RESEARCH HANDBOOK ON EMISSIONS TRADING

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Abbreviations

AAUs	Assigned Amount Units
AB 32	Global Warming Solutions Act
ACER	Agency for the Cooperation of Energy Regulator
ADDs	Anti-dumping duties
ADP	Ad Hoc Working Group on the Durban Platform for Enhanced Action
AR5	Fifth Assessment Report
ARP	(United States) Acid Rain Program
ASCM	WTO Agreement on Subsidies and Countervailing Measures
BAT	best available technology
BAU	business as usual
BCA	border carbon adjustment
CA ETS	California Emissions Trading System
CAA	Clear Air Act
CAIR	Clean Air Interstate Rule
CARB	California Air Resources Board
CAT	Cap-and-Trade
CBDRRC	common but differentiated responsibilities and respective capabilities
CCC	California Chamber of Commerce
CDM	Clean Development Mechanism
CERs	Certified emission reduction units
CFI	Court of First Instance
CJEU	Court of Justice of the European Union
CO_2	Carbon dioxide
CO_{2e}	Carbon dioxide equivalent
COP	Conference of the Parties
CT	Credit Trading
CVDs	countervailing duties
ECJ	European Court of Justice
EEA	European Economic Area
EFTA	European Free Trade Association
EIA	environmental impact assessment
EPA	Environmental Protection Agency
ETS	Emissions Trading Systems

EU	European Union
EUAs	European Emission Allowances
GATS	General Agreement on Trade in Services
GATT	General Agreement on Tariffs and Trade
GHG	greenhouse gases
HCFC	hydrofluorocarbon
ICAO	International Civil Aviation Organization
ICMA	Independent Carbon Market Authority
ILC	International Law Commission
INDCs	Intended Nationally Determined Contributions
IPCC	Intergovernmental Panel on Climate Change
JI	Joint Implementation
KP	Kyoto Protocol
KYC	know-your-customer
LDCs	least developed countries
LMDCs	like-minded developing countries
LMOs	living modified organisms
MEAs	multilateral environmental agreements
MFN	most favoured nation
MiFID	Markets in Financial Instruments Directive
MRA	mutual recognition agreement
MRV	measuring, reporting and verification
MSR	Market Stability Reserve
MTIC fraud	missing trader intra-community fraud
NAM	National Association of Manufacturers
NAP	National Allocation Plan
NO_x	Mono-nitrogen oxides
npr-PPMs	non-product related process and production methods
NT	national treatment
NZ	New Zealand
OECD	Organisation for Economic Co-operation and Development
PSR	Performance Standard Rate
PSRT	Performance Standard Rate Trading
RECLAIM	Regional Clean Air Incentives Market
RGGI	Regional Greenhouse Gas Initiative
SIDS	Small Island Developing States
SO_2	Sulfur dioxide
TMG	Tokyo Metropolitan Government
UNFCCC	United Nations Framework Convention on Climate Change
US	United States of America

VAT	Value Added Tax
WCI	Western Climate Initiative
WPR	windfall profit rate
WTO	World Trade Organization

1. Introduction

Stefan E. Weishaar

THIS BOOK'S APPROACH TO EMISSIONS TRADING

Emissions trading systems (ETS) have grown into a respected tool within the environmental policy arsenal and have proliferated around the globe. They are often employed to fight global warming and were referred to at the international climate conference in Paris in December 2015 where political leaders agreed to hold the global average temperature increase well below 2°C and to pursue efforts to limit the temperature increase to 1.5°C.

Emissions trading is usually traced back in the resource economics literature to Crocker (1966) and Dales (1968). Montgomery (1972) is credited as being the first to provide formal proof of its cost efficiency, and Tietenberg (1985) has established it on the economic research agenda. The law and economics literature by contrast tends to trace emissions trading back to Demsetz (1967), who argues that externalities should be internalized by allocating property rights.

As a wide variety of stakeholders and scholars explore ways to address local or more prominently global environmental challenges and as more and more countries adopt emissions trading, emissions trading research has been described as entering into a stage of maturity. In the eyes of policy makers the hype surrounding emissions trading has subsided to a large degree as the understanding about this policy instrument has developed tremendously over the last decade.

This handbook presents the state of the art of research on emissions trading in selected areas for a variety of readers, including researchers in academia, think tanks and policy makers. This introduction describes the interdisciplinary approach taken to address this subject and gives an overview of the chapters contained in this book.

Because emissions trading is a topic that has received interest from many different sides, this handbook takes a broader approach to incorporate several of these areas of study. Economic theories generate the roots from which emissions trading has emerged; the principles and practice of its economic shape, its acceptance and application; law molds its features in terms of its implementation, execution and occasionally also in terms of its design. Consequently, the book is written both for and by people from a range of disciplines, including economics, law and political science. Contributing authors were encouraged to explain their sophisticated analysis in a manner that a broad range of readers can understand. I hope that a deeper appreciation of the issues involved in emissions trading will enhance the ability to effectively design and implement emissions trading measures and encourage research and interdisciplinary collaboration among different fields.

Bearing tribute to the particular research disciplines and approaches, some authors offer a guide through the literature landscape and set a research agenda, while others review the empirics and offer interesting descriptive insights that help to put the theoretical and policy discussions into perspective. Yet others present their own ideas on policy challenges and choices.

A BRIEF OVERVIEW OF THE BOOK

The book is structured into three parts. Part I presents the economic and legal origins of the book, Part II covers several implementation challenges and Part III presents the international dimension.

Part I: Economic and Legal Origins

Dan Cole reviews the theoretic origins and limitations of emissions trading and implications for ETS design. From a historical perspective the author examines the early applications including the US acid rain program and the gasoline lead treading program. The author identifies three main lessons from the ETS experience: first, ETSs can be (more or less) successfully implemented to improve regulatory practice. Second, regulators are sensitive to the trade-off between compliance costs and environmental protection, and third ETS is not a solution to any environmental problem and must be applied carefully.

Andries Nentjes examines how two generic emission trading designs, cap-and-trade and credit trading compare to each other. The author compares them in terms of their economic impact in the presence of perfect and imperfect competition and how they would work if both designs would be implemented within one jurisdiction. The chapter also examines which design would lend itself to beggar thy neighbor policies at national level. Moreover this chapter examines the implications of the two ETS choices in terms of investment in innovation.

Claudia Kettner presents empirical evidence on trading in the EU ETS. The author first analyses EU allowance trade and the use of international credits for compliance under the EU ETS in the first trading phase (2005 to 2007) and second (2008 to 2012) is addressed on country level as well as on sector level. Subsequently the trading flows on installation and company level are assessed. This is complemented by a literature review of trading on company level and the use of banking and borrowing of EUAs.

Part II: Implementation Problems

Beat Hinterman focuses on market power exercised by individual firms within an existing ETS. Market power is crucial because it undermines the traditional presumption that the market clearing price and eventual abatement are independent of the distribution of allowances and can give rise to welfare losses to society. Hinterman first assesses the theoretical literature on imperfect competition in emission permit markets before reviewing the empirical literature.

Marjan Peeters and Huizhen Chen examine sanction regimes of greenhouse gas emission trading systems from a legal perspective. After presenting the theoretical embedding, the authors examine the legal framework for sanctions of excess emissions in the Chinese emissions trading pilots and the EU by reviewing enforcement approaches and recent case law regarding penalties.

Francesco Gullì examines windfall profits in the EU power sector – a sector that is widely acclaimed to enjoy windfall profits. After presenting the basics of windfall profits and distinguishing between those windfall profits attributable to free allocation of allowances and those profits attributable to price increases, the author continues to present the theoretical and empirical literature on cost pass-through in the power sector. Subsequently he calculates the windfall profits in the EU power sector during the first and the second trading phase of the EU ETS and offers a critical appraisal of a national tax levy to address windfall profits.

Karolin Rogge reviews the theoretical literature and empirical evidence on the innovation impact of the EU ETS. The author then examines the EU ETS's impact on organizational innovation, offers several methodological recommendations for future studies and suggests policy implications for decarbonizing the economy.

Ricardo Pereira and Katherine Nield examine financial crimes in the European carbon markets. They describe the types of frauds that emerged in the context of the EU ETS, in particular the Value Added Tax (VAT) fraud and emission allowance thefts. The authors highlight the characteristics of allowances and the registry system that made the EU ETS vulnerable to such practices. Subsequently the authors discuss the regulatory approaches taken to contain these financial frauds.

Josephine van Zeben examines litigation with regard to emissions trading systems. The author provides an analytical overview of the various types and functions of litigation. Based upon the outcomes of particular cases it is examined how litigation has affected both ETS design and its development. Albeit examining different jurisdictions the author draws lessons from litigation for the implementation of emissions trading systems.

Elena Kosolapova examines the international liability of single major emitters. The author shows that both States and private enterprises – even those operating within or falling under an ETS – are under an 'obligation to prevent significant transboundary harm'. The author examines how the procedural and substantive duties related to this positive obligation can constitute the legal basis for challenging single emissions sources. At the same time, the chapter shows that the international climate regime does not contain liability provisions and that the current approaches to state liability in international law are incapable of addressing climate change-related damages. It also demonstrates that the obstacles to domestic climate change liability remain huge.

Christian de Perthuis and Raphael Trotignon explain and analyse surplus control and the ways supply-flexibility could be brought into the EU ETS allowance market. After examining the current causes of the weak EU ETS price signal the authors examine the importance of market expectations and uncertainty before reviewing the Commission's EU ETS report proposal. The authors then continue by presenting their policy proposal of an independent carbon market authority. This chapter – albeit not constituting a research handbook chapter or providing a literature review – provides an interesting point of departure for addressing current challenges under the EU ETS.

Part III: International Dimension

Andreas Tuerk and Andrej Gubina review the literature on linking on emissions trading systems. The authors present the forms of linkages that are discussed in the literature and their challenges and offer an outlook on the role, options and likelihood for linking trading schemes. They also briefly present the existing emissions trading schemes to date.

Kateryna Holzer reviews the applicability of WTO rules to ETS allowances and examines how particular ETS design elements are assessed from a legal and doctrinal perspective. In doing so, the author examines the critical issues of free allocation (in relation to subsidies and anti-dumping rules), revenue recycling and border tax adjustments. The latter is especially discussed in relation to import taxes and cost rebates. The author also reviews the legal barriers of linking emissions trading systems that derive from WTO law.

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This research handbook started out as a joint project with Edwin Woerdman (Groningen University). Unfortunately, we could not finalize the work together after he retired due to illness. Edwin's engagement in the very early stage and comments on a few of the chapters in the first review round is gratefully acknowledged.

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PART I

ECONOMIC AND LEGAL ORIGINS

2. Origins of emissions trading in theory and early practice

Daniel H. Cole

1. INTRODUCTION

Two instruments dominated the first generation of environmental law: (a) technology-based standards (also known as design standards), which required installation of designated pollution-control technologies; and (b) performance standards, which are numerical, nontradable emissions quotas.[1] Both of these instruments have proven effective in controlling pollution emissions, but have long been denigrated (somewhat unfairly[2]) as unduly expensive forms of 'command-and-control' (see, e.g., Ackerman and Stewart 1988). They should be replaced, economists and legal scholars (e.g., Tietenberg 1985; Stewart 1996; Moran 1995) recommend, by more efficient market-based instruments, including effluent taxes[3] and tradable quotas (often referred to as cap-and-trade emissions trading – in this chapter 'cap-and-trade' is used as a generic term to refer to 'emissions trading' because it represents one of the earliest emission trading designs).

Effluent taxes have been used for several decades, with varying degrees of environmental effectiveness (see, e.g., Faure 2012), throughout the world (though not often in the United States). Meanwhile, policy-makers

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1 In practice, these two instruments are often combined. Numeric emissions quotas often are based on, and presumably cannot be met without installation of, available control technologies. However, the distinction between the two instruments remains significant because performance standards that are not based on technological installations may entail significantly higher monitoring costs.

2 As Cole and Grossman (1999; 2002) showed, technology-based standards can be as or more efficient than either effluent taxes or tradeable permits under certain institutional and technological circumstances – circumstances that are not especially rare, even today, in many developing countries. Moreover, to the extent regulations create more social benefits than costs, which has been the case for example with technology-based standards under the Clean Air Act (see Cole and Grossman 1999: 928), there is little basis for complaining that they are 'unduly expensive.'

3 An effluent tax is a per unit charge on discharges into the air, water, or land (Cole and Grossman 2011: 413).

have also been experimenting, since the mid-1970s, with various forms of emissions trading. While 'command-and-control' regulations continue to predominate US and European environmental policies, cap-and-trade programs have been recommended for all kinds of environmental problems (see World Bank 2014), including climate change (see, e.g., Dudek and Palmisano 1988). Indeed, cap-and-trade has become *the* instrument of choice for climate policy.

This chapter presents the theoretic origins and limitations of emissions trading and implications for ETS design (section 2). From a historical perspective the author examines the early applications including the gasoline lead treading program (section 3) before concluding and offering some lessons from this experience.

2. EMISSIONS TRADING IN THEORY[4]

'Pollution, Property, and Prices'

In 1968, the late Canadian economic historian John Dales elaborated the theory of cap-and-trade in his prescient book, *Pollution, Property, and Prices*.[5] Dales envisioned a 'market' in 'pollution rights' created by the government. First, the government would impose a quota limit on allowable emissions, as it regularly does in ordinary regulation. This quota limit, often referred to as a 'cap,' must be set administratively in order to render available emissions units scarce; otherwise, no market for them would develop. With the cap in place, the government would then issue pollution rights (usually referred to as 'allowances' or 'credits'[6]) equal in number to the cap. Each pollution right would be equal to one unit (usually, a ton) of pollution. 'All waste dischargers would then be required to *buy* whatever number of Rights they need; if a factory dumps 1000 tons of waste per year it will have to buy 1000 Rights' (Dales 1968: 93). All of this presumes,

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4 Parts of this chapter are adapted from Cole (2002) and Cole & Grossman (1999).

5 Dales was not the first economist to recommend tradable permitting. Two years earlier, Thomas D. Crocker (1966) of the University of Wisconsin – Milwaukee suggested the idea. Dales, however, was the first to actually describe how such an approach might be structured.

6 The purpose of this nomenclature is to avoid creating 'rights' enforceable against the government, which may wish to remove credits from the market in order to ensure achievement of its emissions-reduction goal. However, as we shall see in a later section of this chapter, whether labeled as 'allowances,' 'credits,' or 'limited authorizations,' they are still property.

of course, that the emitters and the government both have a way of knowing how much waste per year the emitters emit.

In Dales' original conception, pollution rights would be sold by the government (thus combining a 'tax' instrument with a trading instrument), and remain valid for a term ranging from one year to five years. During that period, even if new pollution sources entered the market, the total number of available pollution rights would remain fixed. Consequently, the price of pollution rights would presumably rise over time, as demand increased relative to the fixed supply of entitlements. But even at a relatively low price of say 10 cents per unit of pollution, Dales believed 'some firms will find it profitable to treat their raw wastes before they discharge them, or to dispose of them in some way other than discharging them into waterways. They will thereby reduce the number of Rights they are compelled to buy.' By the same token, firms might not use all of the pollution rights they previously purchased. They could, instead, sell those rights to other pollution sources or, indeed, anyone else who wanted to buy them, including 'conservation groups' that might want to prevent the pollution rights from being 'used' (p. 93). Thus, Dales foresaw, more than 20 years before the fact, a development that surprised many observers of the acid rain trading program in the United States (US) – that conservation groups and even elementary school classes (see Israel 2007) would enter the market as buyers to retire emissions allowances.

All transactions on Dales' model were to be brokered by the government, so that the government could keep track of each polluting firm's changeable quota to ensure compliance. In addition, Dales expected the government to serve as the 'buyer of last resort,' purchasing excess pollution rights to prevent price collapses resulting from unexpected conditions, presumably including economic recessions. He also expected that the government would reserve some amount of 'issued but unsold' pollution rights to offset unexpected price spikes (p. 95). Dales recognized the need to impose both floors and ceilings on per-unit pollution prices to stabilize the market while maintaining its environmental integrity.

Most importantly, Dales foresaw the central importance of government measurement and monitoring of emissions 'to ensure that waste dischargers did not cheat by buying too few Rights (or discharging too much waste)' (p. 97). What firm, after all, would bother to purchase a pollution right in a market exchange if the government had no way to assess whether or not the firm was emitting within its quota limit? But Dales may have been overly optimistic about the administrative costs of a cap-and-trade regime, presuming inaccurately that they inevitably would be lower than the administrative costs of alternative regulatory schemes, including technology-based standards. At least sometimes, technology-based standards entail lower administrative costs, particularly when point-source monitoring of emissions is either technically impossible, prohibitively costly, or both. In such cases, the installation and operation of the pollution-control technology becomes a more cost-effective measure of compliance (see Cole and Grossman, 2011, pp. 418–423). Indeed, the lack of reliable and cost-effective monitoring equipment, at the time Dales first developed the emissions-trading idea, could explain why it was not quickly implemented.

Cole and Grossman (1999; 2002) have explained that, at the time the 1970 Clean Air Act was enacted, reliable and cost-effective monitoring systems did not exist for measuring point source emissions of conventional air pollutants, such as sulfur dioxide, particulate matter, and carbon monoxide, let alone toxic air pollutants. Before setting its first technology-based performance standards, the US EPA (Environmental Protection Agency) really had very little idea how much air pollution was being emitted from thousands of stationary sources located all over the country; for the most part, the agency was forced to rely on inherently unreliable firm self-reporting of emissions. In that circumstance, technology-based standards made good economic sense because, if actual emissions could not be accurately monitored, at least EPA had the capability to inspect plants on a regular basis to ensure that pollution control equipment was installed and operating.

Subsequent evolution of the theory of cap-and-trade today

The theory of cap-and-trade has advanced since Dales first described the instrument in 1968. No one today believes, for instance, that the government must serve as broker for every transaction in pollution rights in order to keep track of changeable quotas; a mandatory reporting system is sufficient for that purpose. But the central idea of emissions trading remains as valid as ever: by allocating scarce (and enforceable) legal rights to pollute that can be freely traded, the government can enlist market forces to reduce compliance costs of pollution reduction. Although Dales never articulated it in this precise way, the fundamental purpose of allowing trading in pollution rights is not to reduce emissions – that job is performed primarily by the cap – but to minimize the costs of achieving the cap.

The singular advantage of a system of transferable pollution rights over command-and-control regulation is that it accounts for the different cost-structures various firms face, even within a single industry, for controlling pollution. Command-and-control regulations, whether technology-based standards or non-tradable quotas, disregard differential compliance costs in forcing all regulated firms to reduce emissions by the same amount or by using the same equipment. A system of transferable pollution rights, by contrast, pays attention to the likelihood that one source may be able to reduce emissions at far lower cost than another. Because unused emissions rights can be sold, firms with relatively low costs of controlling pollution have a market incentive to reduce emissions below their initial quota limits. They can then sell their excess, unused rights to higher-cost controllers, which in turn save money by purchasing allowances – raising their quotas – instead of reducing emissions to original quota limits. Thus, the market redistributes most pollution rights to firms with relatively high costs of pollution control and the lion's share of the pollution-reduction burden to firms with relatively lower costs of control. While the total emission cap remains unchanged, the social costs of attaining it are minimized.

Theoretical Limitations of Cap-and-Trade

Cap-and-trade is not a panacea (*there are no panaceas*). It does not always minimize the *total* costs of environmental protection; nor is it always preferable, even in theory, to effluent taxes or even design standards (e.g. Faure and Weishaar 2012). Cap-and-trade regimes may be subject to technological and/or institutional constraints that would prevent trading markets from operating efficiently, if at all. As noted earlier, no trading market will get off the ground if the scarcity of emissions allowances cannot be enforced. To ensure scarcity, the regulator must know at all times (a) how much pollution each regulated entity is entitled to emit (that is, its quota) *and* (b) how much pollution the entity is actually emitting. In the absence of cost-effective, accurate, reliable, and sometimes source-specific monitoring technologies, emissions cannot be measured and compliance will not be ensured. If the government cannot accurately assess how much each regulated entity is emitting, the scarcity created by the cap cannot be enforced. No market for emissions allowances would develop because no polluter would pay another to obtain additional emissions allowances if it could simply emit beyond its quota without risk of detection and punishment by the government. The importance of available monitoring technologies to the integrity and success of emissions trading regimes can hardly be overstated, although, as we shall see hereafter, it is sometimes neglected by proponents of cap-and-trade.

The same technological constraints that limit cap-and-trade could also hamper some, but not all, of the other approaches to pollution control. Accurate and reliable emissions monitoring mechanisms are essential to cap-and-trade programs or other policy instruments such as effluent taxes, and performance standards. It is not crucial, however, for design standards (also known as 'technology-based standards'), where installation and operation of the required technology itself becomes the measure of compliance. Even if emissions cannot be accurately measured, inspectors can determine at relatively low cost whether emissions control equipment is installed and operating. They will not know exactly how much pollution is coming out of a smokestack or how much the plant has, or has not, reduced emissions; but they will have some confidence, based on the fact of installation and operation of the pollution control technology, that emissions are being reduced. As Hagevik (1969: 94) observed on the eve of passage of the 1970 Clean Air Act, design standards have '[t]he advantage of permit[ting] the government to take interim steps even though it has almost no idea of relevant measurements.' During the early 1970s, the Los Angeles County Air Pollution Control District (predecessor to the South Coast Air Quality Management District) managed to inspect every major stationary source for compliance at least once each month (Cole and Grossman 1999: 920, n. 96).

Aside from technological constraints, institutional impediments to emissions trading may exist. Many countries, for example, do not have well-functioning market institutions, which are a prerequisite to functional cap-and-trade regimes (as well as effluent taxes). The notion of creating a well-functioning emissions trading market in such countries is hugely problematic. Problems such as lack of an independent judiciary to enforce property rights and contracts, corrupt bureaucracies, and so forth, tend to render government cap-and-trade regimes ineffective and inefficient, just as they hamper markets for ordinary goods and services. In such countries there may be no effective approach to controlling pollution, short of government fiat. Evidence from the former socialist economies of Eastern Europe suggests that-government mandates, including the shutting down of certain facilities, provided far more pollution control than effluent taxes. The taxes had virtually no impact on pollution levels because firms did not face competitive market pressures (in other words, their budget constraints were soft) and the governments, which ultimately owned all significant economic enterprises, regularly compensated polluting firms for pollution tax payments with increased allocations in subsequent budgets (see Cole 1998: ch. 5). In the absence of well-functioning market institutions, market-mechanisms of pollution control could not succeed.

Design Issues for an Emissions Trading Program

In theory, a system of transferable pollution rights is simple to establish. The government sets a pollution-control goal, and determines how much emissions must be reduced to attain it.[7] Those necessary reductions are then

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[7] This first step is in reality quite complex, requiring the government to determine a relation between emissions rates and ambient concentration levels

subtracted from current emissions levels to derive total allowable emissions. Next, the government unitizes and allocates those allowable emissions in the form of transferable allowances or credits among regulated firms (the difference between allowances and credits and other types of emissions trading systems is explained in Chapter 3 by Nentjes). Allocation may be by sale, as Dales presumed, auction, as most economists today would prefer, or gift, which is the most commonly used method simply because it is the most politically expedient – free allowance allocation facilitates 'buy in' on the part of regulated industries. However they are allocated, the total number of rights in circulation should match the emissions level the government deems appropriate to achieve its pollution-control goal. Achievement of that goal depends entirely on the government's ability to enforce the cap, regardless of transferability. As noted earlier, the primary purpose of allowing trading is not to reduce emissions – though transferability may create incentives to reduce emissions below government-mandated levels, depending on market conditions – but to minimize the costs of reducing emissions.

Dales asserted, inaccurately, that the transferability of pollution rights 'automatically ensures that the required reduction in waste discharge will be achieved at the smallest possible total cost to society' (Dales 1968: 107). This overstatement has been repeated ever since by advocates of cap-and-trade (e.g., Tietenberg 1985; Stewart 1996). In fact, transferability ensures the smallest possible aggregate *compliance/abatement* cost, which is but one element in the 'total cost' of any environmental policy. 'Total cost' is the sum of the costs of compliance/abatement, administration (including monitoring and enforcement), and residual pollution (see Cole and Grossman 2002)). As already noted, circumstances exist – for instance, where emissions measurement is impossible or prohibitively expensive – in which technology-based standards may well be less costly to administer than any system based on emissions quotas, whether tradable or not. It is at least possible that the administrative-cost advantages of technology-based standards in those circumstances might offset the presumed compliance-cost advantages of cap-and-trade, resulting in equal or lower *total* costs. Indeed, Cole and Grossman (1999: 914–35) present a strong historical argument that extremely high costs of point-source emissions monitoring (largely due to technological constraints) during early implementation of

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that may be subject to numerous complicating factors, such as chemical reactions in the atmosphere, wind patterns, pollution dispersion rates and topography. Consequently, a given reduction in emissions does not necessarily result in a proportionate improvement in ambient air or water quality.

the 1970 Clean Air Act made technology-based standards a more effective *and more efficient* instrument than either effluent taxes or a cap-and-trade system.

Although the theory of emissions trading is straightforward, designing and implementing an effective transferable pollution rights regime is not always easy (see e.g. Woerdman 2004; Weishaar (2014)). Technological and institutional constraints may disable the government from accurately calculating existing waste levels and necessary reductions. If those calculations are inaccurate, then the government's environmental goals may be met inefficiently or not at all. This is similar to the problem of getting the prices right in a tax-based pollution-control regime (see Baumol and Oates 1971 and Milne and Skou Andersen (2012) for a review). With a tax system, of course, the government can adjust the price up or down (subject, of course, to political constraints) until it achieves the desired incentive effects on polluters. With a cap-and-trade regime, however, the problem is a bit more complex because pollution rights have a property nature. In the US and many other countries, if pollution rights have all the traditional attributes of property, then the government could not remove rights from the market to ensure attainment of its pollution-reduction goal without paying compensation for a taking (US Constitution Amendment V). Under such a rule, all else being equal, governments might have an incentive to undersupply pollution rights in the first place – something that has never, in fact, occurred – which could result in the inefficient over-reduction of pollution emissions. If that did happen, however, the government could fairly easily introduce additional pollution rights into the market.

In his original theory of emissions trading, Dales avoided the 'takings' issue by recommending that transferable pollution rights be limited in duration to one to five years in order to allow the government to make occasional adjustments in the quantity of rights on the market to ensure the attainment of existing or newly adopted pollution-control goals (Dales, 1968: 95). In effect, Dales' pollution rights were leaseholds rather than freeholds.[8] Leases are, of course, valuable property rights, although they are less valuable and secure than fee simple property rights (amounting to full ownership). In addition to limiting the duration of pollution rights, governments can also limit their scope from the outset. For example, they can specify in the legislation creating the pollution rights that those rights

[8] The term 'freehold' designates an estate in land that, in modern terms, amounts to a form of ownership. 'Fee simple' is the largest of the freehold estates, and is the closest equivalent to popular notions of full ownership. A leasehold estate, by contrast, does not amount to land ownership, though it does invest the holder with valuable property rights (see Dukeminier et al. 2010: 222).

have only limited status as property, that is, that the ownership amounts to less than fee simple absolute. The government might retain an express right to remove credits from the market without paying compensation. In that case, the pollution right would amount to full ownership (fee simple absolute) against the whole world (as it were) except for the issuing government. In countries without a constitutional right to compensation for takings, such as the UK, even this simple expedient is unnecessary.

From an economic perspective, the legal characterization of property rights is less important than their incentive effects for market participants. The less secure and complete sellers' property rights are, the less likely potential buyers will be to purchase them (all else being equal). Leaseholds are less valuable, and therefore less costly to obtain, than freeholds precisely because of their more limited tenure and security. Defeasible or otherwise limited pollution rights would have lower market value than absolute pollution rights. If the rights are *too* limited, their market value would fall towards zero, and the market would not function. There is, however, a wide range of economically valuable property interests between fee simple ownership and virtually worthless entitlements (such as revocable privileges). Pollution rights typically are not owned in fee simple, but they are sufficiently strong property rights to retain significant economic value, as evidenced by the functioning of actual emissions trading markets.

In addition to the status of pollution rights as property, the other major and even more controversial design issue concerns the appropriate mechanism for initially distributing emissions allowances. Should they be sold, as Dales recommended? Should they be auctioned? Or should they be given for free to regulated entities? Nearly all economists prefer auctioning of emissions allowances over free allocation for several reasons. For one, free allocation subsidizes polluting behavior and provides windfall profits to regulated entities that subsequently sell their emissions allowances (see, for example, Goeree et al. 2010 and Chapter 7 by Gullì in this volume). For another, auctioning allowances is tantamount to a tax on *all* units of emissions, which provides incentives for further emissions reductions. Auctioning also provides revenues that the government can use in various ways. For instance, it might offset higher energy costs by reducing other taxes (for instance, on income); or it might invest revenues to provide public goods, such as public environmental protection projects (see, for example, Goulder and Parry 2008: 161) or reduce distortionary taxes (double dividend hypothesis) (see Cramton and Kerr 1999). So, why do governments usually freely allocated emissions allowances at the outset of cap-and-trade programs? Free allocation based on historical emissions (a.k.a., 'grandfathering') can facilitate 'buy in' on the part of regulated entities (see, e.g., Revesz and Kong 2011); it is in the nature of a bribe to secure their acquiescence to the cap. Of course, this has more to do with practical politics, as explained by public choice theory, than the economic theory of cap-and-trade.

3. PUTTING THEORY INTO PRACTICE: PRECURSORS TO THE US ACID RAIN PROGRAM

Offsets, Netting, Bubbles, and Banking

In practice, emissions trading evolved in the US, almost entirely, in the context of air pollution control under the 1970 Clean Air Act and its amendments. That statute did not make any provision for the kind of transferable pollution rights system Dales envisioned. As early as 1974, however, the EPA (Environmental Protection Agency) was experimenting with transferable pollution rights programs (Hahn and Hester 1989a: 109). By 1980 the agency had approved four distinct emissions trading schemes in the US (see generally Liroff 1986).

First, in 1974 EPA adopted 'netting,' a policy that allows firms to avoid the application of expensive standards for new and substantially modified sources by netting increased emissions from modernized or expanded existing sources with emissions decreases from other existing sources at the same facility (Hahn and Hester 1989a: 132–3). So long as the net increase in plant-wide emissions does not equal the minimal requirement for a 'major' source, as defined in the Clean Air Act, the modernization or expansion will not be treated as a 'new' or substantially modified source for purposes of the Clean Air Act. Netting can occur in all areas of the country, whether or not they have attained national air quality standards. But netting applies only to internal trades, that is, to trades between sources located at the same facility. Nevertheless, according to Hahn and Hester (1989a: 133) netting has been 'the most commonly used emissions trading activity by a wide margin.' Between 1974 and 1984, as many as 12,000 sources used netting to avoid more onerous regulatory burdens under the Clean Air Act, resulting in cost savings of between $525 million and $12 billion (Hahn and Hester 1989b: 374).

'Offsets' were the second form of transferable pollution rights created by the EPA. By the mid-1970s, the agency had become concerned that many regions in the country would fail to meet air quality standards by the 1977 statutory deadline. If that happened, the question arose, did the Clean Air Act permit the construction of new air pollution sources in these nonattainment areas? A construction ban would have entailed great economic costs for nonattainment areas – including most of the major metropolitan areas in the US – and, consequently, negative political fall-out for state and federal politicians and regulators. To avoid this prospect, the EPA in late 1976 promulgated 'offset' regulations that permitted the construction of new stationary sources in nonattainment areas. New sources could be constructed provided that their emissions would be offset by reductions at existing sources. Under this offset rule '[e]xisting sources are, in effect, given pollution rights equal to their existing emissions, which can then be sold to new sources or to existing sources that wish to increase their emissions' (Stewart and Krier 1978: 593).

Offsets are different from netting in several respects: they apply only in nonattainment regions (and in certain attainment regions, emissions which contribute to nonattainment elsewhere); they are mandatory; and they cannot result in a net increase in emissions. EPA's original offset rule was codified in §178 of the 1977 Amendments to the Clean Air Act, which additionally required that all new emissions in nonattainment regions be *more than offset* by reductions from existing sources. The purpose of this additional requirement was to ensure that new economic development in nonattainment regions would contribute to the attainment of the National Ambient Air Quality Standards. Subsequently, the 1990 Clean Air Act Amendments established precise offset ratios, ranging from 1.1:1 to 1.5:1, which apply depending on the region's level of nonattainment. For example, in 'extreme' nonattainment areas such as Los Angeles, 1.5 tons of Volatile Organic Compound (VOC) emissions must be retired from existing sources for every ton to be emitted from some new source. As of 1988, approximately 2,000 offset transactions had taken place, though only about 10 percent of these were external, involving more than a single facility (Hahn and Hester 1989b: 373). The economic effects of these transactions are difficult to estimate. Offsets are not designed to yield direct regulatory cost savings. The fact that offset transactions occur at all suggests, however, that they must provide some economic benefits both for firms seeking to locate in nonattainment regions and for the nonattainment regions themselves (Hahn and Hester 1989b: 375).

Next, in 1979 EPA permitted regulated firms to use 'bubbles' to avoid more burdensome regulations. A single plant may contain many individual sources of pollution. The 'bubble' policy allows existing plants (or groups of plants under common management) to place their various smokestacks under a bubble, as it were, with a single opening at the top. By treating the entire plant (or group of plants) as a single source with a single emissions target (for each pollutant), plant managers are free to allocate necessary emissions reductions to those smokestacks with the lowest control costs. Instead of having to reduce emissions by a certain amount at each and every smokestack, the plant can reduce emissions more at some smokestacks and less, or not at all, at others. ‘In effect, emissions credits are created by some sources within the plant and used by others’ (Hahn and Hester 1989b: 372). By the mid-1980s the EPA had approved 42 bubbles for firms and various states with EPA-delegated authority had approved another 89, though only two of these involved external trades (Hahn and Hester 1989b: 373, and 1989a: 123–5). The total cost savings from bubbling have been significant. Federally-approved and state-approved bubbles have saved an estimated $435 million in regulatory costs. Although this total is lower than the total cost savings from netting, it reflects a higher average savings per transaction (Hahn and Hester 1989b: 374).

Also in 1979, EPA began allowing regulated firms to bank emissions credits for future use, sale, or lease. This banking system is not really a transferable pollution rights scheme in its own right; it is, rather, a mechanism to facilitate the use of bubbles and offsets. The EPA delegated authority to the states to administer their own emissions credit banks. According to Hahn and Hester (1989b: 373), however, banking has not been well-received by either state administrators or regulated firms. As of September 1986, firms had withdrawn credits from banks for sale, lease, or use only 100 times. Thus, the cost savings realized through banking were ‘necessarily small’ (Hahn and Hester 1989b: 374). One possible reason for the reluctance of firms to use the banking system for emissions reduction credits is the lack of secure property rights in those credits, which can be confiscated by state or federal regulators at any time in order to further environmental-protection goals (Hahn and Hester 1989a: 130).

Gasoline Lead-Content Trading

In 1973, the EPA established a program to phase-out lead content from gasoline (38 Fed.Reg. 33734).[9] Lead is not a natural constituent of gasoline but an additive – tetraethyl lead – first introduced in the 1920s to improve engine performance by raising the octane level of the fuel. Interestingly, lead was not the only additive available for that purpose. Simple ethyl alcohol, today better known as ethanol, was understood to prevent engine

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[9] The US was not the first country to begin phasing lead out of gasoline. Japan started phasing-out leaded gasoline in 1970; by the early 1980s less than 2% of gasoline produced in Japan contained lead; and by 1986 lead was completely phased out in that country. Lead was also completely phased out of gasoline in Canada by 1990, six years before the US completed its phase-out (see Lovei, 1998: 15–16).

'knock' just as well as lead; and any moonshiner (i.e., anyone who operated an illegal still or distillery) could produce it, at least in small quantities, fairly cheaply. However, because ethyl alcohol was too common to be patented, it did not appeal to General Motors and DuPont, which collaborated to produce tetraethyl lead, an additive they could patent. Not long after lead was added to gasoline, scientists began to understand the substantial workplace and public health risks of lead, including blindness, brain damage, kidney disease, and cancer. But even after those risks became better known, lead manufacturers insisted disingenuously that tetraethyl lead was the only available and cost-effective fuel additive (see Kitman 2000). In the meantime, automobiles became the 'dominant source' of environmental lead exposure (Nussbaum 1991).

Why did it take so long for the government to get the lead out of gasoline? Two developments, one institutional and one technological, explain why it did not happen until after 1970. The institutional development was Congress's enactment in 1970 of the Clean Air Act. That Act called for (among other things) a 90 percent reduction in automobile emissions of conventional (that is, non-toxic) air pollutants, including carbon monoxide, nitrogen oxides, and hydrocarbons (42 U.S.C. 7521(b)). When the statute was enacted, no technology existed to accomplish such a massive reduction in tailpipe emissions; but it was not long before the first catalytic converters appeared that could meet the Act's goal cost-effectively. The only problem was that lead in gasoline contaminated and disabled catalytic converters. The lead had to be removed from fuel if the Clean Air Act's most significant motor vehicle goals were to be accomplished. Lead regulation thus served the twin purposes of reducing public health risks from lead emissions *and* enabling the reduction of other air pollutants from automobiles. Together, these institutional and technological factors motivated the federal government to finally get the lead out.

Before the 1973 lead regulation took effect, the standard amount of lead in a gallon of gasoline was 2 grams (Nussbaum 1991: 118). The lead phase-out which began in 1973 gradually reduced the allowable level of lead in a gallon of gas by 95 percent to 0.10 grams. The initial 1973 regulation was in the nature of a 'bubble' policy, limiting the total amount of lead in each gallon of gasoline, averaged across all the leaded and unleaded gasoline produced at each refinery. By averaging lead content across both leaded and unleaded gasoline, the regulation created incentives for refiners to increase production of unleaded fuel (Newell and Rogers 2004: 177). Less stringent requirements were imposed on smaller refineries, for which the average costs of phasing-out lead were expected to be higher (Newell and Rogers 2004: 177–8). It was not until 1982, however, that the EPA authorized the *trading* of lead-content across refineries (47 Fed.Reg. 49322). By then, the lead phase-out already had accomplished an 80 percent reduction in gasoline lead levels (Newell and Rogers 2004: 178).

In 1982, EPA promulgated a new lead-content standard at 1.10 grams per gallon of leaded gasoline, applicable to all refineries. This regulation removed the 'bubble' that previously averaged lead content in both leaded and unleaded gasoline. By 1982, unleaded gasoline and cars that ran on it were established products, so the government could focus exclusively on reducing the amount of lead in leaded gasoline, demand for which was expected to drop as the auto fleet ultimately became more and more dominated by cars running on unleaded fuel. Meanwhile, the laxer small refinery standards were dropped on the presumption that the new nationwide trading system would minimize their costs of compliance with the new uniform standard. A refinery that could cost-effectively reduce lead content below the 1.10 grams per gallon standard could sell its 'unused' lead content to another refinery that might find it more economical to purchase the right to higher lead content than to reduce lead content to meet the standard. Initially, the 1982 trading regulation did not include a banking provision, which would allow a refinery that reduced lead-content below the regulatory standard to save its 'unused' lead content for sale in a later compliance period. Because compliance periods were set as calendar quarters, the lack of banking meant that the trading system was essentially a spot market. The EPA added a banking provision in 1985, when it revised its regulations to reduce lead content in gasoline by a further 91 percent in two stages: from 1.10 grams per gallon to 0.50 grams per gallon on 1 July 1985; then to 0.10 grams per gallon on 1 January 1986. Reductions in lead content below 0.50 grams per gallon after 1 July 1985 could be 'banked' for future use or sale after the more stringent standard took effect on 1 January 1986.

EPA's regulations created a vibrant market in gasoline lead-content. After the 1982 regulations took effect, more than half of all refineries participated in the market, trading up to 20 percent of all lead rights (Ellerman, Joskow, and Harrison, Jr. 2003: 11). After the more stringent caps went into effect beginning in 1985, up to 50 percent of all existing lead rights were traded. Although the volume of trading was never particularly heavy, transactions regularly involved quantities of lead-content on the order of 25 million or 50 million grams, worth $1 million to $2 million (Ellerman, Joskow, and Harrison, Jr. 2003: 122). Meanwhile, the level of banking was higher than anticipated. Starting in 1985, nearly half of refineries reporting to EPA participated in the banking of unused lead content. In total, more than 10 billion grams of lead rights – about two years' worth – were saved in banks for some length of time (Ellerman, Joskow, and Harrison, Jr. 2003: 123). Of those banked lead rights, only 2.3 percent ultimately went unused.

The cap on lead content in gasoline resulted in a 'sharp and rapid decrease' in lead emissions from automobiles (Ellerman, Joskow, and Harrison, Jr. 2003: 121). By 1990, the amount of lead in gasoline had been reduced by more than 99 percent from 1970 levels. Not coincidentally, between 1978 and 1991 blood lead levels in people aged 1 to 74 declined on average by 78 percent. According to an official press release by the US Department of Health and Human Services, 'the removal of lead from gasoline and other environmental sources is primarily responsible for these declines' (National Center for Health Statistics 1994).

Economically, the trading and banking provisions of the lead-content regulation saved refineries more than $260 million, as against quotas that could not be traded or banked. One *ex post* analysis concluded that the benefits of the lead phase-out outweighed its costs by 10 to 1, 'with lead trading and banking significantly lowering those costs' (Newell and Rogers 2004: 179).[10] Because of the scarcity imposed by increasingly stringent caps, the price of lead content rose from 0.75 cents per gram to just over 4 cents per gram during the lead phase-out. As a result, more refineries started substituting a cheaper and coincidentally safer ethanol blend for lead, saving even more lead content for potential sale (Newell and Rogers 2004: 190). Finally, in 1996 EPA banned completely lead-based fuel additives in gasoline (Newell and Rogers 2004: 190).

The lead-content trading scheme is rightly considered a successful early application of cap-and-trade, but one of its most essential attributes is often overlooked: the cost of monitoring and measuring lead content in gasoline was low. The amount of lead contained in refined gasoline is easily measured and remains constant at all fuel levels; that is, no matter how much gas is in the tank, the percentage of lead in the gas remains unchanged. Certainly in 1980, it was simpler and less costly to measure the amount of lead in a gallon of gas than it was to measure the lead emissions from tailpipes. Keeping track of lead-content *transactions* proved a challenge for the EPA (Newell and Rogers 2004: 179), but the actual measurement and monitoring of lead-content itself was not a problem.

4. CONCLUSION

Today, emissions trading is not just a standard instrument in environmental policy, it seems to be treated by many economists as a universal

[10] For *ex ante* economic analysis of the lead regulations, and the influence those regulations had on the political process, see Nichols (1997).

first-best instrument, regardless of circumstances. Indeed, economists, lawyers and policy analysts might wonder why it took so long for emissions trading to get off the ground after J.H. Dales first elaborated the idea in 1968. Cole and Grossman (1999) present several reasons for the lag between conception and the first full-scale experiment in the Acid Rain Program of the 1990 US Clean Air Act Amendments, mostly relating to the lack of available emissions-monitoring technologies to ensure the integrity of trading markets. In any case, it remains quite remarkable that it took only 30 years from publication of *Pollution, Property, and Prices* to the institutionalization in the Kyoto Protocol of a *global* trading regime for greenhouse gas emissions.

The origins and early history of emissions trading recounted in this chapter suggest three clear lessons: (1) economic theories, such as those related to cap-and-trade, can be more or less successfully implemented to improve regulatory practice, albeit with some time lag, which should give some hope to innovative economists; (2) regulators are, in fact, sensitive to the compliance cost concerns of regulated industries and will experiment with mechanisms that can reduce those compliance costs so long as they do not compromise environmental protection goals; and (3) emissions trading is, like all other tools of environmental policy, one of limited utility –not a panacea solution to all regulatory problems.

That third (and most important) lesson, unfortunately, has not been well learned by some economists and policy analysts. The success of most (but not all) early emissions trading experiments, including the Acid Rain Trading Program, generated such enthusiasm for emission trading that cap-and-trade has become the 'go-to instrument' for all manner of pollution-control and resource conservation problems, regardless of institutional and technological constraints. Scholars recommended applications to conserve ocean resources (Tipton 1995), endangered species habitat (Sohn and Cohen 1996), and wetlands (Sapp 1995), for instance. Although such schemes could all work, technological problems of measuring and monitoring, along with the quality of market institutions, especially in developing countries, always must be borne in mind. With that important caveat, there is no question that emissions trading has become, over the course of the past two decades a very useful tool in the instrument mix for pollution control problems ranging from domestic emissions of conventional pollutions, such as sulfur dioxide and nitrogen oxides, to globally polluting greenhouse gases.

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3. Emission targets and variants of emissions trading

Andries Nentjes

1. INTRODUCTION

The idea to contain the use and waste of environmental resources, not by way of command-and-control but through a market where emission rights are traded freely, is approaching its 50th anniversary. In the environmental and resource economics literature, Crocker (1966) and Dales (1968) are usually mentioned as the founding fathers of the emissions trading concept, with Montgomery (1972) as the first to provide formal proof of its cost efficiency, and Tietenberg (1980, 1985) as the one who firmly advocated and established it on the economic research agenda. By contrast, the law and economics literature prefers to trace the emissions trading concept back to Demsetz (1967), who argues that externalities should be internalized by allocating property rights, and ultimately to the exposition of Coase (1960) that bargaining will lead to a cost efficient outcome regardless of the initial allocation of property rights (assuming transaction costs are negligible). About 20 years after the notion of emissions trading had been espoused for the first time, the United States (US) Congress made it actual policy in 1989 by making a cap-and-trade program for sulphur dioxide (SO_2) emissions the cornerstone of its strategy to control acid rain.

The literature knows several types of emission trading systems. In terms of the applied terminology their differentiation is not always clear cut but in the literature the following three types can be distinguished: Cap-and-trade (CAT), Credit Trading (CT) (also referred to as Performance Standard Rate Trading (PSRT), output-based allocation or tradable reduction) and intensity based trading. The latter is a form of emission trading design championed by several Chinese Emissions Trading pilot schemes and so it seems also the emerging Chinese national ETS and does not fit in either of the categories described before (Weishaar, 2014). This chapter, however, only reviews the first two.

A CAT system is based on an environmental target that constitutes the maximum amount of emissions that can be emitted by covered installations. For each unit of pollution installations need to surrender a corresponding amount of emission allowances. The CT is in origin and design quite different from CAT (for an overview see Nentjes and Woerdman, 2012). CT evolved in the US in the 1970s and 1980s as a method to introduce flexibility into the command-and-control type of environmental regulation that frequently employed emissions standards (see Faure and Weishaar (2012)). In essence, CT operates on the basis of an emissions benchmark per unit of production as can also be employed under direct environmental regulation and supplements it with a trading scheme. The amount of emission rights (called credits) that an emitter can sell is determined by reference to the benchmark and the actual output of the installation. CT has been described as trading in emissions in excess of the emission control required by the benchmark (for a full account of the EPA emissions trading program (see Le 2009).

In practice we find a large variety of ETS designs being applied, that mostly differ from the above generic designs (for an overview of the current ETS systems in operation, see Weishaar 2014). A rich body of literature emerged analyzing how CAT and CT differ among other things with regard to (a) their environmental effectiveness and (b) their impact on firms' production costs. This chapter first presents the differences between these ETS variants (section 2). In section 3, their economic impact is examined. Section 4 investigates which generic variant is best in incentivizing progress in emissions control technology. Section 5 concludes.

2. DIFFERENCES BETWEEN DESIGN VARIANTS

In CAT total emissions of covered installations are capped, whereas in CT the total amount of emissions varies with the volume of output. Consequently the three instruments differ in their effectiveness in realizing the objectives of pollution control policy and in their impact on the firms' cost of output, which will be explained hereafter.

2.1 Differences in Effectiveness

With respect to emissions target setting, usually policy makers aim at a minimum level of environmental quality. To achieve this quality standard a cap is set to restrict the total amount of emissions released per period. In a cap-and-trade programme the cap is implemented by issuing a limited number of emission allowances which constitutes the total market supply. Market demand for allowances arises from the obligation of covered firms to surrender allowances equivalent to their unabated emissions. A firm's level of abatement is where marginal abatement cost is equal to the allowance price.

The policymaker's only concern in implementing CAT is to set the emission cap correctly. The allowance market, a monitoring, reporting and verification system and an effective sanctioning mechanism help to ensure the attainment of the cap. In a growing economy, emissions increase together with industrial output. Since the total supply of allowances is 'capped', the rise in demand for allowances drives up their market price and hence incentivises more abatement, ensuring that demand equals supply while the cap is being maintained. In case of an economic downturn CAT works in a reverse way. Emissions decline as industrial output contracts, giving rise to a decline in allowance demand and prices. CAT therefore functions as an economic stabilizer (Koutstaal 1997). Since lower allowance prices also offer less incentives in abatement the EU ETS has been criticized for not setting enough abatement incentives in the aftermath of the 2008 economic downturn. The essential element from an environmental perspective (at least in a static environment) is that the emissions cap is attained.

In the case of CT the policy maker has a more complicated task in keeping total emissions on the envisaged environmental target because such a target must first be translated into an emissions standard per unit of output. Polluters emitting less than the mandated level receive a type of allowance called 'reduction certificate' or 'credit'. A firm exceeding its emissions per unit of output must surrender a corresponding amount of emission reduction certificates. The market price leads to an equalization of demand and supply. If the economy grows the industrial output of covered entities increases and the emission standard may need to be sharpened to protect the environment. Unlike in a CAT the attainment of an environmental target under CT is not automatic.

In case of a contracting economy industrial output falls, emissions per unit of output remain unchanged but total emissions decrease beyond the target. Because emissions per unit of output remain unchanged, also the marginal abatement costs remain unchanged. Different from CAT there is in CT no automatic stabilizer at work to alleviate the cost for firms during the economic bust. It can explain why Fischer and Springborn (2011) find that in a real business cycle model cap-and-trade damps the volatility of output while credit trading does not.

2.2 Differences in the Costs of Residual Emissions

In CAT the authority can allocate allowances for free or auction them. In the presence of perfect competition the two types of initial distribution under CAT are equivalent in their costs to the firm; they do differ however from the cost to the firm of CT.

Free versus auctioned allowances under CAT

In a cap-and-trade scheme a polluter surrenders allowances for each ton of CO_2 emitted. Under auctioning the polluter buys the corresponding number of emission allowances. The allowances are an input into the production process and similar to other inputs constitute a cost that has to be included product. On the surface it looks different when under CAT allowances are allocated to polluters free of charge on a lump-sum basis, for instance by means of grandfathering (free allocation based on historical emissions), but that is deceptive. A covered entity that received allowances for free has the choice between using the allowances to offset emissions or selling them. Going for the first option implies giving up the opportunity to sell the allowances. Apparently using free allowances to offset emissions has an 'opportunity cost'. Not receiving revenue from allowance sales is as much a cost of production as buying the allowances in an auction (e.g. Grafton and Devlin, 1996; Koutstaal, 1997; Dijkstra, 1998; Hargrave, 2000; De Vries, 2003; Woerdman et al., 2009). The use of allowances for offsetting emissions comes at an opportunity cost under both forms of allocation. In a perfectly competitive market for allowances there is no difference in equilibrium market price for allowances that are auctioned and allowances handed out for free on a lump-sum basis, nor is there in market equilibrium a difference in the allocation of allowances across sources to offset their emissions. Under both allocation methods the cost of allowances raises the market price of output in an identical manner, resulting in equivalent market equilibrium.

