place to encourage 'good' biofuels and discourage 'bad' ones (Haigh, 2009: 14.11-3). It did not specify what was meant by 'good' and 'bad', or how policy designs would produce them.

Far from resolving these ambiguities, what the Commission did next simply compounded them. Alongside the Biofuels Progress Report, it published a Renewable Energy Road Map (COM (2006) 848). Recall that the mid-2000s were a period in which the EU was setting increasingly ambitious policy programme-level targets (for details, see Chapter 3). The Road Map duly proposed a 20 per cent target for renewables by 2020 and, crucially, confirmed the need for a legally binding 10 per cent biofuels target, citing its own ex post-evaluation of the 2003

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Directive. The new target proposed by the Commission was an explicit recognition that the national indicative targets mandated in the 2003 Directive had not provided a sufficient stimulus to biofuel production (particularly within the EU) and consumption to fulfil EU-wide goals (Johnson, 2011: 99). More detailed proposals to achieve the 10 per cent target, but for renewable fuels, were subsequently published by the Commission in January 2008 as part of a larger proposal for a Renewable Energy Directive (COM (2008) 19). These were adopted as part of the final new Directive (2009/28/EC) which promoted the use of energy from all renewable sources. Amongst other things, the 2009 Renewable Energy Directive repealed the Biofuels Directive as of the end of 2011, which by that point had only been on the statute book for around six years.

As well as introducing more stringent and binding EU-wide targets, the 2009 Directive included an obligatory template to improve the quality and timeliness of Member State reporting (Howes, 2010: 142). However, the cycle of policy change did not end there – a little over six years later (i.e. in 2015), the ILUC Directive (2015/1513/EU) was adopted which amended the biofuel provisions of the 2009 Directive to address the indirect land use change effects triggered by the 2003 and 2009 Directives. Finally, in 2016, the Commission issued a proposal for a recast Renewable Energy Directive (RED II), which brought biofuel policy into line with the EU's post-2020 targets (COM (2016) 767). When adopted, Directive 2018/2001 set a mandatory 14 per cent target for renewable sources in transport for each Member State. Unlike its predecessor, the recast Directive also capped the use of first-generation biofuels to meet this target at one percentage point above the share of those fuels in 2020. It also set a mandatory target for second- and third-generation biofuels of 3.5 per cent in 2030. The cycles of positive and negative policy feedback that facilitated this long instrument sequence are discussed in much greater detail in Chapter 5.

4.3 The Governance of Large Stationary Emitters: Locking In Emissions Trading?

Emission Patterns

Defining the scope of an emission trading system is not a given: it is a key policy design decision that can have important implications for the policy's effectiveness and durability. At a broad level, an emissions trading system can be designed to be either an *upstream* or a *downstream* policy. In an upstream system, fuel producers and importers of (for example) coal and oil must surrender allowances to cover the emissions embedded in the products they sell. In a downstream system, allowances must be surrendered for greenhouse gas emissions at source by (for example) the

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electricity generation industry (Foundation for International Environmental Law and Development, 2000: 23–26). In the case of the EU (and as explained more fully below), a decision was made relatively early on in the design process to create a downstream system due to fears that an upstream system would directly impact Member State energy systems and so require unanimity voting in the Council of Ministers (Foundation for International Environmental Law and Development, 2000: 23, fn. 21).

In a downstream system, design decisions about policy scope centre on both the types of greenhouse gases and the specific activities that will be covered (hence in the EU, stationary point-source installations such as power stations and/or diffuse sources such as transport). Emissions trading systems around the world have widely varying sectoral coverage, some relatively narrow (emissions from electricity generation and industry) others much broader (including road transport, waste and forestry, etc.; see International Carbon Action Partnership, 2019: 21).

The EU ETS covers emissions from electricity generation, energy-intensive industries such as steel production, and aviation. In 2018, ETS emissions (excluding aviation, which is not included in our study,[4] see Chapter 6) were estimated to be 27 per cent lower than in 2005, taking into account changes in the system's scope (EEA, 2018a, 2019). However, this overall pattern masked significant differences between the various sectors. Fuel combustion, largely for electricity generation, accounted for 63 per cent of emissions in 2018 and had witnessed a 25 per cent reduction since 2005. According to the EEA (2018a: 21), these reductions were the 'the main driver of the decline in emissions' across the entire system after 2013. Emissions from the energy-intensive industrial installations (e.g. steel and cement production) accounted for 33 per cent of emissions in 2018. In large part due to the 2008 financial crisis, industrial emissions fell by 12 per cent between 2005 and 2012, but by 2018 they were actually 4 per cent *higher* than when the system started. Similarly, aviation – while accounting for only 4 per cent of 2018 emissions – saw a 25 per cent increase in emissions between 2013 and 2018.

Target Groups and Other Interest Groups

The main industrial target groups can be placed in three broad categories: the electricity generation industry; the energy-intensive industries; and the aviation industry. The electricity generation industry was an obvious actor for the Commission to target, being a significant point-source emitter of CO_2. Around 40 per cent of total EU generation capacity was owned by seven companies in 2013, down from around 60 per cent in 1990 (Dahlmann *et al.*, 2017: 394). Between 2005 and 2012, seventeen of the twenty highest emitters in the Emissions Trading

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System (ETS) were electricity companies, which were collectively responsible for nearly 40 per cent of emissions during that time (Bryant, 2016: 311). At EU level, electricity generators were represented by the Union of the Electricity Industry (Eurelectric), a well-resourced and well-staffed business association. The industry had been targeted by EU air quality policy for years, chiefly because of its contribution to acid rain. As a result, when emissions trading was proposed the industry was highly engaged from the outset, coordinating modelling exercises and working closely with the Commission (Braun, 2009: 481).

Companies in the electricity industry found that they shared a number of concerns when emissions trading was placed on the EU policy agenda. First of all, they generally did not sell electricity outside of the EU and hence had low vulnerability to global competition. They also calculated that they could 'work within' any EU-wide system; a significant percentage of any additional cost of purchasing emission allowances could be passed on to their customers (see Sijm *et al.*, 2006). Despite some shared interests, high industry concentration and unified EU-level representation by Eurelectric, emissions trading was still expected to generate different effects across the industry, largely depending on how carbon-intensive their operations were. High-carbon electricity companies such as Germany's RWE or Poland's Tauron Polska Energia – which relied on coal and had a relatively high CO_2 intensity of electricity production – preferred less-coercive policy instruments such as voluntary agreements. They were more reliant on allowances being allocated for free instead of being sold (see below) and would be disadvantaged by a high carbon price (Chen *et al.*, 2008). In contrast, low-carbon electricity companies such as France's EDF – which generated electricity from nuclear or renewables and therefore had a low CO_2 intensity of electricity production – perceived that they would be less vulnerable to auctioning and could actually benefit from high carbon prices by raising their electricity prices without significantly increasing their costs (Keppler and Crucini, 2010).

The second major target group was the energy-intensive industries, which included the steel, cement, refining, glass and paper manufacturers. Unlike the electricity industry, this was a much more disparate group of actors operating in many different markets that were relatively exposed to international competition. In some cases, they actively competed with one another: e.g. the steel and aluminium industries fought to supply car producers (Roth *et al.*, 2001). This fragmentation was reflected in the manner in which they were represented in Brussels. For example, a total of twelve EU-level associations representing energy-intensive industries responded to the European Commission's first consultation on emissions trading (see European Commission, 2001c). Initially, they did not place a high priority on participating in policy formulation activities and hence did not produce or actively communicate common policy positions (Wettestad, 2009b;

Skodvin *et al.*, 2010; Meckling, 2011: 38). However, in contrast to its effects on the electricity industry, the EU ETS created shared policy concerns for the energy-intensive industries (Wettestad, 2009b). A large number were intensive users of electricity, traded their products in global markets, benefited from freely allocated allowances and were disadvantaged when carbon prices rose. In Chapter 6 we shall show how the ETS forced them to coordinate more effectively (a policy feedback effect), eventually forming a new coalition in 2005 – the Alliance of Energy Intensive Industries (AEII). The initial preferences of the main EU institutions and NGOs are covered in more detail in Chapter 6.

The Design Space

The theoretical potential of emissions trading has been extensively debated by economists, but in the 1990s it remained a rather unappealing and 'peripheral' policy concept in the EU (Boasson and Wettestad, 2013: 56). It was, after all, a relatively novel instrument globally and ran counter to the EU's policy instrument preference for regulatory instruments.[5] However, these prior regulatory interventions furnished a good deal of usable knowledge about emissions from the sector (note the operation of an interpretive policy feedback mechanism) – detailed knowledge that proved to be directly salient to the Commission's emerging plans for emissions trading (Wettestad, 2005: 4).

Emissions trading had first been employed by US authorities in the 1980s (Voss, 2007; Hansjürgens, 2011: 639), and subsequently diffused to some EU Member States – principally Denmark (from 1999) and the United Kingdom (from 2002). Somewhat ironically, around fifteen years after it began operation, the EU ETS is still the world's only supranational emissions trading system. This is one of the main reasons why it has been described as a 'bold public policy experiment' (Ellerman *et al.*, 2010: 288) and 'a major feat of policy innovation' (Bailey, 2010: 145). According to one systematic analysis of the ETS's first trading period, it 'lifted the environment from the boiler room to the boardroom, from ministries of environment to ministries of finance, and from councils to Cabinet tables' (Ellerman *et al.*, 2010: 1; see also Ellerman *et al.*, 2016 and Wettestad, 2005: 19).