It bears mentioning that auctioning allowances generates public revenues. Auction proceeds can be used to reduce distortionary taxes (e.g. Goulder, 1995; Goulder et al., 1999). Auctioning allowances could thereby yield a 'double dividend'. With free allowances that possibility is foregone.

Allowances under CAT versus certificates under CT

This section compares the effects of allocation (auctioning and grandfathering) under CAT to CT. In CT polluters surrender credits corresponding to their actual output and how their technology compares to the performance standard rate (PSR). Pollution within the limits of the PSR is free of charge.

Under CT expenditures of credit buyers equal the revenue of sellers. Emissions of complying firms are free of charge and do not add to the cost of output.

To conclude: both CAT and CT create flexibility by allowing transfers between emitters. Yet they are distinct instruments of environmental policy. CT is a complementary instrument to make direct regulation through emission standards more flexible by allowing excess emissions of one emitter to be compensated by excess emissions control at another emitter. CAT sets a cap on total emissions and distributes the total of mandated emissions among firms. CAT is not an addition to direct regulation, but a substitute for command-and-control. Section 3 explains how the two variants of emissions trading differ in their impact on economic performance and in their effect on society's welfare.

3. ECONOMIC IMPACTS OF CAT AND CT

Fifteen years ago Dewees (2001) observed that while there was extensive literature on emissions trading generally, there had been little economic analysis comparing performance standard rate trading with cap-and-trade. Although progress has been made in the past 15 years, it still is very much a domain for specialists; but now that the two designs are actually applied next to each other and even are mixed up. The next two sub-sections review the cost efficiency and economic efficiency of CAT and CT. Sub-section 3.3 investigates what happens if the two designs are combined, and 3.4 how CAT or CT fare in terms of international competition.

3.1 Cost Efficiency

Both CT and CAT result in a cost efficient allocation of control of emissions across emitters. Under CT the emitter calculates the emissions by multiplying the difference between his actual emissions and the PSR with his level of output or fuel input and then examines additional abatement opportunities. When the PSR are uniform while firms and their emissions sources are heterogeneous, the costs of emissions control will differ between firms. Empirical studies show that the difference between low cost and high cost sources can run up to 60 percent (e.g. Klaassen, 1996; Pizer et al., 2006). CT introduces flexibility in the command-and-control scheme by offering the firms the possibility to purchase emission credits. Firms with low abatement cost functions have an incentive to obtain credits by abating more than the PSR requires. Trade in certificates thus reallocates emission abatement from high cost sources to low cost sources. For the individual firm, total cost of compliance is minimized by abating up to the level where marginal abatement cost equals the market price of credits. Since this holds for all firms the marginal abatement costs of all sources are equalized and consequently total abatement costs are minimal under CT.

In a CAT scheme a firm can abate emissions or surrender allowances. Similar to CT the individual firm minimizes its total compliance cost by controlling emissions up to the level where marginal abatement cost equals the market allowance price. Because all firms do so the marginal costs of all sources are equalized and consequently total abatement costs of all sources are at the lowest possible level when the allowance market is in equilibrium.

Both CT and CAT are cost-efficient and the total costs of abatement are lower than the cost under command-and-control regulation through emission standards. In a survey of simulation studies for the air pollutants SO_2, NO_x and hydrocarbons (HC) in the US, Klaassen (1996) mentions cost savings varying from 4 to 85 percent. The US sulfur allowance trading program predicted cost savings of 30 to 40 percent compared to the costs of emission standards. In a midway estimate, Carlson et al. (1998) already calculated that actual cost savings might even run up to 60 percent or more.

Providing the proof that CT as well as CAT on a perfectly competitive market brings about a cost efficient allocation of emissions across emitters and firms is one thing; to demonstrate that such a market equilibrium can come about is another. The proof of the dynamic stability of the market equilibrium for allowances and reduction certificates was given by Ermoliev et al. (2000). The authors assume that the agents on the market for emissions are cost minimizers. In bilateral transactions a buyer and a seller agree – for the emissions they trade between them – on a price that lies between their respective marginal costs. Each participant is willing to re-contract in a next transaction when he sees the possibility to lower his costs further. Transactions costs are zero. The trades start from an initial inequality of marginal abatement costs. The transactions between buyer and seller occur in a random-ordered sequence. It is demonstrated the sequence converges towards a Nash equilibrium with a uniform price of traded emissions, equality of all marginal costs and minimum total abatement costs.

3.2 Economic Efficiency in Emissions Trading Design

In this chapter economic efficiency is defined as maximization of welfare, that is, the sum of consumer and producer surplus created by the production of an industry and consumption of the product. In the sub-sections that follow the concept is applied to an industry for which the regulator has set a maximum number of emissions. Comparing CAT or PSRT the economic efficient design realizes the emission target by cutting back the emissions of the industry with minimal loss of the surplus created by the industry's production.

Economic efficiency does not only encompass allocative efficiency (sub-section 3.1) but considers a wider range of options; in particular the possibility to decrease emissions by lowering production instead of abating emissions for a given output. As presented below, the decision of the firm on this issue is in CAT different from the decision in CT. This leads to differences in industry output and abatement.

Decisions at firm level

A firm will restrict output to reduce emissions when this is less costly than other abatement options. Under CT saving on abatement cost by way of reducing output is not a feasible option since decreasing production will not change the average emission per unit of output. CT only rewards the individual polluter for cutting back emissions below the PSR per unit of output.

CAT works differently. Crucial for the scheme with allowances allocated for free is not the trade in allowances but the cap on total emissions of a firm (Dijkstra, 1999). A firm that reduces emissions through cutting back production remains entitled to emissions equal to the cap and receives an equal number of allowances, enabling it to reduce emissions by reducing output. Choosing for the combination of a relatively low level of output and high emissions per unit of output can be profitable, particularly for firms with high marginal abatement costs.

Cap-and-trade is a scheme with capped emissions and the possibility to trade allowances. A firm will abate emissions to the level where its marginal cost of abatement is equal to the market price of allowances. In a cap-and-trade system where the firm has received for free allowances, equal to the cap, the opportunity cost of the allowances used to offset the emissions of the marginal unit of output is equal to the cost of abating the emissions of the marginal unit of output, and simultaneously equal to the market price per allowance multiplied with the number of allowances needed per unit of output (e.g. Koutstaal, 1997; Woerdman et al., 2008). The opportunity cost of free allowances is a component of the marginal cost of the product, as much as expenditure for purchasing in an auction allowances to offset the emissions of output is part of the marginal cost of output. In this respect the free allowances in CAT differ fundamentally from the free mandated emissions in CT. In CT an increase of output entitles the firm to emit as long as it complies with the standard. So they cannot possibly be a part and parcel of the marginal cost of that output.

In a perfectly competitive market equilibrium (assuming away transaction costs) the market price for allowances under full auctioning is identical to the market price under free allocation. In both versions of CAT the cost of allowances to offset the residual emissions of the product raises the market price of output in an identical manner, resulting in equivalent market equilibrium for the level of output.

In CT the firm only pays for emissions above the PSR and not for its actual emissions. The component not reflecting the marginal cost of output is calculated by multiplying the emissions under the PSR of the additional unit of output with either the marginal cost of abatement or with the price of an emission reduction certificate (Boom, 2006: Boom and Dijkstra, 2009). In the literature the non-included abatement cost of marginal output in PSR has been interpreted as an implicit output subsidy (Helfand, 1991). That it also holds for CT has been demonstrated by various authors (e.g. Fischer, 2001; Dewees, 2001; Boom, 2006; Boom and Dijkstra, 2009; Holland, 2012). Bernard et al. (2007) classify the implicit subsidy as a specimen of a class named output-based rebating: 'a popular mechanism for integrating an offsetting subsidy into an environmental policy that raises production costs'.

Market equilibrium and industry output

The difference in how CT and CAT work out on abatement and output of the firm arises from the fact that in CT emissions produced in compliance with the PSR, are not a cost for the firm and do not turn up in its calculation of marginal cost of output. In other words, in CT the marginal cost of output function of the firm and industry lie below the same function in CAT. Similarly the supply curve in CT lies below the supply curve in CAT. In long run equilibrium of a product market with perfect competition the price of output is therefore lower in CT than in CAT and the output of the industry is larger in CT than in CAT.

This has consequences for abatement. If total emissions per unit of industrial output would be equal under both schemes, an expected higher level of output in CT results in higher total emissions in comparison to CAT. In lab experiments by Buckley et al. (2007) this is what actually occurred. However, when the public authority sets one and the same target level for the industry's total emissions, independent of the type of policy instrument, then it has to set a PSR that is more stringent than on average the emissions per unit of output are in CAT. Therefore, the level of abatement of the industry is higher under CT than it is in CAT.

To give the full picture we recall that in CT the output supply curve is lower than in CAT, due to the possibility to emit free of charge below the PSR; but more intensive abatement pushes up the marginal cost of output and by that the supply curve in CT compared to CAT. Boom (2006) and Boom and Dijkstra (2009) have demonstrated that the combined effect of the two opposite forces is to make the marginal cost of output and hence the price under CT lower than it is under CAT. Consequently to attain the same target level of total emissions, output in long run market equilibrium is higher in CT than in CAT, while the emission standard is more stringent than the emission per unit of output under CAT.

As a last result one can derive that due to the higher abatement per unit of output in CT the marginal cost of abating emissions is higher in CT than in

CAT. This implies that the price of one credit in CT is higher than the price of an emissions allowance in CAT (Boom, 2006; Boom and Dijkstra, 2009).

Industry output, market structure and economic efficiency

The difference between the two emission trading designs in terms of levels of output and abatement has consequences for their impact on welfare because of their differences in terms of economic efficiency. First the outcome under perfect competition in the output market is discussed before examining the situation under imperfect competition.

Economic efficiency when competition is perfect A first condition for maximizing social welfare is that the marginal cost of production equals the marginal benefit of the product for each consumer. Second, the marginal cost of production should contain all costs of the product. When the market for output has the structure of perfect competition the first condition is met, both under CAT and CT; but the second condition is only met under CAT. In CAT the supply price of the product (defined by the firms' calculation of marginal cost) signals that abating the residual emissions of the marginal product have a cost. Output is at the level where the marginal benefit consumers have from the product equals the market price. In that market equilibrium the welfare gains from the last unit of output consumed are just equal to the full marginal cost of production. Should a consumer buy an additional unit, then the marginal cost of the product exceeds the marginal benefit and the extra unit decreases welfare. Output and abatement are in CAT at the levels where social welfare is maximized. Consequently CAT is not only cost efficient in allocating abatement among polluters by setting marginal abatement costs equal to the allowance price, but it is also economically efficient by transmitting the shadow price of abatement into the price of output.

In CT the supply price of the product does not signal that abating the residual emissions of the marginal product have a cost. Consumers can get the product at a price that is too low and buy a quantity that is too high. In market equilibrium the welfare gains from the last unit of output consumed is lower than the full marginal cost of producing the product. The total surplus would be higher if output were lower and abatement as well (Boom, 2006). Models assuming uniform firms without emissions trading come to a similar conclusion (e.g. Helfand, 1991; Ebert 1998; Dijkstra, 1999; Fischer, 2001; Boom and Dijkstra, 2009; Holland et al., 2009). In simulations of CT, Boom (2006) finds that the loss of surplus, because of high abatement necessary to offset the emissions of too high output, can be up to 20 percent higher than the loss of surplus in CAT. Considerably more dramatic is the outcome of a simulation of adoption of a low carbon fuel standard to attain a given level of CO_2 emissions in the United States done by Holland et al. (2009). They compare a standard mandating carbon emissions per unit of energy and credit trading added with a cap-and-trade program that sets a ceiling for carbon emissions from energy production. They show that aiming at an emission rate 10 percent more stringent than without regulation, the loss in total social welfare due to abatement cost is about two and a half to five times higher in the CT scenario compared to CAT, depending on the elasticity of fuel supply and demand.

Concluding, when competition on the allowance market and on the product market is perfect, CT is cost efficient, but not economically efficient since the benefits of a part of output are lower than the costs, due to the abatement of the extra mandated emissions of that output. Cap-and-trade is economically more efficient, because it reduces emissions through lower production where that is less costly in terms of loss of social welfare than extra spending on emissions control.

Economic efficiency when competition is imperfect

Industry output under cap-and-trade is lower than under CT, and the emission-to-output ratio is higher, given the target level for total emissions. This is also the conclusion when the allowance market is perfectly competitive while the market for output is characterized by imperfect competition (Boom, 2006; Boom and Dijkstra, 2009). In an unregulated monopolistic or oligopolistic product market output is below the economically efficient level. When the authority then introduces cap-and-trade, the output contracts more than under CT. In a Cournot oligopoly the number of firms may be higher in CT than under cap-and-trade (Dijkstra, 1999; Boom, 2006). Firms will then have less market power, which also leads to higher output under CT. The two impacts would reverse the ranking of the two instruments: here CT brings higher welfare than cap-and-trade. However, there is a third factor working in the opposite direction: CT leads to higher abatement costs than cap-and-trade. The overall effect and by that the ultimate ranking in terms of impact on social welfare depends on the size of the three effects (Boom, 2006; Boom and Dijkstra, 2009).

Boom (2006) has been the first to give a full analysis of the welfare impact of CT when competition on the market for output is imperfect. Earlier publications had given some insights; e.g. Malueg (1990) and Sartzetakis (2004) who argued that an absolute ceiling to emissions is not necessarily a welfare maximizing instrument when the market for output is imperfect. Ebert (1998) had given a short analysis of effects of PSR under imperfect competition. Assuming a target level for total emissions, a perfect allowance market and a Cournot oligopolistic market for output, in a model where a clean product (produced with low emissions) competes with a dirty product (with high emissions in production) de Vries (2003) reaches similar conclusions as Boom (2006). When the difference in emission per product is small, cap-and-trade performs better on social welfare than CT. When the difference is high the ranking is reversed.

Where de Vries (2003) and Boom (2006) in their models maximize the consumer plus producer surplus under the constraint of a total emissions ceiling, Holland (2012) presents a model in which the consumer plus producer surplus minus the environmental damage from emissions is maximized. He interprets the missing emission cost component under a mandated emissions per unit of output standard complemented with CT as a (hidden) consumption subsidy. When there is perfect competition on the market for output the distorting implicit subsidy has to be neutralized by an equal tax on output in order to achieve maximum welfare. In a cap-and-trade scheme the implicit subsidy on output is lacking. Therefore CAT has a welfare maximizing impact on output equivalent to CT with a neutralizing output tax (Holland, 2012). When a CT scheme is implemented while competition on the market for output is imperfect welfare is maximized by setting the 'corrective' tax at a lower level than under perfect competition; even a negative consumption tax might be optimal, if imperfections on the output market are large. Despite the differences in modelling and terminology Holland's conclusions are on a par with those of de Vries (2003) and Boom (2006).

3.3 Combining CAT and CT

As explained above, an industry's total output is higher and emissions per unit of output are lower in CT than *ceteris paribus* in CAT. The abatement effort therefore also have to be higher and consequently marginal cost of abatement is higher in CT than in CAT. In equilibrium, marginal abatement cost is equal to the price of the certificate or allowance, respectively. It follows that under CT the price of an emission reduction certificate is higher than the price of an emission allowance had the industry operated under CAT (Boom 2006; Boom and Dijkstra, 2009). This section examines how the two designs work in combination.

Let us assume an economy with two sectors, sector A subject to CAT and sector B subject to CT. Both can freely purchase and use emission allowances and credits. If necessary the emission target under CT will be adjusted to limit total emissions. Initially the credit price under CT is higher than the allowance price, but trading leads to price equalization. Assuming that the output of both sectors is sold in different markets, sector B will see its average cost of output go down, leading to lower product prices and higher output. By contrast sector A will see its average costs of output go up, leading to higher product prices and lower output. Allowing emissions trading between the two sectors increases the discrepancy in output that existed when the sectors were separated (Boom and Dijkstra, 2009).

Fischer (2003) has also studied the effect of combining CAT and CT. She concludes the trade between sectors will always lead to higher total emissions. It is the consequence of her assumption that the regulator will not set a more stringent emission standard for the expanding CT sector.

3.4 CAT and CT as Instruments of Strategic International Competition

Governments could design emissions trading systems to reap benefits at the expense of other states. The economic literature on the issue conventionally rests upon the assumption that governments aim at maximizing national social welfare, that is, the sum of consumer and producer surplus. Assume the international product market where firms compete in perfect competition. Firms in a large country can affect the terms of trade by reducing output and thus improve its position on the international market. Their joint domestic producer surplus increases more than the domestic consumer surplus diminishes.

Building on Markusen (1975), Krutilla (1991) and Dijkstra (1998), Boom (2006) develops a model in which a national government uses the emission trading design as an instrument of strategic international competition. To achieve the aim of maximizing social welfare under the constraint of not exceeding the national emission target policy makers have the choice between CT and CAT. For the exporting country CAT is the best choice because it has a lower output than CT because it has higher production than CAT. By contrast when the large country imports the good, however, welfare is maximized by increasing the consumer surplus even though some producer surplus has to be sacrificed. This is done by increasing domestic production to reduce imports and to lower the world price of the product. The strategic choice of CT by the importing country decreases world prices and export countries that apply CAT see their producer surplus and welfare shrink. When international competition is perfect, a strategic choice for CT by an importing country is apparently a 'beggar my neighbor' policy (Boom, 2006).

The studies on instrument choice when international competition on the output market is imperfect draw heavily on Brander and Spencer (1983, 1985). In a model of an international duopoly they show that the governments of the two countries, each aiming at maximum national welfare, have an incentive to subsidize R&D of the duopolistic firm established in their country as well as its export. The subsidies give the national firm a price advantage on its foreign competitor. Output and profits are higher and by maximizing the profits of the national industry welfare is maximized. However, since both governments behave in the same way they are caught in a Prisoners' Dilemma. National welfare would be higher in both countries if their governments put a stop to the subsidies.

The model has inspired Ulph (1992) and Barrett (1994) to bring in environmental policy as an instrument of strategic international competition. Boom (2006) made the step to analyze two countries engaged in duopolistic Cournot competition on the international product market, where each government makes the strategic choice between a cap to emissions per firm and an emission per unit of output standard, under the constraint that national emissions do not exceed the target level. As before, national welfare consists of producer surplus plus consumer surplus. With imperfect competition on the international market, a national policy that lowers the marginal cost of output for the national firm leads to a lower market price, resulting in a larger market share and higher output and higher profits for the firm. When there is also domestic consumption of the product the lower price goes hand in hand with higher consumption, thus increasing the consumer surplus. A national government will then strategically choose the emission standard because the instrument generates higher output and by that a higher total surplus than the emission ceiling.

A caveat is in order: the increase in production should not be so large that the increase in the abatement cost for an additional unit of output exceeds the increase in surplus arising from the additional unit of output. Such a situation is not likely when environmental policy is either lax and consequently marginal abatement cost low, or so stringent that mandated emissions per unit of output approach zero. When the two countries are identical, in terms of demand, cost and reaction function, in emission target and in strategy, they end up in a non-cooperative Nash equilibrium: each government chooses the emission standard, whereas the Pareto-optimum would have been to agree on using both the emission ceiling as instrument (Boom, 2006). In the range between lax and very strict environmental policy the optimal national strategy is to go for the lowest output and therefore choose the emission cap per firm. For two identical countries the non-cooperative Nash equilibrium now is a Pareto-optimum (Boom, 2006).

4. EMISSIONS TRADING DESIGN AND INNOVATON IN ABATEMENT

Over the past four to five decades of national environmental policies, pollution control technologies have steadily improved: average and marginal abatement cost have come down and the percentage of technically feasible pollution reduction has gone up. Such technical progress is the spin-off of invention and innovation in pollution control as well as of the adoption (diffusion, penetration) of such innovations. In economic theory, innovation in pollution control is defined as developing and bringing on the market a new technology that shifts the pollution abatement cost function downwards (e.g. Downing and White, 1986). The fall in abatement costs is the major component of the innovation and adoption rent. In innovation the crucial decision is how much to invest in R&D. Adoption is the process of installing and operating a new technology once it has become available. Economic models of innovation and adoption (diffusion) investigate the strength of the incentives to research, develop and adopt new technology, which depend on the rents captured by innovators and adopters.

The survey in this section focuses on models that provide building blocks for answering the question whether and in what respect CAT and CT differ in their incentives to develop and adopt improved control technology.

4.1 Ranking the Instruments

Zerbe (1970) can be taken as a starting point. He assumes the output level is given and does not differ between instruments. Comparing direct regulation, pollution taxes and subsidies, he concludes that market-based instruments provide stronger incentives for innovation than direct regulation. The innovator expects that his new technology will shift the marginal abatement cost curve of the adopter downward. Since output is given and constant the emissions that are mandated under direct regulation are given and constant. So the difference between regulation by way of PSR and regulation by way of cap on a firm's emissions cannot and is not made in the analysis of Zerbe (1970). In a figure showing level of abatement of the firm on the horizontal axis and marginal abatement cost on the vertical axis the required level of abatement under direct regulation is presented by a vertical curve. The expected innovation and adoption rent is the surface between the marginal cost curves before and after innovation, given the required level of abatement. When an emission tax is levied by a regulator who has full information on abatement cost, but knows nothing of potential innovation, he will set the tax rate such that it induces emission control equal to abatement under direct regulation. (Imagine the tax as a horizontal curve.) After innovation and adoption, the marginal cost curve has shifted downwards. Given the unchanged tax level the firm increases abatement to the level where tax and marginal abatement cost are equal again. Since his unabated emissions are lower now he saves on the amount of emission tax to be paid. The total rent of innovation and adoption is now the fall in abatement costs, similar to direct regulation, plus the savings on emission tax. With an emission tax the innovation rent is evidently higher than it is with direct regulation. For the potential innovator, who makes a prognosis based on his expectations of the situation after innovation, the rent is an indicator of the net revenue he can maximally expect to receive from a representative adopter. Also relevant is the number of firms that will adopt the new technology. If firms are similar their adoption decision depends on the premium the innovator will charge to cover his R&D cost and risk taking. Let the premium be a fixed sum F. If the innovator sets F lower than the rent, 100 percent of firms adopt the new technology in the emissions tax scenario, whilst no firm adopts in case of direct regulation. Zerbe's conclusion depends on his implicit assumption of a 'myopic' regulator (Requate, 2005): after innovation and adoption (ex post), the emission standard and emission tax are not adjusted. It makes that ex post total emission reduction of the industry under pollution taxes is higher than it is ex post under regulation.

Downing and White (1986) were the first to include tradable permits in the analysis. Similar to Zerbe they assume a given level of output and therefore do not see the difference between CAT and CT. Probably the authors had a scheme with auction of allowances in mind. Assuming a permit price equal to the tax rate they argued in a way similar to Zerbe (1970) that tradable permits provide innovation and adoption incentives equal to a pollution tax and stronger than direct regulation. The weak spot here is that the permit price is not a policy instrument like a tax but a market outcome. Given the cap on total emissions an innovation that lowers the marginal abatement cost curve will be reflected in a downward shift of the allowance demand curve, which will bring down the allowance price (e.g. Milliman and Prince, 1989; Jung et al.,1996). When all firms have installed the new technology, the same number of allowances will be bought at a lower price. Jung et al. (1996) interpreted the fall in expenditures on allowances due to a lower price as a component of the adoption rent, but that is wrong. The fall in allowance price cannot be counted as an adoption rent because a non-adopter also benefits from the lower price (Keohane, 1999; Requate and Unold, 2003). He does so by purchasing more allowances now that they are cheap, which enables him to reduce expensive abatement. In the extreme case, he would even not abate at all and buy permits to offset emissions, taking a free ride on the low allowances price. The option for non-adopters to purchase cheap permits diminishes the comparative advantage of the adopter and by that the incentive to adopt and to innovate (Malueg, 1990). Comparing the sum of innovation and adoption rents under the two instruments, at an industry level, assuming an unchanged level of output and total emissions, the conclusion then seems to be that under direct regulation the innovation and adoption incentive is stronger than it is under cap-and-trade. So the ranking order would be the reverse of what for quite a time had been common opinion (e.g. Jaffe et al., 2003).

A further point of debate for more than a decade was the innovation and adoption incentive of auctioned allowances versus free allowances. It started with the proposition (Milliman and Prince, 1989; Jung et al., 1996) that auctioned allowances provide greater incentives than allowances distributed free of charge. It was built on the argument that in such an auction all sources are buyers and not so when allowances are distributed for free. The belief was that only purchasers of allowances benefit from lower permit expenditure. The argument has been refuted: there is no difference (Requate and Unold, 2003). The crucial error from Milliman and Prince (1989) to Montero (2002) is not to see that in a scheme of grandfathering a firm that uses free allowances to offset emissions from output has an opportunity cost. It makes the incentive to cut back on the use of free allowances as strong as it is when allowances are auctioned. The innovation and adoption incentive of auctioned and free permits is equally strong or weak.

In a survey article, Requate (2005) has criticized the old view on the rank of incentives. He is in particular critical of the partial equilibrium approach and recommends an analysis in which the output market is included and in which the number of firms that adopt the technology is determined endogenously. He also observes the lacunae that the performance standard, 'one of the most commonly used instruments', is usually not studied. To his criticisms it can be added that the distinction between cap-and-trade and CT is not made in the literature on technical change (e.g. Söderholm, 2010). A discussion of how to rank the innovation and adoption incentives of a uniform emission standard, credit trading, emission cap per firm and cap-and trade is missing. As a contribution to an ongoing debate I present my own view on this issue by comparing the innovation and adoption incentives of credit trading with cap-and trade for the simple case where the firms in the industry have uniform cost functions and competition on the market for output is perfect.

The strength of the incentive depends on the size of the innovation and adoption rent, which we define as the difference in total cost before and after innovation at equivalent level of emission reduction. In CT the emissions below the PSR are free of charge. The rent can only consist of lower cost per unit of emission reduction. In CAT the rent is made up by lower cost per unit of emission reduction and possibly also by lower cost of residual emissions, which is either the expenditure in an allowance auction or the opportunity cost of allowances that have been granted for free.

In both CT and CAT the rent includes the decrease in abatement cost. If output and therefore also the potential (unabated) emissions are equal, the decrease in abatement cost in CAT and CT are equal. However, in CT output is higher than in CAT; and so are potential emissions and emissions abatement. For that reason the savings on abatement costs, thanks to innovation and adoption, are larger in CT than in CAT. But only in CAT there are savings on the cost of residual emissions; they do not appear in CT. So the question is whether the 'surplus' savings on abatement cost in CT compared to CAT are larger or smaller than the savings on cost of residual emissions in CAT. The answer decides which of the two designs has the largest adoption and innovation rent, and that answer depends on circumstances. When output in CT is much higher than in CAT and the limit on total emissions of the industry, set by the authority, is very strict (and equal for both designs) then the surplus savings on abatement cost in CT are high compared to the savings on the cost of residual emissions in CAT. CT has higher innovation and adoption rent than CAT. It therefore provides the strongest incentive for innovation and adoption in emissions control technology. When the difference in output level is small and the limit on total emissions of industry is lax, CAT will better perform than CT in terms of innovation incentives.

Although many empirical papers establish links between environmental policy and innovation, few of them offer a direct comparison of various policy instruments, possibly due to lack of sustained experience with other instruments than direct regulation. One of the few opportunities for comparison is the permit trading scheme for sulfur dioxide emissions in the US, discussed in section 2, that has succeeded direct regulation based on sulfur emission standards in 1995. On the basis of patent data, Popp (2003) finds that there was more patenting of new environmental technology prior to the introduction of the permit scheme and that the programs created different types of technological incentives. Under direct regulation, most new coal-fired electrical utilities could meet the standard by installing flue gas desulfurization units, called 'scrubbers', with a removal efficiency of 90 percent. Popp (2003) finds that under the old command-and-control regime innovations were geared to lowering the costs of operating those scrubbers, and did little to improve removal efficiency. The permit trading regime brought change: innovations did serve to remove a higher percentage of sulfur. However, Popp's finding is at odds with the results of Taylor et al. (2005). They maintain that by 1990 flue gas desulfurization units had reached a removal efficiency of 95 percent. The sulfur allowance trading scheme could meet the cap on total sulfur emissions with that technology. There was no need for higher removal percentages and major research programs were terminated. The authors see this as evidence that emissions trading is not superior to traditional regulation as in inducement for environmental technological innovation. The discussion has brought to light that the theoretical papers missed an opportunity by not making a distinction between cost saving and removal efficiency raising innovation; thus failing to derive propositions that could be tested empirically.

4.2 Which Instrument Maximizes Social Welfare?

The usual economic criterion of ranking of instruments on innovation and adoption incentives is their score on maximizing social welfare. Recent literature is more focused on analyzing the performance of instruments from that perspective. Output is usually made endogenous in the model and a damage function is added. Next to output markets with perfect competition, oligopolistic markets are taken into consideration. Requate and Unold (2001) find that in a model of optimal regulation with perfect competition in both permit and output market, a tradable permit scheme leads to an adoption rate that maximizes welfare. Uniform standards cannot induce the optimal rate of adoption due to their economic inefficiency.

De Vries et al. (2014) is to my knowledge the only publication analyzing the technological impact of CT alongside CAT. In their model the regulator aims at an exogenously given emission level and the interaction between permit market and output market is included in the model. Firms act strategically on the output market, where output produced by means of a conventional dirty technology is in Cournot competition with output produced with a clean technology. Before firms start production, they make their technology choice. Over successive short-run periods firms switch technology, until in the long-run equilibrium profits of clean and dirty firms are equalized. Diffusion of clean technology therefore takes the form of penetration of the clean product as a substitute for the dirty product. For allowances the market structure is assumed to be perfectly competitive. De Vries et al. (2014) analyze the welfare implications of CAT versus CT. Instead of a convex damage function they include the regulator's constraint on total emissions in the model. Welfare consists of three components: consumer surplus, producer surplus and allowance revenue. The allowance price reflects the regulator's marginal valuation of a clean environment. In the long-run equilibrium with free entry and exit CAT outperforms CT in terms of welfare. With CT the size of the clean sector is too large.

4.3 Summary

The current scholarly common opinion, assuming output to be exogenous, still seems to be that direct regulation provides a stronger incentive to innovation and diffusion of control technology than emissions trading. The major argument here is that technical progress lowers the market price of tradable allowances, thus providing firms that stick to the old technology a free ride that shrinks the innovation and adoption rent compared to direct regulation where non-adopters do not enjoy such a benefit. I have argued that in the debate it has been overlooked that in a scheme with a cap on emissions per firm free allowances have an opportunity cost. Innovation in pollution control brings the opportunity cost down which creates an extra innovation rent, next to the rent stemming from the decrease in abatement cost. When output in CT is much higher than in CAT and the limit on total emissions of the industry very strict then the savings on abatement cost in CT are high compared to the savings on opportunity cost in CAT and therefore CT performs best on innovation and adoption incentives. When the difference in output level is small and the limit on total emissions of industry lax CAT comes on the first rank.

5. RECAPITULATION AND CONCLUSIONS

Cap-and-trade with free allowances and tradable emission reduction credits both equalize marginal abatement costs of polluters, thus minimizing total costs of emission control. Yet they are distinct instruments of environmental policy with different histories and dissimilar economic consequences.

Tradable credits have developed as, and still are, an instrument complementary to command-and-control in the form of mandated emissions per unit of output or input. CT infuses flexibility by allowing that emissions in excess of the emission standard at one emission source are compensated by emissions below the standard at another source. Residual emissions are free of cost, as it is in a scheme of emission standards without credit trading. There is no cap on total emissions, since producing output creates its own mandated emissions.

Cap-and-trade started, and continued its existence, not as an addition to command-and-control, but as a substitute. A cap is set on total emissions and allowed emissions are distributed among firms, either through auctioning or by handing out allowances for free. In a cap-and-trade scheme, residual emissions generated in producing output always have a cost: either the price paid in buying the permit to offset emissions, or the opportunity cost of using a permit for production. In a scheme with a cap on the emissions of a firm the opportunity cost of allowances used to offsetting the emission from output arises from the fact that the firm foregoes the opportunity to use the allowances again, either for the production of a next unit of output, or (in capped emissions with trade) for selling or banking the allowances. The opportunity cost is equal to the marginal cost of abating the emissions released in producing the marginal unit of output, which in cap-and-trade are equal to the market value of the allowances used to offset the emissions of the marginal unit of output.

Residual emissions are inputs to output; under credit trading the input is for free, under allowance trading it has a cost. Consequently the marginal cost of output is lower under CT; it results in lower price of output and higher level of industry output compared to allowance trading. A given target level of industry emissions can physically be achieved in two ways: by adjusting output and by adjusting emission abatement per unit of output. Since output is higher in credit trading than in allowance trading emission abatement per unit of output has to be higher. It implies that under CT marginal abatement cost is higher than under tradable allowances. Since in market equilibrium marginal abatement cost is equal to the price of the credit, respectively allowance, it also implies that market price of credits is higher than the price of allowances. The basic proposition then is: under tradable credits output is higher and emission abatement per unit of output is higher than under tradable allowances; and the credit price is higher than the allowance price.

Below I summarize the performance of the two instruments. The outstanding issue discussed in this chapter was: did they perform better in terms of economic efficiency than tradable credits?

5.1 Tradable Allowances

In cap-and-trade all costs of output are signaled in the price of the product, including the cost of the residual, non-abated emissions. It makes cap-and-trade the first-best solution to maximizing national welfare under the constraint of a target level for total emissions, when the markets for allowances and output have perfect competition. Allowance trading is then less costly to society than credit trading. For countries with industries engaged in trade on perfectly competitive international output markets, the Pareto-optimal solution is national cap-and-trade policies, complemented with international permit trading if possible. When there is imperfect competition on the market for output, allowance trading remains the first-best, welfare maximizing instrument, as long as output on the imperfect market is not too far below the output level on a perfect market.

In the process of diffusion of clean technology, cap-and-trade outperforms tradable credits in terms of long run welfare. However the incentive to adopt clean technology is strongest under credit trading.

5.2 Tradable Credits

Despite not being first-best, credit trading has merits of its own. First, the too high level of output under credit trading compared to cap-and-trade corrects, to a certain extent, the market failure of too low production when competition is imperfect. But the higher output also leads to higher costs of emission abatement. If the first effect exceeds the second credit trading comes out as second-best solution. Welfare is lower than in the first-best solution with perfect markets, but higher than under allowance trading when the output market is imperfect. The two effects also occur when the export industry of a country competes on an international oligopolistic product market. If the first effect exceeds the second, credit trading is the chosen instrument of international strategic competition, maximizing national welfare in a second-best solution.

Second, when output under credit trading is considerably higher than it is under cap-and-trade and the target for total emissions of the industry is strict, the innovation and adoption rent arising from lower abatement cost is under credit trade much higher than under cap-and-trade. It can then overrule the innovation and adoption rent arising from the savings on allowance expenditure or on opportunity cost, which is unique for cap-and-trade. Under those conditions credit trading performs better in terms of innovation and adoption incentives than cap-and-trade, either with free or auctioned allowances.

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4. Analyses of allowance transactions – firm behaviour in the first trading phase and learnings from the data

Claudia Kettner

1. INTRODUCTION

In general, economic theory prefers market-based instruments like emission taxes or emission trading schemes to command-and-control regulation like emissions standards. The former is viewed as delivering both environmentally effective and economically efficient outcomes (for example, Grüll and Taschini, 2011; Weishaar, 2014). Many discussions centre on the choice of the optimal market-based instrument for climate change mitigation. According to economic theory, the outcomes of a tax and an emissions trading scheme would be identical assuming the absence of externalities and uncertainty, that is, a world with perfect information, rational agents and the absence of market failures (e.g. Hepburn, 2006; McKibbin and Wilcoxen, 2002). Under uncertainty, both market-based instruments have advantages and disadvantages and policy-makers have to choose between uncertainty about carbon prices (emissions trading schemes) and uncertainty about emission reductions (taxes) (Murray et al., 2009). From an investor's point of view, emissions trading schemes entail higher uncertainty as the price is not fixed but formed on the market. Emissions trading allows, however, the reduction of emissions where it can be achieved at least cost.

In the EU there has been a lively discussion whether to opt for an emissions trading scheme or for a carbon tax as a policy instrument for meeting the emission reduction target as committed to in the Kyoto Protocol. The European Commission has initially been in favour of a carbon tax that could not be adopted due to the resistance of some Member States and the requirement of unanimity in fiscal environmental policies; eventually an emissions trading scheme was set up, partly due to lobbying activities of the industry (for example, Skjærseth and Wettestad, 2008).

The idea of emissions trading reflects the fact that the costs of reducing emissions differ between regulated entities, that is, it is more costly for some installations to comply with a certain emission limit than for others. Actors could benefit in a situation where those with lower emission abatement costs reduce their emissions below their predefined emission limit and sell the surplus reductions to actors with higher abatement costs who will in turn be allowed to emit more than their initial limit. In an emissions trading scheme, differences in marginal abatement costs will hence spur permit trading, leading to their equalization across market participants and aggregate cost efficiency in equilibrium (Montgomery, 1972). The more pronounced the differences in marginal abatement costs between market participants, the higher are the incentives for trading. In view of the broad range of activities covered by the EU ETS, one can expect significant differences in emission reduction costs and hence strong incentives for trade.

Additional flexibility in emissions trading systems can, inter alia, be provided by 'banking' and 'borrowing' on the one hand and the recognition of offsets (that is, permits from activities not included in the scheme) for compliance on the other (see for example, Kettner et al., 2011; Weishaar, 2014). These flexibility elements have to a certain extent been included in the design of the EU ETS in order to reduce compliance costs for the regulated firms.

This chapter presents empirical evidence on trading in the EU ETS. First, the economic theory of emissions trading is briefly laid out (section 2). Then trade with EU Allowances (EUAs, that is, emission allowances issued for compliance under the EU ETS) is analysed and discussed and the use of international credits for compliance under the EU ETS in the first trading phase (2005 to 2007) and the second trading phase (2008 to 2012) is addressed at country level (section 3) and sector level (section 4). In section 5 trading flows on installation and company level are assessed. This is complemented by a literature review of trading on company level and the use of banking and borrowing of EUAs (section 6). Section 7 summarises the main findings and draws conclusions.

2. THE THEORY OF EMISSIONS TRADING AND DESIGN OF THE EU ETS

The idea of emissions trading is based on differences in abatement costs between regulated entities. Vis-à-vis a carbon standard, emissions trading would imply benefits for regulated entities due to increased flexibility: actors with lower abatement costs could reduce their emissions below their predefined emission limit and sell the surplus reductions on the market; actors with higher abatement costs could buy these surplus reductions and in turn would be allowed to emit more than their initial limit. This permit trading would lead to an equalization of costs across market participants and aggregate cost efficiency in equilibrium (Montgomery, 1972).

In a cap-and-trade scheme like the EU ETS, a regulator defines a limit for (greenhouse gas) emissions for installations or sectors of an economy included. Emission permits are then allocated to the regulated entities free of charge based on historical emissions or benchmarks; alternatively permits can be auctioned to regulated entities.[1] In economic theory, the result – that is, the permit price and individual emission reductions – would be independent of the allocation method: allowances that are distributed free of charge also involve costs for the firms included in the scheme, since they represent opportunity costs. The opportunity cost of the certificates can be defined as the 'revenue forgone by refraining from selling the allowances and by employing them in producing output' (Woerdman, Clò and Arcuri, 2008). The distributional effects of the different allocation principles, however, vary (see Hepburn, 2006; Woerdman et al., 2008).

The theory of emissions trading assumes a world with perfect information and no uncertainties. In addition to perfect information, one prerequisite for the cost-effectiveness of an emissions trading system is that emission allowances are fully transferable between the regulated entities, that is, the absence of transaction costs. Transaction costs can either be administrative or related to trading in a narrow sense. Administrative costs include the costs of monitoring, reporting and verification; trading costs comprise for instance price discovery costs. In the presence of transaction costs, the cost-effectiveness of emissions trading is reduced (see for example, Tietenberg, 2006).

In addition to trading, further flexibility in emissions trading systems can be provided by 'banking' and 'borrowing' as well as by the allowance of offsets. Banking allows regulated entities to carry over unused emission allowances from one compliance period to another, while borrowing enables the use of allowances from future periods for current compliance. If the number of allocated allowances exceeds the verified emissions of an installation, that is, if an operator holds a surplus of allowances, the operator can either bank the allowances (keep them) or sell them on the market. If an operator, in contrast, is short of allowances it could either borrow allowances from the allocation of the subsequent year or buy emission allowances in the market. Notably there is an asymmetry between installations holding a surplus of allowances and those exhibiting a shortage: long operators – operators with a surplus of allowances – can behave passively and just keep their surpluses, while short operators – operators with a shortage of allowances – for compliance have to actively decide for either

[1] In the case of auctioning, the regulator has only to decide about the overall cap; the distribution to sectors or installations is left to the market.

borrowing or acquiring allowances on the market. As in terms of climate change not annual GHG emissions, but the cumulative stock of GHGs in the atmosphere are of relevance, environmental effectiveness of an ETS is ensured under banking and borrowing, provided the cumulative emission cap is fixed. Regarding the economic efficiency of an ETS, the option of banking and borrowing will generally reduce compliance costs as it allows for an inter-temporal optimisation of abatement activities (see for example, Tietenberg, 2006).

In the EU ETS, banking of allowances within the first trading phase (2005 to 2007) was provided for, but allowances issued for the pilot phase could not be banked to the second trading phase that started in 2008. Between the second and the third trading phase covering the period 2013 to 2020, banking of allowances is permitted. Borrowing is only implicitly allowed under the EU ETS: installations that are short of allowances in one year can also surrender allowances that were issued for the following trading year, since allowances are allocated at the beginning of the year while the compliance year runs until April. This form of borrowing has, however, not been allowed between different trading phases.

Another option to increase flexibility in an ETS is the recognition of offset credits. Offset credits are permits issued for verifiable GHG emission reductions in sectors that are not covered by the domestic cap-and-trade program ('domestic offsets'), or for qualified projects in other countries. The option of using offsets for compliance, that is, the linking of different carbon markets, increases liquidity and contributes to levelling emission prices.[2]

Installations covered by the EU ETS may also surrender a certain amount of credits from project-based mechanisms for compliance. The linking of the EU ETS with the Kyoto project-based mechanisms, including Joint Implementation (JI)[3] and the Clean Development Mechanism (CDM),[4] is supposed to increase the cost effectiveness of emission reductions. Directive 2004/101/EC, the 'Linking Directive', regulates the use of project-based mechanisms by installations covered by the EU ETS and provides the criteria for the use of Kyoto credits for compliance in the EU

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[2] For more details regarding linking see Türk (2009) and Chapter 14 of this book.

[3] The JI mechanism is defined in Article 6 of the Kyoto Protocol. Credits from project-based emission reductions in Annex-I countries – so called Emission Reduction Units (ERUs) – may be used for compliance in other Annex-I countries.

[4] The CDM mechanism is defined in Article 12 of the Kyoto Protocol. Credits from project-based emission reductions in non-Annex-I countries – 'Certified Emission Reductions' (CERs) – may be used for compliance in Annex-I countries.

ETS that shall ensure the environmental integrity of the scheme. The use of CDM credits has been allowed since 2005 under the EU ETS; since 2008 it has also been possible to surrender JI credits for compliance. The allowed maximum share of CDM and JI credits in the second trading phase was defined in the Member States' National Allocation Plans and generally ranged between 7 and 20 percent of allocated allowances (see for example, Sterk and Wang-Helmreich, 2008).[5]

3. TRADING FLOWS AT COUNTRY LEVEL

Trading flows at the country level can be assessed based on data from the European Union Transaction Log (EUTL).[6] The EUTL includes information on the allowances surrendered for compliance by each installation in the operator holding accounts. This refers primarily to the quantity of allowances surrendered each year as well as to the type of allowance – that is, whether companies used EUAs or credits from project-based mechanisms for compliance. In addition, the country in which the allowances have been issued is reported. With respect to international credits, details on the country of origin are disclosed for all years; for EUAs, in contrast, information on the country of origin of the permits is available only for the period 2005 to 2011; since the introduction of the EUTL in the year 2012 no information on the issuing Member State has been reported.

3.1 EUA Trading at Country Level

For the analysis of EUA trading at country level based on the EUTL three indicators can be used (Kettner et al., 2012): (1) allowance imports, defined as allowances surrendered by an EU Member State originating from another registry; (2) allowance exports, defined as allowances exported by one country and surrendered in another country; and finally (3) net exports of allowances (the difference between allowance exports and imports on Member State level). The analysis of trading flows hence may only include allowances that have been surrendered for compliance. Imports and exports of allowances that were not surrendered but banked by the installations cannot be considered due to data availability.

On average over the first trading period 120 million EUAs originating from another registry were used for compliance every year (Table 4.1). The

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5 The actual use of offset credits at country level is discussed in section 3.2.
6 Formerly Community Independent Transaction Log (CITL).

*Table 4.1 Surrendered allowances, EUA trade and net positions at country level in Phases 1 and 2 p.a.**

	Phase 1 (Annual average 2005–2007)									Phase 2 (Annual average 2008–2011)										
	Surr EUAs	EUA Exports		EUA Imports		EUA Net Exports		Net Position		Surr Units	Surr EUAs		EUA Exports		EUA Imports		EUA Net Exports		Net Position	
	[m]	[m]	[%][1)]	[m]	[%][1)]	[m]	[%][1)]	[m]	[%][1)]	[m]	[m]	[%][1)]	[m]	[%][1)]	[m]	[%][1)]	[m]	[%][1)]	[m]	[%][1)]
AT	32.5	0.8	2.4	1.6	4.9	−0.8	−2.4	0.1	0.3	30.2	29.1	96.2	2.2	7.1	3.4	11.2	−1.2	−4.1	1.5	4.8
BE	54.3	7.0	12.9	4.8	8.9	2.2	4.0	5.3	9.1	49.5	47.2	95.4	6.1	12.3	7.9	16.0	−1.9	−3.7	6.7	17.5
BG										35.9	32.4	90.4	1.0	2.7	0.1	0.2	0.9	2.5	3.0	8.3
CY										5.1	4.9	95.2	0.0	0.0	0.0	0.0	0.0	0.0	0.1	2.5
CZ	84.6	11.3	13.3	1.9	2.2	9.4	11.1	12.3	14.7	76.0	71.9	94.6	9.2	12.2	0.9	1.1	8.4	11.0	10.0	13.6
DE	482.4	9.2	1.9	19.0	3.9	−9.9	−2.0	15.3	3.9	452.5	411.9	91.0	22.4	5.0	37.5	8.3	−15.1	−3.3	−56.2	−12.4
DK	30.0	3.2	10.8	2.8	9.3	0.5	1.5	1.0	2.1	24.7	23.6	95.4	2.0	8.3	2.3	9.1	−0.2	−0.9	−0.8	–
EE	13.4	4.2	31.5	0.0	0.3	4.2	31.2	5.4	40.8	13.3	13.3	99.7	0.1	0.6	2.1	16.0	−2.1	–	−0.5	–
ES	183.3	2.5	1.4	16.2	8.8	−13.7	−7.5	−17.3	−8.6	138.7	121.4	87.5	13.0	9.3	11.9	8.6	1.1	0.8	13.1	9.5
FI	40.1	5.3	13.3	2.0	4.9	3.4	8.4	4.5	11.7	36.7	34.6	94.4	2.8	7.8	5.6	15.3	–	–	0.7	1.9
FR	128.4	15.4	12.0	1.3	1.0	14.1	11.0	21.8	18.9	113.7	103.1	90.6	11.4	10.0	5.4	4.7	6.0	5.3	20.0	17.6
GB	250.2	8.4	3.3	43.9	17.5	−35.5	−14.2	−40.8	−14.2	238.9	230.6	96.5	21.8	9.1	34.1	14.3	−12.3	−5.2	−20.7	−8.7
GR	71.3	0.8	1.1	0.3	0.5	0.4	0.6	−0.2	0.5	62.0	58.4	94.3	2.6	4.3	0.1	0.1	2.6	4.2	1.3	2.1
HU	26.5	3.6	13.4	0.2	0.8	3.3	12.6	4.0	15.5	23.6	22.0	92.9	2.9	12.3	1.5	6.3	1.4	6.0	1.1	4.5
IE	22.2	0.5	2.1	0.8	3.5	−0.3	−1.4	−2.6	−11.5	17.7	16.7	94.3	1.6	9.1	1.1	6.3	0.5	2.9	3.0	16.7
IT	225.1	2.5	1.1	14.5	6.4	−11.9	−5.3	−18.5	−6.1	196.8	184.6	93.8	10.0	5.1	13.8	7.0	–	−1.9	7.4	3.7
LI										0.0	0.0	100.0	0.0	10.9	0.0	0.0	0.0	10.9	0.0	102.1
LT	6.4	4.5	70.2	0.4	6.0	4.1	64.2	5.1	80.5	6.0	4.9	81.8	1.9	32.0	0.7	11.2	1.2	20.8	1.8	30.9

Table 4.1 (continued)

	Phase 1 (Annual average 2005–2007)									Phase 2 (Annual average 2008–2011)										
	Surr EUAs	EUA Exports		EUA Imports		EUA Net Exports		Net Position		Surr Units	Surr EUAs		EUA Exports		EUA Imports		EUA Net Exports		Net Position	
	[m]	[m]	[%][1)]	[m]	[%][1)]	[m]	[%][1)]	[m]	[%][1)]	[m]	[m]	[%][1)]	[m]	[%][1)]	[m]	[%][1)]	[m]	[%][1)]	[m]	[%][1)]
LU	2.6	0.4	13.5	0.0	0.0	0.4	13.5	0.6	22.9	2.1	2.0	93.7	0.1	3.0	0.0	0.0	0.1	3.0	0.3	15.8
LV	2.9	1.1	36.9	0.0	1.1	1.0	35.8	1.2	42.0	2.8	2.6	92.1	0.5	17.3	0.1	3.3	0.4	14.0	1.6	57.6
MT										1.9	1.9	100.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	10.8
NL	79.0	11.3	14.3	6.7	8.5	4.6	5.8	7.5	9.5	82.3	80.2	97.5	10.3	12.5	10.4	12.6	−0.1	−0.1	1.3	1.5
NO										19.3	17.7	91.7	3.7	19.1	10.7	55.4	−7.0	−36.3	−11.3	−58.7
PL	207.7	18.9	9.1	0.7	0.4	18.2	8.7	30.1	14.5	199.5	185.6	93.0	10.2	5.1	4.7	2.4	5.5	2.7	4.5	2.2
PT	33.7	2.8	8.3	0.5	1.4	2.3	6.9	3.3	10.1	26.8	24.8	92.5	3.7	13.7	1.3	4.7	2.4	9.0	4.8	17.9
RO	23.2	0.0	0.0	1.1	4.9	−1.1	−4.9	1.6	6.2	51.4	47.1	91.7	11.8	23.0	0.3	0.7	11.5	22.3	21.0	40.9
SE	19.5	1.7	9.0	0.5	2.4	1.3	6.6	3.1	15.7	20.0	19.2	95.7	2.3	11.6	2.6	12.9	−0.3	−1.3	2.0	10.2
SI	8.9	0.0	0.4	0.4	4.6	−0.4	−4.2	−0.2	−1.6	8.3	7.6	92.0	0.3	3.2	0.2	2.7	0.0	0.6	0.0	−0.5
SK	25.1	4.5	17.9	0.4	1.4	4.1	16.5	5.4	21.4	22.7	20.5	90.3	5.0	21.9	0.3	1.2	4.7	20.7	9.6	42.3
EU ETS	2,053.3	119.8	5.8	119.8	5.8	0.0	0.0	27.7	1.2	1,958.5	1,819.7	92.9	158.9	8.1	158.9	8.1	0.0	0.0	25.5	1.4

Notes:
* For Phase 1, no detailed information on the surrendered units is available for Bulgaria, Cyprus and Malta from the EUTL Operator Holding Accounts. Installations from the EFTA countries joined the EU ETS in 2008.
1) Percent of surrendered allowances.