The Initial Policy Design

The ETS was adopted via a 2003 Directive (2003/87/EC) 'establishing a scheme for greenhouse gas emission allowance trading within the Community and amending Council Directive 96/61/EC'. The 2003 Directive laid the ground rules of the system and allocated governance tasks to the various participants. It established the ETS as a *cap-and-trade* system, capping the emissions from stationary

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emitters by allocating emission allowances, and then making trading of these allowances possible in order to achieve emission reductions at lower cost (Woerdman *et al.*, 2015: 43). Unlike the UK ETS (but like the Danish ETS), it was designed to be a *mandatory* scheme. Thus, installations covered by the system were required to hold a legal permit to emit greenhouse gases and to surrender allowances equal to their annual emissions. If they emitted more than that they would be fined. Article 10 of the Directive stipulated that at least 95 per cent of the emissions allowances would be allocated for free directly to target industries in Phase I (2005–2007), and at least 90 per cent in Phase II (2008–2012; see Woerdman *et al.* 2015: 56). Auctioning allowances to the highest bidder is the preferred allocation method advocated by many economists (Hepburn *et al.* 2006), but in the EU's scheme, free allocation was, as we shall explain in Chapter 6, eventually selected as 'the political price for ensuring' sufficient Member State support (Ellerman *et al.*, 2016: 4). In its first two trading phases, the EU's system was designed to be decentralised – many of the most significant decisions about the total quantity and allocation of the emission allowances were placed in the hands of Member States (Skjærseth and Wettestad, 2010b: 66). Crucially, no formal distinction was made between different types of emitters, e.g. the electricity generators versus the energy-intensive industries, although Member States de facto arrived at such a distinction when they began to allocate allowances (Wettestad, 2009a: 312).

The ETS mainly addresses CO_2 emissions from stationery sources, namely 11,000 power stations and industrial plants listed in Annex 1 (Delbeke and Vis, 2015: 41; DG CLIMA, 2016 : 4). It encompasses activities in 31 European countries – the 28 Member States of the EU plus 3 non-Member States (DG CLIMA, 2016: 96). The aims of the Directive were laid out in Article 1: 'to promote reductions of greenhouse gas emissions in a cost effective and economically efficient manner'. Other than one small reference to encouraging energy-efficient technologies (in the opening recitals of the Directive – number 20), no reference was made to other policy aims which emissions trading is often associated with, such as the nurturing of green technologies (Woerdman *et al.*, 2015: 61).

The system has been organised around a series of 'trading periods' (DG CLIMA, 2016). Thus, the main institutional structures of the policy were established before the initial period of trading in Phase I (2005–2007), later described as a pilot phase (Ellerman and Buchner, 2007; Ellerman *et al.*, 2010). During Phase I, the overall cap was set using emission estimates derived prior to the pilot phase (DG CLIMA, 2016: 4). Phase II ran from 2008 to 2012 and coincided with the first global trading period under the Kyoto Protocol. Phase III extended the lifetime of the system from 2013 to 2020, while Phase IV will cover the time period from 2021 to 2030.

The 2003 Directive allocated a number of important responsibilities amongst the various actors involved. As the system was – at least initially – decentralised, many of these responsibilities went to the Member States. Thus, Article 27 required them to bring into force all relevant legal and administrative provisions by 31 December 2003, draw up a publicly accessible registry of allowances and submit regular monitoring reports to the Commission (Haigh, 2009: 14.13-2). Importantly, Member States were empowered to decide how to allocate the allowances within their respective territories, through the drawing up of National Allocation Plans (NAPs) informed by 11 criteria listed in Annex 3 of the Directive. The EU cap comprised the sum total of the allowances allocated through the NAPs; other than the fines, the cap was in effect the main policy instrument-level *durability device*. In practice, we shall see that the vast majority of Member States opted to allocate their allowances to emitters based on their historical emissions (and indeed were constrained by the Directive from auctioning no more than a small percentage). Member States were also required to decide the fate of the relatively small percentage of allowances that could in theory be auctioned.

The act of continually trading in allowances is meant to guarantee that all trading systems achieve a basic level of dynamic flexibility. However, in the EU ETS two other obligations were placed on the Commission to introduce even more. First of all, the 2003 Directive required it to produce an *ex post* evaluation report on the application of the Directive by 30 June 2006. This report would consider no less than eleven specific issues which were outlined in Article 30 and, if relevant, make recommendations for a new Directive (hence it was a policy instrument-level *flexibility device*). The Commission eventually commenced its review in 2005 (Skjærseth and Wettestad, 2010: 70). Article 30 also gave the Commission the option to propose an amendment to Annex 1 by 31 December 2004 to extend the scope of the emissions trading system to other sectors, providing there was sufficient monitoring information on greenhouse gases – i.e. another policy instrument-level *flexibility clause*. During the policy formulation process, the chemicals, aluminium and transport sectors had been mentioned as possible candidates for later inclusion (Skjærseth and Wettestad, 2010b: 69). Second, Article 14 obliged the Commission to draw up more detailed monitoring and reporting guidelines (Haigh, 2009: 14.13-2), which were eventually formulated through the comitology process, and enacted via technical Regulation 2216/2004 (Haigh, 2009, 14.13-5). These could be thought of as another policy instrument-level *durability device*.

Over time, the 2003 Directive was gradually supplemented with more detailed policy guidance on a range of technical matters such as allowance allocation, monitoring and reporting. In general, these were formulated by the Commission via the comitology process, the aim being to speed up decision making to facilitate faster adjustments. However, in the formulation process, the Member States insisted that more far-reaching changes to the scope and functioning of the system[6] had to be adopted through primary legislation – i.e. via new directives – which of course brought in the other EU institutions. Eventually the 2003 Directive was amended, most conspicuously in 2009 and then again in 2018. With hindsight, the decision to distinguish between these two different routes to making subsequent adjustments was a fateful one, and led to a good deal of acrimony and, paradoxically for something that had sought to facilitate flexibility, significant delay.

The 2009 Directive, the design of which was informed by the 2005–2006 review mentioned above (Skjærseth and Wettestad, 2010a, was eventually adopted in just 457 days – an even more 'speedy' birth than the original 2003 Directive (Skjærseth and Wettestad, 2010b: 66). During Phase III, the overall cap was set centrally to reflect longer-term EU-wide emission reductions targets, 57 per cent of allowances were auctioned and there were more provisions addressing the concerns of the energy-intensive industries such as carbon leakage (Skodvin *et al.*, 2010). A fourth phase is scheduled to start after 2020. At the time of this writing, the system as a whole does not have an end date (DG CLIMA, 2016: 7), although this does not imply that its existence is fully accepted by all concerned.

Policy Implementation and Reform

The process of transposing the 2003 Directive into national law was supposed to have been completed by the end of December 2003, but only the United Kingdom complied on time and the Commission had to resort to extensive enforcement action against many Member States (Wettestad, 2005: 19). Then, the production of the National Allocation Plans (NAPs) proved to be a considerably more complex and time-consuming process than had been originally foreseen (Ellerman et al., 2015: 4), which triggered disagreements and delays; once again, the Commission resorted to legal enforcement measures (Haigh, 2009, 14.13-6). At first, allowance prices in the new system climbed steadily (Skjærseth and Wettestad, 2010b: 69), but then dramatically collapsed in mid-2006 when it became obvious that Member States had over-allocated allowances to the point-source emitters in their jurisdictions. When electricity generators in some countries passed on a substantial proportion of the higher market price of the allowances to their customers, they were accused of generating 'windfall' profits (Energy Intensive Industries, 2004; Woerdman *et al.*, 2015: 66).

The Commission began to tackle some of the fundamental causes of these problems in 2005, informed by the Article 30 implementation report noted above (Skjærseth and Wettestad, 2010b: 65). A formal proposal to amend the 2003 Directive was eventually published in 2008. The aims of the 2009 Directive

(2009/29/EC) were contained in its formal title, namely to 'improve and extend' the existing system. Amongst its most significant design features were:

- *A single, EU-wide cap*: between 2013 and 2020 this declined automatically over time (by 1.74 per cent per year – a downward slope known as the linear reduction factor) to ensure that the EU fulfilled its 2020 emission reduction target (Wettestad *et al.*, 2012: 73). In effect, this was a new, automatic policy instrument-level *durability device*. Accordingly, the decentralised (and manual) process of determining the overall cap (via the NAPs) was discontinued.
- *Much greater auctioning of allowances*: auctioning became the norm for electricity generators in 2013 (with partial derogations for installations in Central and Eastern European Member States), but energy-intensive industries continued to receive free allocation, albeit at a reduced level.
- *A wider scope*: more industries were included (e.g. aluminium production and petrochemicals) and some additional gases covered (DG CLIMA, 2016: 12, 18).

In some respects, the design of the system became significantly more complex (i.e. much greater differentiation between and within sectors), more centralised (i.e. a single, EU-wide cap and auctioning), and more automated (Wettestad *et al.*, 2012: 73–74; Müller and Slominski, 2013: 1437). The Commission was also given more administrative responsibilities. Chief amongst them was the production of carbon leakage lists (Article 10) to guide the allocation of free allowances; they were to be updated every year on the basis of state-of-the-art technology benchmarks. This proved to be a considerable new administrative task – the first list encompassed no fewer than 165 sectors (Müller and Slominski, 2013: 1437) – and quickly become a new focus of target group lobbying (an example of interpretive policy feedback). Member States were also required to make the collection and spending of auction revenues more transparent. In general, it maintained their right to determine how revenues were spent, but pledged that 'at least 50%' would be used to combat climate change (DG CLIMA, 2016: 35). The various other ways in which the 2003 Directive fed back on and in turn affected the design of the 2009 and 2018 Directives are discussed in Chapter 6.

Ex post evaluations of the EU ETS have focused on its first and second order effects on emissions, profits, investment and carbon leakage (Laing *et al.*, 2014: 510). They are technically quite complicated to produce as they require robust data and realistic counterfactuals (Laing *et al.*, 2014: 510; Branger *et al.*, 2015: 10). Confounding factors (the financial crisis, fuel switching, technological innovation, etc.) had to be identified and carefully disaggregated. Most have focused on emission reductions achieved, as that was (and remains) the declared aim of the system. Published evaluations have reported reductions of between 2 and 4 per cent of the total capped emissions, which may not seem substantial but is relatively

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significant given the massive allowance surpluses in Phases I and II (Laing *et al.*, 2014: 516). The effects of the system on private investment and technological innovation are thought to have been quite limited (Laing *et al.*, 2014: 516; Branger *et al.*, 2015: 12), reflecting the relatively low price of allowances.