Source: EUTL; own calculations.

surrender of foreign EUAs continuously increased over the first trading period (e.g. Kettner et al., 2012; Ellerman and Trotignon, 2009) suggesting an increase in trading activity as agents became accustomed to the new market.[7] In 2005 net flows of allowances amounted to 34.6 million; in 2006 and 2007 the net trading flows increased to 85.1 million and 240.3 million respectively (Table 4A.1).

Virtually all countries imported and exported EUAs in the first trading phase, since there were both installations with shortages of allowances and ones with surpluses in all countries (Kettner et al., 2008). The highest country shares in EUA imports occurred in the Netherlands (6.7 million), Italy (14.5 million), Spain (16.2 million), Germany (19.0 million) and the UK (43.9 million). The largest exporters of EUAs in the first trading period were Germany (9.2 million), the Netherlands (11.3 million), the Czech Republic (11.3 million), France (15.4 million) and Poland (18.9 million).

In the first trading period, 16 countries were net exporters of EUAs; the remaining eight countries were EUA net importers. The Baltic States showed the highest relative net exports of EUAs with up to 64 percent of surrendered allowances for Lithuania. The highest relative net imports (14.2 percent) resulted for the UK. As already expected, countries that on average exhibited a net short position or a deficit of allowances compared to verified emissions (that is, Austria, Ireland, Italy, Slovenia, Spain and the UK) have generally been net importing countries of EUAs. Germany and Romania have, however, been net importers of EUAs despite their net long position. The limited correlation between net exports of EUAs and the net long and short positions in these two countries may have three reasons (Kettner et al., 2012): First, the spread of net long and short positions within countries, that is, not all installations with surplus allowances might have sold them on the market and thus imports of allowances might have been necessary for compliance for other installations in these countries. Second, installations that are part of a multinational company and were facing a shortage of allowances may have received transfers from an associate company with a surplus of allowances located in another Member State. Third, installations may have imported more EUAs than they actually needed. This could reflect the expectation of a higher growth of emissions at the time the allowances

[7] Moreover the increase in trading between 2005 and 2006 might reflect the fact that the start of operation of some national registries was delayed (Jaraite and Kažukauskas, 2012).

were acquired. Other underlying factors could be strategic behaviour or price expectations for EUAs.[8]

When comparing net positions and net exports in the EU ETS pilot phase, one can see that in five Member States (Greece, Ireland, Italy, Spain and the UK) net exports exceed the countries' net permit surpluses or net long positions. These data discrepancies result from the fact that allowances distributed to new entrants[9] in the first trading phase were not listed in the EUTL (see e.g. Ellerman and Trotignon, 2009; McGuinness and Trotingnon, 2007).[10] For Phase 1, this leads effectively to higher net long positions and less pronounced net short positions than indicated by the EUTL database.[11]

In the second trading phase, the annual net flows of allowances significantly increased compared to the first trading period (see Table 4A.2): net flows of EUAs peaked in 2009 at 183.5 million and were approximately 145 million in 2008 and 2010; in 2011 163 million EUAs issued in other countries were surrendered for compliance. For the last year of the second phase, 2012, exports and imports of EUAs cannot be estimated as since the introduction of the EUTL details on the country of origin of the allowances are no longer disclosed (see above).

On average, in the period 2008 to 2011 the highest EUA exports showed for Romania (12 million EUAs p.a.), followed by the Czech Republic (8 million EUAs p.a.) and France (6 million EUAs p.a.). The largest net importers of EUAs were Germany (15 million EUAs p.a.), the UK (12 million EUAs p.a.) and Norway (7 million EUAs p.a.).

Up to 2011, 11 countries were net importers of EUAs: Austria, Belgium, Denmark, Estonia, Finland, Germany, Italy, the Netherlands, Sweden, the UK and Norway. The remaining 17 countries were net exporters of EUAs. Norway, which joined the EU ETS in 2008, shows the highest relative net EUA imports; 36 percent of the total number of allowances surrendered in Norway have been imported EUAs. This reflects Norway's decision not to

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[8] Especially firms in the electricity sector might have had an interest in higher carbon prices and hence might have bought additional allowances as they passed on the cost of the certificates to the consumers earning windfall profits since the introduction of the EU ETS (see Laing et al., 2013, and the literature cited therein).

[9] That is, new participants that recently have been included in the ETS.

[10] Also allowance withdrawals from installations are not accounted for in the EUTL allocation and compliance data. These withdrawals can, however, be expected to only account for a small portion of allowances.

[11] These distortions due to unconsidered allocations to new entrants are reduced by basing the analysis of long and short positions on installations for which data on allocated allowances and verified emissions are available for all trading years.

allocate EU allowances to offshore oil companies, which in turn had to buy all of their allowances on the market.

Six countries – Austria, Belgium, Finland, Italy, the Netherlands and Sweden – have been net importers of EUAs despite their net long position in the period 2008 to 2011. As for Phase 1 this could result from the spread of net long and short positions within countries and allowance transfers within companies. Moreover, this behaviour could imply that firms had anticipated higher emissions when they bought the allowances or acquired the allowances in order to bank them to the third trading phase that started in 2013.

Based on the net transfers of EUAs, Ellerman and Trotignon (2009) estimated the financial flows from allowance trading between countries for the first trading phase. As the actual price at which the volumes have been traded is not available from the EUTL, they valued surrendered imported allowances at the average price of the trading year in which they had been surrendered for compliance. The derived estimates provide a rough indication of the financial flows between countries, as fluctuations of the EUA price during the first trading period cannot be captured.

Applying this method to the EUTL trading data yields a total value of trade in the pilot phase of about €1.5 billion, assuming average carbon prices of €20.2 for 2005, €9.5 for 2006 and €0.1 for 2007. For 2005, the estimated financial flows amounted to €700 million; in 2006 they increased to €810 million. Despite the increase in the trading volume in 2007, the estimated financial flows in this year declined to €20 million due to the collapse in EUA prices towards the end of first trading period: While two-thirds of cross-border trading in the pilot phase occurred in 2007, the corresponding financial flows accounted for only 2 percent of the traded value of allowances in the first trading phase.[12] The analysis also highlights disparities at the country level: While Poland was the largest net exporter of allowances in the first trading phase, it received only the third-highest revenues from allowance exports (after France and the Czech Republic). This reflects the fact that most Polish EUA exports occurred in the second half of the pilot phase, since the Polish registry had not been activated before July 2006 (Ellerman and Trotignon, 2009). Germany, in contrast, had been a net importer of EUAs in physical terms but gained in terms of the net financial flows, since the value of exports in the first year overcompensated the costs of the subsequent imports.

[12] The price decline towards the end of the first trading phase reflected the over-allocation and non-bankability of allowances in most Member States (see e.g. Kettner et al., 2008).

For the second trading phase, average annual carbon prices were rather constant between 2008 and 2010 (ranging between €13.6 and €16.1) and declined to €10.4 in 2011. The total value of cross-country EU trade between 2008 and 2011 hence amounted to €8.7 billion. The estimated annual financial flows were €2.3 billion in 2008, €2.5 billion in 2009, €2.2 billion in 2010 and €1.7 billion in 2011. Since average allowance prices were comparably stable there is a stronger correlation between the EUA net exports and the financial flows and no disparities as in the pilot phase can be detected.

3.2 Use of Offsets at Country Level

Linking of the EU ETS with the project-based mechanisms of the Kyoto Protocol should increase the cost effectiveness of the scheme. The use of CDM credits has been allowed since 2005 under the EU ETS; since 2008 it has also been possible to surrender JI credits for compliance. In the first trading phase, however, no project-based credits have been used for compliance under the EU ETS according to the EUTL Operator Holding Accounts. In the second trading phase, in general the share of CER and JI credits at Member State level was limited to 7 (Slovakia) to 20 percent (Germany, Spain) of installations' allocated allowances; only Estonia decided not to allow the use of offsets in its National Allocation Plan.

The European private sector has been the largest buyer on the CER market[13] (see Cappor and Ambrosi, 2009; Kossoy and Ambrosi, 2010; World Bank, 2014). The share of CERs and ERUs in surrendered allowances was, however, relatively small compared to the share of EUAs and the limits for the use of offsets defined in the National Allocation Plans. On average in the period 2008 to 2011, 135 million of CERs and 76 million of ERUs have been submitted by the EU Member States. This corresponds to 6.9 and 3.9 percent of total surrendered allowances respectively.

The highest shares of CERs and ERUs in surrendered allowances accrues to Lithuania (23 percent), Spain (16 percent) and Slovenia (13 percent). In absolute terms, Germany, Spain, Italy and Poland have been the largest importers of credits from JI and CDM projects (Table 4.2).

Towards the end of the second trading phase, the use of offsets for compliance increased considerably (Table 4A.3). This is particularly the case for ERUs of which only 50,000 were surrendered in 2008, but 282 million were

[13] In the second trading phase, EU ETS installations were allowed to surrender a maximum of 1.4 billion of CERs and ERUs (14 percent of EU allocation) for compliance.

Table 4.2 Surrendered allowances and offsets at country level in Phase 2 p.a.

	Surr. Units	Surr. CERs		Surr. ERUs	
	[m]	[m]	[%][1)]	[m]	[%][1)]
AT	30.1	1.8	6.0	1.0	3.4
BE	48.6	2.7	5.6	1.1	2.3
BG	35.8	2.0	5.5	2.7	7.6
CY	4.7	0.2	4.2	0.0	0.1
CZ	74.8	4.0	5.3	3.8	5.0
DE	455.0	34.0	7.5	26.8	5.9
DK	23.6	1.0	4.3	1.5	6.4
EE	13.4	0.1	0.7	0.5	3.4
ES	138.9	16.8	12.1	4.8	3.4
FI	35.4	2.5	7.0	0.8	2.3
FR	113.7	11.4	10.1	3.9	3.4
GB	240.5	11.4	4.7	4.5	1.9
GR	62.9	3.3	5.3	2.3	3.6
HU	23.7	1.4	5.9	0.6	2.5
IE	19.4	1.0	5.2	0.6	3.0
IT	193.8	13.3	6.9	5.9	3.0
LI	0.0	0.0	0.0	0.0	0.0
LT	5.9	0.7	11.3	0.7	11.8
LU	2.4	0.2	8.2	0.0	0.3
LV	2.9	0.2	8.1	0.1	3.6
MT	1.8	0.0	0.3	0.0	0.0
NL	82.0	3.5	4.3	2.3	2.8
NO	19.5	1.4	7.3	0.4	2.3
PL	199.1	13.0	6.5	6.2	3.1
PT	26.8	2.0	7.6	1.0	3.6
RO	51.9	3.2	6.1	3.3	6.3
SE	20.5	1.6	7.8	0.5	2.6
SI	8.1	0.3	3.7	0.9	11.5
SK	22.4	1.9	8.7	0.1	0.3
EU ETS	1,957.7	135.0	6.9	76.3	3.9

Note: 1) Percent of surrendered allowances.

Source: EUTL; own calculations.

surrendered in the last year of the second trading phase, 2012. The use of CERs for compliance increased from 83 million in 2005 to 219 million in 2012. This development reflects limits of banking for offset credits between Phase 2 and Phase 3 of the EU ETS, as defined in Directive 2009/29/EC.

There are four main producers of CERs: China, India, South Korea and Brazil. Imports from these countries accounted for more than 95 percent of total EU CER imports (see for example, Kettner et al., 2012). ERUs surrendered by EU installations mainly originated from the Ukraine which accounted for more than 50 percent of ERU imports. A significant share of ERUs (13 percent) also originated from Russia. One third of surrendered ERUs stemmed from EU Member States (for example, Germany or Poland).

4. TRADING FLOWS AT SECTOR LEVEL

Trading flows at the sector level can also be assessed with data from the EUTL Operator Holding Accounts. For this analysis in general the same data restrictions apply as at country level, that is, information on the origin of offsets is available for the entire second trading phase while information on the origin of surrendered EUAs on Member State level is only available until 2011. As the analysis of trading flows is based on the Operator Holding Accounts it can, however, only include allowances that have been surrendered for compliance due to data availability. This means that only (gross) imports of EUAs, that is, surrendered EUAs from a different country of origin, can be assessed at sector level.

4.1 EUA Trading at Sector Level

The combustion sector that covers 'combustion of fuels in installations with a total rated thermal input exceeding 20 MW' (Directive 2009/29/EC) – that is, mostly power and heat generation but also combustion activities related to other economic activities – is the largest EU ETS sector and accounted for almost three-quarters of surrendered allowances in Phase 1 and Phase 2. The sector, however, played an even more important role in terms of EUA trading (see Table 4.3 and, for more detailed information, Tables 4A.4 and 4A.5 in the Appendix): About 92 percent (90 percent) of EUAs that were imported from another EU Member State were surrendered for compliance by the sector in Phase 1 (Phase 2). This reflects on the one hand that the combustion sector was in a net short position of allowances in both trading phases and had to purchase additional allowances on the market. On the other hand, many electricity generators are international companies which might have favoured international allowance transfers within companies. Moreover, this might explain the larger size of firms in the power sector on the one hand (see above) as well as the sector's trading experience in other markets (Jaraite and Kažukauskas, 2012).

As expected, EUA imports in sectors exhibiting a net shortage of

*Table 4.3 Surrendered allowances, EUA imports and net positions at sector level in Phase 1 and 2 p.a.**

	Phase 1 (2005–2007)					Phase 2 (2008–2011)					
	Surr. EUAs	EUA Imports		Net Position		Surr. Units	Surr. EUAs	EUA Imports		Net Position	
	[m]	[m]	[%][1)]	[m]	[%][1)]	[m]	[m]	[m]	[%][1)]	[m]	[%][1)]
Aluminium	0.2	0.0	0.0	0.2	68.7	0.2	0.2	0.0	0.0	0.2	74.9
Cement and Lime	185.2	3.5	1.9	5.3	2.9	163.9	136.2	3.9	2.4	51.9	31.7
Ceramics	14.9	0.3	2.2	3.0	20.1	10.4	7.1	0.1	1.3	8.6	82.9
Chemicals	5.0	0.0	0.1	1.3	25.9	16.4	13.9	0.3	1.9	2.9	17.8
Coke	14.3	0.1	0.9	2.2	15.5	13.1	11.7	0.8	5.8	2.4	18.3
Combustion	1,507.7	111.2	7.4	−26.9	−1.8	1,432.2	1,262.4	144.0	10.1	−139.2	−9.7
Glass	19.8	0.4	1.9	2.0	10.1	20.2	16.9	0.3	1.4	4.5	22.5
Iron and Steel	121.0	1.3	1.0	23.9	19.8	106.6	82.4	2.7	2.5	74.0	69.4
Opt-in	0.2	0.0	7.3	−6.8	−2859.6	21.7	18.2	0.1	0.4	1.1	5.3
Other metals	6.9	0.0	0.0	7.0	101.7	2.8	2.4	0.0	0.0	0.6	21.4
Other minerals	0.4	0.0	10.7	0.0	−2.8	1.1	1.0	0.1	4.8	0.4	33.9
Pulp and Paper	31.0	0.7	2.2	7.5	24.1	30.5	23.3	0.4	1.3	10.1	33.1
Refineries	146.5	2.5	1.7	9.1	6.2	139.3	124.8	6.4	4.6	7.9	5.7
EU ETS	2,053.3	120.0	5.8	27.7	1.4	1,958.5	1,700.4	158.9	8.1	25.5	1.3

Notes:
* For Phase 1, no detailed information on the surrendered units is available for Bulgaria, Cyprus and Malta from the EUTL Operator Holding Accounts. Installations from the EFTA countries joined the EU ETS in 2008.
1) Percent of surrendered allowances.

Source: EUTL; own calculations.

allowances are generally higher than in the other sectors, since these sectors could not cover their emissions with their initial allocation (see Table 4.3). This does not only apply to the combustion sector, but in the first trading phase also to the sectors 'other minerals' and opt-in installations. Notably, in the second trading phase in three sectors – refineries, other minerals and coke – a considerable amount of imported EUAs was surrendered for compliance despite pronounced net surpluses of allowances. This could again reflect the spread of net long and short positions within sectors, allowance transfers within companies, exaggerated presumptions about production and hence emissions development or the decision to acquire allowances in order to bank them to the next trading phases.

4.2 Use of Offsets at Sector Level

The combustion sector also dominated offset purchases in the EU ETS: 92.3 million CERs (70 percent of ETS-wide surrendered CERs) and 43.6 million ERUs (63 percent of ETS-wide surrendered ERUs) respectively used in the EU ETS were surrendered by combustion sector installations. In terms of the use of offsets for compliance, the combustion sector is, however, underrepresented in comparison with its share in total surrendered allowances. Installations from the sectors ceramics, iron and steel, and pulp and paper have been comparably active in terms of the use of international credits for compliance (Table 4.4 and Table 4A.6 for annual data); in the ceramic sector the share of ERUs and CERs in surrendered allowances amounted to 26 percent; in the sectors pulp and paper and cement and lime the respective shares were 19 and 18 percent. The highest percentage imports of CERs occurred in the iron and steel sector (11.5 percent of the sector's surrendered allowances); the highest relative ERU imports showed for the ceramics sector (17 percent of the sector's surrendered allowances).

5. TRADING FLOWS AT INSTALLATION LEVEL

Trading flows at the installation level can also be assessed with data from the EUTL Operator Holding Accounts, again keeping in mind the same restrictions as for the sectoral analysis. In this section the structure of trading flows at installation level is assessed focussing first on EUA trading and then on the use of offsets. This is complemented by a literature review of trading on company level and the use of banking and borrowing of EUAs in section 6.

Table 4.4 Surrendered allowances and offsets at sector level in Phase 2 p.a

	Surr. Units	Surr. CERs		Surr. ERUs		Surr. Offsets	
	[m]	[m]	[%][1)]	[m]	[%][1)]	[m]	[%][1)]
Aluminium	0.2	0.0	7.2	0.0	8.2	0.0	15.5
Cement and Lime	159.3	14.1	8.8	8.1	5.1	22.1	13.9
Ceramics	10.0	0.9	9.4	1.7	17.0	2.6	26.4
Chemicals	16.4	1.4	8.3	0.7	4.2	2.0	12.5
Coke	12.8	0.6	4.8	0.6	4.5	1.2	9.2
Combustion	1,424.1	92.3	6.5	43.6	3.1	135.8	9.5
Glass	20.0	2.0	10.1	0.6	3.1	2.6	13.2
Iron and Steel	105.3	12.1	11.5	7.2	6.9	19.4	18.4
Opt-in	21.4	0.7	3.1	2.1	9.9	2.8	13.0
Other metals	2.8	0.2	7.6	0.1	4.4	0.3	12.0
Other minerals	1.1	0.1	5.1	0.1	6.6	0.1	11.7
Pulp and Paper	30.1	3.4	11.2	2.4	8.0	5.8	19.2
Refineries	137.5	6.1	4.4	5.5	4.0	11.5	8.4
EU ETS	1,941.1	133.8	6.9	72.6	3.7	206.5	10.6

Note: 1) Percent of surrendered allowances.

Source: EUTL; own calculations.

5.1 EUA Trading at Installation Level

In the first trading phase, 1,880 out of 14,457 installations in the EU ETS (13 percent of all installations) surrendered EUAs originating from another country for compliance. In the period 2008 to 2011, the number of installations surrendering imported EUAs increased to 2,380 or 16 percent of installations respectively (Table 4.5). On average, 8,300 imported EUAs were surrendered for compliance by each installation in the first trading phase, compared to a total of 142,000 allowances surrendered on average each year. In Phase 2, average EUA imports increased to 10,990 EUAs p.a. while the average number of surrendered EUAs decreased slightly to 134,270. The median of surrendered emission allowances also decreased between Phase 1 and Phase 2 both for total surrendered EUAs and EUA imports. This implies that the spread in the size of EUA imports has increased between the two trading phases. The maximum number of imported EUAs surrendered for compliance by an installation amounted to 8.2 million in Phase 1; in the first years of Phase 2 this number even rose to 10.2 million. The maximum

Table 4.5 Surrendered allowances and EUA imports at installation level in Phases 1 and 2 p.a.

	Phase 1	Phase 2
Number of installations		
All installations	14,457	14,457
Installations with imports	1,880	2,382
Installations without imports	12,577	12,075
Average number of surrendered units		
All installations	142,029	134,268
Installations with imports	513,618	408,549
Installations without imports	86,484	80,161
Minimum number of surrendered units		
All installations	0	0
Installations with imports	1	0
Installations without imports	0	0
Maximum number of surrendered units		
All installations	30,125,167	31,602,489
Installations with imports	29,590,525	27,153,539
Installations without imports	30,125,167	31,602,489
Median of surrendered units		
All installations	7,143	5,094
Installations with imports	33,118	16,067
Installations without imports	5,240	3,965
Average number of surrendered imported EUAs		
All installations	8,301	10,991
Installations with imports	63,837	66,706
Minimum number of surrendered imported EUAs	0	0
Maximum number of surrendered imported EUAs	8,153,869	10,204,295
Median of surrendered imported EUAs		
All installations	0	0
Installations with imports	2,820	1,810

Source: EUTL; own calculations.

number of total surrendered allowances has increased from 30.1 million to 31.6 million. Generally, installations that surrendered imported EUAs were bigger in terms of emissions and respectively surrendered allowances than those without EUA imports; the largest installation – a Polish power plant – did however not surrender any EUAs from another EU Member State, but used international credits for compliance (see below).

Table 4.6 Surrendered allowances and offsets at installation level in Phase 2 p.a.

	All installations	Installations surrendering			
		CERs and ERUs	CERs only	ERUs only	No offsets
Number of installations	14,457	2,057	4,506	1,715	6,179
Average number of surrendered units	134,268	411,982	164,714	99,074	29,382
Maximum number of surrendered units	31,602,489	31,602,489	23,731,798	4,867,195	27,153,539
Median of surrendered units	7,058	37,782	18,285	15,031	8
Average number of surrendered offsets	14,281	53,999	16,279	12,851	
Maximum number of surrendered offsets	2,718,314	2,718,314	1,601,630	538,010	
Median of surrendered offsets	400	6,309	2,481	2,064	

Source: EUTL; own calculations.

5.2 Use of Offsets at Installation Level

In Phase 2, more than 50 percent of installations made use of offsets for compliance under the EU ETS: 2,060 installations surrendered both CERs and ERUs, 4,500 installations only CERs and 1,720 only ERUs; the remaining 6,180 installations did not surrender any offsets (Table 4.6). Installations that imported offsets were generally larger in terms of emissions and surrendered allowances respectively: Installations that surrendered CERs as well as ERUs on average surrendered 412,000 allowances p.a., while installations that did not use any offsets for compliance surrendered only 30,000 allowances. On average p.a., all installations surrendered 14,280 offsets per year. The maximum number of offsets surrendered by an installation amounted to 2.7 million.

6. EUA TRADING AT COMPANY LEVEL AND BANKING AND BORROWING

In addition to the comprehensive analysis of emissions trading at country, sector and installation level in Phases 1 and 2 of the EU ETS, in the following section a literature review of allowance trading at the firm level is presented. Some of these analyses (Jaraite and Kažukauskas, 2012; Zaklan, 2013) are based on transactions data from the EUTL that are published with a time lag of three years; hence these empirical studies so far only cover the first trading phase or parts of it. In order to obtain insights at the company level, installation level data has to be matched with company data from other sources, e.g. the AMADEUS database (see for example, Jaraite et al., 2013). Complementary to these quantitative studies, surveys on firm behaviour in the EU ETS have been conducted (Jaraite et al., 2010; Sandoff and Schaad, 2009).

6.1 EUA Trading at Company Level

Jaraite and Kažukauskas (2012) present an econometric analysis of firm trading behaviour and transaction costs in the pilot phase of the EU ETS at firm level in 22 EU Member States. Their analysis showed that about one-quarter of the firms included in the EU ETS were sellers of allowances and about one-sixth of ETS firms were buyers of allowances. The majority of the firms that sold allowances exhibited a net surplus of allowances; 50 percent of the firms that were buyers of allowances exhibited, however, also a net surplus of allowances.[14] This suggests that these firms did not primarily acquire emission allowances for compliance under the EU ETS but might also have bought them for financial speculation. According to Jaraite and Kažukauskas (2012), the average size of the transactions was 214,000 allowances in terms of sold allowances and 227,000 allowances in terms of bought allowances respectively. The largest amounts of allowances per transaction were on average sold by firms from the UK (580,000 EUAs per transaction), the Netherlands (470,000 EUAs) and Estonia (450,000 EUAs); the largest amounts of allowances were on average bought by firms from the UK (760,000 EUAs per transaction), the Netherlands (540,000 EUAs) and Spain (340,000 EUAs). Firms that participated in EUA trading in general were bigger in terms of capital and revenues than the average firm included in the EU ETS; capital and revenues of the firms that were

[14] More than one thousand firms were active on both sides of the market within the same year.

buying allowances were higher than those of the sellers. Moreover, firms that participated in trading consisted of more installations than the average EU ETS firm, with buyers holding on average more installations than sellers. Jaraite and Kažukauskas (2012) show that search and information costs strongly influenced trading decisions in the EU ETS pilot phase: firms comprising a comparably large number of installations participated more actively in trading which suggests that these firms more easily found a trading partner. This is straightforward as allowances could be traded between the different installations of the same firm, which would entail lower search costs compared to firms with a single installation; moreover, larger firms with more installations might have a separate business unit for issues related to EU ETS and pursue a coordinated trading strategy.[15]

Zaklan (2013) analyses firm level trading flows in the EU ETS in the period 2005/2006 based on data from the EUTL that was complemented by information on firm characteristics like size, productivity, profitability and ownership structure from the AMADEUS database. He finds that firms' participation in allowance trading is determined by a combination of firm-specific characteristics (like firm size, company structure or sector) on the one hand and ETS market-specific factors (such as the level of emissions or shortage of allowances) on the other hand. Firms' decisions on the traded volumes seem to reflect largely market-specific concerns. According to Zaklan (2013), larger firms were more likely to purchase EUAs on the market, while the probability of selling allowances on the market was not affected by firm size. Firms that were at least partly owned by the government were found to be more likely to engage in allowance trading, while family-owned firms were less likely to participate in the allowance market. As was suggested by the descriptive analysis presented above, firms from the manufacturing sector were found to be less likely to participate in permit trading than electric utilities. In terms of ETS market characteristics large emitters were more likely to be active in trading as well as firms with a net shortage of grandfathered allowances. With respect to the traded volume of EUAs, the analysis suggests that it is strongly influenced by the value of the firms' free EUA allocation, but also by the relation of firms' allowance endowments and verified emissions. This supports the descriptive results shown before. The analysis by Zaklan (2013) shows that EUA purchases were concentrated at two points of each year, namely at the end of the respective compliance and calendar years: transactions between companies

[15] This study also supports the concerns raised by the European Commission (e.g. see CEC, 2008) that transaction costs might be excessive for smaller participants.

intensified at the end of the calendar years, when EUA future prices were determined. Transfers of allowances within companies, in contrast, concentrated at the end of the compliance years, suggesting comprehensive compliance strategies across installations by firms with multiple installations.

Another analysis was done by Jaraite et al. (2010) who performed a qualitative survey on firm transaction costs of Irish firms in the EU ETS. They distinguish between trading costs on the one hand and the costs of monitoring, reporting, verification and early implementation on the other hand, since – in contrast to the other cost types – trading costs are variable, that is, they depend on the amount of allowances traded. Eleven out of the 27 respondents to the survey had engaged in trading in the EU ETS pilot phase, with six selling and five purchasing allowances on the market; the remaining firms did not engage in trading during the whole pilot phase. By the end of Phase 1 three out of the five companies that bought allowances showed a surplus of allowances as compared to their emissions and seven of the companies that did not participate in trading also showed a considerable surplus of allowances. For the majority of the firms the decision not to participate in trading was motivated by the fact that their allocation had been sufficient to cover their CO_2 emissions and hence trading had not been necessary. Consequently the survey suggests that neither high transaction costs nor cost-effective abatement opportunities explain the covered entities' reluctance towards trading. Some respondents, however, added that when it had become obvious that they had held sufficient allowances to cover their obligations under the EU ETS by the end of the first trading phase, the price had been too low to consider trading as an option. About half of the trading respondents had been trading only with installations from the same business group, and the majority of these trading partners had been located in Ireland as well. The remaining respondents had been trading mainly via financial institutions. Seven out of the eleven respondents had directly engaged in trading, while the remaining respondents had employed an intermediary. The arguments for direct trading included that the volumes of allowances were too small to be traded via third parties, as well as the lower costs of direct trade compared to trade via third parties. Companies that decided to trade allowances indirectly mentioned, inter alia, a lack of in-house trading capacity and the lower cost of indirect trade as an explanation.[16] The survey by Jaraite et al. (2010) concludes that trading costs in the EU ETS have not been a decisive factor for firms' trading decisions. Instead,

[16] In the EU ETS, indirect trading benefitted from low and declining brokerage fees over the first trading phase (see e.g. Convery and Redmond, 2007).

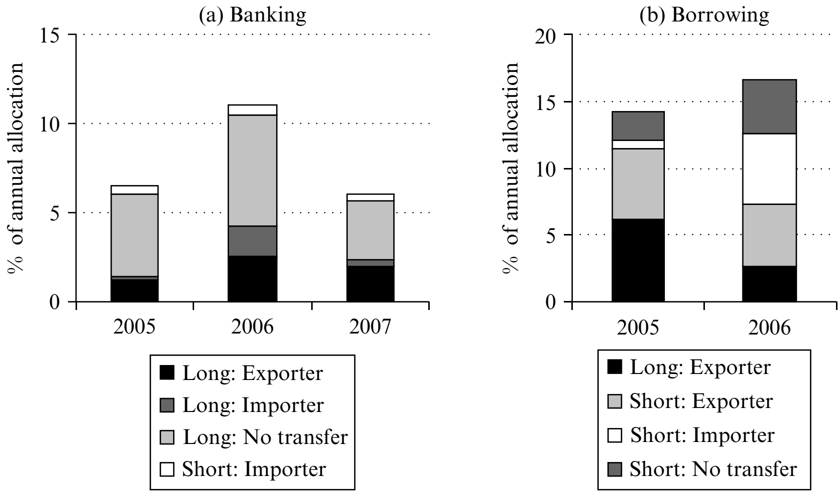
firms' decisions to use allocated allowances exclusively for their own compliance as well as the price collapse towards the end of the pilot phase seem to have shaped firms' decisions not to sell surplus allowances on the market. This matches the results by Ellerman and Trotignon (2009) who detected higher market activity by installations that were short of allowances compared to installations that were long based on their analyses of compliance trading in the first trading phase.

Sandoff and Schaad (2009) conducted a survey about the attitudes and behaviour of Swedish firms included in the EU ETS in the first half of the first trading phase. In terms of their primary trading strategy, respondents most frequently stated that they would trade so that 'the predicted emissions from the first period always are covered'. Another prevalent strategy identified was to purchase allowances only by the end of the year. The dominant motivation hence was a reduction of risk on the one hand and a minimization of administrative efforts on the other hand. As Sandoff and Schaad (2009) concluded, a continuous pursuit of such a trading strategy will impair the efficiency of the EU ETS. A higher stringency of the cap in the EU ETS as well as the possibility of banking allowances towards subsequent trading periods might, however, help stabilise carbon prices and hence mitigate inefficiencies.

6.2 Banking and Borrowing

In their analysis of the first trading period, Martino and Trotignon (2013) address trading as well as banking and borrowing of allowances for a sample of 6,700 installations representing 60 percent of EU ETS operators and 65 percent of EU ETS emissions respectively. The structure of banked and borrowed allowances is illustrated in Figure 4.1.

In 2005 and 2006, 7 percent (91 million) and 11 percent (141 million) respectively of annual allocated allowances were banked to the next trading year (Figure 4.1(a)). The majority of the banked allowances (62 percent) stemmed from installations that exhibited surplus allocation and did not engage in trading at all. Twenty-one percent of the total banked allowances were kept by long installations that had also sold some permits on the market; some installations that were already endowed with surplus allowances acquired additional permits and banked them (accounting for only 11 percent of the total banked allowances). Notably, also some installations that initially faced a shortage of allowances ended up with a surplus of allowances that they banked to the next trading year; the allowances banked by this category of installations accounted for 6 percent of total banking in 2005. In 2007, 81 million excess allowances (6 percent of annual allocation were kept by operators) became worthless by the end of the first



Source: Own illustration based on Martino and Trotignon (2013).

Figure 4.1 Banking and borrowing of allowances in Phase 1

trading phase. Fifty-five percent of these worthless allowances belonged to long installations that had not engaged in trading and 32 percent to installations that had been long but also sold some permits on the market. The remaining 13 percent of allowances were kept by long and short installations that had acquired some permits on the market.

In 2005 and 2006, installations borrowed 197 million (14 percent of annual allocation) and respectively 214 million (17 percent of annual allocation) from the allocation of the subsequent year for compliance. In 2005 especially long and short installations that had been exporting allowances have made use of borrowing for compliance purposes. In 2006, the option of borrowing was used to a similar extent by long installations that had exported allowances as well as by all types of installations that initially had exhibited an allowance deficit.

7. SUMMARY AND CONCLUSIONS

According to economic theory, emissions trading is both environmentally and cost effective in a world with perfect information. In the real world, a cap-and-trade scheme guarantees environmental effectiveness in the sense that the predefined cap is complied with. In terms of cost effectiveness, the performance of an emissions trading scheme depends, however, on the presence of fully rational actors and perfect information as well as on the absence of transaction costs.

Options for analysing trading in the EU ETS are limited due to limited data availability and a time lag of data publication. Empirical analysis of the EU ETS show that the assumptions of the theory of emissions trading are not matched by the real-world setting. Allowance imports (purchases) and exports (sales) showed only a very limited correlation with allowance surpluses and allowance deficits. This phenomenon cannot only be observed at country and sector level, where differences between installations and intra-firm transfers could have been a possible explanation for these discrepancies, but first analyses of trading in the EU ETS show that several companies have bought additional allowances on the market in Phase 1 despite being endowed with surplus allocation and the absence of banking between Phase 1 and Phase 2. Moreover, regulated entities seem not to have considered any opportunity costs of holding grandfathered allowances which contributed to a lower participation of companies with surpluses of allowances in the market. This is also reflected in the low participation in the trade of companies with allowance surpluses. In a survey among Irish firms, Jaraite et al. (2010) found, for example, that most firms that did not participate in trading argued that 'they had been able to meet their CO_2 obligations without engaging in trading'.

Zaklan (2013) shows that firms' participation in allowance trading was determined by a combination of firm-specific characteristics (as firm size, ownership structure) and ETS market-specific factors (as the level of emissions and shortage of allowances) while traded volumes seem to reflect largely market-specific concerns. Transaction costs, that is, search and information costs, have strongly influenced firms' trading decisions in the EU ETS pilot phase (Jaraite and Kažukauskas, 2012). Firms with a comparably large number of installations participated more actively in trading: larger firms more easily found a trading partner as allowances could be traded between the different installations of the same firm on the one hand, and these firms might have a separate unit for issues related to EU ETS and pursue a coordinated trading strategy on the other hand.

While the EU ETS is far from being a perfect market, empirical evidence shows an increase in trading activity since the start of the EU ETS in 2005 as agents became accustomed to the new market. A higher stringency of the cap in the EU ETS as well as the possibility of banking allowances towards subsequent trading periods might help stabilise carbon prices and hence mitigate inefficiencies.

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APPENDIX

Table 4A.1 Surrendered EUAs and EUA exports and imports by country and year in Phase 1

	Surrendered EUAs			EUA Exports					
	2005 [m]	2006 [m]	2007 [m]	2005 [m]	2006 [m]	2007 [m]	2005 [%][1)]	2006 [%][1)]	2007 [%][1)]
AT	33.4	32.4	31.8	0.1	0.3	2.0	0.3	0.8	6.4
BE	55.4	54.8	52.9	1.6	4.3	15.2	2.8	7.9	28.8
CZ	6.3	159.8	87.8	5.7	8.9	19.3	89.3	5.6	22.0
DE	469.9	483.5	493.8	4.8	7.0	15.6	1.0	1.5	3.2
DK	35.4	34.3	20.5	2.6	1.8	5.4	7.2	5.1	26.5
EE	12.6	2.4	25.0	1.2	5.7	5.7	9.5	235.1	22.8
ES	165.2	198.4	186.3	0.3	2.2	5.1	0.2	1.1	2.7
FI	33.0	44.7	42.5	3.5	3.1	9.4	10.5	6.9	22.2
FR	19.7	238.7	126.9	5.6	9.9	30.8	28.5	4.1	24.3
GB	242.4	251.5	256.8	3.3	6.5	15.2	1.4	2.6	5.9
GR	71.3	69.9	72.7	0.0	1.3	1.0	0.0	1.9	1.4
HU	25.4	26.1	27.8	0.1	2.8	7.8	0.3	10.8	28.0
IE	22.4	21.7	22.4	0.1	0.8	0.6	0.2	3.8	2.5
IT	90.3	348.0	237.0	0.0	1.0	6.6	0.0	0.3	2.8
LT	6.6	6.4	6.1	0.5	3.1	9.8	8.1	49.1	159.9
LU	2.6	2.7	2.6	0.0	0.3	0.7	0.0	12.3	28.4
LV	2.9	2.9	2.8	0.2	0.7	2.2	8.4	23.8	78.9
NL	80.4	76.7	79.9	2.6	7.8	23.4	3.2	10.1	29.3
PL	204.8	208.3	210.0	0.1	9.1	47.5	0.0	4.3	22.6
PT	36.5	33.1	31.4	0.3	2.5	5.5	0.9	7.6	17.6
RO	0.0	0.0	69.7	0.0	0.0	0.0			0.0
SE	19.4	18.9	20.2	0.7	1.3	3.3	3.4	7.0	16.1
SI	8.7	8.7	9.2	0.0	0.0	0.1	0.1	0.0	1.1
SK	0.6	50.4	24.4	1.5	4.5	7.5	249.7	8.9	30.8
EU ETS	1,645.2	2,374.4	2,140.3	34.6	85.0	239.8	2.1	3.6	11.2

Note: 1) Percent of surrendered allowances.

Source: EUTL; own calculations.

EUA imports						EUA Net Exports					
2005 [m]	2006 [m]	2007 [m]	2005 [%][1)]	2006 [%][1)]	2007 [%][1)]	2005 [m]	2006 [m]	2007 [m]	2005 [%][1)]	2006 [%][1)]	2007 [%][1)]
0.8	2.6	1.3	2.5	7.9	4.2	−0.7	−2.3	0.7	−2.2	−7.0	2.1
0.6	1.3	12.6	1.2	2.4	23.8	0.9	3.0	2.6	1.7	5.5	5.0
0.0	0.0	5.5	0.1	0.0	6.3	5.7	8.9	13.8	89.2	5.5	15.7
2.3	11.0	43.7	0.5	2.3	8.9	2.5	−4.0	−28.1	0.5	−0.8	−5.7
3.5	0.4	4.5	9.8	1.3	21.8	−0.9	1.3	1.0	−2.6	3.9	4.7
0.0	0.0	0.1	0.2	0.0	0.4	1.2	5.7	5.6	9.3	235.1	22.5
10.0	10.6	28.0	6.0	5.3	15.0	−9.7	−8.3	−22.9	−5.9	−4.2	−12.3
0.0	2.1	3.8	0.1	4.7	8.9	3.4	1.0	5.7	10.4	2.2	13.3
0.2	0.4	3.4	1.0	0.2	2.7	5.4	9.5	27.4	27.5	4.0	21.6
11.9	35.2	84.5	4.9	14.0	32.9	−8.6	−28.7	−69.3	−3.5	−11.4	−27.0
0.0	0.1	0.9	0.0	0.1	1.3	0.0	1.2	0.1	0.0	1.8	0.1
0.0	0.0	0.6	0.0	0.1	2.3	0.1	2.8	7.1	0.3	10.7	25.6
0.5	0.6	1.3	2.1	2.7	5.7	−0.4	0.2	−0.7	−1.8	1.1	−3.3
0.4	17.6	25.4	0.4	5.1	10.7	−0.4	−16.6	−18.8	−0.4	−4.8	−7.9
0.0	0.8	0.3	0.4	12.3	5.5	0.5	2.4	9.4	7.7	36.8	154.3
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.7	0.0	12.3	28.4
0.1	0.0	0.0	2.2	0.0	1.0	0.2	0.7	2.2	6.2	23.8	77.8
2.9	1.1	16.1	3.6	1.5	20.2	−0.3	6.7	7.3	−0.4	8.7	9.1
0.0	0.1	2.1	0.0	0.1	1.0	0.1	8.9	45.5	0.0	4.3	21.7
1.0	0.1	0.3	2.8	0.2	1.0	−0.7	2.5	5.2	−1.9	7.5	16.6
0.0	0.0	3.4			4.9	0.0	0.0	−3.4			−4.9
0.3	0.2	0.8	1.7	1.3	4.1	0.3	1.1	2.4	1.7	5.7	12.0
0.0	0.2	1.0	0.0	2.3	11.3	0.0	−0.2	−0.9	0.1	−2.3	−10.2
0.1	0.5	0.4	21.7	1.0	1.8	1.4	4.0	7.1	228.0	7.9	29.0
34.6	85.0	239.8	2.1	3.6	11.2	0.0	0.0	0.0	0.0	0.0	0.0

Table 4A.2 *Surrendered EUAs and EUA exports and imports by country and year in Phase 2 (2008–2011)*

	Surrendered EUAs				EUA Exports								
	2008 [m]	2009 [m]	2010 [m]	2011 [m]	2008 [m]	2009 [m]	2010 [m]	2011 [m]	2008 [%][1)]	2009 [%][1)]	2010 [%][1)]	2011 [%][1)]	2008 [m]
AT	31.0	27.0	29.7	28.6	3.7	2.6	1.2	1.1	11.8	9.8	4.1	3.9	4.5
BE	54.0	45.5	49.5	39.9	7.4	8.2	3.3	5.5	13.6	18.0	6.7	13.7	9.3
BG	0.0	69.9	30.5	29.3	0.0	0.0	1.4	2.4		0.0	4.7	8.4	0.0
CY	5.3	5.4	4.4	4.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
CZ	78.6	70.7	70.4	68.0	16.9	10.2	3.8	6.0	21.6	14.5	5.5	8.8	0.8
DE	451.4	403.3	417.7	375.3	27.3	25.2	21.1	16.1	6.0	6.3	5.1	4.3	41.6
DK	29.6	24.4	22.5	17.6	2.1	1.7	2.2	2.2	7.2	6.9	9.6	12.3	2.1
EE	13.6	10.1	14.8	14.6	0.0	0.2	0.1	0.1	0.2	1.8	0.5	0.4	0.1
ES	142.4	132.0	106.0	105.1	9.2	17.2	12.6	12.9	6.5	13.0	11.9	12.3	5.6
FI	42.5	30.9	36.2	28.8	2.0	5.5	1.9	2.0	4.7	17.7	5.3	6.9	6.3
FR	118.4	106.9	110.5	76.5	12.6	11.0	9.6	12.4	10.7	10.3	8.7	16.2	4.2
GB	259.9	227.9	229.9	204.6	24.8	28.0	17.4	16.9	9.5	12.3	7.6	8.3	27.7
GR	69.7	63.5	56.3	44.3	0.4	2.9	1.8	5.5	0.6	4.5	3.2	12.5	0.0
HU	32.4	14.7	21.1	19.6	0.1	4.4	2.6	4.5	0.4	29.9	12.1	23.1	0.9
IE	19.7	17.0	16.2	13.8	0.5	1.1	2.4	2.5	2.6	6.3	14.9	17.8	1.0
IT	213.2	177.9	176.6	170.5	2.1	9.4	13.2	15.3	1.0	5.3	7.5	9.0	18.0
LI	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	55.7	194.0	0.0
LT	5.8	4.2	5.4	4.0	1.4	2.6	1.6	2.1	24.3	61.2	28.8	51.3	0.5
LU	2.0	2.2	2.1	1.8	0.0	0.1	0.1	0.1	1.0	2.4	3.4	6.5	0.0
LV	2.6	2.0	3.0	2.8	0.1	0.6	0.4	0.9	3.8	28.3	13.4	31.9	0.1
MT	3.9	1.9	0.0	1.9	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
NL	81.5	80.3	82.4	76.7	17.2	9.3	7.3	7.5	21.1	11.6	8.8	9.7	3.6
NO	19.1	18.7	16.0	16.9	0.3	6.9	3.8	3.7	1.7	37.1	23.7	21.8	12.8
PL	199.4	180.5	184.2	178.2	5.6	10.2	9.4	15.6	2.8	5.6	5.1	8.8	1.3
PT	27.9	26.7	22.6	22.0	3.7	3.7	3.1	4.2	13.2	14.0	13.6	19.3	1.1
RO	62.7	45.3	39.1	41.4	2.5	13.9	15.1	15.8	3.9	30.7	38.8	38.0	1.0
SE	19.5	17.1	21.8	18.3	2.6	2.5	2.3	1.9	13.1	14.9	10.7	10.3	2.5
SI	8.0	7.5	7.6	7.2	0.1	0.4	0.4	0.2	1.8	5.3	4.7	2.4	0.0
SK	23.2	20.4	17.3	21.1	3.0	5.8	5.5	5.7	12.8	28.3	31.9	26.8	0.4
EU ETS	2,017.3	1,833.8	1,793.9	1,633.7	145.6	183.5	143.6	162.8	7.2	10.0	8.0	10.0	145.6

Note: 1) Percent of surrendered allowances.

Source: EUTL; own calculations.

EUA Imports							EUA Net Exports							
2009 [m]	2010 [m]	2011 [m]	2008 [%][1)]	2009 [%][1)]	2010 [%][1)]	2011 [%][1)]	2008 [m]	2009 [m]	2010 [m]	2011 [m]	2008 [%][1)]	2009 [%][1)]	2010 [%][1)]	2011 [%][1)]
0.8	3.8	4.4	14.7	3.0	12.9	15.3	−0.9	1.8	−2.6	−3.2	−2.9	6.8	−8.8	−11.3
8.9	6.0	7.5	17.3	19.5	12.2	18.8	−2.0	−0.7	−2.7	−2.0	−3.7	−1.5	−5.5	−5.1
0.0	0.0	0.3		0.0	0.1	1.0	0.0	0.0	1.4	2.1		0.0	4.5	7.3
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
0.8	1.3	0.6	1.0	1.1	1.8	0.9	16.2	9.4	2.6	5.3	20.6	13.4	3.6	7.8
71.5	6.5	30.5	9.2	17.7	1.6	8.1	−14.3	−46.3	14.6	−14.3	−3.2	−11.5	3.5	−3.8
2.8	2.4	1.7	6.9	11.4	10.8	9.9	0.1	−1.1	−0.3	0.4	0.3	−4.5	−1.2	2.5
0.9	3.9	3.6	1.0	9.3	26.2	24.6	−0.1	−0.8	−3.8	−3.5	−0.8	−7.5	−25.7	−24.3
7.4	10.9	23.7	3.9	5.6	10.3	22.5	3.6	9.8	1.7	−10.8	2.5	7.4	1.6	−10.3
3.1	6.1	6.9	14.9	10.1	16.9	24.0	−4.3	2.3	−4.2	−4.9	−10.2	7.6	−11.5	−17.1
2.5	9.9	4.9	3.5	2.3	9.0	6.4	8.4	8.5	−0.3	7.5	7.1	7.9	−0.2	9.8
42.8	37.0	28.9	10.7	18.8	16.1	14.1	−3.0	−14.8	−19.6	−12.0	−1.1	−6.5	−8.5	−5.9
0.0	0.1	0.1	0.1	0.1	0.2	0.2	0.4	2.8	1.7	5.4	0.5	4.4	2.9	12.3
1.7	1.5	1.8	2.9	11.9	6.9	9.2	−0.8	2.7	1.1	2.7	−2.5	18.1	5.2	13.9
0.7	2.5	0.2	5.3	3.9	15.3	1.7	−0.5	0.4	−0.1	2.2	−2.7	2.4	−0.3	16.1
10.8	11.7	14.6	8.5	6.1	6.6	8.6	−16.0	−1.3	1.5	0.7	−7.5	−0.8	0.8	0.4
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	55.7	194.0
0.5	0.8	0.8	8.8	11.6	15.4	20.8	0.9	2.1	0.7	1.2	15.5	49.5	13.4	30.5
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.1	0.1	1.0	2.4	3.4	6.5
0.0	0.2	0.0	3.4	2.5	7.4	0.6	0.0	0.5	0.2	0.9	0.4	25.8	6.0	31.3
0.0	0.0	0.0	0.0	0.0		0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0
9.0	17.6	11.3	4.4	11.2	21.4	14.7	13.6	0.3	−10.4	−3.8	16.7	0.4	−12.6	−5.0
11.0	8.9	10.1	66.8	58.6	55.7	59.6	−12.5	−4.0	−5.1	−6.4	−65.1	−21.5	−31.9	−37.8
2.2	8.5	7.0	0.6	1.2	4.6	3.9	4.3	8.0	0.9	8.6	2.2	4.4	0.5	4.8
2.6	0.4	0.9	4.1	9.9	1.8	4.1	2.5	1.1	2.7	3.4	9.1	4.2	11.8	15.3
0.3	0.1	0.0	1.6	0.6	0.2	0.0	1.5	13.6	15.1	15.8	2.4	30.1	38.6	38.0
3.1	2.6	2.1	12.8	18.2	11.9	11.7	0.1	−0.6	−0.3	−0.3	0.3	−3.3	−1.2	−1.5
0.0	0.5	0.4	0.0	0.0	6.2	5.5	0.1	0.4	−0.1	−0.2	1.7	5.3	−1.6	−3.2
0.0	0.2	0.4	1.7	0.2	1.3	2.1	2.6	5.7	5.3	5.2	11.1	28.1	30.6	24.7
183.5	143.6	162.8	7.2	10.0	8.0	10.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

Table 4A.3 Surrendered units and CER and ERU imports by country and year in Phase 2

	Surrendered Units					Surrendered CERs					
	2008 [m]	2009 [m]	2010 [m]	2011 [m]	2012 [m]	2008 [m]	2009 [m]	2010 [m]	2011 [m]	2012 [m]	2008 [%][1)]
AT	32.1	27.4	30.9	30.6	29.6	1.1	0.4	1.2	2.0	4.5	3.3
BE	55.5	46.2	50.1	46.2	45.1	1.5	0.6	0.6	5.7	5.3	2.8
BG	0.0	69.9	33.5	40.0	35.3	0.0	0.0	2.3	6.1	1.4	
CY	5.6	5.4	5.0	4.6	2.9	0.3	0.0	0.7	0.0	0.0	5.4
CZ	80.4	73.8	75.6	74.2	70.2	1.8	3.0	4.5	3.1	7.5	2.3
DE	475.1	430.0	455.2	449.6	464.9	23.7	26.0	33.4	41.1	45.6	5.0
DK	30.6	24.5	23.2	20.5	19.5	0.9	0.1	0.7	0.6	2.8	3.0
EE	13.6	10.1	14.8	14.8	13.6	0.0	0.0	0.0	0.0	0.4	0.0
ES	160.1	140.2	121.7	132.6	140.0	17.7	8.2	12.2	20.6	25.1	11.1
FI	44.0	32.3	38.0	32.4	30.4	1.5	1.2	1.6	3.0	5.1	3.5
FR	124.1	111.1	115.6	104.0	113.7	5.8	3.9	4.4	24.2	18.9	4.6
GB	264.4	233.0	237.7	220.6	246.9	4.5	5.0	6.2	14.6	26.5	1.7
GR	69.9	63.7	59.9	54.5	66.6	0.2	0.1	3.7	7.5	5.1	0.3
HU	34.2	16.0	22.8	21.6	23.7	1.8	1.3	1.1	1.3	1.5	5.2
IE	20.4	17.2	17.4	15.8	26.2	0.7	0.2	0.7	1.1	2.2	3.5
IT	220.6	186.5	190.0	190.1	181.9	7.4	8.3	12.9	14.8	23.3	3.4
LI	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
LT	6.3	5.8	6.3	5.5	5.9	0.5	1.1	0.6	0.4	0.8	7.4
LU	2.1	2.2	2.3	2.1	3.6	0.1	0.0	0.2	0.2	0.5	4.1
LV	2.7	2.5	3.2	2.9	3.0	0.1	0.5	0.2	0.1	0.3	3.8
MT	3.9	1.9	0.0	1.9	1.2	0.0	0.0	0.0	0.0	0.0	0.0
NL	83.5	81.1	84.4	80.3	80.6	2.0	0.8	1.9	2.7	10.3	2.4
NO	19.3	19.2	19.3	19.1	20.5	0.2	0.3	3.2	1.9	1.4	1.1
PL	204.1	191.0	199.9	203.0	197.3	4.6	10.3	13.9	19.2	16.9	2.3
PT	29.9	28.2	24.2	25.0	26.7	2.0	1.5	1.3	2.9	2.5	6.6
RO	63.5	49.0	47.5	45.4	54.0	0.9	3.4	4.3	2.1	5.2	1.4
SE	20.1	17.5	22.6	19.9	22.5	0.6	0.4	0.8	1.6	4.6	2.9
SI	8.8	8.1	8.1	8.0	7.7	0.8	0.4	0.1	0.2	0.1	9.0
SK	25.3	21.6	21.7	22.2	21.0	2.1	1.2	4.4	1.0	1.0	8.3
EU ETS	2,100.3	1,915.3	1,931.0	1,887.3	1,954.6	82.9	78.3	117.0	177.8	218.7	3.9

Note: 1) Percent of surrendered allowances.

Source: EUTL; own calculations.

				Surrendered ERUs									
2009 [%][1)]	2010 [%][1)]	2011 [%][1)]	2012	2008 [m]	2009 [m]	2010 [m]	2011 [m]	2012 [m]	2008 [%][1)]	2009 [%][1)]	2010 [%][1)]	2011 [%][1)]	2012 [%][1)]
1.4	3.9	6.4	15.0	0.0	0.0	0.0	0.0	5.1	0.0	0.0	0.0	0.1	17.1
1.4	1.1	12.3	11.7	0.0	0.0	0.1	0.6	5.0	0.0	0.0	0.2	1.2	11.1
0.0	6.8	15.3	3.9	0.0	0.0	0.7	4.6	8.3		0.0	2.1	11.5	23.6
0.0	12.9	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.0
4.1	6.0	4.2	10.6	0.0	0.1	0.7	3.1	15.0	0.0	0.1	0.9	4.2	21.3
6.0	7.3	9.1	9.8	0.0	0.7	4.2	33.2	96.0	0.0	0.2	0.9	7.4	20.7
0.5	2.8	2.7	14.5	0.0	0.0	0.0	2.3	5.3	0.0	0.0	0.0	11.2	27.2
0.0	0.0	0.1	3.2	0.0	0.0	0.0	0.1	2.1	0.0	0.0	0.0	1.0	15.6
5.8	10.0	15.5	17.9	0.0	0.0	3.6	6.8	13.4	0.0	0.0	2.9	5.2	9.6
3.7	4.3	9.2	16.7	0.0	0.1	0.2	0.6	3.2	0.0	0.4	0.4	1.9	10.4
3.5	3.8	23.2	16.6	0.0	0.3	0.7	3.3	15.3	0.0	0.3	0.6	3.2	13.4
2.1	2.6	6.6	10.7	0.0	0.2	1.6	1.3	19.3	0.0	0.1	0.7	0.6	7.8
0.2	6.1	13.7	7.7	0.0	0.0	0.0	2.7	8.7	0.0	0.0	0.0	5.0	13.0
8.0	5.0	5.9	6.4	0.0	0.0	0.5	0.7	1.8	0.0	0.1	2.1	3.2	7.4
1.3	4.2	7.1	8.5	0.0	0.0	0.4	0.8	1.7	0.0	0.0	2.3	5.3	6.4
4.4	6.8	7.8	12.8	0.0	0.3	0.5	4.8	23.8	0.0	0.2	0.2	2.5	13.1
0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
18.7	10.3	6.4	13.6	0.0	0.5	0.2	1.1	1.7	0.0	8.0	2.9	20.	729.2
1.1	8.3	11.8	12.8	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.1
19.3	6.5	1.8	10.6	0.0	0.0	0.0	0.0	0.5	0.0	0.4	0.6	0.7	15.2
0.0		0.0	2.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0		0.0	0.0
0.9	2.3	3.4	12.8	0.0	0.0	0.1	0.8	10.6	0.0	0.0	0.1	1.0	13.2
1.7	16.7	9.8	7.0	0.0	0.2	0.1	0.4	1.5	0.0	1.1	0.7	2.0	7.3
5.4	7.0	9.4	8.5	0.0	0.3	1.8	5.6	23.1	0.0	0.1	0.9	2.8	11.7
5.4	5.3	11.6	9.2	0.0	0.0	0.3	0.1	4.4	0.0	0.0	1.3	0.4	16.3
6.9	9.1	4.6	9.6	0.0	0.3	4.2	1.9	10.0	0.0	0.7	8.7	4.1	18.6
2.5	3.5	8.0	20.6	0.0	0.0	0.0	0.0	2.7	0.0	0.0	0.0	0.1	11.9
4.6	1.5	2.1	0.9	0.0	0.2	0.4	0.6	3.5	0.0	2.1	4.7	7.9	45.8
5.7	20.1	4.6	4.8	0.0	0.0	0.0	0.1	0.2	0.0	0.0	0.1	0.5	0.9
4.1	6.1	9.4	11.2	0.0	3.2	20.1	75.8	282.2	0.0	0.2	1.0	4.0	14.4

Table 4A.4 Surrendered EUAs and EUA imports by sector and year in Phase 1

	Surrendered EUAs			EUA Imports					
	2005 [m]	2006 [m]	2007 [m]	2005 [m]	2006 [m]	2007 [m]	2005 [%][1)]	2006 [%][1)]	2007 [%][1)]
Aluminium	0.0	0.4	0.3	0.0	0.0	0.0	0.0	0.0	0.0
Cement and Lime	127.7	229.8	198.3	0.3	1.9	8.1	0.3	0.8	4.1
Ceramics	11.8	18.2	14.8	0.1	0.2	0.7	0.7	1.0	4.8
Chemicals	3.9	5.5	5.6	0.0	0.0	0.0	0.0	0.0	0.1
Coke	12.9	15.0	15.0	0.0	0.0	0.3	0.3	0.2	2.1
Combustion	1,242.8	1,708.3	1,571.9	33.0	80.6	220.1	2.7	4.7	14.0
Glass	14.9	24.5	20.1	0.0	0.2	0.9	0.2	0.8	4.6
Iron and Steel	96.3	143.6	123.0	0.1	0.5	3.1	0.1	0.4	2.6
Opt-in	0.2	0.2	0.3	0.0	0.0	0.0	0.1	3.8	16.2
Other metals	5.9	7.0	7.7	0.0	0.0	0.0	0.0	0.0	0.0
Other minerals	0.3	0.4	0.5	0.0	0.0	0.1	0.0	4.0	22.7
Pulp and Paper	22.8	39.1	31.1	0.3	0.3	1.4	1.1	0.8	4.6
Refineries	105.6	182.4	151.7	0.8	1.3	5.4	0.7	0.7	3.6
EU ETS	1,645.2	2,374.4	2,140.3	34.6	85.1	240.3	2.1	3.6	11.2

Note: 1) Percent of surrendered allowances.

Source: EUTL; own calculations.

Table 4A.5 Surrendered units and EUA imports by sector and year in Phase 2 (2008–2011)

	Surrendered Units				Surrendered EUAs				EUA Imports							
	2008 [m]	2009 [m]	2010 [m]	2011 [m]	2008 [m]	2009 [m]	2010 [m]	2011 [m]	2008 [m]	2009 [m]	2010 [m]	2011 [m]	2008 [%][1)]	2009 [%][1)]	2010 [%][1)]	2011 [%][1)]
Aluminium	0.3	0.2	0.2	0.2	0.3	0.2	0.1	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Cement and Lime	189.1	157.4	155.3	153.6	180.8	150.2	145.2	119.4	3.5	5.6	3.2	3.1	1.9	3.6	2.1	2.0
Ceramics	13.6	9.6	9.2	9.3	13.0	8.6	7.7	2.0	0.2	0.0	0.1	0.2	1.6	0.4	0.8	2.3
Chemicals	16.8	15.7	16.6	16.6	15.6	15.4	15.2	14.9	0.1	0.1	0.6	0.5	0.8	0.7	3.4	2.7
Coke	14.6	10.9	13.1	14.0	13.8	10.7	12.7	13.2	1.0	0.5	0.9	0.7	6.6	4.5	6.7	5.1
Combustion	1,533.7	1,397.0	1,420.2	1,378.0	1,478.7	1,336.5	1,318.8	1,234.3	131.2	167.5	129.5	147.6	8.6	12.0	9.1	10.7
Glass	21.1	20.0	19.6	20.0	20.2	19.7	18.1	16.3	0.2	0.4	0.3	0.2	0.8	2.1	1.6	1.2
Iron and Steel	125.9	89.9	107.0	103.6	117.0	84.7	99.9	75.9	3.0	2.9	0.1	4.7	2.4	3.3	0.1	4.5
Opt-in	1.8	40.6	20.1	24.3	1.7	40.5	18.0	19.3	0.0	0.1	0.1	0.1	2.4	0.2	0.5	0.4
Other metals	3.2	2.1	2.8	3.0	3.1	2.1	2.5	2.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Other minerals	1.2	1.1	1.1	1.1	1.2	1.1	1.1	1.0	0.0	0.1	0.0	0.1	2.6	4.8	3.3	8.6
Pulp and Paper	32.6	29.2	30.8	29.5	30.0	27.1	27.7	15.0	0.3	0.4	0.5	0.5	0.9	1.3	1.5	1.7
Refineries	146.4	141.6	135.0	134.0	142.0	137.5	128.2	124.4	6.2	5.9	8.3	5.1	4.2	4.2	6.2	3.8
EU ETS	2,100.2	1,915.3	1,931.0	1,887.3	2,017.3	1,834.2	1,795.2	1,638.7	145.6	183.5	143.6	162.8	6.9	9.6	7.4	8.6

Note: 1) Percent of surrendered allowances.

Source: EUTL; own calculations.

Table 4A.6 Surrendered units and CER and ERU imports by sector and year in Phase 2

	Surrendered Units					Surrendered CERs				
	2008 [m]	2009 [m]	2010 [m]	2011 [m]	2012 [m]	2008 [m]	2009 [m]	2010 [m]	2011 [m]	2012 [m]
Aluminium	0,3	0,2	0,2	0,2	0,3	0,0	0,0	0,1	0,0	0,0
Cement and Lime	189,1	157,4	155,3	153,6	141,1	8,3	7,1	8,7	28,0	18,3
Ceramics	13,6	9,6	9,2	9,3	8,1	0,6	0,9	1,0	1,0	1,1
Chemicals	16,8	15,7	16,6	16,6	16,4	1,2	0,3	1,2	1,6	2,6
Coke	14,6	10,9	13,1	14,0	11,5	0,8	0,2	0,3	0,7	1,1
Combustion	1.533,7	1.397,0	1.420,2	1.378,0	1.391,6	54,9	58,4	88,5	105,7	153,8
Glass	21,1	20,0	19,6	20,0	19,1	0,9	0,3	1,5	3,1	4,3
Iron and Steel	125,9	89,9	107,0	103,6	100,2	8,9	5,0	6,0	24,2	16,5
Opt-in	1,8	40,6	20,1	24,3	20,0	0,0	0,1	1,8	1,2	0,2
Other metals	3,2	2,1	2,8	3,0	3,0	0,0	0,1	0,3	0,3	0,4
Other minerals	1,2	1,1	1,1	1,1	1,1	0,0	0,0	0,0	0,1	0,1
Pulp and Paper	32,6	29,2	30,8	29,5	28,5	2,6	2,1	2,7	5,3	4,2
Refineries	146,4	141,6	135,0	134,0	130,7	4,5	3,8	5,0	6,6	10,5
EU ETS	2.100,2	1.915,3	1.931,0	1.887,3	1.871,7	82,9	78,3	117,0	177,8	213,2

Note: 1) Percent of surrendered allowances.

Source: EUTL; own calculations.

					Surrendered ERUs									
2008 [%][1)]	2009 [%][1)]	2010 [%][1)]	2011 [%][1)]	2012 [%][1)]	2008 [m]	2009 [m]	2010 [m]	2011 [m]	2012 [m]	2008 [%][1)]	2009 [%][1)]	2010 [%][1)]	2011 [%][1)]	2012 [%][1)]
0,6	0,9	39,2	0,8	0,5	0,0	0,0	0,0	0,0	0,1	0,0	0,0	1,8	6,2	25,1
4,4	4,5	5,6	18,2	13,0	0,0	0,1	1,3	6,2	32,7	0,0	0,1	0,9	4,1	23,1
4,4	9,9	10,9	11,2	13,4	0,0	0,0	0,5	6,3	1,7	0,0	0,2	5,5	67,2	20,5
7,0	1,8	7,3	9,6	15,6	0,0	0,0	0,2	0,1	3,1	0,0	0,0	1,1	0,8	18,9
5,4	1,4	2,3	5,1	9,6	0,0	0,1	0,1	0,0	2,6	0,0	0,5	1,1	0,3	22,8
3,6	4,2	6,2	7,7	11,1	0,0	2,1	12,9	38,0	164,8	0,0	0,2	0,9	2,8	11,8
4,4	1,5	7,6	15,5	22,5	0,0	0,0	0,0	0,6	2,4	0,0	0,0	0,2	3,2	12,6
7,1	5,5	5,6	23,4	16,4	0,0	0,3	1,1	3,4	31,4	0,0	0,3	1,0	3,3	31,4
1,0	0,1	8,7	5,0	1,2	0,0	0,0	0,4	3,8	6,3	0,0	0,1	1,8	15,7	31,8
1,4	3,7	9,5	10,1	12,7	0,0	0,0	0,0	0,0	0,6	0,0	0,0	0,8	0,0	19,9
3,1	1,3	3,4	7,3	10,4	0,0	0,0	0,0	0,0	0,4	0,0	0,0	0,0	0,0	32,6
8,0	7,2	8,8	17,9	14,9	0,0	0,0	0,4	9,2	2,4	0,0	0,0	1,3	31,3	8,4
3,0	2,7	3,7	4,9	8,1	0,0	0,3	1,8	3,0	22,2	0,0	0,2	1,3	2,2	17,0
3,9	4,1	6,1	9,4	11,4	0,0	2,9	18,8	70,8	270,7	0,0	0,1	1,0	3,8	14,5

PART II

IMPLEMENTATION PROBLEMS

5. Emissions trading and market manipulation

Beat Hintermann

1. INTRODUCTION

An emissions trading scheme (ETS) is an artificially created market that transforms the right to pollute the environment from a production factor in unlimited supply into a scarce resource. After the establishment of an ETS, emission allowances become a necessary production input for firms covered by the scheme, and just like any other input, their price affects the price of the final products. For example, if the price for emission allowances increases, the prices for steel or electricity can be expected to increase as well, unless such price adjustments are not possible due to regulations.

Because of the significant transaction costs associated with monitoring, reporting and verifying emissions (Jaraitė-Kažukauskė and Kažukauskas, 2015), an ETS usually covers large firms only. Such firms tend to be part of sectors that have a history of imperfect competition, such as the electricity sector. For this reason, it is natural that concerns about market power arise in the context of an ETS. Indeed, the literature on market power in emission permit markets, also referred to as allowance markets, dates back almost as far as the literature on emissions trading itself (for example, Sinn and Schmoltzi, 1981; Hahn, 1984; Misiolek and Elder, 1989).

This chapter does not consider the strategic behavior of countries or regions in the process of negotiating an international treaty, such as the Kyoto Protocol. Rather, the chapter focuses on market power exercised by individual firms *within* an existing ETS and reviews the economics literature on this topic.

The predominant share of this literature is theoretical. Perhaps the most fundamental result from the literature on market power is that the method used for allowance allocation matters. This contrasts with the traditional presumption that the market-clearing price for emission allowances and the distribution of abatement across the firms covered by the ETS are independent of the initial distribution of allowances (Hahn and Stavins, 2011). The key to this result is that price-taking firms equate their marginal abatement costs to the (single) allowance price, which equalizes marginal abatement costs across all firms within the ETS. This leads to an efficient outcome in the sense that emissions are reduced at the least economic cost. However, if a firm (or group of firms) wields market power, equating the marginal abatement cost to the allowance price is not the profit-maximizing strategy. Rather, the firm will set the price either above or below marginal abatement costs, and thus under- or over-abate relative to the fully competitive outcome. The market power literature shows that the extent and nature of price manipulation depends on the initial allowance allocation (Hahn, 1984; Misiolek and Elder, 1989).

The divergence between marginal abatement costs and the allowance price creates inefficiency and results in a welfare loss. The underlying reason is that abatement does not happen where it is cheapest, but where it maximizes firm profits, and this is generally not the same under imperfect competition. As a result, society pays more to reduce emissions to a given emissions cap than it would in the absence of ETS market manipulation. The distribution of the economic burden of the excess cost depends on the extent to which firms are able to pass on their costs to consumers, employees and suppliers. Because the determination of an emission cap in the political process depends, among other things, on the likely costs of achieving this cap, an increase in compliance costs may also lead to a less ambitious cap in the long run, and in that sense to a reduction in environmental quality.

While the theory about imperfect competition in emission permit markets has received a fair amount of attention, the same cannot be said for the empirical research on this topic. There are several reasons for this. First, the theoretical literature has focused for many years on the special case of no interaction between markets for emissions and products. This makes the theory more straightforward, but also leads to misguided implications in terms of the direction of price manipulation if product prices are in fact dependent on allowance prices (this issue is discussed in more detail below). Second, there are several data limitations. While data about firms' abatement costs or production at installation level is considered proprietary information and is thus not made public, other information is held back for several years, such as allowance sales and purchases under the EU ETS. Furthermore, much of the publicly available data only exists in a format that requires a significant amount of data processing in order to make it suitable for empirical analyses. And even if these data limitations are overcome, it is virtually impossible to conclusively prove the exercise of market power, as the same pattern of emissions, output and allowance purchase decisions could be due to other factors, such as unobserved expectations or limited rationality. Nevertheless, some attempts at assessing the presence of market power have been made (for example, Ellerman and Montero, 2007; Hintermann, 2015).

This chapter is organized as follows. Section 2 reviews some of the main theoretical results pertaining to market power in emission permit markets. Section 3 discusses the studies that have addressed this issue from an empirical perspective, and section 4 offers some concluding remarks.

2. THE THEORY OF MARKET POWER IN EMISSION MARKETS

This section discusses the two main strands of the theoretical literature to date. First, the earlier models that are based on compliance cost minimization are reviewed, and then the context is analyzed where firms maximize their profits by jointly manipulating the allowance and product markets.

2.1 Models Based on Compliance Cost Minimization

The earliest papers about market power in allowance markets are by Sinn and Schmoltzi (1981) and Hahn (1984). These two papers essentially arrive at the same result, but the paper by Robert Hahn is usually cited as the seminal work. The problem considered is that of compliance cost minimization by a dominant firm that faces a competitive fringe.[1] The main result can be summarized in the following expression (for a derivation, please refer to the Appendix):

$$-C_e = \sigma + (x - a)\frac{\partial \sigma}{\partial x} \tag{5.1}$$

The left-hand side refers to the firm's marginal abatement cost, σ to the allowance price, x to the firm's allowance holdings, and a to the firm's free allowance allocation.

Equation (5.1) states that the dominant firm equates its marginal abatement cost to the allowance price plus an adjustment that depends on (a) whether the firm is a net buyer or seller of allowances, and (b) on its ability to influence the allowance price. A dominant net allowance seller will inflate the price by setting its marginal abatement costs below the allowance price (that is, by under-abating and thus increasing the demand for allowances relative to the competitive outcome), and the opposite applies for a net allowance buyer. The efficient solution can be obtained

[1] The term 'competitive fringe' is a term from the literature on industrial economics and refers to a group of smaller firms which take prices as given, in contrast to the dominant firm which considers the effect of its decisions on prices.

by allocating the dominant firm exactly as many allowances for free as it would demand if it were a price taker. Note that if the firm acts as a price taker, the outcome is efficient regardless of the amount of free allocation.

Liski and Montero (2005; 2011) extend this finding to a dynamic context and show that a dominant seller will drive up the allowance price by restricting supply and delaying the moment in time when it exhausts its allowance stock, and vice versa for a net buyer. If banking allowances for use in later years is allowed but borrowing from the future is not, as is commonly the case in an ETS, the effect of market power on the resulting allowance price is much stronger if the dominant firm is a net seller, because it can credibly withhold allowances and thereby drive up the allowance price. For a net buyer, the ability to manipulate the allowance price downwards is much more limited due to a commitment problem: In the absence of the right to borrow, the fringe firms know that the dominant firm eventually has to increase its allowance demand, or else engage in costly abatement once it has exhausted its stock of banked allowances in order to reach compliance. Based on this insight, Liski and Montero (2011) conclude that market power in allowance markets is mainly a problem if the dominant firm is a net seller. As will be discussed in section 3, the largest firms in the US Acid Rain Program and in the EU ETS actually turn out to be net buyers, which suggests that market power is of limited empirical concern in those markets. However, as will be shown below, this conclusion only holds if firms are not able to pass on their costs to consumers.

Westskog (1996) extends the model to two dominant firms playing a Cournot game, and Malueg and Yates (2009) drop the competitive fringe and consider the situation where the market consists of a group of strategic firms (that is, a setting where all firms have some market power). These extensions change the equilibrium quantities and prices, but the main qualitative finding remains: a net allowance seller will manipulate the allowance price upwards, whereas a net buyer tries to depress the price.

2.2 Joint Maximization of Profits in Allowance and Product Markets

The theory discussed so far focuses on the allowance market alone. This reduces the problem to one that is easier to solve, but which is inconsistent with the empirical finding that carbon costs are passed on to consumers via product prices.[2] Even in cases where prices are regulated, carbon costs

[2] For empirical estimates of cost pass-through in the EU ETS, see Sijm et al. (2008), Zachmann and von Hirschhausen (2008), Lo Prete and Norman (2013), Fabra and Reguant (2014), Fell et al. (2015) and Hintermann (2016).

have to be reflected in the product prices in the long run, because otherwise firms would eventually exit the market. Allowing for cost pass-through and letting the dominant firm maximize its profits in both the allowance the product market leads to the following result (derivation in the Appendix):

A dominant firm will over-purchase allowances and thereby inflate the allowance price if it receives more free allowances than the threshold defined by

$$a^0 = x - \frac{\partial p / \partial x}{\partial \sigma / \partial x} q, \tag{5.2}$$

where p refers to the product price and q is the output quantity; the other variables are as described above. The opposite is true if the free allocation is below (5.2). An allocation that is exactly equal to (5.2) induces the firm to act as a price taker in the allowance market. Importantly, the threshold in (5.2) is below the firm's efficient demand for allowances, such that even net allowance buyers may find it optimal to inflate the allowance price.

The intuition behind this result is as follows. Manipulating the allowance price upwards creates an increase in the firm's compliance costs (it has to purchase more allowances, and at a higher price), but at the same time it increases its revenues in the product market (because the product price reflects the allowance costs associated with production). At the threshold, the two effects exactly offset each other such that the firm has no interest in manipulating the allowance price. If, however, the firm receives a free allocation that exceeds the threshold, the revenue effect exceeds the compliance cost effect, which in turn creates an incentive for the firm to increase the allowance price by over-purchasing allowances and under-abating emissions, relative to the efficient solution. Lastly, note that firms for which (5.2) is negative will have an incentive to increase the allowance price even if they receive no free allocation at all.[3]

The assumption of market power in the allowance market, but not in the product market, is difficult to justify if the allowance market includes several different sectors, because in this case the product markets are a

[3] To provide some intuition for this result, suppose that the product price contains the full marginal carbon cost at any given time. If a firm's average carbon intensity is below the average intensity of the marginal production unit, then the price increase in the product market due to an increase in the allowance price exceeds the compliance cost increase, such that the firm profits from a higher allowance price even if it has to purchase all allowances on the market. Importantly, this applies to electricity firms that generate a portion of their output via nuclear power. This lowers their average carbon intensity to below that of the average marginal unit, which is usually fossil-based.

subset of the allowance market. Allowing for price-setting power in both the allowance and product market, Misiolek and Elder (1989), Disegni Eshel (2005) and Hintermann (2011) show that a dominant firm will manipulate both markets jointly in order to maximize its profits. These findings are closely related to the literature on 'raising rivals' costs', the principal result of which is that firms that are dominant in both markets can find it optimal to increase the input price if they are affected less by the cost increase than their competitors.[4]

Allowing for market power in the product market leads to different equilibrium levels of output, abatement and emissions, but the threshold of allocation beyond which a dominant firm profits from an allowance price increase remains determined by (5.2). Being able to manipulate both markets rather than just the allowance market is therefore not a necessary condition for a net allowance buyer to have an incentive to manipulate the allowance price upwards (although having price-setting power in both markets may increase the extent of the resulting price distortion). Similarly, it is not necessary that the dominant firm engages in 'exclusionary manipulation', a term used by Misiolek and Elder (1989). This rather extreme form of market power, in which the dominant firm excludes competitors by hoarding allowances, would arguably be difficult to maintain in a real-world ETS, because the largest emitters are usually net allowance buyers, whereas smaller firms tend to be net sellers and can therefore not be excluded from the allowance market. However, rivals need not be excluded in order for market power to be of regulatory concern. A dominant firm can profit from increasing the allowance price even if rivals remain in the market (de Feo et al. 2012; 2013).

The effect of including the product market in allowance price manipulation in a dynamic context is not clear, as no such model has been published in the literature to date. However, given a sufficiently generous free allocation and a possibility for cost pass-through, the result should mirror the findings in the static case: even if a dominant firm is a buyer in the allowance market, the profits generated in the product market from increasing the allowance price may more than outweigh the increase in compliance costs. This would then lead to a situation where a net allowance buyer (defined over the time period during which banking takes place) increases its allowance demand beyond the efficient level, thus moving the time point

[4] The seminal article is by Salop and Scheffman (1983). Sartzetakis (1997a; 1997b) applied this theory to emission permit markets. De Feo et al. (2012; 2013) show in the context of upstream-downstream strategic competition that allowance prices generally exceed marginal abatement costs.

when the banked allowances are exhausted forward rather than backward in time.

3. EMPIRICS OF MARKET POWER

This section reviews the literature that analyzes the issue of market power in allowance markets from an empirical perspective.[5] First, laboratory evidence will be given and then evidence will be reviewed from the US Acid Rain Program, RECLAIM in California, and the EU ETS. We are not aware of empirical investigations in other emission markets.[6]

3.1 Laboratory Experiments

A series of laboratory experiments has examined the effect of various features in emission permit markets, including that of market power. Ledyard and Szakaly-Moore (1994) study the effect of market power using different types of auctions and find that when using a double auction format (in which trades happen sequentially and clear at different prices, as is the case for actual allowance trading on exchanges), the effect of market power on prices and welfare is small. In contrast, monopolists were found to capture almost all monopoly rents when the experiment used a sealed-bid auction format. Similarly, Carlén (2003) and Cason et al. (2003) report that a double auction format yields an efficient result despite the presence of market power. Muller et al. (2002) find evidence for persistent market manipulation in the double auction, but again the welfare loss due to imperfect competition is small. In these papers, market power was examined strictly in the context of compliance cost minimization, while ignoring any price changes in the product market.

Brown-Kruse et al. (1995) find that welfare is reduced significantly in treatments where firms/subjects are able to manipulate prices in both the allowance and product market. The resulting welfare was below command-and-control levels, implying that market power can more than offset the gains from instituting an ETS in the first place. In a series of follow-up experiments, Godby (2000; 2002) reports that price manipulation emerges

[5] For an earlier review of empirical investigations of allowance market manipulation, see Montero (2009).

[6] Besides the EU ETS, the US Acid Rain Program and RECLAIM, emission markets currently exist in some of the eastern States of the USA (e.g., RGGI) and in California, as well as in Canada, China, Kazakhstan, New Zealand, South Korea, Japan and Switzerland.

and persists, and that welfare is significantly reduced if firms/subjects are able to manipulate both markets. Dormady (2014) also uses an experimental setting where firms make decisions on both the allowance and product markets and finds that in treatments where firms hold market power, the auction price for allowances is lower than in treatments where all firms are price takers. This result is driven by dominant firms' restricting the output in the product market, which reduces overall emissions and thus the demand for allowances.[7]

Bohm (2003) reviews the experimental literature and concludes that market power in contexts involving 'single' market manipulation does not lead to significant distortions as long as a double auction mechanism is used, whereas in experiments where firms manipulate both markets, the effect of market power is much stronger. Bohm further concludes that exclusionary market power is unlikely to be empirically relevant because firms will find it difficult to prevent competitors from obtaining allowances, or to control both the upstream and the downstream market in the long run. To our knowledge, no experiment has used the setting where firms perceive direct market power in the allowance market alone, but cost pass-through occurs such that the product price is a function of the allowance price.

Note also that with cost pass-through and free allowance allocation, the rivals of the dominant firm (be they price takers or themselves oligopolists) profit from a higher allowance price along with the dominant firm, at the expense of consumers and taxpayers. This means that they have no reason to collude against the dominant firm's price-setting policy, which has been suggested as the underlying mechanism of the efficiency result generated by the double auction (Smith, 1981).

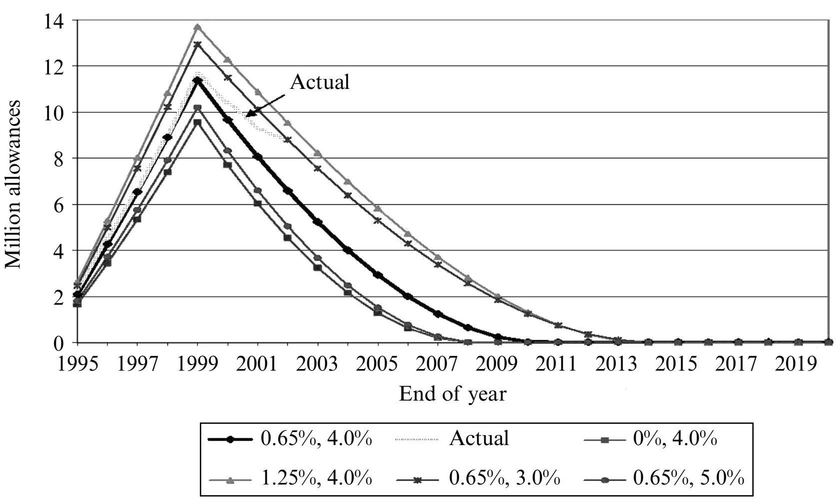
3.2 The US Acid Rain Program

Ellerman and Montero (2007) examine the banking behavior of firms covered by the US Acid Rain Program, which regulates sulfur dioxide emissions from power generators in the United States. Firms accumulated a large stock of emission allowances during Phase I of the program

[7] Dormady (2014) interprets this as empirical evidence against the theory laid out by Misiolek and Elder (1989), Disegni Eshel (2005) and Hintermann (2011). Theory predicts, however, that the dominant firm will set its marginal abatement cost below the allowance price and thus inflate it *conditional* on the restricted output in the product market. In order to test the theory laid out in Section 2.2, the treatment would have to be that firms perceive market power in both markets, relative to the control where firms are able to influence the product price alone.

(1995–1999), which they started to deplete once the scope of the program was increased and the cap tightened relative to unrestricted emissions in Phase II, which started in 2000. Some commentators interpreted the fact that firms banked allowances rather than emitting more as evidence of market manipulation, whereas others believed the amount of banking to be evidence for an over-estimate of initial business-as-usual (BAU) emissions in combination with irreversible abatement decisions. Ellerman and Montero (2007) derive a theoretical model of efficient banking and calibrate it using data from the US Acid Rain Program. They show that adjustments in expectations about initial BAU emissions only have a minor impact on the amount of allowances banked during the first year, since the optimal banking path is determined mainly based on expectations about future emissions and (relative) abatement costs. Based on their calibrated model, an over-estimate of initial BAU emissions by 20 percent would increase the amount of banked allowances in 1995 only marginally, from 2.09 to 2.11 million.

Using a range of values for firms' projected discount rates and expectations about the growth rate of future BAU emissions, Ellerman and Montero (2007) compute various efficient time paths for firms' allowance holdings during the program, under the assumption of profit maximization under perfect competition. These paths are shown in Figure 5.1. The



Source: Ellerman and Montero (2007).

Figure 5.1 Actual and computed allowance banking paths in the US Acid Rain Program solid line in the center corresponds to the path based on the central estimate with a 0.65 percent emissions growth rate and a 4 percent discount rate, whereas the other lines correspond to different parameter values. The central estimate is very close to the actually observed path during Phase I. The authors interpret this as indirect evidence that market power was not exercised.

Firms' banking behavior in Phase II is consistent with a situation where firms lower their discount rate. The real discount rate decreased, as presumed, in the period 2000–2003, but not enough to justify the less-than-expected depletion of the banked allowances. As an alternative explanation, the authors propose that firms may have revised their expectations about future BAU emissions upwards. Another possible explanation, which was not suggested by the authors, would be that firms engaged in over-abatement, thus depleting their reserve of accumulated allowances more slowly in order to increase the allowance price, which would be optimal if they could pass the costs on to consumers. However, there is no way to test one hypothesis against the other, and no evidence for or against cost pass-through is given. Based on the data, Ellerman and Montero (2007) conclude that there is no evidence to indicate price manipulation by the participating firms.

Liski and Montero (2011) use data on emissions and emission allowance allocation to the largest four electricity firms that, combined, represent 43 percent of total allocation during Phase I. The authors show that all of these firms were net buyers. According to their theoretical model of dynamic compliance cost minimization, this implies that (i) firms would use their market power, if any, to suppress demand (and thus prices), which delays the moment when the allowances banked during the Phase I are used, and (ii) that market power should be of limited concern because a net buyer cannot significantly decrease prices due to the commitment problem discussed above. Therefore, the finding that the largest firms act as net allowance buyers is interpreted as an indication that market power is not likely to be a relevant concern in this market.

However, given the large amount of free allowances, even a limited ability to pass allowance costs on to electricity prices could have incentivized these firms to increase the allowance price as discussed in section 2.2, rather than depressing it. Liski and Montero do not provide any evidence in favor or against cost pass-through. During the time period under consideration, some of the regional electricity markets in the United States were in the process of being liberalized, and thus it is not clear to what extent input costs affect electricity prices.

As a second indication against market manipulation, Liski and Montero (2011) cite allowance sales conducted by two of the largest firms. A net buying firm that aims to minimize compliance costs and thus to depress the allowance price would never sell allowances in the market. Likewise, a net buyer that aims to drive up the allowance price in order to maximize overall profits in the presence of cost pass-through would also not be expected to sell allowances, because this would put downward pressure on the allowance price. Note, however, that Liski and Montero do not have access to complete firm-level allowance transactions, such that the two instances of reported allowance sales may be offset by even larger allowance purchases at different points in time as part of firms' hedging strategies.[8]

3.3 The RECLAIM program

Holland and Moore (2012) examine the nitrous oxide market within California's Regional Clean Air Incentives Market (RECLAIM).[9] This market differs from other ETS's due to limits imposed on banking (an allowance gives the holder the right to emit one metric ton of NOx within a 12-month period but not thereafter), repeated adjustment to the total amount of available allowances, and quarterly rather than annual emissions accounting. Taking these features into consideration, the authors derive a set of equilibrium implications, some of which are testable. For example, firms trade allowances across time and sell any allowances they cannot use themselves before they expire. Holland and Moore find that most firms either used or sold all allowances before their expiration. However, some firms allowed a significant number of allowances to expire unused, especially during the period after California's electricity crisis in 2000–2001.[10]

Although the authors do not offer an explanation for this apparent deviation from efficiency, excess allowance holdings could be a sign of firms trying to manipulate the allowance price upwards. However, this makes sense only to the extent that firms are able to pass on their carbon

[8] In the EU ETS, large firms, which were net buyers in aggregate, were active both as sellers and buyers in the market (Hintermann, 2015). For example, a firm may take a short position in a futures contract at one point in time, but later decide to adjust its allowance position downwards in response to changing prices.

[9] The RECLAIM program also covers SO_2 emissions. Because this separate market is very thin, Holland and Moore (2012) restrict their analysis to the NOx part of the program.

[10] During the electricity crisis in California there was a shortage of electricity supply caused by a mixture of market manipulation, shut-downs of pipelines and price regulation. During this period, California suffered from repeated blackouts.

costs in the product market. Since the California electricity market was regulated during a part of the observed time frame, cost pass-through would have had to take place via regulation rather than via market forces. Kolstad and Wolak (2013) investigate the allowance purchase and holding decisions of electricity firms that supplied some or all of their output to the California electric market during the crisis years. They find that these firms paid higher prices for NOx allowances than other firms during these years (but not during previous years), and allowed a significant number of allowances to expire unused. Furthermore, the electric output of installations owned by these firms exceeded theoretical predictions based on the least-cost dispatch model developed by Borenstein et al. (2002), suggesting that firms did not fully price in the cost of the NOx allowances used for electricity generation.

The argument put forward by Kolstad and Wolak (2013) is that electricity suppliers paid excessively high allowance prices in order to 'cost-justify' their bids to join the California electricity market.[11] This particular form of price manipulation leads to price differentiation between market participants, which requires collusion between brokers and firms that seems difficult to sustain in the long run. In addition to paying excessively high prices, firms also bought more allowances than needed to cover their emissions, presumably with the aim of increasing the overall allowance price. For a firm that holds a surplus of allowances due to strategic over-purchasing, the opportunity cost of these surplus allowances is zero, which would explain the failure to include allowance costs in the supply bids. This is very much in the spirit of the theory discussed in section 2.2: although the strategy of purchasing allowances at an excessively high price and letting them expire unused is obviously inconsistent with a policy of minimizing the costs of compliance, this may have been a profitable strategy if firms were more than compensated for these costs in the product market.

3.4 The EU ETS

Figure 5.2 presents the EU allowance price path for the first two market phases. Note that allowances from Phase I (2005–2007) could not be banked in Phase II (2008–2012), such that the allowance prices in those

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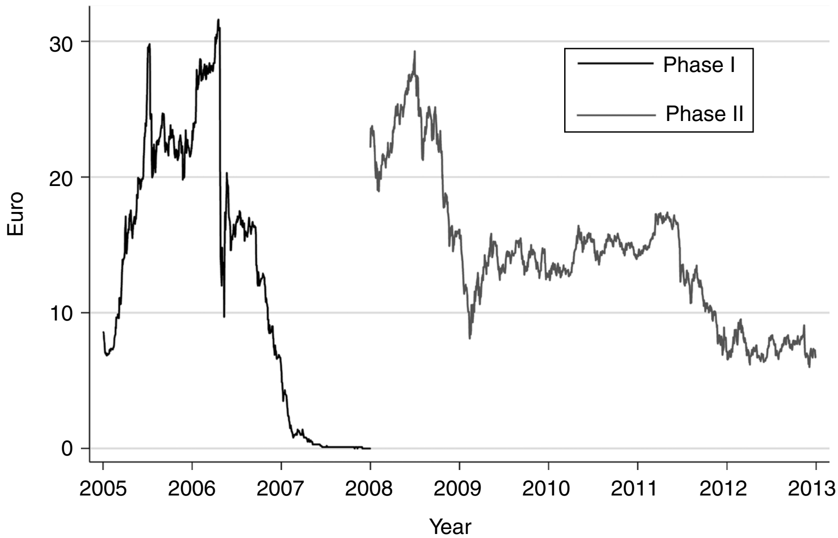
[11] Since electricity prices in California are regulated, firms need to justify price increases with corresponding cost increases. However, because the price then applies to the entire output and not just the output supplied by the marginal generating unit that experiences the cost increase, a price increase may lead to a substantial increase in economic rents.



Source: Thomson Reuters Datastream.

Figure 5.2 EU Allowance prices during Phases I–II of the EU ETS

phases are not linked. The largest firms in this market are electricity suppliers, most of which have been net buyers since the inception of the market and would have manipulated the price downwards (according to the results based on compliance cost minimization). However, inspection of the price paths and the fact that both phases were characterized by an over-allocation/excess supply of allowances suggests that prices were too high, rather than too low (at least with the benefit of hindsight), especially in the beginning of each phase.

To our knowledge, there are only two papers that analyze the issue of market power in the EU ETS from an empirical perspective, which may be partly explained by data availability.[12] Hintermann (2015) examines emissions, allowance allocation and electric output by the largest ten electricity

[12] In the EU ETS, all data is published on the installation level. Since economic decisions are usually taken on the firm level, this requires that the data are aggregated on the firm level and that additional information is added, e.g. from business reports, before conducting the analysis. For Phase I, aggregate data has been generated and made available by Jaraite et al. (2013), but no similar dataset exists for later phases. Furthermore, firm-level output is not always available.

firms (in terms of fossil electric generation) during Phase I of the EU ETS[13] and concludes that all of these firms received a free allocation well in excess of the threshold defined by equation (5.2), which makes their profits an increasing function of the allowance price. This is confirmed by empirical investigations by Oberndorfer (2009) and Bushnell et al. (2013), who find a positive correlation between the allowance price and firms' stock prices. The underlying reason is the generous provision of free allocation, but also because firms were able to pass on most or all of their costs to consumers.[14] This does not prove, however, that firms (i) had market power and (ii) used this power to increase the allowance price. Although the EU ETS is in fact fairly concentrated, in the sense that 1 percent of firms are responsible for close to 60 percent of total emissions, market concentration as measured by the Herfindahl-Hirschmann index is rather low (Hintermann, 2015).

Nevertheless, large firms may have been able to manipulate the market for two reasons. First, many small firms abstained from the market in the beginning, for instance due to transaction costs, and delayed activation of some national registries, such that the 'market-relevant' cap was much smaller than the total number of distributed allowances.[15] Second, since all large electricity firms profited from a high allowance price, their interests were aligned. This would make collusion a profitable strategy, albeit one that would be difficult to enforce given the impossibility of observing trades in real time.

Figure 5.3 displays allowance holdings over time of the largest five firms (in terms of emissions), relative to their total emissions during Phase I.[16] With the exception of Suez-Electrabel, all firms held more allowances than they needed to cover their entire emissions not only at the very end, but during a significant part of Phase I. Hintermann (2015) further shows that

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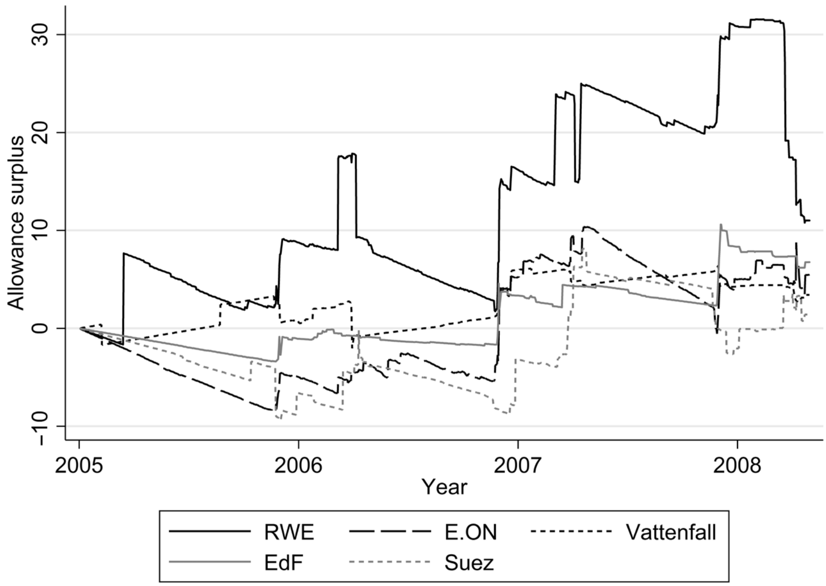
[13] The focus is on Phase I, because transfer information is published by the EU with a 5-year delay. Note that due to the no-banking provision, holding a surplus of allowances at the end of Phase I cannot be explained by price speculation, as the allowances lost their value in the beginning of 2008.

[14] See e.g., Zachmann and von Hirschhausen (2008), Lo Prete and Norman (2013), Fabra and Reguant (2014), Hintermann (2014) and Fell et al. (2015) for estimates of allowance cost pass-through to electricity prices.

[15] The sum of the allowances that were allocated to installations that never engaged in any trade was 50% by the end of the first compliance year (April 2006), and around 30% by the end of the second year. Weishaar et al. (2012) show that the 50 largest firms accounted for the majority of trades in 2005–2006.

[16] Allowance holdings in this figure include the full free allocation over the entire period of Phase I, plus net purchases. Note, however, that the data only contains actual allowance transfers, whereas the price signal could have occurred earlier in a futures trade.



Source: Hintermann (2015).

Figure 5.3 Allowance holdings and emissions in final year of Phase I

the excess allowance holdings for two of these firms (RWE and Vattenfall) could not be explained by risk aversion or uncertainty about future emissions, leaving over-purchasing as a likely reason. However, as indicated above, alternative explanations cannot be ruled out, because firms' expectations and the resulting strategies are inherently unobservable.

The second empirical contribution about market power in the EU ETS is by Weishaar et al. (2012), who focus on short-term price manipulation perpetrated by large firms from the beginning of 2005 to mid-2006. This is conceptually quite different from manipulating the allowance price level, which is the subject of the previously discussed literature. Their data analysis shows that the largest 50 firms were responsible for over half the trading volume, whereas the corresponding number on the global ultimate owner level is even larger.[17] The authors argue that these firms, and especially

[17] The largest 50 firms accounted for 66.5% of all purchases and 58.3% of all sales, whereas the corresponding percentages for the global ultimate owner (who may hold a majority position in several firms) are 81.3% and 67.6%, respectively.

those in the energy sector, have better knowledge about relevant parameters in the allowance market (for example, energy use) and can therefore be expected to use this information to engage in strategies that allow them to make a short-term trading profit by buying low and selling high. In particular, the authors examine whether large energy firms were able to profit from a 'pump and dump' strategy.[18] The authors do not find evidence that energy firms successfully executed this strategy, and interpret this as indirect evidence against market power being exerted.

However, the analysis is subject to a fundamental data problem: given that the European Union Transactions Log (EUTL) only records physical transactions and that many allowance trades are made on forward markets, the timing of the financial trade associated with a transfer is essentially unknown.[19] A single transfer recorded in the EUTL could be the sum of a series of smaller forward trades that occurred over months or even years, but which expired on the same day. A succession of two large transfers of opposite sign could be the result of a hedging strategy in which firms continuously adjust their net forward position in response to new information, rather than evidence for a 'pump and dump' strategy.

4. CONCLUSIONS

This chapter has provided a review of the economics literature that pertains to market power in emission permit markets. The theory is well developed, but few empirical applications exist to date. The early contributions focus on compliance cost minimization and abstract from price changes in the product market. Since emission allowances are a necessary input for production, this assumption clearly does not hold if firms are able to pass on some of their costs to consumers. Allowing for an interaction between allowance and product market prices will alter the results: although it holds that under compliance cost minimization net buyers will always use their power to depress the allowance price, this is no longer the case if firms maximize joint profits in the allowance and product markets.

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18 In such a strategy, a large firm temporarily increases market prices by making a large purchase, thereby inducing less-informed traders to buy as well, and consequently benefitting by selling the same (or similar) volume of allowances at the higher price (or vice versa, by temporarily reducing the price with a large sale in order to buy in more at a lower price).

19 For the relative share of spot vs. future trading, refer to Ellerman et al. (2016). The limited amount of future transactions during the inception of the first trading period of the EU ETS is also presented by Bredin et al. (2012).

Instead, a net allowance buyer may find it optimal to increase the allowance price upwards, because the increase in revenue more than compensates the increase in compliance costs. Neglecting this interaction between markets, when it is present, can lead to an ill-informed testable hypothesis. For example, if the largest firms are net allowance buyers (which has been the case for all ETS's to date), one might search for evidence of downward price manipulation, without considering that market power would actually result in an allowance price increase.

Evidence from laboratory experiments suggests that market power can indeed lead to significant deviations from efficiency and to a resulting loss in welfare, provided that the allowance and product markets are considered jointly. The empirical evidence from actual markets is less clear. The papers focusing on the US Acid Rain Program report evidence of efficiency based on firms' banking behavior, and the analysis by Weishaar et al. (2012) finds no evidence for short-term trading gains by large energy firms covered by the EU ETS. On the other hand, analyses of longer-term allowance holdings in the RECLAIM and EU ETS markets suggest that large firms may have engaged in allowance over-purchasing in order to drive up the price.

Because the power of any single firm, or colluding group of firms, decreases with an increasing number of actively participating firms, market power may be a temporary phenomenon that is especially relevant for new emissions trading schemes, when only a subset of all firms are active in the market. Due to reasons of political acceptability, it is precisely during these early years that free allocation tends to be more generous, which further increases the scope for upward price manipulation. Regulators of future schemes, such as the planned national ETS in China, are therefore well advised to be mindful of market power especially in the beginning. The policy recommendations that can be derived from the reviewed literature are (a) keeping free allowance allocation to a minimum, and (b) engaging in measures that aim to increase transparency and overall market participation. Such measures could include the publication of aggregate emissions at a monthly or quarterly frequency, making trades (and thus allowance holdings) public with little or no time delay, and generally lowering transaction costs to the minimum attainable level. This would decrease the scope for market manipulation and improve the performance of emissions trading schemes.