The policy feedbacks created by the ETS have, by contrast, received relatively little attention in the existing literature (but see Skjærseth, 2018; Wettestad and Jevnaker, 2019: 6, 18). The electricity generators have proven particularly adept at passing through costs to their customers, although cost pass-through has occurred in all sectors (Laing *et al.*, 2014: 514), the precise extent being a function of sectoral- and firm-level characteristics (Convery, 2008: 128; Skodvin *et al.*, 2010: 860; Skjærseth, 2013: 46). Although modest at first, these feedbacks effects have, over time, became more pronounced and politically consequential, causing the 2003 Directive to affect the design of subsequent directives (see Wettestad, 2009b: 310), and encouraging actors such as the Commission to push for the scope to be expanded (see Graichen *et al.*, 2017). From 2005 to 2012, the system covered carbon dioxide emissions from electricity generation and industrial processes. Starting in 2012, it was expanded to cover aviation emissions from flights wholly within the EU. In 2013, the scope was again expanded to include several additional industries (chemicals and aluminium production) as well as emissions of non-CO_2 greenhouse gases from a specific, relatively limited set of industrial processes. Its scope has also expanded geographically as a result of the accession of new Member States – Bulgaria and Romania in 2007 and Croatia in 2013 – as well as its expansion to the EEA in 2008. If these extensions in scope had been in effect in 2005, the scope of the original 2003 Directive would have been approximately 15 per cent greater (Graichen *et al.*, 2017: 7). Further details of these changes in scope, stringency and timeframe are provided in Chapter 6.

4.4 The Governance of Car Emissions: From Voluntary Action to Regulation?

Emission Patterns

When, in the late 1990s, the EU came under international pressure to back up its emission reduction pledges with internal policies, political attention inevitably focused on the transport sector. A 1997 European Commission Communication noted that greenhouse gas emissions from the sector had risen by 10 per cent between 1990 and 1995 (Volpi and Singer, 2002: 143). More significantly, it predicted that without stronger internal policies, transport would account for nearly 40 per cent of total EU CO_2 emissions by 2010 (Haigh, 2009: 14.2-3). The Commission concluded that from both a political and an environmental

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perspective, the policy *status quo* was patently unsustainable. In the late 1990s, it warned that if the transport sector continued on the same trajectory, it would not only imperil the EU's international climate leadership ambitions, which significantly increased in the 2000s (see Chapter 3), but require other sectors to take up the difference (which it argued would be unfair to those sectors).

Fast-forward fifteen years to 2013 and the transport sector still accounted for around 25 per cent of the EU's total greenhouse gas emissions, of which passenger cars alone contributed 43 per cent (EEA, 2015a: 28). The Commission's earlier warning about rising emissions were thus well-founded. In fact, transport remained the only sector from which emissions increased year-on-year between 1990 and 2013 (EEA, 2015a: 6–7). The EEA has estimated that between 1990 and 2013, emissions rose by almost 20 per cent (against an EU-wide target of a 40 per cent *reduction* by 2030); from road transport, they had climbed by almost 17 per cent (EEA, 2015a: 8). In other sectors (principally electricity generation – which falls within the scope of the emissions trading system), emissions have fallen.

Cars, heavy goods vehicles, road fuels and spare parts are all traded across borders. As a result, the Commission has repeatedly argued that the EU should be involved in significant policy design decisions. However, there were and still are other large and powerful incumbent players, not least the fuel and car-production companies, many of which have existed for well over a century (Urry, 2008). One of the running themes of the ongoing politics in the transport sector has been the battle between the fuel and car industries to shape policy designers' perceptions of the prevailing design space (Weale *et al.*, 2000: 405). Politicians have certainly struggled to reach durable decisions on who should enjoy the benefits and shoulder the costs of transport-related climate policy: fuel producers, car producers and/or consumers?

The interaction between these aspects has done much to constrain the politically feasible design space. Thus the fuel economy of private cars improved significantly after the 1970 oil crisis, but declined throughout the 1980s and 1990s as the economy strengthened and consumers opted to purchase larger and heavier vehicles (HM Government, 2013: 13). This trend partly reflected consumer tastes (which the car companies arguably worked hard to influence), technological shifts and also, in part, related EU policies tackling urban air pollutants, noise and driver safety (Keay-Bright, 2000: 14). Against this backdrop, the EEA repeatedly argued that 'significant additional measures' (EEA, 2015a: 10) were needed to ensure the sector played its part in fulfilling the EU's decarbonisation ambitions. In principle, transport emissions occur along the whole supply chain, offering many potential points at which policies could be targeted. However, the direct emissions from car tailpipes were quickly identified as the key target for climate policy design activities: they were relatively easy to quantify and were already the subject of existing EU policies on localised pollutants.