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APPENDIX

Derivation of Hahn's (1984) result:

Let q refer to a firm's final output (e.g., electricity), e to emissions, and $C(q,e)$ to the cost function, which is increasing in output and decreasing in emissions.[20] Furthermore, let x be the number of allowances a firm holds at the end of a compliance period, a the amount of free allocation, and s the market price for an emission permit. Minimization of compliance costs means that the dominant firm solves the following constrained optimization problem:

$$\min_{x,e} \; C(q,e) + (x - a)\sigma \quad s.t. \; e \leq x \tag{5A.1}$$

Substituting the constraint for an interior solution, taking the derivative w.r.t. emissions and setting to zero leads to:

$$-C_e = \sigma + (x - a)\frac{\partial \sigma}{\partial x} \tag{1}$$

If the firm's free allocation exceeds its permit demand (i.e., if it is a net seller of allowances), the last term on the RHS is positive, and vice versa if it is a net buyer.

Derivation of the result in Hintermann (2015):

Let q refer to the price of the dominant firm's output. The dominant firm chooses output, emissions and allowances to maximize its profits under an emissions constraint:

$$\max \; \Pi = pq - C(q,e) - (x - a)\sigma \quad s.t. \; e \leq x \tag{5A.2}$$

The optimality conditions from this model are given by

$$C_q = p \tag{5A.3}$$

[20] More specifically, in order to generate the results shown here, the cost function must satisfy $C_q>0$, $C_{qq}>0$, $C_e<0$, Cee>0, $C_{qe}<0$ and CqqCee-(Cqe)2>0 (see Hintermann, 2011). The model by Hahn (1984) excludes output, but q is introduced to facilitate comparison with the model that includes both markets.

$$-C_e = \sigma + (x - a)\frac{\partial\sigma}{\partial x} - q\frac{\partial p}{\partial x} \tag{5A.4}$$

Eq. (5A.3) states that the firm equates its marginal production cost to the output price, which is the standard result under perfect competition.[21] Eq. (5A.4) describes the distortion in the permit market and differs from Hahn's result by the last term on the right-hand side, which measures the effect of the dominant firm's permit purchases on the output price. This indirect effect is present even if the firm acts as a price taker in the output market and is due to the fringe's response to a change in the permit price. Defining Q^f as the total output of the fringe firms, the marginal effect of the dominant firm's permit purchase on the output price can be written as

$$\frac{\partial p}{\partial x} = \frac{\partial p}{\partial Q^f}\frac{\partial Q^f}{\partial\sigma}\frac{\partial\sigma}{\partial x} > 0. \tag{5A.5}$$

The efficient solution can be obtained by allocating the dominant firm the amount of allowances that makes it set its marginal abatement cost equal to the permit price. Solving eq. (5A.4) for the level of free allocation that makes the last two terms cancel each other yields

$$a^0 = x - \frac{\partial p/\partial x}{\partial\sigma/\partial x}q \tag{2}$$

The threshold (2) can also be interpreted in the context of assessing whether firms' profits increase with the introduction of a permit market. Taking the derivative of (5A.2) with respect to the allowance price shows that firms' profits increase with the permit price (and by extension, with the introduction of a permit market, which increases the price for emissions from zero to some positive number) if they receive a free permit allocation in excess of

$$\tilde{a} = x - \frac{dp}{d\sigma}q \tag{5A.6}$$

[21] This follows directly from the assumption that the dominant firm perceived market power in the allowance market, but not in the product market. Note that even though eq. (5A.3) is the standard condition for a competitive market, the firm's marginal production costs depend on the level of both output and emissions. With distorted permit prices, the marginal production costs and thus the firm's output generally differ from the outcome under perfect competition in both markets.

Note that this applies to all firms, including those that do not wield market power. If the dominant firm's output is held constant, the thresholds in (2) and (5A.5) are the same.[22]

[22] To see this, we totally differentiate the output price and the permit price to obtain $dp=\partial p/\partial x\ dx + \partial p/\partial q\ dq$ and $d\sigma=\partial\sigma/\partial x\ dx + \partial\sigma/\partial q\ dq$. Dividing the former by the latter and setting $dq=0$ yields $dp/d\sigma = (\partial p/\partial x)/(\partial\sigma/\partial x)$.

6. Enforcement of emissions trading – sanction regimes of greenhouse gas emissions trading in the EU and China

Marjan Peeters and Huizhen Chen

1. INTRODUCTION

This chapter aims to provide new insights into the role of law for establishing and maintaining a well-operating emissions trading scheme for which an adequate enforcement strategy is essential.[1] Since emissions trading is an instrument for environmental policy, compliance of participants with the emissions trading scheme is of imminent importance in order to avoid unacceptable damage to the environment. Indeed, while the government may decide on the level of environmental protection by determining an emissions cap (which in fact determines the level of allowable pollution under the emissions trading regime), proper compliance is needed to ensure that this environmental target will not be superseded. Consequently, a trustworthy *measurement* of emissions has to take place, together with a submission of a number of tradable allowances exactly corresponding to the amount of emissions an operator has produced. In addition, *sanctions* need to be established in order to ensure that:

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[1] This chapter (finished in March 2015) builds on several publications of the authors, among which are: Marjan Peeters: 'Enforcement of the EU Greenhouse Gas Emissions Trading Scheme', in K. Deketelaere and M. Peeters: *EU Climate Change Policy: The Challenge of New Regulatory Initiatives*, Cheltenham, UK and Northampton, MA, USA: Edward Elgar, 2006, pp. 169–87; 'Inspection and Market-based Regulation through Emissions Trading: The Striking Reliance on Self-monitoring, Self-reporting and Verification', *Utrecht Law Review*, 2(1), June 2006; 'The Enforcement of Greenhouse Gas Emissions Trading in Europe: Reliability Ensured?' in Lee Paddock, Du Qun, Louis Kotze, David L. Markell, Kenneth J. Markowitz, Durwood Zaelk, *Compliance and Enforcement in Environmental Law: Toward More Effective Implementation*, Cheltenham, UK and Northampton, MA, USA: Edward Elgar 2011, 407–430. Huizhen Chen: 'Inspection and Enforcement in Chinese Carbon Emissions Trading: Progress, Problems, and Prospect,' *Environmental Law Reporter*, 44(7), July 2014, pp. 10596–606; and also: 'Towards a market-based climate regime in China? A legal perspective on the design and implementation of greenhouse gas emissions trading' (PhD thesis defended at Maastricht University in October 2015).

(1) unlawful action – meaning action against the rules of the emissions trading regime – will be deterred; and
(2) in case unlawful emissions took place (meaning emissions not covered with a corresponding emission allowance submitted to the government), these so called 'excess emissions' will be compensated by the infringing operator so that finally the environmental target will be reached.

A one-size-fits all blueprint for the monitoring and sanction rules for an emissions trading regime does not exist, although some convergence of approaches among several jurisdictions may happen. A range of different design options for monitoring and enforcement can be developed, which may also depend on the legal system in which an emissions trading regime will operate. When discussing how monitoring and enforcement has to be set up within an emissions trading regime, a balance has to be found between on the one hand developing an effective approach that ensures that infringements and particularly excess emissions will be avoided, and, on the other hand, taking account of the legal rights private actors enjoy. Meanwhile, examination of practice may furthermore contribute to the further understanding of how to enforce emissions trading, since in a number of jurisdictions the instrument has been applied for several pollution problems.

Remarkably, legal scholarship has thus far paid limited attention to the enforcement dimension of emissions trading. This chapter aims to further the debate but since the topic would merit a book instead of a chapter, a choice is made to concentrate on sanction regimes. Section 2 first sets the scene by pointing at the need of an adequate enforcement approach and related legal scholarship. Then section three will present the specific case of the EU ETS, established with Directive 2003/87/EC from 2003, while section 4 turns to the recently developed greenhouse gas emissions trading pilot projects in China. Sections 3 and 4 particularly focus on sanctions for excess emissions, thereby pointing at recent case law regarding penalties for emissions trading within the EU and at specific enforcement approaches in the emerging emissions trading regimes in China. Section 5 concludes.

2. LEGAL AND ECONOMIC PERSPECTIVES ON ENFORCEMENT OF EMISSIONS TRADING

2.1 The Need for an Adequate Sanction Regime

As a starting point, we assume that the price of the allowances not only creates an incentive to attain environmental protection at low cost, but at the same time may serve as an incentive for firms to avoid compliance with the rules.[2] The latter is particularly a risk in the absence of an effective enforcement system.[3] How to set up an enforcement approach has been considered to some extent in economic literature, particularly also by Tietenberg who has manifested himself as an expert on the emissions trading instrument.[4] From an economic point of view, the emitters – as rational economic persons – generally make up decisions of compliance or non-compliance by weighing the costs and benefits of each choice, in order to minimise their own costs.[5] Economic literature argues that cost-minimising emitters would choose a level of compliance at which the marginal cost of compliance is equal to the expected marginal cost of non-compliance, which is defined as the likelihood that a non-compliance will be detected and a sanction levied, multiplied by the marginal sanctions (such as the fines).[6] In this respect, economists propose that regulators can raise the expected cost of non-compliance by paying attention to: (a) the likelihood that violations will be detected and punished, and (b) the level of the sanctions.[7] In this context, it should be ensured that the expected cost of non-compliance such as the fine per unit of excess emissions is (sufficiently) higher than the allowance's price.[8] The two conditions mentioned above should not only stimulate the operator to submit truthful emissions reports, but also guarantee that each operator will hold sufficient allowances to cover his emissions.[9]

In order to reduce and to remove the incentive for non-compliance, the instrument of a *financial penalty* imposed on offenders with excess emissions is widely considered to be a necessary and effective approach to be included into an emissions trading enforcement regime. However, the financial penalty may be ineffective if it exceeds the financial capacity of the offenders and thus additional approaches and penalties would be necessary.[10] Particularly the compensation of the excess emissions may become a problem when the company goes bankrupt. Also the possibility to impose criminal sanctions for specific non-compliance including false

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2 Peeters (2006a) p. 179; also Epiney (2012) p. 27.
3 OECD (2004), p. 25.
4 Tietenberg (2006), pp. 171–6, particularly p. 171.
5 *Ibid*, p. 171.
6 *Ibid*.
7 *Ibid*, p. 173.
8 Stranlund et al. (2002), p. 346.
9 Tietenberg (2006), p. 175.
10 Becker (1968); Shavell (1985), referred to by Peeters (2011), p. 421. An evaluation report of the EU ETS mentions that in Germany several cases of insolvent operators have been the case, see Verschuuren, Fleurke (2014) p. 33.

reporting and other illegal conduct is suggested.[11] It is generally believed that when optimal dissuasion can be achieved equally through either fines or criminal sanctions, fines are preferred as they are less expensive to administer.[12]

2.2 States as Participants to an Emissions Trading Regime

The need for an adequate enforcement regime also applies in the situation where *states* are the participants to an emissions trading scheme, which is for instance the case with the Kyoto Protocol (article 17) and the EU Effort Sharing Scheme (Decision 406/2009/EC). Within the latter EU scheme, emission reduction targets are addressed to Member States. In view of reaching those targets transfer of emission reduction achievements may happen among states, next to other flexibility options like the use of Clean Development Mechanism (CDM) credits (Certified Emission Reduction Credits (CERs)).[13] To what extent Member States will make use of the trading options remains to be seen: the compliance period runs from 2013 until 2020. In order to let the trading operate efficiently, it is of imminent importance that the Commission will start infringement actions against Member States that have not reached their annual emission reduction obligations, otherwise potential non-complying states may be less interested in using trading opportunities in order to cover the shortfall of their target. However, the possibilities for keeping Member States in compliance with EU law are not strong; the infringement procedure that the European Commission may undertake for holding Member States to account is not very effective.[14]

On a more global scale, the issue of how to ensure that states comply with emission trading rules is a concern of the Kyoto Protocol, which includes the project-based mechanisms known as Joint Implementation (JI) and CDM next to interstate trading of so-called Assigned Amount Units (AAUs). The enforcement branch of the Compliance Committee to the Kyoto Protocol has some means to supervise proper compliance with the set emission reduction obligations.[15] The number of parties with emission reduction targets under the Kyoto Protocol (including the Doha amendment from 2012) is limited and mainly consists of EU Member States. Nonetheless, experiences with the Kyoto Protocol Compliance Committee

[11] Tietenberg (2006), p.182.
[12] See Polinsky, Shavell (1979), referred to by Faure (2010), p. 266.
[13] See for a more elaborated discussion Peeters, Stallworthy (2012).
[14] Peeters (2013) section 3 (compliance).
[15] See about the specific features this compliance committee for instance Loibl (2010) and Oberthür, Lefeber (2010).

are useful for gaining further understanding of what is needed and what kind of strategies are suitable for holding states in an international emissions trading regime compliant. Also the EU, particularly its experience with the Effort Sharing Decision, provides a source for further insights on how states as participants to an emissions trading regime will act and how compliance can be reached. Next to considerations of the behaviour of EU Member States as participants to an emissions trading regime, a proper implementation by Member States of the EU ETS – which covers emissions by industries – is crucial for ensuring that no excess emissions take place. Within the EU ETS, Member States are primarily responsible for monitoring and enforcing the behaviour of the covered industries, and the European Commission may start infringement procedures if Member States fall short regarding this. Part of the problem is that the Commission may not always have sufficient information on possible lacks in implementation and/or execution by a Member State.[16] Another issue regarding compliance by states arises if emission trading regimes are linked between several countries. This is for instance the case for Norway, Iceland and Lichtenstein that have joined the EU ETS. In order to protect the integrity of the EU ETS, it is of course necessary that sufficient guarantees are established to ensure that these three countries apply an adequate monitoring and enforcement approach.[17]

2.3 A Need for More Legal Scholarship on Enforcement of Emissions Trading

This section will briefly review the limited appearance of legal literature concerning enforcement of emissions trading, particularly regarding the EU ETS. As such, quite some legal literature has emerged after the introduction of greenhouse gas emissions trading in the EU in 2003. The bibliography in a book that critically reviews the development of legal scholarship on the EU ETS covers 21 pages.[18] A review of this list, together with other English legal literature that is known to us, confirms that the question of how to ensure compliance within the EU ETS has thus far got limited attention.[19] We can however point to two relatively recent important publications:

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[16] Epiney (2012), p. 27.

[17] As far as is known to us, this issue has not yet been discussed in depth in legal literature.

[18] Bogojević (2013).

[19] We tried to get a comprehensive overview and are grateful to Dionysios Stivas, student fellow at the Metro institute of Maastricht University in 2013–

- Pablo Mendes de Leon (2012) has discussed the legality of the sanctions applicable to aviation emissions covered by the EU ETS. This article takes a firm position against the inclusion of international aviation emissions. The author argues that the EU, including the Court of Justice of the EU, has acted disrespectfully towards international law, more specifically the international agreement regarding civil aviation.[20] Particularly the legality of the operating ban, a sanction of the EU ETS system that is provided only for aircraft operators, may be at risk according to this scholar.
- With regard to (the much needed) empirical studies, a study, commissioned by the EU, was published in 2014. This study, developed by Verschuuren and Fleurke, discusses experiences from six Member States with monitoring and enforcement of the EU ETS. The study (further discussed in section 3.3) finds inter alia that EU ETS compliance practice in the six investigated Member States varies greatly.[21] This illustrates the need for additional comparative research discussing different enforcement approaches (also those from other EU Member States), including the need for considering further harmonisation of national enforcement practice.

3. SANCTION REGIME OF THE EU ETS

3.1 The Legislative System of the EU ETS

Achieving full compliance with the EU ETS depends on, first, the activities by the covered entities, second, the due performance of the verifiers, and, third, the enforcement strategies of the Member States and the three states that have joined the EU ETS. The EU ETS relies heavily on self-monitoring and self-reporting by covered entities, backed up by a third party verification of the compiled report.[22] The reliance on private verifiers means that competition may take place among these verifiers in order to

2014, who did a literature search in spring 2013. See for an interesting comparative overview Kruger, Egenhofer (2006). Since the implementation of the enforcement provisions of the EU ETS has to be done by Member States, it may be that in the languages of these countries interesting case law and legal literature has emerged.

[20] Mendes de Leon (2012). See Case C-366/10 *Air Transport Association of America and Others v Secretary of State for Energy and Climate Change*, Judgment of the Court (Grand Chamber) of 21 December 2011 (not yet reported).

[21] Verschuuren, Fleurke (2014) p. 2 (executive summary).

[22] Peeters (2006b).

win contracts by the covered entities; this adds to the market-based regulatory character of the emissions trading instrument. The idea of using private actors for compliance control is not necessarily limited to the EU ETS but can also be considered for command and control approaches. The use of private verifiers means that a system has to be set up to control the verifiers, and to ensure that fraud by verifiers can be effectively addressed. While admittedly much more research is needed to understand in depth how the combination of self-monitoring, self-reporting and the use of third-party verifiers works, and whether this strategy is to be preferred above governmental control, this chapter focuses on the sanctions regime since regarding this issue even less legal literature has been published.[23] This section aims to provide some modest input to this much needed scholarly attention.[24]

The EU ETS Directive prescribes Member States, first, to establish and implement a national legislative framework with penalties regarding infringements of national provisions needed to implement the Directive.[25] As a general rule, it is provided that these penalties must be effective, proportionate and dissuasive. This instruction leaves some discretion to Member States as to how to design the national legislative framework. Next to this, the directive prescribes two specific sanctions.[26] These sanctions relate to the core rule that no emissions may happen without a corresponding surrendering of emission rights.[27] It concerns the following sanctions:

(1) The imposition of a financial penalty, called the 'excess emissions penalty'.[28] The level of this penalty is prescribed in the Directive, meaning that it shall be €100 for each tonne of carbon dioxide

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[23] Epiney (2012) discusses the (limited) possibilities for individuals to enforce correct compliance by Member States in the EU (pp. 27–9).

[24] Within the EU ETS, several criminal activities have been employed such as, in particular, VAT fraud and money laundering. These activities as such, most likely, do not threaten the environmental integrity of the scheme which consists of the idea that no emissions may happen without allowances. See for a court decision related to the case of stolen allowances and liability of the EU (claim rejected): Case T-317/12 (18 September 2014).

[25] Article 16(1) EU ETS directive.

[26] Article 16(2) EU ETS directive (naming and shaming) and art. 16(3) (financial penalty).

[27] This rule is established in art. 12(3) EU ETS directive. See furthermore also art. 15 EU ETS directive establishing that in case an emissions report has not been verified as satisfactory cannot make further transfers of allowances.

[28] Art. 16(2) EU ETS directive.

equivalent emitted for which the operator (including aircraft operators) has not surrendered allowances.[29] From 1 January 2013 onwards the level of the excess emissions penalty shall increase in accordance with the European index of consumer prices.[30] Payment of the excess emissions penalty shall not release the operator from the obligation to surrender an amount of allowances equal to those excess emissions, which may be done when surrendering allowances in relation to the following calendar year.[31]

(2) On top of the excess emissions penalty, Member States shall ensure publication of the names of operators (including aircraft operators) who are in breach of requirements to surrender sufficient allowances.[32] There is no instruction as to how this publication has to be done. Moreover, the publication is only needed in the language(s) of the specific Member States (and hence, for instance, in the specific official journal of the Member State) which may be a barrier for easy insight into the names of offenders.

Next to this, and specifically for aircraft operators, Member States *may request* the Commission to impose an operating ban to an aircraft operator for which the national enforcement measures have not resulted in compliance.[33] The fact that it is the Commission, and not a national authority, that imposes such a ban means that access to the court for aircraft operators that want to fight against such a ban will be at the EU level. All other legal actions by industries against Member States' enforcement decisions have to be submitted to the national courts.

[29] See about a potential legal barrier for Member States to impose a more severe penalty, Jans, Vedder (2012) p. 118.

[30] Article 16(4) EU ETS directive.

[31] Article 16(3) EU ETS directive.

[32] See for a critical discussion Peeters (2006a) pp. 180–81. In-depth research is needed to obtain a better understanding of the usefulness and acceptability of the naming and shaming approach. One can also wonder whether information about compliance by the operators should already be publicly available, upon request, in view of Directive 2003/4/EC.

[33] Article 16(5) EU ETS directive. Mendes de Leon (2012) states that not imposing this sanction on stationary installations is disputable in view of the principle of equal treatment (p. 297). He furthermore argues that exercise of this sanction by the Commission on aviation companies from third countries (and not by an administering Member State with which a third country has treaty relationships) might be illegal (p. 301).

3.2 Case Law on the EU ETS

In the meantime, case law has shown that the CJEU favours a strict enforcement approach regarding the excess emissions penalty.[34] The case concerned a failure by two Swedish industries to surrender sufficient allowances before the deadline of 30 April, and as a consequence the excess emissions penalty was imposed.[35] The industry, however, held that it was due to an error, and not unwillingness, to surrender before the deadline, stating that it had sufficient allowances on its account but failed to surrender them. The referring Swedish court wanted to know, first, whether the financial penalty has to be applied irrespective of the cause of the omission, and, second, whether there may be a possibility to waive or reduce the penalty in case of an administrative error or a technical problem while the operator had a sufficient amount of allowances. The Court considers that art. 16(3) and 16(4):[36]

> must be interpreted as precluding operators who have not surrendered, by 30 April of the current year, the carbon dioxide equivalent allowances equal to their emissions for the preceding year, from avoiding the imposition of a penalty for the excess emissions for which it provides, even where they hold a sufficient number of allowances on that date.

Interestingly, the Advocate General favoured a more flexible approach, based on the reasoning that given the specific circumstances of the case (an administrative error by the operators) the excess emissions penalty does not apply, but the more general rule laid down in art. 16(1) EU ETS Directive obliging Member States to provide adequate penalties.[37] Accordingly, one could also consider that in order to avoid administrative mistakes (that can lead to huge financial consequences for the operators) administrative authorities could establish a system of warnings in order to ensure that operators are aware of their duty to surrender their allowances

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[34] CJEU Case C-203/12 (17 October 2013).

[35] Case C-203/12, para 17 and 18: 'As at 30 April 2007, the Billerud companies, companies governed by Swedish law holding carbon dioxide emission allowances, had not surrendered the allowances equal to their emissions for 2006 (10,828 and 42,433 tonnes respectively). Consequently, the Naturvårdsverket imposed the penalty provided for by Law No 2004:1109 implementing Directive 2003/87, in the amount of SEK 3. 959.366 for one company and SEK 15.516.051 for the other (€433,120 and €1,697,320).'

[36] As applicable by the time of the offence.

[37] Case C-203/12, Opinion of Advocate General Mengozzi, delivered on 16 May 2013, para. 33.

before the deadline.[38] Basically, the Court chooses to follow a strict interpretation leaving no room to take into consideration the circumstances that led to the breach. In theory, the application of emissions trading does not necessarily require such a strict enforcement approach, but incorporating the possibility to consider the proportionality of the sanction of course may imply more governmental capacity.[39] In the specific EU context, moreover, discretion in defining the level of the penalty embodies the danger of sparing the industry.[40] In the meantime, one can see that on the national level some exceptions are provided to the application of the excess emissions penalty. Germany allows an exemption in case of force majeure.[41] UK legislation provides that the excess emissions penalty will be reduced to €20 if an operator who initially failed to surrender the correct allowances, but advised the regulator of their mistake and surrendered the correct allowances before the regulator noted their non-compliance.[42] The question of legality of these national measures may be submitted to the courts.

Another question for preliminary ruling was submitted to the CJEU in February 2014, and the Court has yet to provide judgment.[43] While in the Swedish case discussed above the possibility of mitigating or waiving the financial penalty is being discussed, this case concerns the question of whether the financial penalty has to be applied when the emissions report was approved by the verifier but later is found by the public authority to be incorrect in the sense of showing an insufficient amount of emissions.[44] The question submitted for preliminary ruling reads as follows:

> Must Article 16(3) and (4) of Directive 2003/87/EC be interpreted as meaning that the excess emissions penalty must also be applied in the case where an operator has, by 30 April of a given year, surrendered a number of allowances corresponding to the total emissions stated in its report on emissions from the installation for the preceding year, and that report has been assessed as satisfactory by the verifier, but where the competent authority, after 30 April, has established that the verified emissions report had erred by understating

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[38] Peeters (2014). For instance in Poland, a national authority sends reminders before each important deadline: Verschuuren, Fleurke (2014) p. 61.

[39] Peeters (2014).

[40] Peeters (2006a) p. 182.

[41] Verschuuren, Fleurke (2014) p. 32.

[42] Verschuuren, Fleurke (2014), p. 68, see for the UK text http://www.legislation.gov.uk/uksi/2012/3038/regulation/54/made (available 13 July 2016).

[43] Case C-148/14; see for the decision of the German Federal Administrative Court (Bundesverwaltungsgericht) http://www.bverwg.de/entscheidungen/entscheidung.php?ent=200214B7C37.11.0 (available 13 July 2016).

[44] German Federal Administrative Court Decision, para. 4.

> the total quantity of emissions, the report was duly corrected and the operator surrendered the additional allowances within the new period for surrender?

The German Federal Administrative Court who referred this question to the CJEU elaborates the potential positive or negative answer to the question, and seems to be in favour of a negative answer, both in view of the text of the Directive and the principle of proportionality. The referring decision considers inter alia that the Directive does not prescribe that the excess emissions penalty has to be applied when after approval of the verifier a governmental authority deviates from this decision.[45]

3.3 Discovering Non-compliance in the EU ETS

Epiney (2012) has stressed some potential enforcement problems with the EU ETS. She particularly states that the Commission may not always have sufficient information on possible lacks in implementation and/or execution by Member States.[46] Also researchers face dilemmas when trying to understand the level of compliance of operators with the EU ETS. Naturally, they cannot do investigations regarding the behaviour of industries themselves, and are hence dependent on information provided by industries or by national authorities. Obviously, operators and national authorities may have reason to stay reluctant to provide comprehensive and correct information, since disclosure of non-compliance by operators, or disclosure of short-falling enforcement strategies by Member States, may have consequences in the form of negative publicity and sanctions, including infringement actions by the Commission towards Member States. Any evaluation research by academics regarding compliance with the EU ETS (and other environmental legislation) hence meets quite serious barriers for finding the truth and may mostly be based on indirect information. The evaluation report regarding the effectiveness of the compliance mechanism of the EU ETS, published in 2014, is for instance largely based on information from existing written sources and interviews with key players in the compliance mechanism (particularly national authorities from six Member States). Based on these materials, the report states that compliance with the EU ETS is very high.[47]

The report provides a string of more detailed and interesting insights into how compliance with the EU ETS works in practice. One finding is for instance that the number of staff employed in the national emissions

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45 *Ibid*., para. 23.
46 Epiney (2012) p. 27, see also Peeters (2012) pp. 423–4.
47 Verschuuren, Fleurke (2014) p. 78 para. 6.

authorities differs enormously;[48] this may give rise to assumptions that in countries with low capacity some non-compliance will stay undetected. Moreover, and strikingly, the researchers have found that in one of the six Member States (Hungary) there was 'unsatisfactory access to the necessary Hungarian sources'.[49] The researchers also point at the fact that the information provided by Member States in reports pursuant to art. 21 of the EU ETS directive do not provide a complete picture regarding actual compliance and enforcement of the EU ETS in the Member States. These findings illustrate the barriers to getting full insight into the rate of compliance by operators with the EU ETS.

Regarding naming and shaming, the report concludes that this sanction is not actively applied in the six researched Member States. Two interesting facts are been reported:

- The names of installations that did not surrender sufficient allowances are far from easy to find.[50]
- German practice shows that, at least according to the competent authority, NGOs do not yet follow up on the naming and shaming information.[51] This could be explained by the fact that most of the detected infringements are not qualified as deliberate fraud actions, but are merely in the sphere of mistakes due to the complexity of the rules.[52]

These short examples show that more research is needed towards the usefulness and application of naming and shaming. If a legislator wants to use naming and shaming in order to apply any deterrent effect, there should be provisions that the names will be known to those who for instance might want to buy allowances (or the products, since consumers may want to boycott industries that do not obey the climate law rules). However, it remains to be seen how deterrent this information disclosure is, and whether it will be used by ENGO's.

The report furthermore discloses that Member States provide a range of different administrative and criminal sanctions. The authors suggest some further harmonisation, for instance by broadening the scope of the Directive 2008/99/EC on the protection of the environment through

48 *Ibid*, p. 78
49 *Ibid*, p. 76.
50 *Ibid*, p. 79.
51 *Ibid*, p. 37.
52 *Ibid*, p. 78.

criminal law to the EU ETS.[53] An important forward look given by the evaluation report is that enforcement challenges may become more serious if the price of the allowances increases.[54] The compliance mechanism of the EU ETS will then be put at the test.

4. SANCTION REGIMES OF THE CHINESE CO_2 ETS

China has set up a national carbon intensity reduction target binding the Chinese governments in the 12th Five-Year Planning (2011–2015), intended to reduce national carbon dioxide (CO_2) emissions by 17 per cent per unit of GDP, compared to that of 2005.[55] In order to facilitate the achievement of this target, seven pilot projects for CO_2 ETS, with the approval of the National Development and Reform Committee (NDRC), are being established and implemented in five municipalities and two provinces.[56] These pilots take place in view of a possible national ETS in the future.[57] The enforcement packages established within these sub-national ETS pilots contain various penalties for non-compliant behaviour, primarily related to causing excess emissions and infringing MRV-rules. In addition, a target-based accountability system regulating local governments and governmental officials aims to stimulate proper governmental enforcement action.

4.1 Regulatory Frameworks Establishing Sanctions in China

This section will introduce first the sanction regimes of the pilot projects, with a focus on penalties for excess emissions, after which the specific Chinese governmental accountability system will be presented. A discussion of these two approaches takes place in sections 4.2 and 4.3.

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[53] *Ibid*, p. 78.
[54] *Ibid*, pp. 33–34.
[55] Chapter 3, Guidelines of the Twelfth Five-Year Planning for National Economic and Social Development [hereafter referred as the 12th Five Year Planning (2011–2015)], National People's Congress, 16 March 2011 (in Chinese).
[56] The five municipalities are Beijing, Shanghai, Tianjin, Shenzhen and Chongqing, and the two provinces are Guangdong and Hubei.
[57] NDRC, The Notice on Initiating Pilot Programs of Emissions Trading, 29 October 2011 (in Chinese).

Penalties for excess emissions[58]

Each pilot project has established its own enforcement regime, including sanction regimes for excess emissions. The discussion below takes the Shenzhen pilot project as a starting point,[59] supplemented by an explanation of major differences in other pilot projects.

For excess emissions in the Shenzhen pilot project, the competent authority, the Shenzhen Development and Reform Committee (DRC), shall order the emitter to surrender allowances to cover the excess emissions within a specific time period or will deduct the allowances directly from the holding account, including, if necessary, from next year's holdings.[60] In addition, a financial penalty in an amount equal to three times the average market price of the last six continuous months will be imposed on the emitter with excess emissions.[61] Moreover, causing excess emissions also invokes other penalties, including:

- 'naming and shaming' by putting the non-compliance into the credit record[62] of the offender and disclosing it to the public;
- disqualification in applying for relevant governmental financial funding for five years;
- notification of the non-compliance record of a State Owned Enterprise (SOE) to the municipal state-owned assets supervision and administration commission and including it into the performance evaluation system for the SOE (including its leading cadres).[63]

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[58] The pilot projects also prescribe various sanctions for other non-compliance behaviour. The focus of this chapter goes to penalties for excess emissions.

[59] The reason mainly relies on the fact that Shenzhen pilot is the first CO_2 ETS pilot started, and, moreover, relatively detailed provisions regarding enforcement have been provided in this pilot.

[60] Article 75, Interim Measures on Carbon Emissions Trading Management in Shenzhen, Shenzhen Municipal People's Government, 19 March 2014 (in Chinese).

[61] *Ibid*; see also art. 8, Provisions of Controlling Carbon Emissions for Shenzhen Special Economic Zone, Shenzhen People's Congress Standing Committee, 10 November 2012 (in Chinese).

[62] The credit record of an enterprise contains its basic registration information, commercial credit information, and other relevant information that may affect the enterprise's credit, such as the illegal behaviour record that has been punished by the government. See art. 43 of Administrative Measures of Shenzhen Municipality on Credit Information Collecting and Credit Rank Evaluating of Enterprise, 19 November 2002 (in Chinese).

[63] See art.65, Interim Measures on Carbon Emissions Trading Management in Shenzhen.

Similar sanctions for excess emissions have been established by most of the other pilot projects but with different designs and intensities. For instance, twice the number of the shortfall of allowances will be deducted from the allowances of next year's holding by the provincial DRC in the pilot projects of Guangdong and Hubei.[64] In contrast, other pilot projects such as Shanghai and Tianjin only provide that the governmental authorities would 'order the emitters to make the correction' without any specificity as to how.

Imposing a financial penalty for excess emissions is a standard option in most pilot projects, but the level of the penalty and the way in which it is determined differs. The fine for the excess emissions in Beijing ranges from three to five times the average market price of the allowances.[65] However, in other pilots the financial penalties have not been directly linked to the market price and the level of the fines differs: the excess emissions will result in a fixed fine of 50,000 Chinese Yuan (CNY) in Guangdong[66] while the fines range from 50,000 to 100,000 CNY in Shanghai.[67] In the Hubei pilot project, the fine ranges from one to three times the market price for the excess emissions, with a total maximum of 150,000 CNY.[68]

Target-based accountability system for local governments/cadres

The pilot projects to be managed by the selected local governments are aimed at contributing to the achievement of sub-national carbon intensity reduction targets that have been determined for these local governments. At the same time, these local governments are subjected to a target-based accountability system, holding the leading cadres of local governments responsible for the fulfilment of these targets.

According to the 12th Five-Year Planning (2011–2015), the results of the performance evaluation concerning the fulfilment of the carbon intensity reduction target, including the sub-national targets, will provide

[64] Article 37, Trial Measures of Guangdong Province on Carbon Emissions Management, Guangdong Provincial Government, 15 January 2013; see also art. 46, Interim Measures of Hubei Province on the Management and Trading of the Carbon Emission Right, Hubei Provincial government, 4 April 2014 (in Chinese).

[65] Decision on Implementation of Carbon Emissions Trading Pilot on Premise of Strict Total Control of Carbon Emissions, Beijing People's Congress Standing Committee, 27 December 2013 (in Chinese).

[66] Article 37, Trial Measures of Guangdong Province on Carbon Emissions Management.

[67] Article 39, Trial Management Measures for Carbon Emissions in Shanghai, Shanghai Municipal People's Government, 18 November 2013 (in Chinese).

[68] Article 46, Interim Measures of Hubei Province on the Management and Trading of the Carbon Emission Right.

an important basis for the election, promotion, punishment and awards for the leading cadres.[69] In this respect, the NDRC has formulated specific performance assessment measures regarding the implementation and completion of the distributed carbon intensity reduction targets at the provincial levels.[70] Similar to the accountability system applied in the domains of economic development and social management, the leading cadres may be punished by disciplinary sanctions including warning, demotion, or even dismissal once any failure of a specific target achievement is detected.[71] By means of the incorporation of the target achievement into the cadre system, the Five-Year Planning has significant binding effect on the local governments, and has in this respect a potential powerful effect.[72]

Furthermore, according to the amendment of the Environmental Protection Law in 2014, a target-based responsibility system and performance assessment system for environmental protection will be implemented in the future, requiring the government to incorporate the completion of environmental protection targets, as an important basis for assessment of the competent departments and persons in charge thereof.[73] This amendment suggests the linkage between the political accountability mechanism and the legal system, but the exact design and detailed implementation of this new regime is not yet further explained. While this regime may stimulate the local governments to apply an ETS with stringent sanctions on excess emissions that may impede the fulfilment of the carbon intensity reduction target, it deserves further discussion in order to understand to what extent it will contribute to the effective implementation of the pilot projects, particularly also the imposition of excess emissions penalties.

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[69] Section 3, Chapter 61, 12th Five-Year Planning (2011–2015).

[70] The Notice of the NDRC on Printing and Distributing the Performance Assessment Measures Regarding the Carbon Intensity Reduction Target Responsibility, NDRC, 6 August 2014 (in Chinese).

[71] This is usually called 'One-vote veto' policy (*yi piao fou jue zhi*) in China, since these targets have the 'veto power', the failure to achieve which can trump other accomplishments. See for instance Deng (2011).

[72] Lin (2012), p.11; see also Mol, Carter (2006), p. 157.

[73] Article 26, Environmental Protection Law of China, promulgated by Standing Committee of the National People's Congress, 26 December 1989, revised 24 April 2014, effective as of 1 January 2015.

4.2 Comparison and Discussion of the Sanction Regimes in China

Legal basis for penalties in the pilot projects

In order to ensure compliance and instil confidence in the ETS, it is necessary to establish a binding and sufficiently stringent sanction regime. The legal basis and specific implementation procedures for penalties to be applied by local governments in the pilot projects should obviously be consistent with the Chinese legal system. Because most of the penalties that would be imposed by the pilot projects will be administrative penalties, the institution and implementation of sanctions for non-compliance ought to be designed in accordance with the Administrative Penalty Law of China to ensure that the sanctions will be enforceable.[74] Administrative penalties may be created only through formal legislations including laws, administrative regulations, local regulations, and specific rules of ministries and local governments.[75] In light of the fact that greenhouse gases are still not regulated as pollutants in China, the sanctions provisions established in the Environmental Protection Law cannot be applied directly to the CO_2 ETS pilot projects. Up to the end of November 2014, most of the regulatory documents relating to the pilot projects were issued in the form of 'other normative documents' (*qi ta gui fan xing wen jian*) adopted by local governments that do not constitute formal legislation in China. Important exceptions are the local regulations adopted by the Shenzhen and Beijing authorities, and four rules of local governments promulgated for the pilot projects of Shanghai, Guangdong, Shenzhen and Hubei.[76] Hence, one may raise questions as to whether the existing regulatory frameworks established in the pilot projects, in particular those of Tianjin and Chongqing that have not yet adopted local legislation, have sufficient legal status to impose specific penalties.

The Administrative Penalty Law as well as the Legislative Law in China permits the imposition of most administrative penalties through local regulations, with the exception of restriction on personal freedom

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[74] Article 3, Administrative Penalty Law of China, promulgated by National People's Congress, 17 March 1996.

[75] *Ibid*, at art. 14.

[76] According to the Chinese legislative structure, the provincial people's congresses and their standing committees may enact local regulations, while the provincial government can adopt rule of local government in accordance with laws; the legal effect of the local regulations is higher than that of the rules formulated by local governments at and below the corresponding level. See arts 63, 73, 80, Legislation Law of China, promulgated by National People's Congress, 15 March 2000, revised 15 March 2015 (in Chinese).

and withdrawal of a business license of an enterprise.[77] However, there are limitations provided by the Administrative Penalty Law as to the type of administrative penalties that may be imposed under specific rules of local governments; for example, with regard to violations of an administration order for which no laws or regulations have been enacted, the administrative penalty of a disciplinary warning or certain level of fine may be all that is permitted.[78] Therefore, some of the administrative penalties necessary for the ETS could be implemented through local regulations and rules of local government adopted in the pilot projects. However, criminal liability for non-compliance can only be established at the national level.

Fines (like the excess emissions penalty) can be created by law, administrative regulation, or local regulation.[79] In addition, a certain level of fine could be imposed based on the rules of ministries and provincial governments.[80] In this respect, the Standing Committees of the People's Congress of Shenzhen and Beijing as well as the municipal/provincial governments of Shanghai, Shenzhen, Guangdong and Hubei have the authority to establish a financial penalty for non-compliance through local regulations and rules of local government. At the same time, the absence of local legislation is also the main reason for the lack of financial penalties in the pilot projects of Tianjin and Chongqing, which have only adopted the governmental normative documents rather than relevant local legislation yet. This deficiency exists while, in line with the Chinese legislative system, there are not many legal barriers for provincial (and certain municipal) authorities, such as the local governments implementing the pilot projects, to impose a fine on excess emissions through local regulation and rule of local government.

Effectiveness of the financial penalty

The financial penalty, particularly the excess emissions penalty, is assumed to play a key role in ensuring the compliance of the ETS-participants.[81] However, one may question whether the excess emissions penalties established in the Chinese pilot projects are stringent enough.

In the pilot projects of Shenzhen and Beijing, financial penalties for excess emissions established in the local regulations are indexed to the prevailing market price. Because the amount is prescribed, the local authority

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77 Article 11, Administrative Penalty Law of China.
78 *Ibid*, at art. 13.
79 *Ibid*, at arts 9–11.
80 *Ibid*, at arts 12–13.
81 Cheng (2013), p. 32.

does not need to determine the specific amount of the fine, apart from gathering data on the prevailing market price. Meanwhile, imposing a fine equal to three (or up to five) times the market price on the violator, demonstrates the government's attempt to raise the cost of breaking the law. Emitters, as rational economic players, might be expected to choose to buy allowances to cover excess emissions from the market at market prices rather than paying three (or five) times as much in fines.

In contrast, the Shanghai and Guangdong pilot projects permit the competent authorities, usually the provincial DRC, to determine the fine within a certain range stipulated in the rules of local government. As for the Hubei pilot project, the fine is connected to the market price but limited to a cap of 150,000 CNY.[82] It is still an open question whether the penalties applied in the pilot projects, such as a cap on the penalty of 100,000 CNY in Shanghai, will serve as an effective deterrent. Especially when the cost in reducing emissions or buying allowances is higher than that of the capped fine, emitters may simply choose to pay the fine. However, the local governments still have quite limited legislative competence to create the financial penalties, the specific maximum amount of which shall be laid down by the standing committees of the provincial people's congresses.[83] In this respect, for instance, the maximum for the fine that can be established by Hubei provincial government has already been raised largely from 30,000 to 150,000 CNY by the standing committee of the people's congresses of Hubei province,[84] although it may remain relatively low for some big industries.

Although the revised Environmental Protection Law cannot provide a solid and direct legal basis for the pilot projects as mentioned above, the strict sanctions for illegal discharge of pollutants established in the amendment to the Environmental Protection Law in 2014 may pave the way for a stringent financial penalty punishing the excess emissions covered by the ETS, if specific GHG is explicitly labelled and regulated as a pollutant in the future. To be specific, if an emitter is fined and ordered to make a correction due to the illegal discharge of pollutants but refuses to make such a correction, the competent authority may impose the fine again on a daily

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[82] The 'cap' seems to be the maximum amount of fine for excess emissions, even when the price of the allowance is very high, the excess emission is very large, or when the operator is a relatively small company.

[83] See art. 13, Administrative Penalty Law of China.

[84] See The Regulation of the standing committee of the people's congresses of Hubei province on the maximum amount of the administrative fine established by governmental rule, adopted on 21 September 1996, revised on 23 May 2013 (in Chinese).

basis consecutively, equal to the original amount of the fine multiplying the days commencing from the date immediately following the date when it is ordered to make a correction.[85] There will be no cap on such a fine until the correction is made by the emitter. Moreover, it is stipulated that local regulations may increase the types of illegal acts subject to the fine consecutively on a daily basis in light of the actual needs of environmental protection.[86] In this sense, there seems to be a possibility for the local ETS pilot projects to impose this strict fine on the excess emissions in relevant local regulations.

Implementing the enforcement measures

Because the excess emissions penalty is deemed to be an administrative penalty, the decision-making has to be in conformity with the Administrative Penalty Law, which requires the administrative body to conduct an investigation in a comprehensive, objective and fair manner and to collect relevant evidence.[87] However, the detailed provisions of these procedures are still missing in some of the local regulations or other relevant rules in the pilot projects to date.

Some other challenges may occur in the pilot projects. For instance, the Beijing DRC attempts to limit the discretion for imposing administrative penalties by adopting detailed standards for decision-making, in the form of an 'other normative document' instead of formal legislation, but whether this will turn out effective remains to be seen.[88] On the basis of the fact, seriousness and relevant factors of the illegal act, the non-compliance is categorised into four groups subjected to (1) impunity, (2) lighter punishment, (3) normal punishment and (4) heavier punishment. Moreover, the specific legal basis, types and ranges of the penalties for non-compliance emerging in the Beijing pilot have been stipulated in a set of trial executive standards in the form of a detailed table.[89] This decision together with the executive standard is supposed to control the exertion of the administrative discretion.[90] Because of its legal status, however,

[85] Article 59, Environmental Protection Law of China.

[86] *Ibid.*

[87] Article 36, Administrative Penalty Law of China.

[88] Decision on the regulation of the discretion in implementing the administrative penalties in the carbon emissions trading, Beijing DRC, 6 May 2014 (in Chinese).

[89] For instance, with regard to the financial penalties for excess emissions ranging from three to five times of the average market price, the lighter punishment will be three times, while the normal punishment will be four times and the heavier punishment will be five times. *Ibid.*

[90] *Ibid*, at arts 1, 4.

this decision and the standard can only be considered as a kind of policy provision. Nonetheless, this provision appears to be a self-regulatory document binding the Beijing DRC *per se*, aiming to reduce and to avoid abuse of discretion. It will be important to observe whether and how the application of such standards for administrative discretion (*cai liang ji zun*) lead to effective and proportional deterrents in the sanction regime for the Chinese ETS.[91]

Furthermore, under the Administrative Penalty Law administrative counterparts, including citizens, legal persons, and organisations, have the right to refuse to accept an administrative penalty and may apply for administrative reconsideration or bring an administrative lawsuit to appeal the penalty.[92] Emitters participating in emissions trading have the same right to appeal the administrative decision in court, although it remains questionable whether such case law will emerge in China. If that is the case, the case law will provide a source with further information on how the sanction regime can be applied in practice and what legal protection ETS participants may get. Especially in view of the powerful government but weak judicial system in China, in particular the fact that the courts are funded by the governments at the same level, the limited role of the judicial system (and the possible lack of independency) is a concern.[93] Nonetheless, it would be important to ensure the private actors can challenge the government's sanction decision before independent courts, if China indeed wants to follow the rule of law.

4.3 Concluding Remarks

In addition to the legal enforcement provisions in the ETS pilots, non-legal mechanisms will be used for achieving a reliable ETS. The penalties for the participants' non-compliance elaborated above have shown that the sanctions for the SOEs are connected with the cadre system. In view of the fact that ETS pilot projects are meant to contribute to compliance with the carbon intensity reduction target, the cadre responsibility system is expected to have a significant influence on the leaders of the local government, stimulating them to apply an effective ETS pilot project with stringent punishment on excess emissions that may endanger the achievement of the carbon intensity reduction target. In this sense, the cadre system

91 For more on this topic Yu (2008); Zhou (2007) (Chinese literature).
92 Article 6, Administrative Penalty Law of China.
93 Du (2009), pp. 441–2.

remains an important approach, controlling the local governments and SOEs in view of ensuring well-functioning ETS pilot projects.

However, it remains questionable to what extent the enforcement package of the pilot projects can ensure compliance. The pilot projects are encouraged to explore feasible approaches to emissions trading, including the establishment of an effective enforcement package. It is inevitable that diversity of specific enforcement rules developed by local authorities seeking to accommodate different local circumstances will develop. However, the effectiveness of a sanction regime will largely depend on the legislative competence of the local authorities, and in this respect there are limits concerning the types and intensity of the sanctions regime established in the pilot projects. Furthermore, the question remains whether this decentralised approach for sanctions by local authorities will lead to problems of a 'race-to-bottom' among the regions in which the ETS projects are applied. This could happen in order to attract more industries to stay in the local territory, aiming to ensure local economic growth in view of the Chinese GDP-oriented evaluation system. In this respect, a harmonised or unified sanction regime at the national level deserves further consideration, in particular if China wants to move to a national ETS. However, a rule regarding the establishment of a possible national ETS, adopted by the NDRC recently, indicates that the sanctions for non-compliance will be mainly designed and implemented by the provincial authorities.[94] The specific design and implementation of the decentralised approach under the national scheme needs to be observed and the lessons from the pilot projects will be relevant.

5. CONCLUSION

This chapter has focused on sanction regimes within the EU ETS and the Chinese emissions trading pilot projects, and has shown that a financial penalty to be imposed if excess emissions have taken place forms a core sanction in these regimes. Both in the EU and in the Chinese pilot projects, the financial penalty is backed up with additional sanctions, like naming and shaming in the EU, and exclusion from governmental funding in China.

Regarding the EU ETS, some case law is developing particularly regarding the extent to which an administrative authority may (or has to) take into consideration the circumstance which has led to the infringement by

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[94] See arts 40–42, The Interim Measures for Management of Carbon Emissions Trading, NDRC, 10 December 2014, effective as of 10 January 2015 (in Chinese).

the operator. In a judgment, and contrary to the opinion of the Advocate General, the CJEU does not allow a flexible approach regarding the imposition of the excess emissions penalty (which could, in essence, be a more proportional approach). In the meantime, some Member States (Germany and the UK) have included in their national legislation some possibilities to deviate from a strict imposition of the financial penalty but such provisions are, given the Court's decision, at risk of being incompatible with EU law. Some further discussion will take place since a German court has referred a new case to the CJEU, which prolongs the discussion about how to enforce the EU ETS particularly in view of a positive (but incorrect) assessment by the verifier of the emissions report.

Within China, a country that aims to move to the rule of law but which still suffers from the relatively weak enforcement of legislation and heavily depends on strong political steering by the central government, local governments that are selected to establish pilot emissions trading programmes face competence problems with establishing rules that would support a comprehensive enforcement, including adequate sanctions. At the same time, through the cadre system, local governmental officials can be held responsible for the performance of the carbon intensity reduction target, which then has significant impact on their specific design choice and application of the emission trading instrument. In line with the decentralised experimental approach in establishing the pilot projects, the diverse design features of the sanction regimes at the local level may however imply the risk of a 'race-to-bottom' endangering the integrity of an ETS, as far as a national scheme is concerned. Since the pilot projects are approved by the national government with a view to establishing a national ETS system, it will be interesting to see whether and how the local pilot projects will be evaluated in order to derive learning points for moving to a national scheme. Such evaluation can also shed some further light on how enforcement actually works in the specific Chinese context: on the one hand, it needs to be discussed whether the practice of imposing excess emission penalties will be effective enough, and, on the other hand, how the imposition of penalties will be done in view of the Chinese rule of law and how, in this respect, operators may have access to independent courts. One aspect that has not been discussed in this chapter, but which deserves certainly also scholarly attention, is how Environmental Non-Governmental Organisations could play a role for holding operators compliant.[95]

Taking also a forward look in the case of the EU ETS, a potential increase of the price of the allowances will put the compliance regime to

[95] McAllister (2010).

the test and may provide further insight into the practice and legality of imposing sanctions on participants to the EU ETS, including the operating ban to aviation companies. At the same time, doing proper evaluations of emissions trading regimes, particularly where it concerns compliance and enforcement, is a challenge in itself, not only for governments, but also for academics who want to gain further insights into how emissions trading regimes work in the practice of different legal systems. Most likely, barriers exist to getting real insight into non-compliant behaviour. In conclusion, access to information regarding compliance and enforcement with an emissions trading regime needs to be further explored, not only for China but also for the EU.

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7. Windfall profits in the EU ETS power sector

Francesco Gullì

1. INTRODUCTION

When policymakers have to choose among different tools of environmental policy (standard, subsidies, taxes, emissions trading), they generally look at the following three attributes: (1) effectiveness (ability to induce the decrease in pollution); (2) efficiency (cost of achieving the environmental target); and (3) equity (cost distribution between firms and consumers).

Theoretically, an emissions trading scheme (ETS) can be designed to be environmentally effective and efficient.[1] Also fairness may depend on how the scheme is designed. In particular, unfairness is perceived when the environmental cost is mostly charged to consumers whereas the polluters gain windfall profits (profits due to the environmental regulation).