Target Groups and Other Interest Groups

The car industry is large, interconnected and mature, employing millions of people either directly or in its expansive supply and servicing chains (Mikler, 2009). By the early 2010s, annual global car sales had climbed to some €2 trillion (Wells, 2010: 2). In the EU alone, around 13 million new cars were registered in 2010 (DG CLIMA, 2011). The industry is part of a much larger 'regime of automobility' encompassing the production and sale of new cars, through to their fuelling, maintenance and disposal (Smith *et al.*, 2010: 440). Twelve million people (around 6 per cent of total EU employment) currently manufacture and service them in the EU (European Automobile Manufacturers' Association (ACEA), 2017: 1; COM (2017) 676: 1). Moreover, the car is not just 'a technology': it is an entire 'way of life, an entire culture' for millions of people (Urry, 2008: 347), extensively underpinned by what Pierson (1993) would presumably recognise as policy lock-ins. These are not, one might think, conducive to the adoption of new policy instruments that trigger extensive positive feedback. On the contrary, history suggested that the sector would fight hard to 'lock out' disruptive policy change.

The car industry certainly offered few 'silver bullet' solutions to the challenge of rising emissions. More often than not, it presented the continuation of the dominant existing technology – the fossil fuel-powered internal combustion engine (Ntziachristos and Dilara, 2012) – as a given. The industry sought to preserve this core technology by supporting and often securing the adoption of policies that facilitated incremental or 'drop-in' technological responses. The most common examples included the fitting of catalytic converters (to address local air pollution challenges) and (in relation to climate change) making existing diesel engines more efficient, reducing air and rolling resistance, and introducing devices that automatically switch off engines at traffic lights. Additional incremental technologies suggested by the sector included biofuels (see above) and hybrid electric-diesel engines (Unruh, 2002: 318; de Wilde and Kroon, 2013: 2). However, while drop-in solutions might fulfil the EU's short to medium-term mitigation targets (i.e. until the late 2020s), environmental groups repeatedly maintained that more radical innovations, including hydrogen or electric vehicles, will eventually be required to fully decarbonise the sector (de Wilde and Kroon, 2013: 2; see also Chapter 3). At present, however, these innovations reside in technological niches: e.g. electric and plug-in hybrid vehicles accounted for only 1.5 per cent of total new car sales in the EU in 2017 (EEA, 2018b).

The Design Space

When climate change first rose up the EU's political agenda in the 1990s, the policy designers in the Commission had to decide which policy design levers to pull without becoming mired in a prolonged battle with powerful incumbent

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industries. The fact that the regulation of air pollution from vehicles represented one of the oldest sub-sectors in EU environmental policy created its own path-dependent effects. In fact, the most feasible design options that were available to the Commission in the 1990s were essentially the same as they had been in the late 1960s when the EU first became involved in governing the car industry (Weale *et al.*, 2000: 398). The first option was to focus on well-to-tank emissions by altering the quality of fossil fuels or switching to alternative sources such as electricity or hydrogen (hence primarily an issue for the fuel-supply industry). The second option was to focus on tank-to-wheel emissions and set car emission standards (hence primarily a concern for car manufacturers). The third was to modify driver behaviour (e.g. through education campaigns and altering the layout of roads). Given the balance of power between the various actors, it was almost inevitable that the process of selecting amongst these options would generate conflict, pitting EU institutions and Member States against one another, often backed by powerful interest groups (Wurzel, 2002: 134).

To complicate matters further, the main target groups, whilst united on broader issues, regularly disagreed on specific policy details. Intense disagreements between producers of large, premium vehicles (BMW, Porsche) and producers of smaller, more economical alternatives (Peugeot, FIAT, etc.) first surfaced in the 1970s and 1980s (Friedrich *et al.*, 2000: 608). Indeed, the European industry organisation, the European Automobile Manufacturers' Association (ACEA), was established in 1991 to give the industry a common (and, in particular, a more environmental) face (Wurzel, 2002: 141). By the late 1980s, the Commission had come around to accepting that a more collaborative approach would achieve environmental goals more quickly than resorting to *dirigiste* forms of regulation (Wurzel, 2002: 162). Dubbed the 'Auto-Oil' programme, it formally commenced in 1991 with the aim of building a more harmonious and forward-looking relationship between the main policy designers (Friedrich *et al.*, 2000: 609).

It was at this point that climate change started to rise up the EU's political agenda. Although the actors and the technologies (and hence the preferred solutions) were essentially the same, greenhouse gases nonetheless constituted a new category of pollutants, and hence a somewhat different policy design problem. Following a historic October 1990 joint meeting of the Energy and Environment Councils (at which the EU offered, for the first time, to adopt a collective emissions target – see Chapter 3), the Commission commenced detailed design work. Almost immediately, it ran into strong political opposition from ACEA, whose members lobbied their national governments to lock out radical new policy and technological innovations. Fearing a prolonged battle, the Commission opted instead to follow the grain of existing policy designs and focus on fuel quality and/or car emissions – the two main foci of the then ongoing Auto-Oil discussions.

But even so, the design process moved extremely slowly. The outcome – summarised in Chapter 7 – was one of the EU's first voluntary agreements in the environment sector, which sought to reduce CO_2 emissions from all new cars produced in the EU.