In relation to an ETS, the literature generally distinguishes two components of windfall profits (WP):

(1) *Regulation-induced windfall profits (RWP)*. RWP are additional profits accruing to entities falling under an ETS due to the free allocation of emission allowances. Free allocation is adopted to make the environmental regulation more acceptable to producers. If producers are able to, they will seek to pass the value of the allowances they have to use in their production process on to consumers as they forego the possibility to sell them. Economists describe such costs as 'opportunity costs' that operators naturally take into account. Producers thus receive allowances for free but charge consumers for them; such windfall profits due to free allocation can be perceived as unfair (by the consumers).
(2) *Price-induced windfall profits (PWP)*. Operators can also enjoy additional profits due to ETS-induced changes in production costs and prices, so called PWP. This category of additional profits does not depend on the allocation method and (at least in principle) arises even if all companies have to buy their allowances (for example, at auction).

[1] See Chapter 3 by Nentjes in this volume.

> It should not be considered as unfair. In fact, when we look at spot markets (for example, wholesale spot electricity markets) prices are set by the marginal technologies whose marginal cost includes the pollution cost. Prices may increase and the infra-marginal technologies (technologies with production costs lower than those of the marginal technology) may gain an additional profit if their pollution cost is lower than the increase in price. Therefore, as long as the implementation of an environmental regulation is expected, this kind of windfall profit is not a surprise. Furthermore, unless the environmental regulation determines an increase in market power (in imperfectly competitive markets), this change in profit should not be perceived as unfair. Rather, it is a legitimate environmental rent whose role it is to promote cleaner supply.

This chapter presents the economic treatment and literature on windfall profits and regulatory approaches to address them. In addition we carry out a simple analytical model finalised to identify and estimate the windfall profit rate (WPR). This indicator measures the share of the pollution cost converted to additional profits. As such, it allows us to understand how much windfall profits are likely to arise independent of the carbon price.

The focus of this chapter is on the first and second trading periods of the EU-ETS[2] (the largest ETS worldwide) and on the power generation sector, which is the most important sector in terms of carbon dioxide emissions and the sector where windfall profits are more likely to arise given the current organisation of the wholesale power markets. The effect of allocation on the electricity sector and the creation of windfall profits is well documented in the literature[3] while analysis of whether energy intensive industries could pass on their carbon costs seem to be less frequent as this emerged later.[4] Both of the aforementioned categories of windfall profits (price-induced and regulation-induced windfall profits) are examined and their specific contribution due to the free allocation of allowances will be determined in the context of the EU power sector.

The chapter is organised as follows. Section 2 presents the main determinants of windfall profits in the EU power sector, including some theoretical issues that are useful to understand how windfall profits depend on

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[2] Directive 2003/87/EC on emissions trading specified that for the first period 2005–2007 at least 95% of the allowances should be handed out for free and at least 90% for the second period 2008–2012.

[3] See for example Sijm et al. (2005), Sijm et al. (2006), Neuhoff et al. (2006), Sijm et al. (2008b).

[4] For an examination of other sectors being able to pass on costs to consumers see Smale et al. (2006) or CE Delft (2010a), CE Delft and Oeko-Institut (2015).

market organisation and regulation. Section 3 provides a critical review of the literature, both theoretical and empirical, on carbon cost pass-through, which is one of the main sources of windfall profits. Section 4 estimates the different types of windfall profit rates with a focus on the EU power sector in the first and second phase of the EU-ETS (2005–2012). Section 5 focuses on regulatory approaches to windfall profits (especially the Spanish carbon levy). Section 6 concludes.

2. WINDFALL PROFITS: BASICS

This section identifies the main determinants of a carbon ETS-induced windfall profit rate (WPR) in the power generation sector. The WPR is commonly referred to as the share of marginal carbon cost (the carbon cost of the marginal technology in the wholesale spot market) converted to windfall profits.

For this purpose, it is necessary to explain the fundamental mechanisms through which power generation costs are affected by the carbon price. Three conceptual steps have to be considered.

- First, ETS creates a market for emissions allowances. Since the allowances have a value, their use generates an opportunity cost (hereafter the carbon cost) equal to the allowance price, τ, multiplied by the emission rate, r_j
- Second, ETS causes an increase in the unit variable cost equal to the carbon cost. This cost arises even if the public authority allocates allowances free of charge (because of the opportunity costs of free allowances).
- Third, the value of the freely allocated allowances constitutes a 'gift' to the generator.

By following these steps, we are able to calculate the WPR (the share of the marginal carbon opportunity cost converted to additional profit). This is given by the following formula (see Appendix A):

$$WPR = \left(PTR - \frac{r_j}{r_i}\right) + \frac{r_j}{r_i}FAR = PWPR + RWPR \qquad (7.1)$$

Where WPR is the windfall profit rate (the share of the marginal carbon opportunity cost converted to additional profit), PTR is the marginal pass-through rate and FAR is the free allocation rate (the share of emissions allowances allocated for free). r_i is the emission rate of the marginal technology i (the technology-setting prices in the spot market) and r_j is the emission rate of the infra-marginal technology *j* (the technology whose bid price is lower than that of the marginal technology).

We assume that the regulator allocates the allowances on the basis of the expected future production of the different kinds of power generating technologies.[5] Note that the free allocation rate is equal to the allocation rate (the amount of allowances allocated divided by verified emissions), since we assume full free allocation.

Equation (7.1) highlights that windfall profits would depend on three factors: (1) the degree of free allocation (the amount of allowances allocated for free compared to the verified emissions); (2) the pass-through rate (the extent to which the marginal carbon cost is passed through to power prices) and (3) the ratio of emission rates. The third factor is given (exogenous). The first factor depends on policy decisions. The second factor also depends on how markets are organised and on their structure (fully or imperfectly competitive markets).

Then the following considerations arise:

1. The WPR includes two components. One price-induced windfall profit rate (PWPR) does not depend on how the allowances are allocated (freely or auctioned). The other regulated-induced windfall profit rate (RWPR) depends on the kind of allocation (namely, the degree of free allocation).
2. The PWPR depends on the pass-through rate and on the ratio of emissions rates. It is positive if the pass-through rate is higher than the ratio of emissions rates. This means that a positive marginal pass-through rate (PTR) is not a sufficient condition for windfall profits. The WPR may be negative even if the allowances are allocated for free (provided that the PTR is sufficiently low).
3. The RWPR depends on the free allocation rate and on the ratio of allocation (the number of allocated allowances divided by the expected emissions). This component may be positive unless all allowances are sold (full auctioning) and/or the infra-marginal technologies are carbon-free. This finding raises the question of how the windfall profits induced by free allocation could be addressed by regulators, for example by means of regulatory instruments such as for example, a levy applied to producers. To the extent to which the PTR is independent of the free allocation rate (FAR), producers should only pay

[5] We adopt this assumption for the sake of simplicity. Other solutions can be adopted. For example, the regulator might also assume an emissions rate lower than the current one, alone or combined with the expected emission or assuming a pure grandfathering (historical emissions). However, our assumption does not undermine the significance of the model.

for the allowances allocated for free and not for the additional profits due to the possible increase in prices. Furthermore, all carbon free technologies should be exempted from the regulatory instrument to capture windfall profits. Finally, producers have to pay for the windfall profits induced by free allocation (that is, the RWPR) even if they do not generate electricity. Moreover more polluting technologies should pay more than the less polluting ones.

4. The WPR may be positive, even if allowances are fully auctioned, if there is over-allocation; that is, if the allocated emissions are more than the verified emissions. However, note that in this case, if the over-allocation is generalised, the allowance price will collapse and consequently the absolute value of windfall profits will be very low.
5. Even if power generators are not carbon free (and even if they are the marginal technologies), they can gain windfall profits if the PTR and/or the FAR are sufficiently high.
6. Finally, for the reasons just described, free allocation is neither a necessary nor a sufficient condition for windfall profits. In particular, if allowances are allocated for free, windfall profits arise only if the pass-through rate is sufficiently high.

The last remark highlights that, before focusing on the WPR, it is worthwhile to analyse in depth the economic literature on the carbon cost pass-through rate and to distinguish between the theoretical and empirical contributions.

3. CARBON COST PASS-THROUGH TO ELECTRICITY PRICES

3.1 Theoretical Literature

The theoretical literature on carbon cost pass-through is limited.[6] Just a few authors deal with theoretical issues.[7] Among them, Sijm et al. (2012)

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[6] For a review, see also Gullì and Chernyav'ska (2013) or RBB Economics (2014). Independent of the electricity sector several determinants of cost pass-through have been identified: these include capacity utilization (CE Delft, 2010b), exposure to international trade (Varma et al., 2012), product differentiation (Reinaud, 2008), and non-traded costs and markup adjustments (Goldberg and Hellerstein, 2008). Nominal price rigidities might have a limited effect and delay cost pass-throughs (for studies on this see Nakamura and Zerom, 2010; Goldberg and Hellerstein, 2013).

[7] See for an application of this theory with regard to international competition Smale et al. (2006) but also CE Delft (2010b).

and Gullì (2013) provide the most comprehensive analyses focusing on how carbon cost pass-though is related to market structure.

Sijm et al. (2012) explain firms' behaviour under quantity competition and explore different conditions of demand and supply, such as linear or iso-elastic demand, constant or variable marginal costs, and so on.

In particular, first they analyse the relationship between output market structure and carbon cost pass-through under two extreme conditions of market structure, namely full competition and monopoly. Second, the authors analyse the oligopolistic framework by assuming quantity (Cournot) competition in output (electricity) markets.

The main finding of this contribution is that, if the demand curve is linear, the carbon cost pass-through is less than under full competition (and even converges to the fully competitive outcome when the number of competing firms becomes very high). This result is not surprising. It is straightforward that under monopolistic conditions (the classic situation of imperfect competition) pass-through to prices is less than the change in cost. In fact, the carbon pass-through under Cournot competition follows the general rule of pricing under this kind of firms' interaction: when the number of firms in the market increases, prices converge to the fully competitive equilibrium. However, the contribution by Sijm et al. (2012) has an important merit. It helps us to reject the widely held misunderstanding that the carbon cost pass-through rate under imperfect competition is always greater than that under full competition.

If the demand curve is linear, the pass-through under imperfect competition can be lower than under perfect competition. Nevertheless, this is not a conclusive finding, for the following reasons.

First, it might help us to explain part of the pass-through variability shown by the empirical literature, but cannot explain why pass-through may be nil or even negative.

Second, it is based on quantity competition which is not well-suited to describe the electricity spot markets, most of which are organised in the form of price auctions. In these markets, price competition models seem to be more suited to simulate the interaction between firms.

Third, empirical evidence shows that generally firms do not maximise profits as assumed by standard models of competition. They pursue strategies besides profit maximisation.

The contribution by Gullì (2013) takes into account these two latter factors and for this reason is complementary to the analysis by Sijm et al. (2012).

In particular, Gullì (2013) takes into account the following important features of electricity markets:

(1) Power systems include several types of power generating units with different variable costs and different emissions rates.
(2) Power demand varies cyclically over time within a reference time-period (day, year). This feature is crucial in order to understand how firms pass-through the carbon price to energy prices. In fact, if demand varies over time, this implies that strategic firms do not always prefer to bid above competitive prices. In some periods a strategic firm may prefer to engage in Bertrand competition with its rivals by bidding the rivals' marginal cost or by bidding its own marginal costs (pure Bertrand competition). This implies that in some periods the pass-through may be higher or lower than the fully competitive pass-through depending on the emissions rates of power plants operating in the market.
(3) Since price elasticity of electricity demand is very low, firms might not maximise profits because of the regulatory pressure exerted by competition and sector-specific authorities (authorities' monitoring and possible intervention to avoid excessive profits). In this case firms pursue strategies besides profit maximisation. In particular, because of the above-mentioned regulatory pressure, firm's prices may be constrained to be below some (implicit) threshold. In other words: firms may restrain themselves to bid above some presumed threshold (implicit price cap) in order to avoid the risk of more restrictive regulation. A typical situation in which firms' offer prices are below their profit-maximising level is when those firms try to meet a profit target, namely a supposed equitable target minimising the risk of regulatory intervention. This latter may be a short-term (for example, one-period) target or a long-term target (for example, multi-period target). The short-term target consists of keeping constant the profit period by period. The long-term target consists of keeping constant the profit over time while minimizing price volatility. In the latter case, firms try to keep constant the multi-period average profit. The time-period considered is the settlement period for the carbon price (e.g. daily auctions) or a multiple of these periods. In other words, power firms do not bid the price maximising profit. They bid a (lower) price consistent with their profit target (lower than the maximum possible profit).
(4) Also the input market (for power generation), and not only the output one (electricity market), might be imperfectly competitive. This feature is crucial in order to account for the fact that the main input of power generation in several countries is natural gas. Imperfect competition in the gas market is likely (especially in the EU markets where the number of natural gas suppliers is low). A suitable way of accounting for this is assuming that gas firms set prices according to

the net long run marginal cost of alternative technologies to natural gas installations in the main consumption segments (referred to as the 'market value principle').

By taking into account these features, Gullì (2013) finds that:

(1) If the input market is imperfectly competitive, the marginal cost of gas fired power generation is not related to its own emissions rate. It is related to the emissions rate of the alternative fuel to natural gas.
(2) As a consequence, when the input market is imperfectly competitive, the pass-through can be lower than that under full competition (related to the emissions rate of a natural gas fired plant). It is true even if the output market is fully competitive and provided that the alternative technology to gas fired power installations is low polluting (that is, less polluting than natural gas). This may explain empirical evidence for very low pass-through even when electricity markets (but not gas markets) appear to be very close to full competition.
(3) When the output market is imperfectly competitive and firms pursue a short-term profit target, the pass-through may be much higher than that under full competition (even considering linear demand curves). In this case, keeping profits constant implies a pass-through that is much higher than the increase in cost.
(4) If firms pursue a profit target, instead of maximising profits, pass-through depends on the kind of allowance allocation, whether auctioned or allocated free of charge and in the latter case also on the type of free allocation.
(5) When both markets are imperfectly competitive and firms pursue a long-term profit target, pass-through may be much lower than that under full competition and even negative especially if (but not only if) allowances are allocated free of charge.

In sum, the theoretical literature includes two core contributions helpful to interpret the empirical results described below. The major differences among the theoretical models are the following.

Sijm et al. (2012) assume quantity (Cournot) competition in output markets and find that the pass-through can assume a wide range of values (lower and higher than one) depending on the structural features of output markets, namely the shape of the demand curve and its price elasticity as well as the number of firms active in the market. In our opinion, their model has three limitations. First, quantity competition is not well suited to simulate spot power markets. Second, it does not also take into account imperfect competition in the input markets (input for power generation).

Third, this model cannot explain why empirical analyses find negative pass-through to electricity prices.

Gullì (2013) assumes price competition, which is more suited to simulate electricity wholesale spot markets. This contribution also attempts to evaluate what happens when also input markets (for power generation) are imperfectly competitive. In line with the analysis by Sijm et al., Gullì finds that the pass-through can assume a wide range of values (even if electricity markets are fully competitive, provided that the input market is imperfectly competitive). In addition, Gullì's model is able to explain why the pass-through can even be negative. The major limitation of this theoretical analysis is that it assumes a dominant firm model while the electricity market structure would be better described by an oligopolistic framework.

In sum: the theoretical literature suggests that the range of possible values of PTR is very wide. The PTR may be lower and higher than one, and even negative, depending on a number of factors related to the configuration of power markets.

3.3 EMPIRICAL LITERATURE

In contrast with the theoretical literature, the empirical literature on carbon price pass-through is very diverse.[8] Most contributions are based on econometric analyses whilst just a few contributions use analytical approaches and 'visual' interpretation (just looking at graphical paths). Almost all focus on the first and second phases of the EU ETS.

Table 7.1 presents the summary of the estimated pass-through rates (PTR) on the power markets analysed in the literature. The results of this literature are commented upon below, in light of the theoretical contributions reported above.

Summing up, the values in Table 7.1 are significant in all cases and suggest that the bandwidth for pass-through can be quite large, varying between countries and periods. The estimates range from values much lower than one and much higher than one, either if we refer to the average value or to the values in the peak and off-peak hours. Furthermore, the analysis of how the PTR is distributed over time (among peak and off-peak periods) would seem to suggest that there is not the same behaviour everywhere (that is, a general rule). The PTR is higher during peak than off-peak hours in some countries, but vice versa in other countries. The

8 For a review, see also Gullì and Chernyavs'ka (2013).

Table 7.1 Carbon cost pass-through rate: empirical results

Country and study	Methodology	Price	Period	Average	Peak	Off-peak	
						Mid-merit	Very off-peak
Finland (Honkatukia et al., 2008)	Econometric VEAC and AR-GARCH	Wholesale spot	2005 ÷ 2006	0.5 to1.0			
France (Solier and Jouvet, 2011)	Econometric autoregressive	Wholesale spot	2005 ÷ 2006		0.17 to 1.75	0.65 to 1.05	
France (Solier and Jouvet, 2011)	Econometric autoregressive	Wholesale spot (forward)	2008 ÷ 2010		–0.49 to 0.27 (1.14 to 1.71)	–0.46 to −0.21 (1.04 to 1.57)	
Germany (Bunn and Fezzi, 2008)	Econometric VEAC	Wholesale spot	2005 ÷ 2006	0.52			
Germany (Sijm et al., 2008a)	Econometric OLS	Wholesale spot (forward)	2005 2006		0.60 0.57	0.41 0.64	
Germany (Solier and Jouvet, 2011)	Econometric autoregressive	Wholesale spot	2005 ÷ 2006		–0.34 to 1.18	0.47 to 1.03	
Germany (Solier and Jouvet, 2011)	Econometric autoregressive	Wholesale spot (forward)	2008 ÷ 2010		–0.66 to 0.48 (1.22÷1.86)	–1.29 to 0.15 (1.05 ÷ 1.61)	

Table 7.1 (continued)

Country and study	Methodology	Price	Period	Average	Peak	Off-peak	
						Mid-merit	Very off-peak
Italy-whole (Chernyavs'ka and Gullì, 2008a)	Load duration curve approach	Wholesale spot	2006		1.1 to 1.5	1.2 to 1.5	0.9 to 1.1
Italy (Solier and Jouvet, 2011)	Econometric autoregressive	Wholesale spot	2008 ÷ 2010		–6.39 to −1.23	–5.43 to 1.01	
Spain (Solier and Jouvet, 2011)	Econometric autoregressive	Wholesale spot	2005 ÷ 2006		1.29 to 2.03	–0.18 to 0.67	
Spain (Solier and Jouvet, 2011)	Econometric autoregressive	Wholesale spot	2008 ÷ 2010		–2.98÷3.43	–0.76 to 4.24	
The Netherlands (Solier and Jouvet, 2011)	Econometric autoregressive	Wholesale spot	2005 ÷ 2006		0.33 to 0.79	–0.3 to 0.99	
The Netherlands (Solier and Jouvet, 2011)	Econometric autoregressive	Wholesale spot (forward)	2008 ÷ 2010		–4.36 to 4.56 (0.38 to 0.73)	–0.74 to 0.53 (1.25 to 1.51)	

The Netherlands (Sijm et al., 2008b)	Econometric OLS	Wholesale spot (forward)	2005		1.34	0.40
			2006		1.10	0.38
United Kingdom (Solier and Jouvet, 2011)	Econometric autoregressive	Wholesale spot	2005 ÷ 2006		0.83 to 1.12	0.57 to 1.66
United Kingdom (Solier and Jouvet, 2011)	Econometric autoregressive	Wholesale spot (forward)	2008 ÷ 2010		2.83÷3.69 (0.52 to 2.32)	–0.97 to 0.37 (1.25 to 1.51)
United Kingdom (Bunn and Fezzi, 2008)	Econometric VEAC	Wholesale spot	2005 ÷ 2006	0.30		

Note: VEAC = Vector Error Correction Model; AR-GARCH = Autoregressive General Autoregressive Conditional Heteroskedasticity Model; OLS = Ordinary Least Square.

Source: Gullì and Chernyavs'ka (2013).

overall picture, therefore, would seem to support the conclusions of the theoretical analysis, that is, we cannot know in advance how an ETS will impact power prices (whether the pass-through rate is low or high, more or less than one) without carefully accounting for the structural features of the power markets in the analysis.

Furthermore, the estimates have to be interpreted with due care as, to some extent, they depend on (shortcomings of) the data, the methodologies used and the assumptions made. Concerning the methodological issues, many empirical methods can be used in order to estimate carbon price pass-through rates. Each of them shows strengths and weaknesses, so that one approach cannot be considered as definitively preferable to another. Nevertheless, it is at least worth stressing some important differences between the econometric and non-econometric techniques.

The econometric approach uses sophisticated statistical tools in order to measure carbon price pass-through rates. It is based on the statistical elaboration of time-series of either forward or spot prices (of both electricity and carbon) and estimates the impact of the ETS on the average price, eventually distinguishing between the peak and off-peak hours. The specifications of these models are generally quite simple. The set of drivers commonly includes fuel costs and temperature (Sijm et al., 2012; Bunn and Fezzi, 2008). Only one model (Honkatukia et al., 2008) uses additional variables (namely, the production capacity and the utilization of the transmission capacity). Furthermore, models neither consider the real marginal technology hour by hour nor are suited to capture (by using appropriate drivers) the effect of market power. They assume that during the observation period power prices are set by a single (marginal) technology with a fixed, generic fuel efficiency. In other words, the econometric models are very useful to provide a precise (statistically significant) value of the carbon pass-through, but they are not able to justify this value, that is, to explain why a pass-through rate is high, low, nil or even negative.

The non-econometric approach (Chernyavs'ka and Gullì, 2008a) allows us to obtain the pass-through rate hour by hour. It shows several advantages compared to the econometric one. First, as pointed out before, it provides a detailed analysis of the pass-through over time, on an hourly basis. As a result, it seems to be well suited to describe the impact of market power whose extent depends on the level of power demand (and hence on the time of consumption). Moreover, by using this approach, market power can be effectively simulated by means of a theoretical model assuming oligopolistic competition or a dominant firm framework. Second, the non-econometric approach allows us to take into account other important structural factors which can hardly be included in econometric models, such as the technological mix and the available production capacity in the market. However, unlike the statistical approach, it does not provide a precise value of the pass-through rate but only a range of its variability. In this sense, the two approaches are complementary. The non-econometric one is useful to improve the specification of the statistical models and to interpret their results.

The synergy between the econometric analysis carried out by Solier and Jouvet (2011) and the non-econometric and theoretical analyses by Chernyavs'ka and Gullì (2008a) and Gullì (2013) is an emblematic example of this complementarity.

As pointed out before, Solier and Jouvet (2011), trying to explain negative pass-through in the second phase of the EU ETS, conclude that negative pass-through rates derive from market instability due the economic crisis that started in 2008 with its negative impact on power demand and on power price volatility.

The theoretical contribution by Gullì (2013), combined with the non-econometric empirical analysis carried out by Chernyavs'ka and Gullì (2008b), improves our ability to interpret this result and helps us to estimate actual carbon cost pass-through rates.

This contribution shows that the overall change in prices can include two components. One is independent of environmental regulation and is due to the economic crisis that started in 2008. Because of this crisis, the years 2008 to 2010 were characterised by excess power generation capacity. In most countries, this led to a sharp decrease in market power and consequently to a sharp decrease in prices. The other component is related to the ETS, namely to the change in market power due to the increase in carbon price during the transition from the first to the second phase of the EU ETS (from 2007 to 2009).

By identifying and explaining how environmental regulation can impact upon market power, Gullì (2013) provides the analytical tool for separating these two effects (one due to the economic crisis and the other due to the ETS). This may help us to isolate the 'ETS effect' and, as a consequence, to estimate the actual carbon cost pass-through rate.

Therefore, if econometric models (to estimate carbon cost pass-through) do not account for the change in structural conditions of power markets, as suggested by Gullì (2013), they risk attributing the drop in power prices to the change in carbon prices (negative pass-through rates). Consequently, what empirical analyses estimate is not so much a pass-through of carbon prices to electricity prices but rather the effect of the change in market structure due to exogenous factors, namely market instability due to the economic crisis. This confirms that econometric models alone may lead to incorrect representations of the correlation between carbon and energy markets as long as they are not able to adequately (and directly) capture the underlying impact that market structure has on strategic behavior of firms and on price formation.

This critical review of the economic literature seems to suggest that econometric analyses should always be supported by effective theoretical frameworks.

4. WINDFALL PROFITS IN THE EU POWER SECTOR

In this section we aim to estimate the windfall profit rate (WPR) that is the share of the marginal carbon opportunity cost converted to additional profit (see equation (7.1)). To calculate the windfall profit rate, we need three kinds of information: (1) the pass-through rate; (2) the ratio of emissions rates (infra-marginal rate divided by marginal rate); and (3) the rate of free allocation.

The pass-through rate is estimated by using the results of the existing literature on this topic (see Table 7.1).

The rate of free allocation is equal to the amount of allowances allocated for free divided by the verified emissions for each country (average values in the first and second trading periods of the 2005–2012 EU ETS). We should also take into account the share of allowances auctioned, but this share is only around 10 percent in Germany, the Netherlands and United Kingdom in the second trading period. Therefore, keeping the assumption of full free allocation (see appendix equation (7A.4)) does not undermine the significance of the analysis.

The ratio of emissions rates is equal to the emissions rate of the representative infra-marginal technology divided by the emissions rate of the representative marginal technology, for each year and each country. The former is the average emissions rate of the entire power generating system. The second is the average emissions rate of the mix of marginal generating units (the units which set prices hour by hour in each country). Table 7.2 reports the total average WPR and the related components (price-induced windfall profit rate (PWPR) and regulated-induced windfall profit rate (RWPR) in each trading period.

As can be noted, in the first period (2005–2007) of the EU ETS:

(1) the PWPR (price-induced windfall profit rate) contribution is ambiguous. Except for Italy and France, it can be either positive or negative reflecting the wide range of PTR already shown in Table 7.1;
(2) the RWPR (regulated-induced windfall profit rate) is the most

Table 7.2 Windfall profit rate (EU ETS phases I and II)

Country	Phase	PTR	Free allocation rate (FAR)	rj/ri	PWPR=PTR-rj/ri	RWPR=rj/ri FAR	Total
Finland	I	0.51	1.10	1.55	–1.04	1.71	0.67
France	I	0.37 to 1.31	1.23	0.40	0.03 to 0.91	0.49	0.46 to 1.40
Germany	I	0.18 to 1.17	1.02	0.57	–0.39 to 0.60	0.58	0.19 to 1.18
Italy	I	1.10 to 1.30	0.87	1.00	0.10 to 0.30	0.87	0.97 to 1.17
Netherlands	I	0.06 to 0.98	1.01	1.00	–0.94 to –0.02	1.01	0.07 to 0.99
Spain	I	0.65 to 1.45	0.74	0.85	–0.20 to 0.60	0.63	0.43 to 1.23
United Kingdom	I	0.35 to 1.47	0.80	1.00	–0.65 to 0.47	0.80	0.15 to 1.27
France	II	–0,42 to 0.06	1.16	0.40	–0.82 to –0.34	0.46	–0.36 to 0.12
Germany	II	–0.87 to 0.44	0.75	0.56	–1.43 to –0.12	0.42	–1.01 to 0.30
Italy	II	–5.80 to 0,06	0.98	0.86	–6.66 to –0.80	0.84	–5.82 to 0.04
Netherlands	II	–2.26 to 2.45	0.93	0.98	–3.24 to 1.47	0.91	–2.33 to 2.38
Spain	II	–1.60 to 4.04	0.85	0.57	–2.17 to 3.47	0.49	–1.69 to 3.95
United Kingdom	II	2.93 to 3.79	0.83	0.92	2.01 to 2.87	0.76	2.77 to 3.63

important component in all countries. Consequently, windfall profits are mostly due to the decision to allocate the allowances for free;

(3) there is a trade-off between PWPR and RWPR. This trade-off is due to the fact that either the PWPR or the RWPR depends on the ratio of allocation (the PWPR inversely and the RWPR positively). There is no correlation between the pass-through rate and the degree of free allocation, as expected;

(4) the RWPR is able to compensate the negative values of the PWPR everywhere. Thus, regulated-induced (free allocation) windfall profits occurred in all EU Member States. In particular, on average more than the half of the carbon cost was converted to regulated-induced windfall profits.

The framework significantly changes in the second period (2008–2012) of the EU ETS:

(1) Except for the United Kingdom, negative values of PWPR seem to be more likely (the range of negative value is larger).

(2) At the same time, the RWPR is lower than in the first period everywhere.

(3) As a consequence, except for the United Kingdom (where it is always positive), the WPR is generally ambiguous. It ranges from negative to positive values, depending on the variability and uncertainty of the pass-through rate (PTR) estimations.

Overall, these results suggest that under the EU ETS the power generators may be able to gain windfall profits. However, this additional profit (if any) is mainly regulation-driven. If any, windfall profits are mostly due to the policy decision to allocate the emissions allowances for free. However, the price-induced windfall profits seem to be ambiguous because of the uncertainty and variability of the pass-through rate (PTR) of carbon costs.

5. 'NEUTRALISING' WINDFALL PROFITS IN THE POWER SECTOR: THE IMPORTANT CASE OF THE SPANISH LEVY

As discussed above, windfall profits can be perceived as unfair. This is the reason why, in some EU countries, policymakers and regulators discussed the possibility of 'neutralising' this kind of (considered undeserved) profits. In the literature the increasing use of auctioning or the use of a windfall profit tax is being proposed (see CE Delft (2010b) p. 43 for examples of such taxes).

In the Spanish electricity market emissions costs are almost fully passed through to electricity prices (Fabra and Reguant, 2014). The Spanish levy, set up in 2006, is one of the most interesting cases.[9] This levy was equivalent to the surplus revenue obtained by electricity suppliers as a consequence of the integration of the value of the free allowances in their costs. The levy applied to all installations in the ordinary regime, thus excluding renewable energies and co-generation. Two different formulations were applied to generating units under the ordinary regime, depending on whether they were covered by the ETS or not (that is, on whether received CO_2 allowances or not). For those units to which no allowances were allocated, the amount to be paid was determined by multiplying the energy produced by the carbon cost of a combined cycle unit (this is given by the allowance price multiplied by the emission rate to the combined cycle). The generating units to which allowances were allocated paid an amount equal to the load factor (expressed as the number of days during which the installation sold electricity during the reference period divided by 365) multiplied by the quantity of allocated allowances for 2006, the average allowance price and the ratio between the emission rate of a natural gas combined cycle installation and the emission factor of the corresponding generating unit.

Looking at these approaches, we observe that: (1) the levy was paid also by free carbon technologies; (2) the levy depends on the price-through rate (assumed full and equal to the carbon cost of a combined cycle unit); (3) the higher the carbon emission rate of the generating unit the lower the amount to be paid.

On the basis of the analysis carried out in this chapter, none of these provisions would be correct. In fact this analysis suggests that only the regulated-induced windfall profits (RWPR) should be neutralised. Thus: (1) the levy should not depend on the pass-through rate (PTR) included only in the price-induced windfall profits (whose existence is independent on free allocation); (2) all the free carbon technologies (whose emission rate is nil) should not pay the levy; (3) the higher the emission rate the higher the amount of levy to be paid.

As can be noted, what the theoretical framework seems to suggest is opposite to what the Spanish levy implied. This means that the attempt to absorb windfall profits may be risky, leading to inefficient (and unfair) solutions from an economic point of view. Abandoning free allocation in

[9] For an interesting discussion about the legal issues of this case, see Rodriguez (2014).

favour of other solutions (for example, auctioning) surely has been a good choice to avoid this risk and consequently to improve the efficiency of the EU ETS.

6. CONCLUSIONS

We define the windfall profit rate (WPR) as the share of the environmental cost converted to additional profit for firms. This indicator allows us to understand how much windfall profits are likely to arise independent of the carbon price.

This chapter explores the determinants of the WPR when the environmental regulation is based on an emissions trading scheme (ETS). The focus is on the EU ETS and on the power generation sector (one of the most important polluting sector in terms of carbon emissions).

Two categories of WPR have been identified and estimated:

(a) price-induced WPR (PWPR), which may arise independently on how emissions allowances are allocated (either for free or auctioned);
(b) regulation-induced WPR (RWPR), arising only if allowances are allocated free of charge.

Focusing on the EU ETS power sector during the years 2005–2012, we find that firms are able to gain windfall profits. The analysis shows that these additional profits are mainly regulation-driven in the sense that they are mostly due to the policy decision to allocate the emissions allowances free of charge. Furthermore, the price-induced effect (independently on how allowances are allocated) seems to be ambiguous because of the variability of the carbon cost pass-through rate (PTR). This variability is the consequence of either the different structural conditions of the power markets or the uncertainty about the related estimations.

The fact that free allocation could imply windfall profits was not a surprise. This effect was known in the literature when Directive 2003/87/EC was designed and approved. However, policymakers still chose free allocation.[10] Why? That choice was presumably based on three pre-assumptions (or preconceptions), namely that: (1) by reducing the impact on production costs, free allocation would help to preserve the competitiveness of the industrial sectors engaged in international

[10] For a discussion on the political reasons of this choice, see also Woerdman et al. (2009)

competition; (2) free allocation would not affect the ability to meet the environmental target (which depends on the cap and not on the method of allocation); (3) probably, the pass-through rate would be nil or very low thanks to free allocation.

After the first year of the operation of the EU ETS, these assumptions have been largely criticised for these reasons.

- First, allowances have been allocated for free also to firms belonging to sectors which are not engaged in international competition (for example, power sector). This provision suggests that the free allocation has been mainly a political decision: making the scheme more acceptable for producers.
- Second, the fact that allowances are allocated free of charge does not imply that the carbon cost is not passed through to prices. Regardless of the kind of allocation, the ETS creates an opportunity cost which may be correctly passed on to consumers, to some extent. This effect became clear after the first year of the ETS, especially in the electricity wholesale markets. Therefore, the preconception that free allocation would reduce the impact on prices is not economically correct.
- Third, it is true that (at least in principle) the free allocation does not undermine the ability to meet the environmental target. However, if the polluting firms can gain a windfall profit and pursue a profit target (rather than profit maximisation), they might have less incentives to reduce pollution (to abate emissions beyond their specific compliance target). Consequently, the target would be met but the overall compliance cost would not be minimised (which is the main economic objective of an ETS).
- Finally, windfall profits might be 'neutralised' by applying a levy to producers. Such a levy should not be paid by all carbon free technologies (even if they gain price-induced windfall profits), but the more polluting installations should also pay more, whereas the levy should be independent of the carbon cost pass-through to prices. If we look at the Spanish levy, we observe that these conditions were not met. This highlights that the attempt to absorb windfall profits may be risky, leading to inefficient (and unfair) solutions from an economic point of view.

Given these justified criticisms, we believe that the policy decision to abandon free allocation in the 2013–2020 EU ETS (from 2013 in the power sector and gradually in other industries, keeping some exceptions) could be considered as appropriate. Auctioning of emissions allowances makes the environmental regulation more acceptable for consumers.

However, it is worth underlining that the controversy about the allocation method was exaggerated in the past. Perhaps the debate about this question overshadowed other important issues, which appeared more relevant over time, namely: how to deal with the risk of over-allocation when the expected emissions are highly uncertain; should we adopt a (regulated) minimum (and/or maximum) allowance price; should we include other polluting industries (for example, transport sector) and, first of all, should we insist on a 'quota' mechanism (cap-and-trade regulation) rather than adopting a price mechanism (carbon tax), given the current weaknesses of the cap-and-trade regulation, which are beyond the choice of the allocation method.

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APPENDIX

The WPR is referred to as the share of marginal carbon cost (the carbon cost of the marginal technology in the wholesale spot market) converted to windfall profits.

To explain the fundamental mechanisms through which power generation costs are affected by the carbon price, three conceptual steps have to be considered.

First, ETS creates a market for emissions allowances. Since the allowances have a value, their use generates an opportunity cost (hereafter the carbon cost) equal to the allowance price, τ, multiplied by the emission rate, r_j. Free allowances have an opportunity cost when they are used for covering the emissions. Instead of using the allowances, the firm could have sold them. These costs are part of the cost price and thus have to be incorporated in the electricity price. An energy producer will not sell his allowances unless he can earn the revenue forgone via the electricity price.

Second, ETS determines an increase in the unit variable cost equal to the carbon cost. This cost arises even if the public authority allocates allowances free of charge (because of the opportunity costs of free allowances).

Third, the value of the freely allocated allowances constitutes a 'gift' to the generator. Consequently, if we want to calculate the long run marginal cost of producing electricity (which includes fixed components), the unit value of these allowances (the value per unit of electricity generated) should be deducted from the overall cost of power generation.

Then the total cost of power generation (net of fixed costs) will be:

$$TC_i(q_i, r_i, \tau) = c_i q_i + \tau r_i q_i - \tau \overline{E}_i \tag{7A.1}$$

where:
c_i = variable costs of electricity generation (fuel cost)
q_i = amount of generated electricity
$\overline{E}_i$ = amount of allowances allocated free of charge.
As a consequence, the marginal cost:

$$MC_i(r_i, \tau) = c_i + \tau\, r_i \tag{7A.2}$$

includes the entire carbon cost regardless of whether allowances are allocated free of charge or auctioned. This implies that, in fully competitive spot markets, the ETS-impact on product prices (that is, the so-called pass-through rate) does not depend on the method of allowances allocation. The fact that the allowances are allocated free of charge does not imply lower pass-through rates. In other words, at least in principle, the degree of free allocation and the pass-through rate are not inter-correlated.

Therefore, the change in profit of power firms will also depend on how the carbon cost will be passed-through to energy prices. To estimate the change in profits, we use a simple model based on two periods (period 1, before the ETS, and period 2, after the ETS) and two technologies: the marginal technology *i* (the technology-setting prices in the spot market) and the infra-marginal technology *j* (the technology whose bid price is lower than that of the marginal technology).

Let p_1 and p_2 be the power prices before and after the ETS implementation, respectively. Then, assuming that the power demand is completely inelastic and that there is full free allocation, the change in profit of the infra-marginal technology *j* will be:

$$\Delta\pi_j = (p_2 - p_1)q_j - (r_j\tau_j q_j - \overline{E}_j) \tag{7A.3}$$

Where:
q_j = production
r_j = emission rate
τ = allowance price
$\overline{E}_j$ = value of the allowances allocated free of charge

By dividing equation (7A.3) by the carbon cost of the marginal technology and by q_j, we find the share of the marginal carbon cost converted to additional profit of the technology *j*. This is the windfall profit rate (WPR):

$$WPR = \left(PTR - \frac{r_j}{r_i}\right) + \frac{r_j}{r_i}FAR = PWPR + RWPR \tag{7A.4}$$

Where $WPR = \Delta\pi_j/(r_j\tau q_j)$ is the windfall profit rate (the share of the marginal carbon opportunity cost converted to additional profit), $PTR = (p_2 - p_1)/r_i\tau$ is the marginal pass-through rate and $FAR = \overline{q}_i/q_i$ is the free allocation rate (the share of emissions allowances allocated for free). We assume that the regulator allocates the allowances on the basis of the expected future production of the different kinds of power generating technologies.[11] Note that the free allocation rate is equal to the allocation rate (the amount of allowances allocated divided by verified emissions), since we assume full free allocation.

[11] We adopt this assumption for the sake of simplicity. Other solutions can be adopted. For example, the regulator might also assume an emissions rate lower than the current one, alone or combined with the expected emission or assuming a pure grandfathering (historical emissions). However, our assumption does not undermine the significance of the model.

8. Reviewing the evidence on the innovation impact of the EU Emission Trading System

Karoline S. Rogge

1. INTRODUCTION

At the international climate conference in Paris in December 2015 the political leaders of the world agreed to hold the global average temperature increase well below 2°C and to pursue efforts to limit the temperature increase to 1.5°C.[1] Translated into global greenhouse gas emissions this implies that these should peak as soon as possible and thereafter be rapidly reduced so as to achieve the decarbonization of the economy in the second half of this century. This massive transformation calls for a radical redirection and acceleration of technological change towards low- and particularly zero-carbon solutions, which in turn necessitates policies inducing such changes. Global carbon pricing is seen as a key enabler for such a decarbonization of the economy, and the European Union's Emission Trading System (EU ETS) as the world's largest and first multi-country greenhouse gas emission trading system may serve as pilot and starting point for implementing such carbon pricing. Therefore, in this chapter I review the empirical evidence on the innovation impact of the EU ETS, based on which I discuss the role such a cap-and-trade instrument can play in achieving a radical transformation towards a decarbonized economy.

Under the EU ETS, a certain absolute number of greenhouse gas emission allowances (EUAs) are allocated per year, where one EUA gives the right to emit one ton of CO_2e.[2] Operators can trade these EUAs at the market, and have to surrender the number of allowances equivalent to the amount of CO_2 emissions caused by their installations during the previous year. Ideally, such a cap-and-trade approach ensures that emissions are reduced where it is cheapest to do so, and that the market price for EUAs reflects the scarcity of allowances in the system. Eventually, the market

[1] *See* UNFCCC, Paris Agreement 2015.

[2] Carbon dioxide equivalent. For simplicity in the remainder of the chapter I will simply refer to CO_2.

mechanism ensures that all participants face the same marginal abatement costs so that overall reduction costs are minimized *(static efficiency)*.

In addition, the EUA price also sets monetary incentives to adopt new technologies or implement new processes with lower emissions and to invest in research and development (R&D) in low-carbon technologies *(dynamic efficiency)*. This direct innovation impact can be differentiated into the effect occurring at EU ETS firms and the one relevant for other actors conducting R&D on low-carbon technologies, such as technology providers, universities or research institutes. If the additional costs for CO_2 emissions are passed on and included in the product prices of EU ETS firms, emission trading may also induce indirect innovation effects on the demand side where those products are used as inputs. However, in this chapter I focus on the scheme's direct impact on innovation.

Focusing on the impact of the EU ETS on low-carbon innovation is particularly well justified in the context of long-term transformative change such as the envisioned decarbonization of the economy. Indeed, environmentally-friendly technological innovation has already early on been identified as 'perhaps the single most important criterion on which to judge environmental policies' in the long haul.[3] In addition, the EU Commission states as one of the major goals of the EU ETS the promotion of global innovation to act against climate change.[4] It is therefore of utmost importance for policy evaluation studies to judge the EU ETS against this criteria.

Yet, judging the impact of the EU ETS on innovation is no straightforward exercise, as innovation is a complex and systemic phenomenon.[5] Given the difficulties involved in measuring innovation as a dynamic, interactive and uncertain process it may not come as a surprise that studies evaluating the innovation impact of a single instrument typically follow a rather linear understanding of the innovation process – often separating it in the three stages of invention, innovation and diffusion, or into innovation and diffusion.[6] As is done in other contexts, studies on the innovation impact of the EU ETS utilize different input- and output-based indicators of the innovation process, such as expenditures for research and

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[3] *See* AV Kneese and CL Schultz, *Pollution, Prices and Public Policy* (Brookings Institute 1978).

[4] *See* European Commission, 'EU Action against Climate Change: EU Emissions Trading — an Open Scheme Promoting Global Innovation' (2005).

[5] *See* J Fagerberg, DC Mowery and RR Nelson, *The Oxford Handbook of Innovation* (Oxford University Press 2005).

[6] *See* JA Schumpeter, *Capitalism, Socialism and Democracy* (Harper and Brothers 1942).

development (R&D), innovation activities, patents, or innovations. Both quantitative and qualitative as well as mixed method research designs are applied for studying the innovation impact of the EU ETS, including econometric analysis of patent data, regression analysis of company survey data, case study analysis based on interviews with company representatives, and expert interviews.

In this chapter, I follow the OECD Oslo Manual in defining innovation as 'the implementation of a new or significantly improved product (good or service), or process, a new marketing method, or a new organizational method in business practices, workplace organization or external relations'.[7] Given the chapter's focus on the EU ETS and climate change mitigation, I investigate findings on the impact of the EU ETS on low-carbon innovations which reduce greenhouse gas emissions. In the context of this chapter I further differentiate these into low-carbon technological and organizational innovation. The former covers both low-carbon product and process innovations, such as significant improvements in technical specifications, components and materials, or other functional characteristics, and in production and delivery methods. The latter refers to the implementation of new organizational methods, such as changes in business practices and in workplace organization which may facilitate the reduction of greenhouse gas emissions.

Before presenting the results of existing studies on the innovation impact of the EU ETS, in section 2, I will first provide an overview of the expected innovation impact of the EU ETS. In section 3, I will examine the empirical evidence of the impact of the EU ETS on technological innovation, and in doing so will differentiate between its three trading phases (2005–2007, 2008–12, 2013–2020). In section 4, I will then present the findings on the scheme's impact on organizational innovation. In section 5, I conclude the chapter with some final observations on the innovation impact of the EU ETS as one instrument in the climate policy mix. I also offer some methodological recommendations for future studies and suggest implications for policy makers interested in the decarbonization of the economy.

2. EXPECTED INNOVATION IMPACT

Economists have long argued for the superiority of market-based instruments, such as the EU ETS, in terms of cost efficiency and their continued

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[7] *See* p. 46 in OECD and Eurostat, 'Oslo Manual: Guidelines for Collecting and Interpreting Innovation Data' (3rd edn, OECD and Eurostat 2005).

provision of innovation incentives.[8] In addition to these theoretical claims the empirical literature examining market-based environmental policies has delivered important insights into actual innovation incentives by studying the first applications of permit trading in the United States for pollutants such as SO_2, NO_x, and lead.[9] Building on the theoretical environmental economics literature[10] and empirical evidence from US trading schemes,[11] the innovation impact of the EU ETS was expected to be rather low, at least in its first phase.[12] In addition, some early studies have estimated the potential innovation impact of the EU ETS by identifying design features that could be important in determining this effect.[13]

This attention to the scheme's design features rests on the increasing recognition that rather than the instrument type, what seems to be more influential for innovation are its design features,[14] such as its stringency,[15]

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[8] *See* WJ Baumol and WE Oates, *The Theory of Environmental Policy* (Cambridge University Press 1988); T Requate, 'Dynamic Incentives by Environmental Policy Instruments – a Survey' (2005) 54 *Ecological Economics* 175.

[9] For an overview, *see* B Hansjürgens, *Emissions Trading for Climate Policy – U.S. and European Perspectives* (Cambridge University Press 2006).

[10] *See*, for example, C Fischer, IWH Parry and WA Pizer, 'Instrument Choice for Environmental Protection When Technological Innovation is Endogenous' (2003) 45 *Journal of Environmental Economics and Management* 523; CH Jung, K Krutilla and R Boyd, 'Incentives for Advanced Pollution Abatement Technology at the Industry Level: An Evaluation of Policy Alternatives' (1996) 30 *Journal of Environmental Economics and Management* 95; DA Malueg, 'Emission Credit Trading and the Incentive to Adopt New Pollution-Abatement Technology' (1989) 16 *Journal of Environmental Economics and Management* 52.

[11] *See*, for example, S Kerr and RG Newell, 'Policy-Induced Technology Adoption: Evidence from the US Lead Phasedown' (2003) 51 *Journal of Industrial Economics* 317; D Popp, 'Pollution Control Innovations and the Clean Air Act of 1990' (2003) 22 *Journal of Policy Analysis and Management* 641.

[12] *See* F Gagelmann and M Frondel, 'The Impact of Emission Trading on Innovation – Science Fiction or Reality?' (2005) 15 *European Environment* 203.

[13] *See* F Gagelmann and B Hansjürgens, 'Climate Protection through Tradable Permits: The EU Proposal for a CO_2 Emissions Trading System in Europe' (2002) 12 *European Environment* 185.

[14] *See* R Kemp and S Pontoglio, 'The Innovation Effects of Environmental Policy Instruments — A Typical Case of the Blind Men and the Elephant?' (2011) 72 *Ecological Economics* 28; KS Rogge and K Reichardt, 'Towards a More Comprehensive Policy Mix Conceptualization for Environmental Technological Change' (2013) S 3/2013, Fraunhofer ISI.

[15] *See* NA Ashford, C Ayers and RF Stone, 'Using Regulation to Change the Market for Innovation' (1985) 9 *Harvard Environmental Law Review* 419; M Frondel, J Horbach and K Rennings, 'What Triggers Environmental Management and Innovation? Empirical Evidence for Germany' (2008) 66 *Ecological Economics*

predictability[16] or flexibility.[17] These design features have been analyzed in detail, for example, for the pilot phase of the EU ETS from 2005–2007 and for its second trading phase 2008–2012.[18] Based on Schleich and Betz (2005), Table 8.1 provides a summary of those design features which may be most relevant for the innovation impact of the EU ETS: (1) the cap or emission budget, (2) the rules of banking from one period to the next, (3) the allocation method for existing installations, (4) the treatment of new entrants, including transfer rules from existing to new installations, (5) allocation rules for the closure of installations, and (6) information provided about future allocations.[19]

Based on an analysis of its design features for its pilot phase a limited innovation impact of the EU ETS is anticipated. This can be traced back to the scheme's lenient cap, generous links to the project-based Kyoto Mechanisms Clean Development Mechanism (CDM) and Joint Implementation (JI) which further reduce EUA prices, the prohibition of banking allowances into the second trading period and the negligible role of auctioning as an allocation mechanism. In addition, the free allocation to new entrants based on differentiated benchmarks, the termination of free allocations to closing plants as well as uncertainty about future rules are all said to weaken the scheme's expected innovation impact. However, incentives for low-carbon innovation generated by the EU ETS have improved between phase 1 and phase 2, for example through tighter emission caps and the gradual introduction of auctioning. Some of the remaining shortcomings in the scheme's design have been addressed within

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153; M Frondel, J Horbach and K Rennings, 'End-of-Pipe or Cleaner Production? An Empirical Comparison of Environmental Innovation Decisions Across OECD Countries' (2007) 16 *Business Strategy and the Environment* 571.

[16] *See* VH Hoffmann, T Trautmann and J Hamprecht, 'Regulatory Uncertainty: A Reason to Postpone Investments? Not Necessarily' (2009) 46(7) *Journal of Management Studies* 1227.

[17] *See* I Hascic, N Johnstone and M Kalamova, 'Environmental Policy Flexibility, Search and Innovation' (2009) 59 Finance a úvěr – *Szech Journal of Economics and Finance* 426.

[18] *See* R Betz, K Rogge and J Schleich, 'EU Emissions Trading: An Early Analysis of National Allocation Plans for 2008–2012' (2006) 6 *Climate Policy* 361; R Betz, W Eichhammer and J Schleich, 'Designing National Allocation Plans for EU-Emissions Trading – A First Analysis of the Outcomes' (2004) 15 *Energy & Environment* 375.

[19] *See* J Schleich and R Betz, 'Incentives for Energy Efficiency and Innovation in the European Emission Trading System' (2005) ECEEE 2005 Summer Study.

Table 8.1 EU ETS design elements relevant for the innovation effect of the EU ETS[20]

No.	Element	Innovation effect
1	Cap	The lower the total quantity of allowances allocated to installations, the higher the price, the higher the innovation incentive
2	Banking	Banking from one period to the next accelerates innovation
3	Allocation method for existing installations	Auctioning tends to have stronger innovation effects than grandfathering
4	New entrant rules, including transfer rules	Highest innovation incentive if new entrants have to buy allowances on the market; when benchmarking is used the innovation incentive is greatest for undifferentiated product-specific benchmarks because they do not limit the innovation incentive to specific sub-groups, such as certain fuels or technologies
5	Closure rules	Termination of allowances issuance within the period of plant closures results in too long operation times for old plants and postponements of new investments
6	Information about future rules	Clarity reduces investment uncertainty which is beneficial for innovation

its third phase.[21] Despite these improvements several studies point out the general insufficiency of an emission trading system like the EU ETS for promoting the development of breakthrough technologies, but see its strength in achieving short-term cost minimization, for example, by getting commercially available technologies off the shelf.[22]

[20] Summarized from p. 1497 ff. in ibid.

[21] *See* J Schleich, K Rogge and R Betz, 'Incentives for Energy Efficiency in the EU Emissions Trading Scheme' (2009) 2 *Energy Efficiency* 37. For an example of the largely distortive replacement incentives generated by the allocation rules of the EU ETS for the power sector in 2008–12 *see* KS Rogge and C Linden, 'Cross-Country Comparison of the Incentives of the EU Emission Trading Scheme for Replacing Existing Power Plants in 2008–12' (2010) 21 *Energy & Environment* 757.

[22] *See* BA Sandén and C Azar, 'Near-Term Technology Policies for Long-Term Climate Targets — Economy Wide versus Technology Specific Approaches' (2005) 33 *Energy Policy* 1557; C Egenhofer et al., 'The EU Emissions Trading Scheme: Taking Stock and Looking Ahead' (2006); C Philibert, 'Technology Innovation, Development and Diffusion' (2003); D Montgomery, 'Creating Technologies to Reduce Greenhouse Gas Intensity: Public Options & Opportunities' (2005).

In conclusion, the literature-based, ex-ante examinations of the potential impact of the EU ETS on innovation expect modest incentives at best, which, however, should increase in the second and third trading period, given some improvements in instrument design. However, the improvements in the stringency of the emissions cap have been largely compensated by the unforeseen emissions reductions associated with the financial and economic crisis, resulting in continuously low EUA prices. Therefore, with hindsight of this crisis driven lack of scarcity in EUA an even smaller innovation impact of the EU ETS is to be expected. In the following sections I review the empirical evidence on the scheme's actual innovation impact, starting by examining the impact of the EU ETS on technological innovation (section 3) and then taking a closer look at its impact on organizational innovation (section 4).

3. IMPACT ON TECHNOLOGICAL INNOVATION

In this section I review the empirical evidence of the impact of the EU ETS on technological innovation. In doing so, I exclude its impact on investment in new plants or modernization of existing plants even though some of these adoption decisions may represent low carbon solutions new to the firm, which, however, most studies do not clarify. Given these blurry boundaries between innovation and diffusion, in this section I only include studies with an explicit focus on innovation. These studies use patents, R&D expenditures or introduced low-carbon product or process innovations as proxies for innovation.

3.1 Evidence for Phase 1 (2005–2007)

Early studies on the impact of the EU ETS on technological innovation draw a moderately positive picture of the scheme's incentives for low-carbon research and development (R&D). One of the earliest insight results from a survey conducted by McKinsey and Ecofys in June-September 2005 across EU Member States and across all sectors regulated by the EU ETS.[23] Based on the responses of 147 firms the study finds that more than half of the respondents (53 percent) claim that the EU ETS has a strong or at least medium impact on company decisions to develop innovative low-carbon technologies. In contrast, less than one-fifth of

[23] *See* McKinsey and Ecofys, 'Review of EU Emissions Trading Scheme: Survey Highlights' (2005).

respondents (16 percent) stated that all R&D decisions are made independently of the EU ETS and thus see no innovation impact of the EU ETS at all. The study shows striking differences across sectors, which range from all companies saying that the EU ETS has no impact at all on the development of innovative technologies (aluminum industry) to over two-thirds of firms claiming a strong innovation impact (steel industry). According to this study, a positive impact of the EU ETS on technological innovation should be expected, with the strongest impact for steel (84 percent), refineries (60 percent), other ETS sectors (59 percent) and power generation (55 percent), and the weakest in chemicals (41 percent), pulp and paper (33 percent) and aluminum (0 percent).

This fairly positive finding of the innovation impact of the EU ETS during the pilot phase is confirmed by a large-scale cross-sectoral study specifically evaluating the innovation impact of the EU ETS in its first trading phase conducted by Borghesi et al. (2015) for the manufacturing industry in Italy.[24] This study links firm-specific data on eco-innovation – aiming at energy and CO_2 reduction – of the Italian Community Innovation Survey (CIS) from 2008[25] with sector-specific data regarding the sector's coverage by the EU ETS – using an ETS dummy for paper and paper products, coke and refinery, ceramics and cement and metallurgy.[26] In addition, each of these sector's EU ETS stringency – in terms of the sector's ratio of emissions to allocated EU allowances (EUA) – is considered. On the one hand, the regression results – using the ETS dummy – show that in phase 1 ETS sectors were more likely to innovate than non-ETS sectors (n=6,483). On the other hand, for the ETS sectors (n=1,613) the study finds a statistically significant albeit negative link between ETS stringency and eco-innovation,

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[24] That is, energy sectors are explicitly excluded from the analysis. *See* S Borghesi, G Cainelli and M Mazzanti, 'Linking Emission Trading to Environmental Innovation: Evidence from the Italian Manufacturing Industry' (2015) 44 *Research Policy* 669.

[25] The eco-innovation dummy becomes 1 if a company has introduced a product, process, organizational or marketing innovation with an environmental benefit in the period 2006–2008 – the study differentiates between reduced energy use per unit of output (ECOEN) and reduced CO_2 'footprint' (total CO_2 production) by an enterprise (ECOCO). In total, the study finds 17.6% ECOEN innovators and 14.1% ECOCO innovators (n=6,483).

[26] That is, a firm must not necessarily be participating in the EU ETS, e.g. if its production is below the ETS thresholds, but it is classified as EU ETS sector if it belongs to a sector that is covered by the EU ETS. Also, note that manufacture of chemicals and chemical products is not included as EU ETS sector, nor is manufacture of machinery and equipment, thereby omitting potential effects at firms supplying production equipment for EU ETS firms.

indicating an increased likelihood to innovate in sectors with lower EU ETS stringency. One of the reasons the study provides for this surprising finding is that innovative firms may have reacted early in anticipation of the introduction of the ETS which would confirm the innovation impact of expected environmental regulation. Another reason suggested are sector specific weaknesses in adopting eco-innovations which seem to be particularly pronounced for cement and ceramic firms, and thus for sectors with high ETS stringency indicators. However, the evidence may also be due to the use of sector specific instead of firm specific data on EU ETS stringency. Therefore, the study provides mixed evidence on the innovation impact of the EU ETS. However, when only considering whether a sector is covered by the EU ETS or not, then the analysis provides clear support of a positive effect of the first phase of the EU ETS on the adoption of energy and CO_2 saving eco-innovations. Yet, the underlying data does not allow for a differentiation of technological and organization innovation.

The analysis conducted by Anderson et al. (2010) for all EU ETS sectors in Ireland provides further evidence of a positive albeit very moderate impact of the EU ETS on technological innovation.[27] This result is derived from a mail survey conducted with Irish EU ETS firms during the first trading phase (n=27, representing a response rate of 40 percent), and was supplemented with follow-up interviews with seven of the participating firms. Based on descriptive statistics and qualitative data analysis the study concludes that the introduction of a CO_2 price has increased firms' appetite for low-carbon innovation, but that the EUA price is too low, the EU ETS is too uncertain and that energy prices tend to be more important than the EU ETS.[28] As a matter of fact, the large majority of firms reported that the EUA price had no effect on their decision relating to machinery and equipment (74 percent), process change (70 percent) and fuel switching (78 percent), but that these were mainly encouraged by rising energy prices. The study concludes that in the pilot phase of the EU ETS mainly low-cost and low-risk abatement opportunities were employed, such as process changes and fuel switching. However, the study also highlights that Irish firms tend to be technology takers buying new technologies from external

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27 *See* BJ Anderson, F Convery and C Di Maria, 'Technological Change and the EU ETS: The Case of Ireland. Working Paper n. 43' (2010) SSRN Electronic Journal.

28 For example, the study finds that 74% of respondents have undertaken process changes leading to a reduction of energy use and emissions, but despite being short firms claimed to be more driven by energy than EUA prices. In addition, only a small share of R&D spending is related to CO_2, and the little that is done focuses on process innovations.

suppliers (92 percent) rather than developing them internally (8 percent), and thus their innovation responses may not be representative for EU ETS firms in other EU Member States.

In addition to these cross-country and/or cross-sectoral studies Pontoglio (2010) provides early evidence regarding the innovation impact of the EU ETS in the Italian pulp and paper industry.[29] Her unit of analysis is not EU ETS firms but EU ETS plant operators of which 38 (of 163) participated in a survey conducted in May–June 2006. The study finds a wait-and-see strategy of operators: they addressed the typical shortage in allowances by making use of borrowing (and banking) provisions of the EU ETS. That is, most pulp and paper producers followed a conservative and cautious approach to decision making, with so far only 13 percent of them having invested in technological innovation aimed at reducing CO_2 emissions. However, one-third of respondents (35 percent) indicated that they were developing CO_2 and energy saving innovation projects to be implemented in subsequent years. Based on these findings and additional interviews with industry experts Pontoglio concludes that the EU ETS in its pilot phase either did not encourage or, at best, only modestly encouraged technological innovation.

Finally, two studies of German power generators provide in depth insights in the innovation impact of the first phase of the EU ETS, and thereby further supplement the aforementioned cross-country, cross-sectoral quantitative studies. The study by Hoffmann (2007) is based on five company case studies for which 20 interviews with senior managers of German electricity providers were conducted in March–July 2006.[30] It finds that in its first trading phase the EU ETS only had limited effects on R&D efforts, mainly by accelerating selected R&D activities within the fossil fuel regime. Most predominantly, the EU ETS was found to provide additional incentives for ongoing R&D projects aiming at an increase of the energy efficiency of fossil-fuel power plants. It was also shown to increase companies' interest in carbon capture and storage technologies (CCS), despite their detrimental effect on energy efficiency and the associated high regulatory and technological uncertainties. The study of Cames (2010) supports and sheds further lights on these findings of the limited innovation impact of the EU ETS in the German electricity

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29 *See* S Pontoglio, 'An Early Assessment of the Influence on Eco-Innovation of the EU Emissions Trading Scheme – Evidence from the Italian Paper Industry' in A Mazzanti and M Montini (eds), *Environmental Efficiency, Innovation and Economic Performance* (Routledge 2010).

30 *See* VH Hoffmann, 'EU ETS and Investment Decisions' (2007) 25 *European Management Journal* 464.

sector by drawing on qualitative panel analysis among 20 German power generators.[31] In this before-after research design interviews took place in 2004 – and thus before the start of the EU ETS – and were then repeated in 2007, that is, 2.5 years after the scheme's introduction. The study finds that, before the inception of the EU ETS, there were mainly organizational innovations, while hard innovations involving larger investments were postponed. But even by 2007, the EU ETS had not generated enough incentives to trigger substantial investments in R&D activities, with the exception of clean coal and particularly CCS. Cames also notes an increased interest in renewable energy technologies, as the EU ETS has enforced the perception that renewable energy will play an important role in the future electricity system.

Overall, these studies suggest a positive but moderate impact of the EU ETS on the development of low-carbon technologies which occurred across multiple sectors and different countries. Most studies also point out that the effect would have been stronger with a higher CO_2 price and lower regulatory uncertainty about the future of the EU ETS.

3.2 Evidence for Phase 2 (2008–2012)

The evidence on the impact of the EU ETS on low-carbon technological innovation in its second trading phase (2008–12) is less positive, as revealed by two cross-sectoral, cross-country studies and a few more focused sectoral studies. Calel and Dechezleprêtre (2012) provide a comprehensive analysis based on low-carbon patents (up to 2010) of 743 EU ETS firms and 1,019 non-EU ETS firms in 18 EU Member States.[32] While the study finds an increase of the overall share of low-carbon patents since the introduction of the EU ETS in 2005, their more sophisticated econometric estimations – based on the combination of matching methods with difference-in-differences – suggest that the EU ETS had 'virtually no impact at all on low-carbon technological change' (p. 18). This finding is partly alleviated by Martin et al. (2011) who find mixed results regarding the innovation impact of the EU ETS.[33] Their regression analysis is based

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[31] *See* M Cames, 'Emissions Trading and Innovation in the German Electricity Industry' (TU Berlin 2010).

[32] *See* R Calel and A Dechezleprêtre, 'Environmental Policy and Directed Technological Change: Evidence from the European Carbon Market' (2012) CCCEP Working Paper 87.

[33] *See* R Martin, M Muûls and U Wagner, 'Carbon Markets, Carbon Prices and Innovation: Evidence from Interviews with Managers', *Paper presented at the EAERE conference 29 June–2 July 2011* (2011).

on phone interviews conducted in August–October 2009 with 800 manufacturing firms across six EU Member States – among them 446 ETS firms. It finds that the EU ETS in its second phase had a positive impact on the development of cleaner products, but not on cleaner production processes – using a dummy variable for measuring if a firm is part of the EU ETS or not. However, when looking at the perceived stringency of the EU ETS – measured by the amount of allowances companies receive for free in the EU ETS – the study finds the opposite result, namely that higher stringency of the firm-specific cap is associated with more clean process innovations but is irrelevant for product innovations. Hence, while the EU ETS may not have a measurable impact on patenting behavior of EU ETS firms, it seems to have partly influenced clean product and process innovations. However, in a comparable study based on 190 manufacturing firms in the UK – 33 of them subject to the EU ETS – interviewed in January–March 2009, Martin et al. (2012) do not find support of a positive impact of the EU ETS on low carbon product or process innovation, but only on general R&D.[34]

The lack of evidence of a positive innovation impact of the EU ETS in its second trading phase is partly corroborated but also slightly modified in sector-specific analyses. Most of these sector-specific analyses have been performed for the electricity sector. Based on company data gathered in an online survey of electricity generators and technology providers in six EU Member States conducted in November–December 2009, that is, just before the Copenhagen climate summit, Schmidt et al. (2012) find that the EU ETS in its first two trading phases neither had a positive impact on R&D in renewable nor in fossil fuel electricity generation technologies (n=130).[35] A preceding qualitative study by Rogge et al. (2011) based on in-depth interviews conducted with 19 companies in the electricity sector from June 2008 until June 2009 had already alluded to this limited innovation impact of the EU ETS.[36] However, this study also shows that the innovation impact of the EU ETS varies tremendously across technologies

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[34] However, the significance of this relationship vanishes when more covariates are included, which, however, could be due to the small sample size, *see* R Martin et al., 'Anatomy of a Paradox: Management Practices, Organizational Structure and Energy Efficiency' (2012) 63 *Journal of Environmental Economics and Management* 208.

[35] *See* TS Schmidt et al., 'The Effects of Climate Policy on the Rate and Direction of Innovation' (2012) 2 *Environmental Innovation and Societal Transitions* 23.

[36] The sample included 7 electricity generators, 10 technology providers, and 2 project developers, s*ee* KS Rogge, M Schneider and VH Hoffmann, 'The Innovation Impact of the EU Emission Trading System — Findings of Company Case Studies in the German Power Sector' (2011) 70 *Ecological Economics* 513.

and firms, with the largest impact occurring among the most carbon-intensive technologies and among incumbents with large-scale coal power generation technologies in their portfolios. More precisely, the study finds that the largest impact of the EU ETS on corporate R&D occurs for CCS and improvements of the energy efficiency of coal technologies. This effect appeared particularly pronounced for power generators for whom the EU ETS apparently signaled the beginning of fundamental change in their business environment to which they decided to respond, among others, with an increased engagement in CCS R&D projects with German and international technology providers and chemical industry players. The reason for this not showing up in the findings of other studies, such as in the regression results by Schmidt et al., may simply be that only a handful of large companies are involved in such R&D activities on CCS. In addition, as Rogge et al. point out, it is not only the EU ETS which has driven the increase in R&D on CCS but other factors have also played a role, including the prospects of stringent long-term climate policy, debates about the introduction of performance standards for thermal power plants, public research funds, and a lack of public acceptance for coal.

Comparable qualitative results for the pulp and paper sectors in Germany, Italy, Sweden and Norway suggest that the EU ETS in its second trading phase has not impacted technological innovation in the paper industry. Based on case study interviews and a survey of German pulp and paper producers (n=19) as well as technology providers (n=17) conducted between June 2008 and September 2009, Rogge et al. (2011) conclude that the EU ETS has hardly impacted corporate innovation activities.[37] Instead, market factors and here particularly the price and demand for paper have been singled out as key for innovation activities in the German pulp and paper industry. In addition, and in contrast to the electricity sector, the regulatory pull effect of the EU ETS has barely trickled down from companies regulated by the EU ETS to those providing the equipment for producing paper and pulp. Gasbarro et al. (2013) arrive at a similar conclusion on the absence of an innovation impact of the EU ETS for the Italian pulp and paper industry.[38] Based on interviews with six Italian companies conducted from December 2010 until March 2011 the study finds that pulp and paper producers have

[37] *See* KS Rogge et al., 'The Role of the Regulatory Framework for Innovation Activities: The EU ETS and the German Paper Industry' (2011) 11 *International Journal Technology, Policy and Management* 250.

[38] *See* F Gasbarro, F Rizzi and M Frey, 'The Mutual Influence of Environmental Management Systems and the EU ETS: Findings for the Italian Pulp and Paper Industry' (2013) 31 *European Management Journal* 16.

not undertaken any additional investment in technological innovation in response to the EU ETS. Reasons for this include, among others, low and volatile carbon prices, the EU ETS just being one of many investment factors, and long-time horizons of investments – with most recent ones having been planned prior to the introduction of the EU ETS. Finally, Gulbrandson and Stenquist (2013) show that the EU ETS has also not triggered a search for innovative low-carbon solutions in the pulp and paper industry in Sweden and Norway.[39] This insight is based on interviews with one pulp and paper producer in Sweden and one in Norway and three complementary interviews conducted from June 2010 until October 2011. Taken together, this evidence allows for the conclusion that the impact of the EU ETS on technological innovation which was limited in the electricity sector is even weaker, or non-existent for the pulp and paper industry.

Empirical evidence for the link between the EU ETS and innovation for other industry sectors is limited, but the few studies that do exist confirm the existence of sectoral differences in companies' responses to environmental regulation in general, including the EU ETS.[40] One of the few studies of the innovation impact of the EU ETS in industry (other than in the paper industry) was conducted for the German cement industry by Schleich et al. (2010), using evidence from interviews conducted with company representatives of four cement manufacturers and four technology providers between October 2008 and July 2009.[41] This study finds that the EU ETS has led to a somewhat stronger focus of R&D activities on energy, given the add-on effect of costs for allowances to energy costs. The EU ETS is also one of several factors supporting some engagement of cement producers in R&D activities on CCS. It also seems to reflect positively on ongoing research on green cement, but in general product innovations tend to be incremental and largely driven by customer demands

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[39] *See* LH Gulbrandsen and C Stenqvist, 'The Limited Effect of EU Emissions Trading on Corporate Climate Strategies: Comparison of a Swedish and a Norwegian Pulp and Paper Company' (2013) 56 *Energy Policy* 516.

[40] *See* S Borghesi et al., 'Carbon Abatement, Sector Heterogeneity and Policy Responses: Evidence on Induced Eco Innovations in the EU' (2015) 54 *Environmental Science & Policy* 377. Based on 29 interviews conducted with industry associations in eight EU Member States in June–July 2013 this study provides some evidence on the limited but varying role of the EU ETS as one of several other policy instruments on the following sectors: ceramics and cement, paper, energy, coke and refinery.

[41] *See* J Schleich et al., 'Wirkungen Neuer Klimapolitischer Instrumente Auf Innovationstätigkeiten Und Marktchancen Baden-Württembergischer Unternehmen' (2010) Fraunhofer ISI.

which are not yet paying much attention to CO_2. The study also notes that the innovation impact of the EU ETS on technology providers is negligible since demand for new cement plants is largely located outside of Europe where climate policies play a much smaller role.

Overall, it can therefore be concluded that despite the improvements in the EU ETS design it did not generate any significant impact on technological innovation in its second trading phase; the only exception to this is an increased interest in R&D on carbon capture and storage technologies, particularly in the electricity sector.

3.3 Outlook for Phase 3 (2013–2020)

While there is yet no empirical study which has investigated the actual innovation impact of the EU ETS in its third trading phase, several of the studies conducted during the second trading phase also provide indications on the expected innovation impact of the EU ETS up to 2020. All of these studies suggest that the impact of the EU ETS on technological innovation is going to increase in its third phase.

In their cross-sector, cross-country study Martin et al. (2011) find no significant link between the EU ETS and technological innovation – neither for clean product nor clean process innovation – when only using a dummy variable for the EU ETS which captures whether a company expects to be subject to the EU ETS in its third trading phase or not.[42] Similarly, firms' expectations of the CO_2 price by 2020[43] are not significantly related with higher levels of low-carbon innovation, that is, higher price expectations do not seem to be associated with more clean product nor clean process innovation. In contrast, the study finds that firms expecting their future allowance allocation to be more stringent pursue more clean product innovation, and in some models also more clean process innovation. This suggests that not the price of CO_2 but rather the actual costs – rather than opportunity costs – associated with CO_2 emissions stimulate low-carbon innovation. That is, free allocation seems to disincentivize low-carbon innovation, while paying for – at least a part of – CO_2 emissions leads to more low-carbon innovation, with particularly strong evidence for clean product innovation.

For their cross-country study of the electricity sector Schmidt et al. (2012) find that firms' perceptions of the EU ETS negatively affect their R&D investments for non-emitting power generation technologies, in

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[42] *See* Martin, Muûls and Wagner (2011).

[43] The median price expected for 2020 by companies was €30, the mean €40.

particular for renewable energies.[44] That is, those companies which perceive the EU ETS in its third trading phase as more negative increase their R&D in non-emitting technologies, that is, in renewable energies. In contrast, no significant link is found for overall R&D or R&D in emitting technologies, as was already the case for the first two phases of the EU ETS. Since all power generators are required to purchase all of their allowances this implies that particularly power generators with higher emitting technologies in their portfolio are incentivized to spend more on R&D in clean technologies, suggesting a redirection of innovation activities in this sector towards low-carbon solutions.

Finally, for industry sectors, two studies on the German pulp and paper sector and the cement sector indicate that despite its so far negligible impact on technological innovation firms expect the relevance of the EU ETS for R&D to increase by 2020.[45]

However, these expectations were largely resting on the assumption of rising stringency and allowance prices of the EU ETS. Yet, EUA prices have remained low, and have remained low after the landmark agreement at COP 21 in Paris. This suggests that companies do not yet believe that the global climate agreement reached in Paris will be translated into strengthening the stringency and thus the allowance prices of the EU ETS. Should these expectations turn out to be true, then future studies on the impact of the EU ETS on technological innovation in its third trading phase are unlikely to lead to results which differ significantly from the very limited effect found for its second trading phase.

4. IMPACT ON ORGANIZATIONAL INNOVATION

Studying the impact of the EU ETS on organizational innovation has been somewhat neglected in the literature when compared to technological innovation, both in terms of scope of analysis and methodological rigor. First, there is no study which dedicatedly addresses the impact of the EU ETS on organizational innovation. Rather, studies either treat it as a side aspect alongside technological innovation,[46] address only selected aspects of organizational innovation – often in a non-systematic manner – or

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44 *See* Schmidt et al. (2012).
45 *See* Rogge et al. (2011); Schleich et al. (2010).
46 As a matter of fact, the evidence presented here on the impact of the EU ETS on organizational innovation largely draws on some of the studies already reviewed within the section on technological innovation, but is complemented with a few specialized studies focusing on organizational aspects only.

address organizational change (or elements thereof) rather than organizational innovation.[47] Second, many studies stop short at identifying organizational innovation but do not investigate the role the EU ETS played for the observed changes. As a consequence, insights on the causal link between the EU ETS and observed organizational innovation remain limited. Finally, evidence is largely based on qualitative case studies based on interviews and often limited to one particular sector and country. The few quantitative studies including some selected aspects of organizational innovation only use descriptive statistics in analyzing their data, making their contribution to the evidence base rather small, at least in comparison to some of the rigorous case study analyses. Yet, despite these limitations the combination of insights of the identified studies reveals a relatively clear picture regarding a positive impact of the EU ETS on organizational innovation which in most instances has already occurred in the first trading phase.

In reviewing the evidence on the impact of the EU ETS on organizational innovation, I will structure this section according to the relevant aspects provided by the Oslo Manual's definition of organizational innovation as 'the implementation of a new organizational method in the firm's business practices, workplace organisation or external relations'.[48] Organizational innovations in *business practices* comprise 'new methods for organising routines and procedures for the conduct of work'. This component of organizational innovation is by far the most widely studied in the context of the EU ETS. It is, of course, often connected to innovations in *workplace organization* which capture 'new methods for distributing responsibilities and decision making among employees for the division of work within and between firm activities (and organisational units), as well as new concepts for the structuring of activities, such as the integration of different business activities'. Here, many studies allude to the role of top management in dealing with the EU ETS, but also address other structural changes. Finally, organizational innovation also includes innovations in *external relations*, that is, 'new ways of organizing relations with other firms or public institutions'. While several studies find such new collaborations, integration with suppliers, outsourcing or subcontracting

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[47] According to the Oslo manual the difference between organizational change and organizational innovation is that the latter 'has not been used before in the firm and is the result of strategic decisions taken by management', *see* p. 51 in OECD and Eurostat.

[48] *See* p. 51 f. in ibid. Note that the reviewed studies typically do not apply this standard definition of organizational innovation which complicates a systematic analysis of the findings.

in the context of the EU ETS, they typically do not address this explicitly as an organizational innovation. Also, the empirical evidence suggests that EU ETS-triggered innovations in business practices go along with changes in workplace organization and/or external relations. However, for analytical purposes these different aspects of organizational innovations will be discussed in turn rather than in an integrated manner.

4.1 Innovations in Business Practices

Organizational innovation in business practices covers new routines and procedures associated with the introduction of the EU ETS. Some studies refer to this as procedural change[49] and others suggest differentiating between those practices concerning physical CO_2 management and those addressing financial CO_2 management.[50] Innovations in *physical CO_2 management* cover, among others, the introduction of carbon emission monitoring, verification and reporting as well as the set up of other compliance procedures for the novel market-based instrument EU ETS. Cross-sectoral anecdotal evidence collected during the first phase of the EU ETS by Kenber et al. (2009) from nine companies located in different countries indicates that firms have significantly expanded and improved their CO_2 monitoring and cost-assessment capabilities, including the introduction of precise monitoring of CO_2 emissions and the implementation of a carbon accounts for investment decisions.[51] Similarly, for Ireland an early cross-sectoral survey conducted by Anderson et al. (2010) finds that the EU ETS has led to the adoption of verifiable emissions accounting and a greater awareness of CO_2 emissions reduction possibilities in companies subject to the EU ETS.[52] In line with these findings Sandoff and Schaad (2009) report how Swedish companies subject to the EU ETS had implemented the novel instrument in their organization.[53] For 2006, they find that of the

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[49] *See* Rogge, Schneider and Hoffmann (2011); Rogge et al. (2011); Schleich et al. (2010).

[50] *See* Gasbarro, Rizzi and Frey (2013).

[51] This is based on interviews with nine large companies based in different countries and belonging to different sectors directly or indirectly covered by the EU ETS conducted during the pilot phase of the EU ETS, s*ee* M Kenber, O Haugen and M Cobb, 'The Effects of EU Climate Legislation on Business Competitiveness: A Survey and Analysis', (Climate and Energy Paper Series 09 2009).

[52] *See* Anderson, Convery and Di Maria (2010).

[53] These insights are based on a web-based company survey conducted among the 221 EU ETS firms in Sweden in April–July 2006 (response rate of 52%), *see* A Sandoff and G Schaad, 'Does EU ETS Lead to Emission Reductions through

multiple organizational innovations involved the measurement and verification of emissions is the most time consuming of the compliance-related activities.[54] On a more strategic level Anderson et al. (2010) report that for almost half of the Irish EU ETS firms (46 percent) the scheme has influenced how investments in capital and infrastructure are analyzed.[55]

The most detailed account on the dynamics and scope of organizational innovation in response to the EU ETS is provided by Cames (2010) for the German electricity sector.[56] He finds that a few months prior to the introduction of the EU ETS the large majority of the 22 interviewed power generators had already introduced CO_2 emission scenarios and about half of them had established continuous monitoring of CO_2 emissions as well as implemented a tool for comparing EU allowances and CO_2 emissions. In contrast, only a few power generators had started to work with avoidance cost curves and only a minority had introduced risk management procedures. However, towards the end of the pilot phase of the EU ETS more companies had implemented organizational innovations. For example, by 2007 all power generators had introduced a tool comparing EUA with emissions, almost all were working with emission scenarios and the majority of companies had adapted their risk hedging strategies. A comparable study conducted in 2008–09 by Rogge et al. (2011) confirms the strong impact the EU ETS had on the business procedures of German power generators and finds that overall the EU ETS has led to a change in companies' CO_2 cultures.[57] The findings also suggest that organizational innovations were first introduced regarding operational aspects and continuously moved towards more strategic aspects, such as the integration of CO_2 into all investment appraisals.

The impact of the EU ETS on organizational innovation has also been investigated for the pulp and paper industry – by means of company interviews conducted in four different countries during the second trading

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Trade? The Case of the Swedish Emissions Trading Sector Participants' (2009) 37 *Energy Policy* 3967.

[54] Sandhoff and Schaad (2009) found a gap between companies' perception on the administrative efforts required due to the introduction of the EU ETS by companies and themselves. They themselves judged the labor time employed to be fairly moderate and did not see the EU ETS as raising questions regarding the administrative efficiency of the EU ETS. A similar complaint about the time and effort needed to comply with the EU ETS was reported for both the German paper and cement industry, see Schleich et al. (2010).

[55] *See* Anderson, Convery and Di Maria (2010).

[56] Note that Cames uses the term 'soft' institutional innovations, *see* Cames (2010).

[57] *See* Rogge, Schneider and Hoffmann (2011).

phase of the EU ETS.[58] For Italy, Gasbarro et al. (2013) showed how the physical management of CO_2 has been integrated for all but one of the six Italian paper producers studied into existing environmental management systems. However, none of the analyzed paper producers introduced specific procedures for a systemic management of emission reduction opportunities, but instead has treated this within existing procedures for managing investments. Similarly, Gulbrandsen and Stenqvist (2013) find an increase of resources being put into site-level administration and reporting of greenhouse gas emissions data since the introduction of the EU ETS for the Norwegian and Swedish paper producers studied.[59] The descriptive survey results by Rogge et al. (2011) for the German paper industry allow some further insights on the implementation of organizational innovation between 2005 and 2009.[60] By 2009, almost 70 percent of paper producers had started to integrate CO_2 and climate policy as a factor when constructing future scenarios. Surprisingly, however, the new cost factor CO_2 had only been taken into account as a new factor in operative business areas by 42 percent of paper producers. This share decreases further (to 37 percent) when considering the integration of CO_2 as standard factor in investment analysis and product development processes, and goes down to 21 percent of companies for the integration of CO_2 as standard factor when planning R&D.[61] The latter is supported by Gasbarro et al. (2013) who find that only one of the six Italian paper producers surveyed systematically engaged in ETS-oriented investment planning for their R&D department. The limited strategic importance of the EU ETS is also underlined by the EU ETS being seen as one among many factors influencing firm strategies, and certainly not one having more importance than others. In addition, the influence of allowance prices on investment choices still seemed to be unclear to Italian paper producers. This situation somewhat improves for German cement producers subject to the EU ETS for whom Schleich et al. (2010) find that the costs for emitting CO_2 are now seen as important factor in investment appraisals.[62]

While the large majority of EU ETS firms seem to have introduced new business practices regarding physical CO_2 management, the picture is

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58 *See* Gasbarro, Rizzi and Frey (2013) for Italy; Gulbrandsen and Stenqvist (2013) for Sweden and Norway; Rogge et al. (2011).

59 *See* Gulbrandsen and Stenqvist (2013).

60 *See* Rogge et al. (2011)

61 In contrast, the two paper producers from Sweden and Norway interviewed by Gulbrandsen and Stenqvist had integrated CO_2 prices in investment appraisals, see Gulbrandsen and Stenqvist (2013).

62 *See* Schleich et al. (2010).

less favorable for *financial CO_2 management*. Innovations in financial CO_2 management cover, among others, the implementation of a carbon trading strategy, of establishing routines for EUA trading and introducing practices for CO_2 market monitoring.[63] As suggested by cross-sectoral, cross-country survey data from 2009 by Martin et al. (2011) more than half of the 446 EU ETS participants surveyed did not engage in trading EUAs.[64] Furthermore, 30 percent of EU ETS firms did not consider allowances as a financial asset and rather focused on compliance with the EU ETS, although this share differs significantly across sector (but not countries). This insight on the reluctance towards an active trading strategy confirms findings of an earlier study by Sandoff and Schaad (2009) who surveyed the Swedish EU ETS participants in 2006 and found that almost 80 percent of companies only traded once a year. The study suggests that trading was mainly conducted to minimize risks and for compliance purposes, rather than as a market opportunity.[65] This finding was further supported by almost half of respondents (46 percent) claiming to reduce a potential EUA shortage within the second trading phase of the EU ETS through internal measures, such as improving and developing new production processes (18 and 56 percent, respectively) and developing new products (18 percent). They also note that back then JI/CDM was of marginal importance. Overall, it can thus be argued that financial CO_2 management has become more widespread over the first two phases of the EU ETS, but a significant share of companies has remained reluctant.

This hesitancy towards trading is confirmed by Gasbarro et al. (2013) for the Italian pulp and paper industry.[66] In general, their findings suggest a higher orientation of EU ETS firms with compliance rather than trading, and confirm a very limited interest in JI/CDM. Not surprisingly, then, the study points to companies pursuing internal emission reductions as main strategy. In contrast, for the electricity sector Cames (2010) shows that German power generators had already integrated CO_2 trading into their existing trading floors from the very beginning of the EU ETS and started to actively engage with JI/CDM toward the end of the pilot phase.[67] The findings for the German cement industry by Schleich et al. (2010) suggest that cement producers could be positioned in the middle of this spectrum of trading strategies, with particularly the larger ones having implemented CDM projects – typically with subsidiaries located in developing

63 *See* Gasbarro, Rizzi and Frey (2013).
64 *See* Martin, Muûls and Wagner (2011).
65 *See* Sandoff and Schaad (2009).
66 *See* Gasbarro, Rizzi and Frey (2013).
67 *See* Cames (2010).

countries.[68] In addition, almost all companies with CDM projects planned to use the project-based CDM credits for EU ETS compliance purposes up to the limit of 22 percent valid in the second trading phase. However, actual trading varies largely across cement producers, with some companies banking any access EUAs for the third trading phase, others having created a new business unit on CO_2 trading and those with internationally active companies having delegated any trading activities to their headquarters which tend to cooperate with banks. These sectoral differences point to a discrepancy regarding financial CO_2 management between sectors in general, and the electricity and industry sectors in more particular, thereby corrobating the need for cross-sectoral studies.[69]

4.2 Innovations in Workplace Organization

Many of the aforementioned innovations in business practices were associated with innovations in workplace organizations. However, there is limited systematic evidence regarding the impact of the EU ETS on this aspect of organizational innovations. Still, the existing evidence allows for one general and two sector-specific observations regarding novelties in the distribution of responsibilities and decision making among employees as well as new concepts for the structuring of activities in response to the introduction of the EU ETS. The first cross-sectoral finding concerns the engagement of top management with climate policy, in general, and with the new cost factor CO_2, in particular. For example, already Kenber et al. (2009) note that the EU ETS has led to a shift in management awareness towards climate change which has arrived in the boardroom as a new topic.[70] Similarly, in their cross-sectoral study Sandoff and Schaad (2009) find that the EU ETS has quickly become a top management issue in Swedish companies.[71] In particular, the study notes that trading decisions are taken by top management in almost two-thirds of EU ETS firms (64 percent) and even actual trading was conducted by management in more than 40 percent of companies (41 percent). Yet, only a third of the companies (37 percent) have introduced a CO_2 reduction target, suggesting limited managerial attention to climate change, despite it being a top management issue.

The second group of observations concern innovations in workplace

68 *See* Schleich et al. (2010).
69 *See* Borghesi et al. (2015).
70 *See* Kenber, Haugen and Cobb (2009).
71 *See* Sandoff and Schaad (2009).

organization in the electricity sector. In his study of the German electricity sector Cames (2010) shows that already prior to the start of the pilot phase of the EU ETS most of the 22 power generators interviewed had already established a task force coordinating the implementation of the novel policy instrument into corporate practices.[72] Interestingly, three years later these task forces were meeting less often as the new task arising from the introduction of the EU ETS had been integrated into daily business routines. According to Rogge et al. (2011) this decentral integration of the EU ETS even goes as far as some power generators stating that they did not have a specific person responsible for climate policy, as everyone plays a part.[73] In addition, Cames' second round of interviews showed that by 2007 the large utilities had set up new departments for sourcing project-based CO_2 certificates from the Clean Development Mechanism and Joint Implementation projects (CDM/JI). This finding on the establishment of new organizational units for CDM/JI sourcing is also confirmed by Rogge et al. (2011) who find that this organizational innovation is directly and predominantly driven by the EU ETS.[74] In contrast, the EU ETS only indirectly contributes to the establishment of new business units for renewables in German utilities seen towards the end of the pilot phase of the EU ETS. Rather, this innovation in workplace organization leading to the build-up of new competencies is driven by vision changes of companies regarding internal 2020 renewables and greenhouse gas emission targets, which in turn have been shown to result from the impact the implementation the EU ETS and the existence of policy support for renewables in the form of feed-in tariffs had for companies' perceptions on the much increased credibility of the EU's 2020 targets.[75] Despite this only indirect link of the EU ETS with several workplace organizations it can be clearly stated that in the electricity sector the EU ETS has led to the attention of top management to climate change issues.

The third group of observations concerns changes in workplace design within the paper industry. For the Italian paper industry Gasbarro et al. (2013) report the introduction of new functions, the hiring of partly dedicated ETS staff, and the coordination of EU ETS activites.[76] More precisely, two of the six paper producers employed new ETS-dedicated staff, although in different functions. Interestingly, it was these firms who also

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72 *See* Cames (2010).

73 *See* Rogge, Schneider and Hoffmann (2011).

74 *See* ibid.

75 These long-term targets of the EU encompass a 20% reduction of CO_2 emissions, an increase in the share of renewable energies to 20%, and a 20% improvement of energy efficiency to be reached by 2020.

76 *See* Gasbarro, Rizzi and Frey (2013).

stood out in terms of a more active engagement with trading, particularly in one company which introduced a specific function for EUA trading. Similarly to findings from earlier cross-sectoral studies the authors further find that in the majority of companies the executive management was actively involved in EUA trading decisions, although the extent of involvement varies greatly among companies. These insights are confirmed and complemented by a study by Rogge et al. (2011) for the German pulp and paper industry which finds that the large majority of companies (84 percent) had appointed a responsible party or coordinator for the topics of CO_2 and climate policy and had become more involved with these topics at management level.[77] However, only less than a third of companies (31 percent) had set up new strategic departments in the field of climate protection. That is, the attention of top management to the EU ETS has led to ample vision changes, which however, do not yet fully translate into operational changes. A reason for this may be the difference between real costs vs opportunity costs, the low stringency of the EU ETS, but also operational slack and transaction costs.

4.3 Innovation in External Relations

Organizational innovation is also given if a company implements new ways of organizing relations with other firms or public institutions in response to the EU ETS. Cross-sectoral evidence for this is largely limited to the study by Sandoff and Schaad (2009) conducted for Swedish EU ETS firms.[78] It finds that over half of paper producers uses brokers for CO_2 tradings (60 percent) while only a third (36 percent) of the large companies use a CO_2 exchange. Of course, all EU ETS firms were required to establish a link with the national administrative body responsible for the implementation of the EU ETS, but this is not explicitly studied.

For the electricity sector Cames (2010) finds that by 2007 some large German power generators had started cooperating with smaller utilities by offering them market access.[79] In addition, when taking into consideration the impact of the EU ETS on the sectoral innovation system for power generation technologies Rogge and Hoffman (2010) identify new linkages of power generators with technology providers active in the chemical industry.[80] These new external relations are a direct result of the EU ETS

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77 *See* Rogge et al. (2011)
78 *See* Sandoff and Schaad (2009).
79 *See* Cames (2010).
80 *See* KS Rogge and VH Hoffmann, 'The Impact of the EU ETS on the

which led power generators to jointly conduct R&D on CCS, thereby effectively broadening the boundaries of the innovation system.

Finally, for the paper industry Gasbarro et al. (2013) find that rather than extending its environmental management system regarding EU ETS related activities one Italian paper producer worked with a consulting service for regulatory updating, for annual emission communication, for relationships with the Authority, and for the calibration of instruments. Three other companies outsourced the calibration of instruments, and one decided to conduct its carbon market monitoring and allowance trading through a 100 percent controlled energy service subsidiary.[81] In a similar spirit of changes in external relations Rogge et al. (2011) find that in the German pulp and paper industry one-fifth of companies (21 percent) had intensified climate relevant R&D partnerships after the introduction of the EU ETS.[82]

4.4 Outlook

In conclusion, the EU ETS seems to have triggered – or at least contributed to – various organizational innovations, with the evidence base being largest for innovations in business practices. While keeping in mind the limits of the largely qualitative evidence base two key patterns for this impact of the EU ETS on organizational innovation emerge. First, the companies in the electricity sector seem to have been faster and more thorough in implementing the full range of organizational innovations. Key reasons for this may include the EU ETS affecting core production processes and the fertile ground prepared through the process of liberalization of the electricity sector.[83] Second, the implementation of organizational innovations was found to be much more pronounced for EU ETS firms than for their counterparts supplying them with production equipment. Again, this varies across sectors, with the EU ETS having led to some organizational innovations in other parts of the value chain in the electricity sector – particularly for large diversified power generation suppliers. In contrast, such a trickle through effect has remained largely irrelevant for suppliers of production equipment in the cement industry.[84] Similarly, in the pulp and paper industry organizational innovations seem to be much

Sectoral Innovation System for Power Generation Technologies' (2010) 38 *Energy Policy* 7639.

81 *See* Gasbarro, Rizzi and Frey (2013).

82 *See* Rogge et al. (2011).

83 *See* Rogge and Hoffmann (2010); Borghesi et al. (2015).

84 *See* Rogge, Schneider and Hoffmann (2011); Schleich et al. (2010).

less pronounced among technology providers when compared to paper producers.[85]

Overall, these findings on organizational innovations driven by the EU ETS are in line with general insights by Borghesi et al. (2015) on the impact of environmental policy on innovation which has shown that organizational innovations have been important in most sectors.[86] They can be seen as operating as a leading force in technological innovation, and thus as an important precondition for wider and deeper technological changes should the design of the EU ETS and the policy mix into which it is embedded be improved.

5. CONCLUSION

In this chapter I have reviewed the evidence on the innovation impact of the EU ETS and have found a very limited effect on technological innovation but clear signs of the scheme having stimulated organizational innovation.

Regarding the very moderate impact on technological innovation I found initially high expectations regarding its innovation impact which, however, have dissipated after the scheme's lack of stringency became apparent and prices have collapsed accordingly. The innovation impact varied across sectors and technologies, with the strongest effects occurring for the electricity sector and carbon capture and storage technologies. However, with hindsight the spike of innovation in CCS needs to be seen as a temporary phenomenon, which has been reduced significantly in recent years, among others due to low carbon prices in Europe and other factors, such as a lack of public acceptance for storing CO_2 in Germany, and changed political priorities leading to the cancellation of a demonstration program in the UK. So far, the impact of the EU ETS on technological innovation is likely to remain low, despite attempts of strengthening the scheme's design in its third trading phase (2012–2020).

In contrast, there is clear evidence that the EU ETS has been a key driver in various organizational innovations, such as incorporating CO_2 into business practices, making climate change a top management issue or building external relations to address the challenge of climate change, may that be with consultants, CO_2 exchanges or new R&D partners. While many of these organizational innovations may still be in place – the exception

85 *See* Rogge et al. (2011), p. 266.
86 *See* Borghesi et al. (2015).

perhaps being the shut-down of new business units primarily established in large companies for sourcing JI/CDM credits – these organizational innovations have so far had only limited effects on shifting corporate strategies towards low-carbon solutions. Reasons for this include low carbon prices, the relatively high share of free allocations in industry sectors, and more pressing business concerns.

The findings of this review point to three main patterns of the innovation impact of the EU ETS. First, the impact of the EU ETS on innovation seems to be more pronounced for the electricity sector than for industry sectors. This could both be observed for technological innovation and for organizational innovation. While there may be good reasons for these sectoral differences, such as the importance of CO_2 as a cost factor in the production process or a higher share of auctioning in the electricity sector, it points to the need of focusing future efforts in stimulating low-carbon innovation towards industry sectors. Of course, there remains quite some diversity within industry sectors themselves, as the examples of the pulp and paper industry compared to the cement industry have shown. Second, the innovation impact of the EU ETS was found to be much stronger for those firms regulated by the EU ETS than suppliers of these firms' production equipment. This suggests that the trickle through effect of the EU ETS to other parts of the value chain remains limited, particularly in industry sectors but also for suppliers of power generation technologies based on fossil-fuels which have largely remained locked-in. Given the relevant sectoral patterns of innovation found in the EU ETS sectors this limited trickle through effect is likely to be a major limitation for triggering innovation in the related innovation systems. This calls for a wider innovation system perspective in future research and policy.[87] Finally, for many of the observed technological innovations – typically incremental – the EU ETS was shown to be a contributing factor among others – including the broader policy mix but also the wider business environment. In some instances it has even only played an indirect role for certain innovations. This complexity of the causal link between the EU ETS and innovation has both methodological and policy implications.

However, before moving on to these implications it needs to be noted that the evidence reviewed here did not include the impact of the EU ETS on diffusion. Clearly, several studies have investigated the impact of the EU ETS on investment decisions, such as for modernizations, fuel switching and new plants. The evidence suggests that the EU ETS has mainly

[87] *See* K Pavitt, 'Sectoral Patterns of Technical Change: Towards a Taxonomy and a Theory' (1984) 13 *Research Policy* 343; Rogge and Hoffmann (2010).

contributed to incremental process innovations (including fuel switching), typically strengthening the effect of energy prices. For example, in Ireland over two-thirds of EU ETS companies (74 percent) implemented process or behavioral changes, half of them (48 percent) employed new machinery or equipment and a good third (41%) switched fuels, thereby contributing to reductions in CO_2 emissions.[88] Another example concerns the electricity sector where the impact of the EU ETS on investment seems to have been most pronounced for retrofitting of existing plants. Regarding the investment in new plants the EU ETS with its free allocation for new entrants was shown to have initially led to more investments in polluting plants, while not influencing adoption decisions on non-polluting plants.[89] In this context it was shown that incentives considerably depend on the specific design of the EU ETS, such as the overall cap, share of auctioning, allocation rules for incumbents and new entrants, or closure provisions and transfer rules.[90]

This limitation notwithstanding, this review finds a so far very limited impact of the EU ETS on technological innovation, but a fairly strong impact on organizational innovation. Two main policy implications regarding an innovation-proof design of the EU ETS – and other emission trading schemes – arise from these findings: (1) increase of the carbon price and (2) increase of the share of auctioning. First, this review finds that a higher carbon price provides higher incentives for innovation. For this to happen the scarcity of EUA needs to be increased. Two possible key mechanisms to achieve this are the further reduction of the current and future cap of the EU ETS and the permanent retirement of excess allowances from earlier trading phases. In addition, the introduction of a minimum price should be reconsidered so that the currently very weak carbon price would be strengthened. Second, since free allocation was shown to be detrimental for innovation the share of auctioning should be continuously increased. This also implies reconsidering industry exemptions based on competitiveness concerns. A promising way forward in this regard could be the development of a gradual phase-out strategy for free allocations – coordinated with other emission trading systems, such as the soon to be implemented Chinese ETS, in order to address carbon leakage concerns while at the same time strengthening innovation incentives. Resulting auctioning revenues could be earmarked to further

88 *See* Anderson, Convery and Di Maria (2010).

89 *See* Hoffmann (2007); Cames (2010); Rogge, Schneider and Hoffmann (2011); Schmidt et al. (2012).

90 For a detailed empirical examination of the innovation incentives arising from different design features of the EU ETS for the German electricity sector, *see* Cames (2010).

stimulate radical innovations in low-carbon solutions in industry sectors. Clearly, these policy implications for the redesign of the EU ETS raise difficult questions regarding both political and legal feasibility, and thus tackling them remains a major challenge – but arguably a smaller one than introducing a EU wide carbon tax. Without addressing the current shortcomings the EU ETS cannot play its foreseen role in guiding the decarbonization of the European economy for which innovations in low-carbon solutions are a fundamental requirement.

Despite the limited impact of the EU ETS on technological innovation different indicators for low-carbon innovation, such as patents or innovation expenditures, indicate a positive trend. This increasing pattern of low-carbon innovation is in contrast to Taylor's (2012) observation of declining inventive activity after the introduction of permit trading schemes for sulfur dioxide and nitrogen oxide control in the US.[91] But if it is not the EU ETS which is driving this positive development, the question remains what else is behind it. One important part of the answer to this question is provided by those studies which did not only consider the EU ETS as political driver of low-carbon innovation but also included elements of the wider policy mix[92] in their analysis.[93] For example, the empirical evidence gathered for the electricity sector suggests that long-term emission reduction targets, such as the EU's 2020 targets, have been an important determinant of corporate innovation activities in non-polluting technologies, that is, in renewable energies. In addition, the existence of technology-specific instruments promoting the diffusion of low-carbon solutions, such as feed-in tariffs for renewable energies, have been found as another important element of the policy mix and as such complement the EU ETS.[94] The relative importance of long-term targets and concrete policy instruments differs among innovation dimensions, with long-term targets being particularly important for R&D activities of technology providers.[95]

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[91] *See* MR Taylor, 'Innovation under Cap-and-Trade Programs' (2012) 109 PNAS 4804.

[92] *See* KS Rogge and Reichardt, K, 'Policy mixes for sustainability transitions: An extended concept and framework for analysis' (2016) 45(8) *Research Policy* 1620–1635.

[93] *See* Borghesi, Cainelli and Mazzanti (2015); Borghesi et al. (2015); Hoffmann (2007); Schmidt et al. (2012); Rogge, Schneider and Hoffmann (2011); Rogge et al. (2011).

[94] In particular, *see* the findings of the cross-country study conducted by Schmidt et al. (2012).

[95] *See* KS Rogge, TS Schmidt and M Schneider, 'Relative Importance of Different Climate Policy Elements for Corporate Climate Innovation Activities' (2011) Berlin: Climate Policy Initiative.