The Initial Policy Design

When it was eventually adopted in 1998, the voluntary agreement on CO_2 from cars was hailed as a policy innovation because it departed from the EU's established preferences for regulatory instruments (see Chapter 3). It committed European manufacturers to reduce average CO_2 emissions from all new passenger cars sold on the EU market from an average of 186g/km in 1995 to 140g/km by 2008, roughly equating to a 25 per cent cut over ten years (COM (1998) 495: 3; see also Bongaerts, 1999: 102). In turn, the agreement was expected to contribute around 70 per cent of the total reductions required to achieve a more ambitious longer-term target of 120g/km by 2012 (Recommendation 1999/125/EC). However, it took over three years to negotiate – the slowest adoption process of the fifteen policy instrument changes analysed in this book.

The agreement's objectives could be thought of as an instrument-level *durability device,* which aimed to encourage manufacturers to produce cleaner vehicles. To ensure that it remained relevant, it also incorporated several *flexibility devices.* For example, the emission target was time-specific (i.e. to be achieved by 2008–2012). It also incorporated a *flexibility clause* which committed ACEA (i.e. *not* the Commission), its main co-signatory, to review (by 2003) the potential for further improvements to be made (by 2012) to achieve a fleet average of 120g/km. This clause was, however, connected to a *durability device* (an intermediate target to achieve 165–170g/km by 2003), although the connection was ambiguously worded. Thus, if – and only if – ACEA failed to achieve the interim target (the degree of underachievement was not fully specified), would the Commission formally review the agreement (i.e. undertake an *ex post* evaluation – another type of *flexibility device*). Only then would it 'consider drawing up a proposal for binding legislation' (COM (1998) 495: 5). Furthermore, 'some European manufacturers' (no further details on their identify were given) were expected to produce cars that were capable of achieving the tougher 120g/km standard by 2000 (COM (1998) 495: 5). At first blush, the looseness of this commitment did not appear to put manufacturers under much pressure to make large, upfront investments in the durability of the policy. Finally, the Commission undertook to negotiate similar agreements with Japanese and Korean manufacturers to prevent them from securing an uncompetitive advantage – one amongst many pre-conditions that ACEA laid down before signing the agreement.

The agreement, incorporating ACEA's conditions (now re-termed 'assumptions'), was eventually published as an EU Recommendation in 1999, more than three years after the commencement of the policy design process (1999/125/EC). It was signed by the board of ACEA on behalf of its individual members, who, crucially, only 'endeavoured to contribute to' its implementation (Bongaerts, 1999: 102). It was presented as the centrepiece of a larger package of instruments addressing transport emissions, proposals for which were published by the Commission in December 1995 (COM (95) 689; see Bongaerts, 1999: 101; Haigh, 2009: 14.8-3). The other three elements in the package were designed to make up the missing 30 per cent reduction (i.e. to 120g/km) noted above. As Chapter 5 will show, this missing percentage turned out to be politically significant, because it equated to the difference between the reduction target sought by the Environment Council as far back as 1994 and the target (140g/km) which was included in the agreement after the prolonged, three-year design process.

An important additional part of the Commission's 1995 transport package sought to ensure that all new cars were better labelled so that consumers had access to more information on fuel usage and CO_2 emissions (these labelling requirements were adopted as Directive 1999/94/EC). The Commission hoped that this would inform purchasing decisions, which in turn would eventually incentivise manufacturers to produce cleaner and more efficient cars. After a great deal of discussion, an additional instrument was eventually adopted in 2000, which ensured that the right data was collected from the car manufacturers (Decision 1753/2000/EC). This Decision, which was in effect a polity-based *durability device*, also mandated the Commission to collect information about how the agreement was performing. However, the producers proved extremely reluctant to release it (largely, they claimed, to preserve their commercial confidentiality). For the first three years of the agreement (i.e. to 2003), ACEA was therefore only willing to supply data to the Commission; after 2003, it was independently verified by the Commission and released to the public in an anonymised form (Volpi and Singer, 2002: 150). The final part of the package was to have been another directive to stimulate green car purchases by adjusting national tax levels. However, this too foundered and was eventually shelved, blocked by Member States who were opposed to the EU becoming more involved in national tax affairs (Haigh, 2009: 14.8-7), leaving the whole proposal rather less package-like than the Commission had originally hoped.

Policy Implementation and Reform

At first, the implementation of the agreement proceeded smoothly and the car manufacturers met their targets with room to spare. But progress soon faltered and despite a late rally, the 2008 target was missed by some distance (Jordan and Matt, 2014). Having failed to achieve emissions reductions voluntarily, in 2009 the Commission claimed that it had no option but to change direction. It exploited the agreement's *flexibility clause* to formulate and then adopt a completely different policy instrument – a regulation – which was not only more coercive[7] and more finely tuned (i.e. through company-specific targets), but also had considerably stricter, binding standards set for 2015 and 2020. The instrument-level *durability devices* that were eventually inserted into the design of what came to be known as the 2009 Cars Regulation (443/2009) included fines for individual manufacturers that did not meet their company-specific targets. As the durability devices in the Regulation bore down on the producers, they were forced to make more significant investments in the long-term success of the EU's policy and more significant emission reductions eventually began to accrue (EEA, 2015a: 8). In fact, the time-specific *durability device* that required manufacturers to achieve a fleet average of 130g/km by 2015 delivered compliance a full two years ahead of schedule. Considerably more technological progress will need to be made to deliver on the 2020 target of 95g/km.[8] Designers were unable to agree on how to achieve such significant reductions when the Regulation was formulated in 2007–2008. Article 13 (5) of the 2009 Regulation thus committed the Commission to work with producers to review 'the modalities for reaching the 2020 target' by 1 January 2013 (i.e. a policy instrument-level *flexibility clause*) and issue a formal proposal to amend the 2009 Directive. A new policy proposal was duly published in July 2012 (COM (2012) 393), debated and discussed and eventually adopted as Regulation 333/2014.

After two decades of policy design, the core issues of responsibility and political feasibility are still as deeply contested amongst the major players as they were in the late 1980s. As will become clear in Chapter 7, even something as apparently self-evident as how to measure the tail-pipe emissions from particular vehicle types has proven difficult to agree upon. The 2009 Regulation reported emissions under something known as the standardised test cycle, which was applied by national authorities and certified by the Commission. Environmental groups claimed that it significantly under-reported emission levels – claims that were vindicated when (in 2015) the world's largest manufacturer of diesel engine vehicles – Volkswagen (VW) – was found to have deliberately cheated on them to achieve emission reductions at lower cost. This revelation triggered a worldwide scandal known as 'Dieselgate', after which the sale of new diesel cars in the EU plummeted and VW was forced to fight costly legal actions brought by car owners in the US. The whole affair shone an unflattering light on the lobbying activities of the car manufacturers in Brussels and eventually encouraged the EU to adopt tighter emission reduction targets and establish a new testing regime that better reflected 'real world' driving conditions (EEA, 2015a: 8).

When, as outlined in the previous chapter, the EU adopted its post-2020 emission reduction commitments in 2014, the Commission embarked on another reformulation of the EU's car emissions policy. It issued a new regulatory proposal in November 2017 that set an interim ('by 2025') target of a 15 per cent reduction from 2021 levels and a 30 per cent reduction by 2030 (COM (2017) 676). In December 2018, the Council and the Parliament agreed to an amended regulation which kept the Commission's proposed 15 per cent by 2025 target but set a more stringent target of 37.5 per cent below 2021 levels by 2030 (Regulation 2019/631). This equates to average emissions of 59g/km which, if achieved, would represent a 68 per cent reduction in the emissions from new cars since 1995 (COM (1998) 495: 3) – far more than had been envisaged when the voluntary agreement was adopted in 1998.

4.5 Conclusions

In this chapter we have introduced the three policy instruments – a regulation, an emissions trading system, and a voluntary agreement – that constitute the main foci of Chapters 5–7 respectively. For each instrument, we have outlined the pre-existing pattern of emissions, and the main designers and target groups. Then, we summarised the policy design space in each policy sub-area and noted the most salient design features of the instruments that were adopted. On closer inspection, these features include a complicated mix of durability and flexibility devices, operating at the level of policy instrument goals and policy instrument calibrations. If nothing else, we have confirmed that, to paraphrase Chapter 1, policy instrument design is more or less the essence of everyday governance. Policy instrument design processes in the real world often do take a long time to accomplish and, as we have shown, generate a good deal of political conflict. However, these three instruments were just the starting instruments in the respective sequences. Throughout this chapter, we have hinted at the existence of certain resource/incentive and interpretive feedback mechanisms and effects and noted them as priorities for more in-depth analysis in the next three chapters.

Endnotes

1 Biomass, biofuels and other non-fossil organic fuels are collectively known as bioenergy (Environmental Audit Committee, 2008: 5).
2 For a detailed list, see Annex IX of the 2015 Directive (OJ L239, 15.9.2015: 28-9).
3 Including all support mechanisms such as excise tax exemptions, capital grants and R&D funds.
4 Policy changes affecting the aviation industry are generally decided separately and the industry is still treated somewhat differently than other sectors (e.g. with a separate type of emission allowances). For further information, see Andlovic and Lehmann (2014).
5 The Large Combustion Plant Directive of 1988 and the Integrated Pollution Prevention and Control Directive of 1996 being especially relevant examples (for more examples, see COM (2000) 87: 8).

6 For example, the Linking Directive (2004/101/EC) linked the system to the other Kyoto Protocol flexibility mechanisms. In 2008, the EU adopted an Aviation Directive (2008/101/EC) which extended the system to the aviation sector. Because of space constraints, we mainly concentrate on the three core directives: the original 2003 Directive (2003/87/EC), and the two amending Directives adopted in 2009 (2009/29/EC) and 2018 (2018/410/EC) respectively.
7 And recall that, unlike directives, EU regulations are immediately effective (i.e. they do not have to be transposed in national legislation). See Chapter 2 for details.
8 In 2017, average emissions increased slightly for the first time since the 2009 Regulation came into effect (EEA, 2018b).