This review allows for deriving a number of methodological challenges in evaluating the innovation impact of the EU ETS which should be tackled by future research. First, evaluating the innovation impact of the EU ETS calls for dedicated data collection, for example in the form of specialized company surveys or rigorous case studies. Alternatively, the utilization of existing data, such as patent data or data originating from the Community Innovation Survey, requires the creation of meaningful proxies capturing the EU ETS and its design features. Second, given the different strengths but also limitations of the employed methodological approaches it seems most promising to combine different methods into a multi-method research design combining qualitative and quantitative methods. Third, studies attempting to establish the innovation impact of the EU ETS should pay closer attention to the different sectoral patterns of innovation and thus extend the boundaries of their investigation accordingly. Typically, this will imply assuming an innovation system perspective in which not only EU ETS firms will be found to perform innovative activities but also other key actors in the system. Finally, as the EU ETS is embedded in a broader policy mix studies should pay greater attention to the role played by long-term targets and other policy instruments as well as their interaction. However, such a thorough analysis of the innovation impact of the EU ETS in its third phase – ideally covering multiple sectors and countries – would be best postponed until the major shortcomings of the scheme have been addressed. In the meantime, these methodological recommendations may be also valuable for the evaluation of the innovation impact of the many national and regional CO_2 trading schemes and carbon taxes implemented around the world.[96]

Finally, I want to emphasize that while the impact of the EU ETS on technological innovation has remained limited, its positive impact on organizational innovation should not be underestimated. The reason for this is that organizational innovations – as by now widely acknowledged in the innovation studies literature – provide a necessary precondition for future technological innovations.[97] In this regard it is promising to see that companies aware of climate change – an awareness for which the EU ETS has been the main driver – introduce more low-carbon innovations than companies without such an awareness.[98] In addition, Martin et al. (2012) show that managers aware of climate change tend to introduce firm-internal targets, e.g. regarding the reduction of greenhouse gas

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96 *See* World Bank and Ecofys, 'State and Trends of Carbon Pricing' (2015).

97 *See* OECD and Eurostat (2005).

98 *See* Martin et al. (2012).

emissions – something to which the EU ETS has been shown to contribute, alongside other elements of the policy mix. These internal targets, in turn, have been shown to positively impact low-carbon innovation. As several of the studies reviewed in this chapter have shown it is indeed the EU ETS which has made managers more aware of climate change. However, the scheme currently does not provide sufficient incentives for turning corporate aspirations into lucrative business opportunities, something that needs to be urgently corrected if policy makers are taking the Paris Agreement seriously.

Indeed, it is the momentum of COP 21 in December 2015 and the global agreement reached there to hold the global average temperature increase well below 2°C which provides European and national climate policy makers with the mandate to think creatively how to find the political majorities to increase the stringency of the EU ETS. As the studies surveyed in this review have shown this implies first and foremost increasing the scarcity of EUAs, thereby contributing to higher yet predictable carbon prices. The permanent retiring of EUAs parked in the market stability reserve, the strengthening of the reduction factor, and the establishment of an intelligent mechanism which guarantees a minimum EUA price are three possible avenues for such a rescue mission. Making the EU ETS more stringent as outlined above would not only unleash its transformative power by generating greater incentives to invest in low-carbon innovation, but it would also send out a strong political signal that Europe takes the agreement struck in Paris seriously and implements it accordingly. It is exactly this line of thinking in terms of consistent and credible policy mixes made up of ambitious long-term targets which are implemented by a combination of well designed demand pull, technology push and systemic instruments which is needed for successfully governing the decarbonization of the European economy.[99]

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[99] See KS Rogge et al., 'Green Change: Renewable Energies, Policy Mix and Innovation' (2015).

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9. Financial crimes in the European carbon markets

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1. INTRODUCTION

The European Union Emissions Trading Scheme (the 'EU ETS') has grown rapidly since its inception. Now the largest carbon market in the world, it covers over 11,000 power stations and industrial plants in 31 different countries,[1] and was worth an estimated €56 billion in 2012.[2] The sophistication and complexity of the associated market for tradable allowances has mirrored this growth, with market participation extending beyond those interested purely in the emissions compliance use of the underlying commodities. Financial intermediaries quickly joined the EU ETS market, both to profit from buying and selling emissions allowances on their own accounts and to provide trading and risk management services for compliance entities. Sophisticated trading platforms have emerged as the market has matured, with the majority of trade now carried out on electronic trading platforms, through which buyers and sellers can enter their orders and carry out trade in emissions allowances anonymously and rapidly.[3]

In addition to basic contracts for the immediate delivery of emissions allowances (spot contracts), financial intermediaries have made use of a variety of more complex derivative products for trade in emissions allowances. 'Forwards' and 'futures', for example, represent contracts for the delivery of a set volume of a commodity on a specified future delivery date, whereas 'options' represent a contract which provides the option to buy a set volume of a commodity on a specified future date. These derivative products help firms manage their price risk by being certain of the price they will be paying (or receiving) for allowances in the future.

The broad involvement and diversity of products within the market has

[1] This includes the 28 EU Member States, Iceland, Liechtenstein and Norway.

[2] European Commission, *The EU Emissions Trading System (EU ETS)*, (EU Factsheet, October 2013) available 14 July 2016 at http://ec.europa.eu/clima/policies/ets/index_en.htm.

[3] Ellerman et al., *Pricing Carbon: The European Emissions Trading Scheme* (Cambridge, UK: Cambridge University Press, 2010), 135–7.

been important in driving liquidity and market efficiency and has itself fuelled further market growth.[4] But as the EU carbon market has grown in size, value and complexity, it has become an increasingly attractive market for fraudsters. Fraud has materialised on this market in a variety of sophisticated forms, including Value Added Tax (VAT) carousel fraud and thefts in emissions allowances. In recognition of the specific vulnerabilities of the EU ETS trading system to fraud, significant reforms to the way that emissions allowances are traded were introduced to the Registry Regulation in 2013, as well to the EU financial markets regulations in 2012–2014.

This chapter aims to examine the major forms of fraud that have affected the EU ETS and the effectiveness of the regulatory measures adopted at the EU and national level in response to their emergence. Initially, this chapter highlights the specific characteristics of emissions allowances and the registries system through which they are traded; and, assesses the extent to which they have made the EU ETS especially vulnerable to value-added tax fraud and thefts in emissions allowances. In addition, this chapter analyses the effectiveness of the regulatory reforms that have been implemented at both EU and national levels to address these vulnerabilities, in particular the reforms to the EU ETS registry system. Finally, this chapter examines the reforms to the EU financial markets oversight regulations adopted *inter alia* in response to fraud in the European carbon markets.

2. FRAUD IN THE EU ETS (I): VALUE-ADDED-TAX (VAT) FRAUD

Domestic rules dictating the amount of VAT charged on emission allowance transfers, as well as how and from whom this tax is collected, are not harmonised across the EU. Therefore, they vary from Member State to Member State. For VAT purposes, the transfer of emissions allowances is treated as a taxable supply of services. Under the VAT Directive, on the domestic supply of services (that is, when the supplier and buyer of services are based in the same Member States), VAT charged on the transfer is payable by the supplier. On the cross-border supply of services (that is, when the supplier and the buyer of services are based in different member

[4] See Daskalakis, G., G. Ibikunle and I. Diaz-Rainey, 'The CO2 Trading Market in Europe: A Financial Perspective', in *Financial Aspects in Energy: A European Perspective* (A. Dorsman, W. Westerman, M. Karan and O. Arslan eds, 2011), Springer-Verlag GmbH, 51–67.

states), however, the buyer is obliged to remit VAT charged on the transfer (this is referred to as a 'reverse charge').[5] The application of this tax regime to the trade of emissions allowances allowed the exploitation of the EU ETS by fraudsters.

By buying emissions allowances from a company in another country at a price non-inclusive of VAT (since cross-border transactions are reverse-charged) and then selling them on domestically at a price inclusive of VAT, large amounts of money can be raised fraudulently by a trader who does not surrender this VAT 'profit' to the treasury of the state in which the sale was made,. This type of activity is commonly known as 'missing trader intra-community' (MTIC) fraud, as it involves a trader disappearing before it can be traced by the authorities. The amount of VAT acquired in this way can be augmented by an organised group of traders or companies acting in concert to trade allowances in a series of 'carousels' (see Figure 9.1 below). By repeatedly trading emissions allowances through a circle of conspirator companies or individual traders, the amount of VAT that can be charged and not surrendered is multiplied each time the allowances are circulated.[6] This form of MTIC fraud is known as 'VAT Carousel fraud'.

2.1 The Emergence of VAT Fraud on the EU ETS

MTIC fraud has historically been concentrated in the markets for mobile telephones, computer chips and other high value, low volume goods, due to their ease of transportation and the high VAT revenues that can be generated.[7] EU emissions allowances also share similar characteristics that make them an attractive vehicle for this type of fraud. With a high value and no physical volume, emissions allowance transfers can be completed electronically on the spot market in as little as 15 minutes, avoiding the cost and delay involved in physical delivery.[8] As a result, fraudulent traders can transfer large volumes of allowances and conduct multiple 'carousels' before being traced by the authorities, which ultimately allows them to maximise the amount of VAT 'profit' made.

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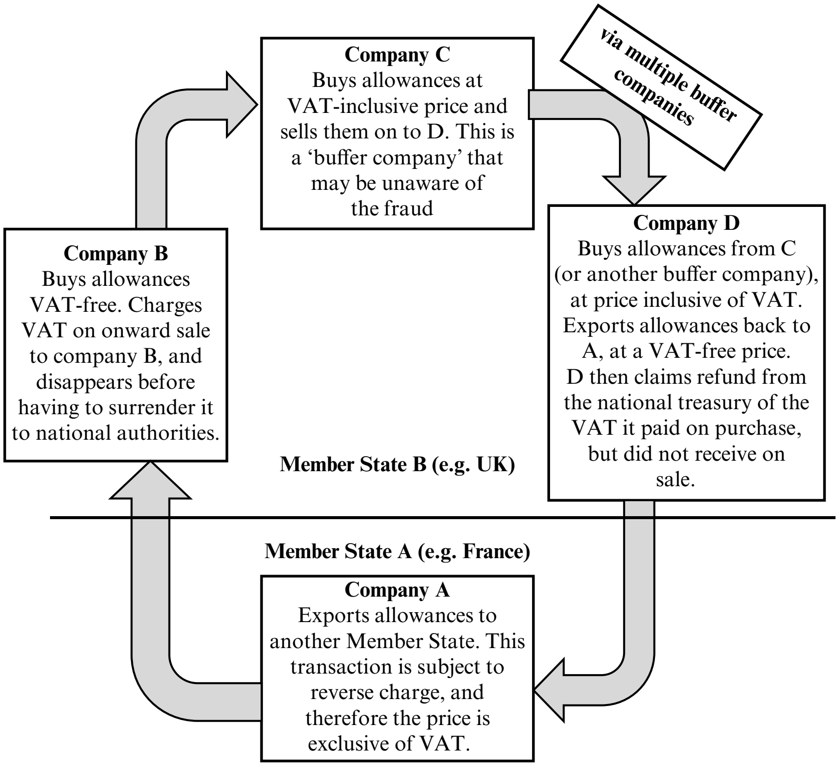
[5] Ainsworth, R., 'CO2 MTIC fraud – technologically exploiting the EU VAT (again)' (2010), *Boston School of Law Working Paper* 10(01), Boston, USA); Article 196 Council Directive 2006/112/EC of 28 November 2006 on the common system of value added tax (as amended).

[6] *ibid*

[7] Eurojust, 'Fraud', *Eurojust News*, Issue No. 4, July 2011, Eurojust, The Hague, the Netherlands.

[8] Ainsworth, R. (2010), above, n. 5.



Source: This diagram is adapted from Keen, M. and S. Smith, 'VAT fraud and evasion: what do we know and what can be done?' (2007), IMF working paper WP/07/31, International Monetary Fund, New York, USA.

Figure 9.1 VAT carousel fraud

Suspicions that the EU ETS market was being used as a vehicle for VAT fraud first arose following an unprecedented rise in emissions allowance spot trading volumes towards the end of 2008. This trading volume peaked on 2 June 2009, when a record 19.8 million metric tons of CO_{2e} were traded on the BlueNext spot exchange (which was then the largest carbon spot exchange in Europe, based in Paris).[9] Rumours that these volumes were being driven by VAT carousel fraud prompted BlueNext to close its spot

[9] Reuters UK, 'France makes CO2 credits VAT-exempt to avoid scam', [2 June 2009], Reuters UK Online, [online] available 1 August 2016 at http://uk.reuters.com/article/us-carbon-bluenext-idUSTRE55726W20090609

exchange.[10] Before re-opening the exchange, the French authorities classified domestic sales of emissions allowances as transactions involving financial products. As such, transactions in France were VAT exempt, because VAT could no longer be charged on domestic transfers of emissions allowances traded as financial instruments. This effectively blocked the ability of fraudsters to conduct MTIC fraud in France. When trade resumed, BlueNext's daily spot trading volumes had plummeted by over 85 per cent, leveling out at roughly 2.5 million tons CO_{2e}, thus strengthening suspicion that the surge in trading volume had been driven by MTIC carousel activity.

2.2 Effects of VAT Fraud

In 2009 Europol estimated that VAT fraud on the EU ETS had thus far cost taxpayers across all member states roughly €5 billion through lost tax revenues.[11] In addition to these significant tax revenue losses, it is thought that VAT fraud has had impacts on the functioning of the market as a whole and its effectiveness as an emissions abatement tool. Forensic econometric studies suggest that during its peak in 2009, VAT carousel fraud was driving spot trading volumes as much as ten times higher than would have normally occurred.[12] These inflated trading volumes had a distorting effect on the spot market carbon price signal, which in turn could have compromised the market's efficiency and ability to incentivise emissions abatement. This price-distorting effect has also been seen on the Italian spot exchange which, following suspicious surges in trading volume in November 2010, was trading EU emissions allowances at a significant price discount.[13]

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[10] Frunza, M., D. Guegan and A. Lassoudiere, 'Missing trader fraud on the emissions market', (2010) 18(2) *Journal of Financial Crime* 183.

[11] Europol, 'Carbon credit fraud causes more than 5 billion euros damage for European tax payer', [2009], Europol website, available 1 August 2016 at: https://www.europol.europa.eu/content/press/carbon-credit-fraud-causes-more-5-billion-euros-damage-european-taxpayer-1265.

[12] Frunza et al. [2010], above, n. 11.

[13] At the time, Italy was one of the few countries which had not yet implemented a reverse charge mechanism for emissions allowances and it is suspected that this surge in trading volume resulted from a shift in fraudulent activity away from the countries in which such a mechanism had been implemented. Reuters, 'Italian bourse sees surge in spot EU carbon trades', [2010], Reuters news website, available 1 August 2016 at http://uk.reuters.com/article/2010/11/24/carbon-gme-idUKLDE6AN1VM20101124; FERN [2011] EU Emissions Trading Scheme: Memorandum submitted by FERN to the House of Commons (ETS 33).

2.3 The Reverse Charge Mechanism: A Temporary Fix?

Following France's adoption of a domestic VAT exemption for the supply of emissions allowance, a number of other EU Member States similarly altered their domestic VAT treatment of emissions allowances as an interim measure to prevent VAT fraud within their own jurisdictions. In July 2009, the UK made domestic transfers of emissions allowances zero-rated.[14] The Netherlands introduced a reverse-charge mechanism, whereby the buyer rather than the seller is the party responsible for surrendering VAT on domestically traded emissions allowances.[15] These changes in domestic VAT rules in France, the Netherlands and the UK effectively put a stop to VAT fraud within jurisdictions containing three of the largest emissions trading exchanges in Europe at the time, that is, Bluenext (France), Climex (Netherlands) and ICE ECX (UK).

The European Commission also took centralised action to protect the EU ETS market from VAT fraud. In March 2010, revisions to the 2006 VAT Directive[16] were adopted which provided Member States with the option to introduce a reverse charging mechanism on the VAT charged on domestic sales of emissions allowances.[17] This optional reverse charge regime was only intended as a temporary fix to MTIC fraud on the EU ETS pending a long-term comprehensive solution.[18] Indeed, under the

[14] Council Decision 2007/250/EC, authorising the United Kingdom to introduce a special measure derogating from Article 193 of Directive 2006/112/EC on the common system of value added tax, O.J. (L 109) 24; HMRC, Revenue and Customs Brief 46/09 (2009), available 14 July 2016 at: http://webarchive.nationalarchives.gov.uk/20140109143644/http://www.hmrc.gov.uk/briefs/vat/brief4609.htm. Note that there are key differences between the supply of a good or service being zero rated and being classified as exempt from VAT. Although both regimes result in no VAT being payable on the supply of that specific good, if the supply is VAT exempt, suppliers cannot recover VAT incurred on related costs. If the transaction is zero-rated, however, such costs can be recovered.

[15] This reverse charge system passes the obligation to pay VAT on purchased allowances on to the buyer, rather than including the VAT in the purchase price and leaving the seller responsible for the payment of this amount to the authorities.

[16] Council Directive 2006/112/EC. Article 199 outlines the goods and services to which Member States are permitted to apply a reverse charge mechanism.

[17] Article 1, Council Directive 2010/23/EU.

[18] Recital 8, Council Directive 2013/43/EU. This Directive introduced further amendments to Directive 2006/112/EC, extending the period of application of the reverse charge mechanism from 2015 to 2018, and extended the scope of application of the reverse charge mechanism to trade in other VAT-fraud susceptible goods and services (including mobile telephones, tablet PCs, telecommunications services and precious metals).

2006 VAT Directive Member States are only permitted to maintain this reverse charge mechanism until 2018.[19] Furthermore, as the Directive only imposed the *option* for Member States to adopt this regime, as yet not all Member States have introduced a reverse charge mechanism.

As discussed in Section 5, emissions allowances are due to become classified as financial instruments under the revised Markets in Financial Instruments Directive (MiFID II),[20] which must be applied within Member States by January 2017. This classification will provide a more harmonised VAT treatment for the domestic trade of emissions allowances, as in general the domestic supply of financial instruments is VAT exempt under EU law.[21] It is thus expected that a consistent market-wide application of this VAT exemption will act to protect the EU ETS from MTIC fraud.

Although it was the tax treatment of emissions allowances that ultimately created the scope for MTIC fraud on the EU ETS, particular vulnerabilities also existed within the regulation of the EU carbon market and Registries system which left the EU ETS especially open to this type of fraud. These vulnerabilities are further explained in Section 4, which analyses recent reforms to the regulation of the EU ETS Registries system.

3. EU ETS FRAUD (II): EMISSIONS ALLOWANCE THEFTS

3.1 Phishing and Account Registry Hacking

'Account takeover' is a form of fraud, prevalent in the banking and credit industries. It occurs when a fraudster, posing as the genuine account holder, gains control of an account and initiates unauthorised transactions. Access is usually gained by 'phishing' for account identity and password information. This can be done by deceptive email requests, or by more aggressive cyber-hacking methods. This form of fraud has been

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[19] Article 1, Council Directive 2013/43/EU. Member States that have adopted the reverse change mechanism are obliged to submit an evaluation report to the European Commission which assesses the level of fraud both before and after the application of the mechanism. An overall assessment report incorporating the results of reports submitted by individual Member States is due to be published by the Commission by January 2018.

[20] This includes spot trading of emissions allowances.

[21] Article 135, Council Directive 2006/116 of 28 November 2010 on the common system for value added tax.

used to facilitate the theft of emissions allowances from EU ETS registry accounts.

The first recorded instance of emissions allowance thefts occurred on 28 January 2010, when a widespread phishing attack targeted emission traders in Germany. Phishers posing as registry administrators sent emails to thousands of EU ETS registry account holders instructing them to disclose their user identification numbers and passwords on a fake registry website infected with a phishing virus. The fraudsters subsequently used the phished access information to gain control of the victims' accounts and fraudulently authorise the transfer of emission allowances to their own accounts (or those of unwitting third parties) from which they could be freely traded. An estimated 250,000 allowances, worth over €3 million euros, were allegedly stolen from six German companies in this way.[22]

A second instance of more sophisticated hacking attempts followed in late 2010–early 2011 (see Table 9.1 below). In November 2010, allowances were stolen from accounts in both the Romanian and Italian registries. In January 2011, accounts in the Austrian, Czech and Greek registries were also fraudulently accessed, resulting in the theft of over two million allowances. The European Commission reacted by suspending spot trading from accounts in all national registries on 19 January 2011.[23] Registries were only permitted to reopen once sufficient evidence was provided to prove they met minimum security standards. Registries in some Member States took months to do so, and only opened again in mid-April 2011.

3.2 Effects of Emissions Allowance Thefts on the EU ETS Market

The total number of emissions allowances stolen during this bout of thefts (approximately three million) represented only 0.003 per cent of the total number allocated at the time, and were stolen from the accounts of only a handful of companies. Thus, the direct financial implications of the thefts on the market as a whole were minimal, with impacts being localised to a few very unfortunate individual companies. The direct financial effects of the thefts were further alleviated, as a large number of stolen allowances

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[22] *The Guardian*, 'Carbon trading fraudsters steal permits worth £2.7m in "phishing" scam', available 14 July 2016 at http://www.theguardian.com/environment/2010/feb/04/carbon-trading-fraudsters-steal-permits.

[23] European Commission, 'Announcement of transitional measures: the EU ETS registry system', [2011], European Commission Climate Action News Archive, available 14 July 2016 at: http://ec.europa.eu/clima/news/articles/news_2011011901_en.htm.

Table 9.1 Allowances stolen via registry hacks in late 2010/early 2011

National registry targeted	Number of allowances stolen*	Company account targeted	Date	Number of allowances returned or traced*
Romania	1,600,000	*Holcim*	16 November 2010	*600,000 returned* (from Lichtenstein)
Italy	267,911	*TCEI*	24 November 2010	(figures not available)
Austria	488,141	*Austrian government account*	10 January 2011	*All returned* (from Lichtenstein and Sweden)
Czech Republic	950,000	*Blackstone Global Ventures; CEZ*	18 January 2011	*225,001 returned* (from Estonia)
Greece	300,000	*Halyps*	18 January 2011	figures not available

Note: * These figures may not be exact and represent estimated figures only, as published by the Greek registry (2011), Czech registry (2011), Italian registry (2011), Austrian registry (2011) and Dutch emissions authority (NEA, 2011).

were traced and returned to their original owners (see Table 9.1). However, hundreds of thousands of stolen allowances remain unaccounted for and remain in circulation on the EU ETS market. These circulating stolen allowances have had significant market-wide effects on market confidence and trade liquidity due to the legal uncertainties associated with inadvertently purchasing them.

Within the EU ETS system, each emissions allowance is allocated a unique unit identification code.[24] Historically, these unit identification codes were only visible to owners of allowances.[25] Following the allowance thefts of 2009–10, a number of victim companies and national registries published lists of allowances allegedly stolen from their accounts, based on the unit identification codes visible to them.[26,27] Yet under the regulations

[24] Annex VI para 3, Regulation No. 2216/2004.
[25] Annex XVI para 14, Regulation No. 2216/2004.
[26] Holcim, 'List of stolen allowances', [2010], Holcim website, [online] available 14 July 2016 at http://www.holcim.ro/fileadmin/templates/RO/doc/EUA_identification_numbers.pdf.
[27] OTE, List of allowances from illegal transactions on 18 January 2011, available 14 July 2016 at http://www.ote-cr.cz/about-ote/file-news/blocks-cz-20110118-public.pdf.

in place at the time, the unit identification codes of allowances subject to transactions were not officially made *publicly* available until 5 years after the completion of the transaction.[28] Therefore, at the time no official lists of serial numbers of allowances fraudulently transferred from registry accounts were available, and market participants had no reliable source against which to test the accuracy of the unofficial lists published by victim companies. Despite numerous pleas, the European Commission refused to publish an official list of stolen allowances and publicly identify the companies holding them.[29] Market participants were therefore unable to reliably ascertain whether the allowances they purchased on the market had been stolen. Although stolen allowances would be identifiable by the victim of the theft (by the serial number made visible to them), any subsequent purchaser would not have been able to recognise those allowances as stolen.

3.3 Legal Uncertainty Surrounding Risks of Purchasing Stolen Allowances

The European Commission has confirmed that stolen allowances are still valid for contribution towards EU ETS compliance obligations.[30] The compliance value of purchased allowances is therefore unaffected by the fact that they may have been stolen and subsequently traded. The risks involved with the purchase of stolen allowances circulating on the market instead resulted from uncertainties as to whether valid ownership could be acquired and the potential liability of inadvertent purchasers.

Although the law relating to stolen goods and the acquisition of good title is well established in many EU Member States, there are significant variations between them.[31] There is no EU-level harmonisation of the rules relating to whether a third party purchaser of stolen goods can be liable to the party from whom the goods were originally stolen. In some

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[28] Article 10(1) and Annex XVI, para 12, Regulation No. 2216/2004.

[29] This decision that was later upheld by the General Court of the Court of Justice of the EU in *Holcim (Romania) v Commission* [2014] EU ECJ T-317/12.

[30] Delbeke, J., ‘Statement on the recent incident of unauthorised access to EU ETS registry accounts in Romania, Statement made by Director-General, DG Climate Action’, (2011), European Commission website, available 14 July 2016 at http://ec.europa.eu/clima/news/articles/news_2010120302_en.htm.

[31] Schwartz, A. and Scott, R., ‘Rethinking the laws of good faith purchase’, (2011) 111 Columbia Law Review 1332.

jurisdictions, such as France[32] a good faith purchaser is only required to return stolen goods to the owner in return for compensation.[33] In countries such as England[34] and Germany,[35] however, a good faith purchaser of stolen goods may be liable to return the goods to the original victim of the theft without compensation. Under certain jurisdictions, an inadvertent purchaser of stolen emission allowances may therefore be liable to return those allowances to the initial victim of the theft, with no compensation for their value, and therefore risk losing their initial investment.

A further layer of complexity arises from the historic lack of a clear legal definition of emissions allowances. How a certain good or commodity is defined legally within a Member State will determine the legal doctrines that apply to its sale and purchase, how legal title is treated in cases of theft and the remedies available to victims of theft against inadvertent third party purchasers. Prior to the 2013 Registries Regulation, the legal nature of emissions allowances had not been defined in EU legislation,[36] and few national courts had provided further clarification on how emissions allowances should be treated in individual Member States. Therefore, the question of whether a buyer could obtain legitimate ownership rights to stolen emissions allowances lacked legal clarity. Indeed, the English courts were the first national courts to have directly considered the legal nature of emissions allowances, and the claims available against a purchaser of stolen allowances (see Box 9.2).

The difficulty of assessing the legal risks of participating in the EU ETS market, combined with the inability of market participants to identify whether the emissions allowances they are purchasing had been the object of theft, meant that market participants were unable to assess or protect themselves from these risks (yet see Box 9.1 for examples of tools offered by various exchanges and service companies to assist market participants in avoiding these risks).

32 French Civil Code, Article 2279.
33 *Ibid.*
34 Sale of Goods Act 1979, ss. 21–24.
35 Burgerliches Gesetzbuch, paras 932 and 935.
36 Directive 2003/87/EC provided the following definition for emissions allowance 'an allowance to emit one tonne of carbon dioxide equivalent during a specified period, which shall be valid only for the purposes of meeting the requirements of this Directive and shall be transferable in accordance with the provisions of this Directive'. This definition cast very little light on what type of legal property that emissions allowances represent.

BOX 9.1 PREVENTATIVE MEASURES TAKEN BY TRADING PLATFORMS AND EXCHANGES AND SERVICE COMPANIES[37,38]

Some exchanges and service companies established short-term solutions to help market participants protect themselves from the legal risks involved with participating in the EU ETS. For example in May 2011 the BlueNext exchange opened a 'safe trading zone', in which only allowances that have had their chain of title traced back to the source of issuance and verified not to have been stolen can be traded. The creation of this 'verified spot' helped the recovery of confidence in the spot market to some extent. The market analyst company Tschach Solutions also offered an 'allocation identifier tool' which claimed to enable companies to identify allowances that were part of their counterparty's initial allowance allocation (if trading with a firm with compliance obligations). These allowances carry a reduced risk of having been stolen. Indeed, if they are still in the account of the firm they were initially allocated to, it is less likely they have ever been traded. Although these tools provided useful ways in which trading entities could manage their market participation risks, they came at a cost and by no means represented long-term comprehensive solutions to protect the market from the confidence-disabling impacts of allowance thefts.

3.4 Impacts of Allowance Thefts on Market Confidence, Trading Volumes and Liquidity

The inability of market players to accurately evaluate or avoid the risks of trading on the EU ETS had a crippling effect on market confidence and trading volumes on the spot market. Following the allowance thefts of early 2011, the closure of national registries and spot exchanges inhibited all spot transactions from taking place. But even after registries resumed trading, trading volumes failed to recover fully. In August 2011, four months after the complete reopening of the registries, the BlueNext spot exchange daily trade volume had levelled out to roughly 200 KT/day, compared to the near 800KT/day figure the exchange was functioning at prior

[37] Point Carbon, 'Spot EUA discount to futures plummets on new safeguards' (2011), Point Carbon news website, available at: http://www.pointcarbon.com/news/1.1533192

[38] Bloomberg, 'Tschach Solutions Offers Software to Cut Carbon-Theft Risks', [2011], Bloomberg Online, available 1 August 2016 at: http://www.bloomberg.com/news/articles/2011-05-13/tschach-solutions-offers-software-to-cut-carbon-theft-risks

BOX 9.2 ARMSTRONG DLW GMBH V WINNINGTON NETWORKS LTD[39]

Armstrong, an operator of two EU ETS installations in Germany, was the subject of a phishing attack in January 2010 during which its registry account username and password were fraudulently obtained by an unknown third party. Having accessed Armstrong's account, the phishers initiated the transfer of 21,000 emissions allowances to the UK registry account of an established trader of emissions allowances, Winnington. Winnington accepted the allowances, before selling them on and paying the phishers for the initial transfer. Winnington was unconnected with the phishers responsible for initiating the transfer of allowances from Armstrong's account and claimed to be unaware that the transfer had been initiated fraudulently. Armstrong claimed, however, that Winnington had undertaken insufficient due diligence before agreeing to accept the emissions allowances and had either known the transfer was fraudulent or at least closed its eyes to the possibility of it being so. Armstrong therefore claimed that under English law it could recover the value of the stolen allowances from Winnington.

On the facts, the court found that Winnington had failed to carry out sufficient due diligence on the transferring party in very suspicious circumstances and therefore had indeed 'closed its eyes' to the possibility of the transfer being fraudulent. The court's assessment of the remedies subsequently available to Armstrong required consideration of the legal nature of emissions allowances. The judgment concluded that emissions allowances were a form of 'intangible property', and that based on this determination, Armstrong was able to recover the value of the stolen allowances from Winnington.

As the transfer in question took place directly from the party who had conducted the theft, its circumstances were very suspicious and warranted a high level of scrutiny. Although this case highlights the level of due diligence that is expected in such circumstances to avoid a restitutionary claim under English law, it is important to stress that this case casts limited light on the risks of unwittingly purchasing stolen allowances in good faith in less suspicious circumstances. The judgment does, however, provide the first in depth analysis of the legal nature of emissions allowances by a domestic court.

to the thefts. This decreased liquidity was reported to have cost the market €110 million.[40]

The blow to spot market confidence was most marked in the trading behaviour of firms that participate for purely financial rather than compliance reasons. Some banks, such as Barclays Capital, withdrew completely

[39] *Armstrong DLW GmBH v Winnington Networks Ltd* [2012] EWHC 10 (Ch).

[40] See K. Nield and R. Pereira, 'Fraud on the European Union Emissions Trading Scheme: Effects, Vulnerabilities and Regulatory Reform' (2011) 20 *European Energy and Environmental Law Review*, Issue 6.

from spot market trading following the allowance thefts of January 2011.[41] This had a severe impact on the trading volume and liquidity of the spot market.

Although the spot market only accounts for a small proportion of the overall EU ETS market (approximately 10–20 per cent), its significance must not be underestimated. The spot market remains an important tool for firms with compliance obligations under the EU ETS, especially small industrial players. Trading on the spot market allows these firms to quickly sell or buy emissions allowances to meet their compliance obligations, or cash in excess allowances.[42] Furthermore, the spot market is not completely isolated from the derivatives market (which makes up the remaining 80–90 per cent of the EU ETS market). Futures and options contracts, for example, are themselves settled by spot transactions at the time of the delivery dates specified in the contract.

4. THE EU ETS REGISTRY SYSTEM: VULNERABILITIES TO FRAUD

4.1 Historic Vulnerabilities of the EU ETS to Fraud

The EU ETS registry system keeps account of the ownership of emissions allowances, tracks allowance trade transactions and records the verified emission levels of individual installations. In this way, these electronic databases keep track of the emissions compliance of installations covered by the scheme, as well as the trading activities of all those involved in the EU ETS market. In order to trade and participate in the EU ETS, a company or individual must open a registry account. The rules governing how these registries operate are therefore integral to determining how the market itself functions, as they not only determine who can gain access to and participate in the market, but also govern the way in which allowances are transferred between accounts. Furthermore, the level of security surrounding the access and operation of these registry accounts determines the vulnerability of the market to fraudulent activities, in particular allowance thefts. This section outlines the main characteristics of the registries system that have made the EU ETS susceptible to VAT

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41 *Financial Times*, 'EU spot carbon market set for partial restart' (2011), available 14 July 2016 at http://www.ft.com/cms/s/0/c5d1392e-2efa-11e0-88ec-00144feabdc0.html#ixzz1P9fZ5h9t.

42 Ellerman et al., *Pricing Carbon: The European Emissions Trading Scheme* (Cambridge, UK: Cambridge University Press, 2010), 135–7.

fraud and allowance thefts as discussed in the previous sections of this chapter.

Until 2012, each EU Member State was responsible for running its own EU ETS registry. Every EU ETS operator and market participant held an account in their Member State's registry, and each individual registry was responsible for recording the issue, transfer and cancellation of emissions allowances of the operators or traders who held accounts with them.

Although rules regarding how these national registries should be run were laid out in EU Regulations,[43] the way in which these rules were implemented and the amount of resources available to do so varied significantly between each national registry. Some registries, therefore, were less secure and more vulnerable to fraudulent attacks than others. As each national registry represented a separate target for criminals and the fraudulent activities conducted through each registry could have significant impacts across the wider market, the whole EU ETS system was only ever as strong as its weakest 'registry link'.

Another problem is that while an open-access registry can foster liquidity, it makes the scheme particularly vulnerable to fraud. One such vulnerability arises from the fact that emissions allowances are not material physical goods *per se*, but represent tradable electronic permits that have been created entirely by legislation.[44] They require no physical paper proof of ownership, but exist only electronically within registry accounts. In order to own EU emissions allowances one therefore needs to have a registry account in which to store them. As a result, the market is a contained one. Unlike commodity markets for physical goods, there is no way that emissions allowances can escape the EU ETS system, as they can only be traded from one registry account to another. One cannot therefore steal emissions allowances, or conduct VAT fraud without first setting up a registry account or fraudulently gaining access to someone else's. The restrictions determining who can open a registry account are therefore an important level of upstream control to prevent fraudsters or thieves accessing the market.

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[43] Commission Regulation (EC) No 2216/2004 of 21 December 2004 for a standardised and secured system of registries pursuant to Directive 2003/87/EC of the European Parliament and of the Council and Decision No 280/2004/EC of the European Parliament as amended by Commission Regulation (EC) No 994/2008 of 8 October 2008.

[44] Note that there has been much debate as to what type of property emissions allowances represent legally. This is a question that is yet to be fully resolved, however. Although see *Armstrong DLW GmBH v Winnington Networks Ltd* [2012] EWHC 10 (Ch) for an analysis by the English courts as to their definition as a form of intangible property.

The EU 2003 Directive which initially established the EU ETS explicitly states that 'any person may hold allowances'.[45] Historically there were very few additional barriers to opening a registry account and entering the market. An account could easily be opened online via any national registry's website. Initially, the only information required by the registry administrator was the name, address, email and telephone number of the person requesting to open the account, as well as evidence to support the identity of that person.[46] This open-access regime was aimed at fostering liquidity and growth in the nascent EU ETS market. Unfortunately, however, this left room for some national registries to be particularly lenient in the 'know-your-customer' (KYC) checks they carried out before approving account applications, thus allowing criminals relatively easy access to the market.

In response to the rise of fraud in the market in 2009–2011, some national registries took independent action to improve the KYC checks carried out on account holders. For example, following the peak of VAT fraud activity in 2009, the Danish Registry reportedly introduced a basic but effective additional KYC filter by asking all their registry account holders the simple question: 'what is the purpose of you holding this account?'. Registry administrators received answers from only 10 per cent of all account holders, and closed down the accounts of the remaining 90 per cent. Some of these accounts were registered under suspicious email addresses linked to fast food restaurants and garages – establishments whose legitimate involvement in the market would be difficult to justify.

Furthermore, the European Commission has taken action to tighten access to the EU carbon markets. In 2010 additional registry account access requirements were imposed on national registries. These included additional KYC checks requiring applicants to provide specific types of proof of identity in order to open an account,[47] and the power of registry administrators to close accounts if they believe account holders to be engaging in suspicious activities.[48] These new minimum access requirements entered into force in October 2010, but by January 2011 they

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[45] Commission Directive 2003/87/EC of 13 October 2003 establishing a scheme for greenhouse gas emission allowance trading within the Community and amending Council Directive 96/61/EC, Article 19.

[46] Annex I, Commission Regulation 994/2008 of 8 October 2008 for a standardised and secured system of registries pursuant to Directive 2003/87/EC of the European Parliament and of the Council and Decision No 280/2004/EC of the European Parliament and of the Council.

[47] *Ibid*, Article 13 and Annex IV.

[48] *Ibid*, Article 27.

had been implemented by only half of the national registries.[49] During the height of the emissions allowance thefts of 2010–11 it was therefore still relatively easy for anyone with fraudulent motives to open and start trading from a registry account, especially in countries which retained more lenient requirements.

Yet the scheme remained vulnerable to the low levels of account security. Historically, account holders and authorised account representatives only required one username and password to gain access to their registry account.[50] No further authentication was required to initiate allowance transfers to other accounts. When compared to the substantial security requirements present in other financial markets, or even online personal banking, this single level of security represented a small hurdle for hackers to overcome in order to illegitimately gain access to and transfer emissions allowances from other individuals' accounts. Although amendments were introduced to the regulation of national registries in 2010 which required secondary authentication for account access and transaction approval, these security improvements were not made obligatory for all EU Member States, and many Member States remained reluctant to implement them due to the costs involved. The vulnerability of registries that had failed to implement these changes was soon exposed during the allowance thefts that occurred in 2010–11.

Moreover, due to the dematerialised nature of emissions allowances, transactions on the spot market are virtually immediate as they are conducted electronically with no need for physical delivery. Allowances can be rapidly transferred from one account to another, and were historically not subject to any additional delay, with spot transactions on the BlueNext exchange taking roughly 15 minutes to complete.[51] Although this transaction speed allowed for rapid and liquid trade, it left the spot market open to illegitimate use. Allowances could be quickly stolen and sold-on before the theft was detected, and transactions facilitating VAT fraud could be rapidly conducted. This allowed multiple carousels to be carried out before coming to the attention of the authorities or account holders.

In some cases of emissions allowance thefts, the effects were minimised by the rapid and coordinated response of the targeted registries. For example, the estimated 488,141 allowances stolen from the Austrian registry in 2011 were all traced to and returned from accounts in Lichtenstein

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[49] CDC Climat Research, 'Closing the door to fraud on the EU-ETS', (2011), Climate Brief No. 4.

[50] Regulation 994/2008, Article 80(3).

[51] Ainsworth (2010), *supra* n. 5.

and Sweden. Many national registries, however, lacked a comprehensive protocol with which to appropriately respond to the detection of allowance theft, and to coordinate with other registries in order to track their onward trade.

This vulnerability is exemplified by the reported reaction to the theft of allowances from Holcim's account in the Romanian registry. The registry administrators were only contactable between 9am and 1pm, with no available emergency contact phone number.[52] By the time the registry administrators became aware of the thefts, many of the stolen allowances had already been subject to a number of complex onward trades, had infiltrated the market and were therefore difficult to trace and recover.

Under the previous Registries regulation, registry administrators were able to suspend access to registries in the case of a security breach.[53] However, mechanisms by which registry administrators could be rapidly alerted to security breaches were not obligatory. Although the obligation on national registry administrators to provide a help desk through which assistance and support could be provided to account holders was introduced in 2010, it did not specify how or when this desk should be accessible to account holders.[54] These details were left to the discretion of Member States, allowing some registries to maintain much less robust response mechanisms than others.

4.2 Recent Reforms to the EU ETS Registry System

Weaknesses in the way that registries were run and the way trade in emissions allowances were conducted were brought to light during the registry attacks in early 2011. In response to these events, a centralised Union registry was introduced in 2012 to replace the national registries system, and further changes to the EU legislation dictating how this new central registry is structured and run were introduced in 2013 (the '2013 Registries Regulation').[55] These changes aimed to reduce the risk of fraud on the EU ETS, improve response mechanisms to fraudulent attacks and avoid the market disruptive effects of fraud.

As part of the Commission's 2007 Climate and Energy Package, in 2009

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[52] Romanian Emission Trading Registry Secretariat, 'The Romanian Emissions Trading Registry Procedures' (2007), available 14 July 2016 at http://rnges.anpm.ro/files2/Annex%208.2.6_%20-%20Roxmanian%20procedures%20for%20ERT_20095.pdf.

[53] Articles 83 and 84, Regulation 994/2008.

[54] Article 60, Regulation 920/2010.

[55] Commission DRAFT Regulation, above n. 10.

amendments were introduced to the EU ETS Directive in order to consolidate the decentralised EU ETS registries system into a single central EU registry operated by the European Commission (the 'Union Registry').[56] This transition was given effect in 2012 with the aim of increasing the security of the registry system and avoiding delayed implementation of security measures. All accounts and allowances are now held on the Union Registry online database. Each Member State retains its own national administrator and national registry section within the Union Registry. These national administrators are responsible for receiving applications to open registry accounts, and collating and verifying supporting KYC documentation.

Strengthened security in trading in emissions allowances

The 2013 Registries Regulation strengthened the KYC checks and documentation requirements which make up the registry account application process, introducing an obligation on applicants to provide the following minimum information to national administrators prior to being able to open a registry account:

- proof that the individual or company requesting the account has an open bank account in a Member State of the EEA;
- company bank account details;
- confirmation of company VAT details;
- a copy of the company's annual report of latest audited financial statements;
- criminal records of the company's directors or of the individual requesting the account.[57]

The 2013 Regulation further provides that registry administrators may refuse ownership of an account if:

> the prospective account holder, or if it is a legal person, any of the directors of the prospective account holder, is under investigation or has been convicted in the preceding five years for fraud involving allowances or Kyoto units, money

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[56] Article 21, Directive 2009/29/EC of the European Parliament and of the Council of 23 April 2009 amending Directive 2003/87/EC so as to improve and extend the greenhouse gas emission allowance trading scheme of the Community.

[57] Article 18 and Annex IV, Commission Regulation No 389/2013 of 2 May 2013 establishing a Union Registry pursuant to Directive 2003/87/EC of the European Parliament and of the Council, Decisions No 280/2004/EC and No 406/2009/EC of the European Parliament and of the Council and repealing Commission Regulations (EU) No 920/2010 and No 1193/2011.

laundering, terrorist financing or other serious crime for which the account may be an instrument;[58]

or if they have 'reasonable grounds' to believe that the accounts may be being used for these purposes.[59]

These provisions should help prevent access to the market by convicted or suspected criminals. Moreover, requiring that those details are disclosed to national administrators should make access more difficult for illegitimate companies or individuals who attempt to join the market for the sole purpose of conducting fraud on the EU ETS. In addition, access to these details should assist the central registry in efforts to track criminals following the detection of fraudulent activity.

Although these additional KYC restrictions undoubtedly represent a step forward for market security, the additional information disclosure requirements outlined above do not drastically alter the market's general regime. The minimum documentation requirements generally represent a low standard to opening a registry account, and the introduction of measures aimed at preventing those convicted of fraud from participating in the carbon markets are unlikely to be effective in restricting market access to a significant extent. Therefore, there has been mixed reaction amongst stakeholders as to whether these measures will sufficiently protect the EU carbon market from fraud. Barclays Capital and a number of Member States have suggested that the European Commission restricts the EU ETS market to compliance entities and regulated firms (that is, financial traders regulated under the EU financial regulations), claiming that the related licensing requirements would restrict access to the market to those with legitimate interests in participating in it. Such proposals contradict the Commission's willingness to attract further liquidity to the market to optimise competition and market efficiency. This conflict highlights that the question of EU ETS market access is not merely a technical discussion on how to maximise security, but prompts more fundamental political questions regarding the balance between open market competition and the minimisation of systemic market risk.

The higher levels of account security can be illustrated by the requirement for two-factor authentication for account holders to access their registry accounts under the 2013 Registry Regulation.[60] In addition, in

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[58] *Ibid*, at Article 22(b).
[59] *Ibid*, at Article 22(c); note that the regulation does not provide any further clarification of what 'reasonable grounds' might consist of however, and what level of suspicion or evidence would be required to refuse the opening of an account. This discretion is left to national administrators.
[60] Article 95(3).

order to initiate transactions from a registry account, an out-of-band confirmation must be provided.[61] To initiate transfers to accounts not on the trusted accounts list of the account holder (see further below), an out of band approval from a second account representative is also required.[62] This 'four-eyes' principle requiring confirmation over two different networks by two different authorised individuals is aimed at limiting the ability of hackers to fraudulently initiate transfers, as the account access details of two separate representatives would have to be obtained and inputted over two separate networks.

There is no doubt that these measures have strengthened to a significant extent the technical security of the registry system. Yet as the sophisticated tactics used by cyber criminals are constantly evolving,[63] it is essential for the integrity of the EU ETS that the registry security system continues to evolve to address future security risks.[64]

Another measure to improve security was the adoption of a 26-hour delay and the definition of trade between 'trusted accounts' and 'holding accounts'. The 2013 Registry Regulation introduced the differentiation between 'holding accounts' and 'trading accounts'.[65] These two registry account types are subject to differing restrictions with respect to the counterparty accounts they are permitted to transfer allowances to and the time delays imposed on transactions initiated from them. Holding accounts may only transfer allowances to accounts listed as 'trusted accounts'. Accounts can be added to an account holder's trusted account list following authorisation by two account representatives and a seven-day delay.[66] In addition, a 26-hour delay is imposed between initiation and finalisation of all transfers from holding accounts. Allowances held in trading accounts, however, may be transferred to trusted accounts with no transaction delay, or to accounts outside of the holder's trusted account list subject to a

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61 Article 39.

62 Article 23(3) and Article 39.

63 House of Lords, 'The EU Internal Security Strategy' (2011), 17th Report of the Session 2010–2012, European Union Committee, House of Lords, London, UK.

64 On the implications of fraud in the EU ETS to EU criminal law cooperation, see K. Nield and R. Pereira (2011) above note 40.

65 Annex 1, Commission Regulation No 389/2013 of 2 May 2013 establishing a Union Registry pursuant to Directive 2003/87/EC of the European Parliament and of the Council, Decisions No 280/2004/EC and No 406/2009/EC of the European Parliament and of the Council and repealing Commission Regulations (EU) No 920/2010 and No 1193/2011.

66 *Ibid*, at Article 26.

26-hour delay.[67] Therefore, under the 2013 Registry Regulation, the only instantaneous transfer of allowances that can be made from any registry account is from a trading account to a counterparty account on the initiating account holder's trusted accounts list. The two-level authorisation and seven-day delay required to add the counterparty account to this trusted accounts list aims to ensure that such immediate transfers are only made to accounts assessed by the account holder as 'trusted', thus decreasing the risk of the rapid fraudulent transfer of emissions allowances out of hacked registry accounts.

The 26-hour delay imposed on all transfers from holding accounts, and transfers from trading accounts to 'non-trusted' accounts was imposed by the 2013 Registry Regulation to provide a period of time in which fraudulent transfers could be identified and prevented prior to the transaction becoming final. Within the first 24 hours of the transaction delay, if an account representative suspects that a transfer was initiated fraudulently they may request the national administrator cancel the transfer.[68] This trusted account and transaction delay system aims to overcome the historic vulnerability of the EU ETS spot market to VAT fraud and allowance thefts due to the speed of allowance transfers possible under the previous registry system, whilst maintaining trade flexibility by enabling the immediate transfer of allowances to trusted accounts.

Legal status of emissions allowances and title transfer rules

In an attempt to attenuate the market disruptive consequences of the circulation of stolen allowances on the EU ETS, the 2013 Registries Regulation aims to clarify and harmonise the legal title of inadvertent purchasers of stolen allowances.

The 2013 Registry Regulation defines emissions allowances as 'fungible, dematerialised' instruments.[69] The definition of allowances as fungible renders all emissions allowances completely substitutable. The 2013 Regulation further states that allowance transactions are final and irrevocable, and that 'no law, regulation, rule or practice on the setting aside of contracts or transactions shall lead to the unwinding in the registry of a transaction that has become final and irrevocable under this Regulation'. Importantly, Article 40 also states that purchasers acting in good faith acquire good title to purchased allowances despite any defects in the ownership title of the seller – meaning that inadvertent purchasers of stolen

67 *Ibid*, at Article 39(3).
68 *Ibid*, at Article 39(4).
69 *Ibid*, at Article 40.

allowances would retain ownership of those allowances. The meaning of 'good faith' however, is left as a question of national law.

By confirming the good title to purchases in good faith, and preventing the unwinding of transactions involving stolen allowances, the Regulation aims to attenuate the risks associated with inadvertently purchasing stolen allowances. Yet uncertainties remain as to the application of Article 40 of the Regulation and contradictions between its provisions and domestic law in some Member States, leading to difficulties in implementation. The recognition of good title to inadvertently purchased stolen allowances, for example, would directly contradict national law in Member States such as the UK. This became evident in the *Armstrong* judgement discussed in Box 9.2 above, where the English courts ordered the restitution of stolen allowances to the original victim of a phishing attack as the due diligence conducted by the inadvertent purchaser was not sufficient in the circumstances.

5. EU FINANCIAL MARKETS OVERSIGHT REGULATIONS

EU-level financial market oversight regulation applies to commodity trading in general as well as specifically to trade in emissions allowances. According to the European Commission, the purpose of market oversight regulations is to enable fair and efficient trading conditions for all participants, as well as to prevent the inappropriate use of markets for fraudulent activities.[70] Yet the risk of market misuse posed by different segments of commodity markets can differ considerably, depending on the nature of the trading product or type of transaction (whether derivatives or spot), the platform through which the trade is executed (whether, for example, in exchanges or multilateral trading facilities); or the entity carrying out the trade (for example, a financial intermediary or a EU ETS compliance entity). As such, the weight of EU market oversight regulation applied to different segments of commodity markets vary according to different factors.

It should be noted that emission allowances differ significantly to most other physical commodities in that they are dematerialised instruments with a high value and zero volume. This may call for different levels of

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[70] European Commission, '*Emissions trading: Questions and answers on enhanced market oversight for the European carbon market*' (2010), available 14 July 2016 at http://europa.eu/rapid/pressReleasesAction.do?reference=MEMO/10/697.

regulatory burden and market oversight to the EU carbon markets than that applicable to other commodities sectors. The extent of fraud on the EU ETS over the past few years has exposed the additional risks of market misuse – especially on the spot market – thus calling for more effective regulation and supervision of the spot market segment.

This section of the chapter discusses the application of EU financial regulations to the EU carbon markets; and analyses the extent to which recent legislative reforms adopted by the EU will enable it to address the regulatory weaknesses that have led the EU carbon markets to become vulnerable to fraudulent activities.

5.1 The Derivatives and Spot Markets: Vulnerabilities to Fraud

The derivatives market has grown from a value in 1998 of less than US$100 trillion to a peak figure of nearly US$700 trillion at the end of 2007.[71] This market consists of forwards and futures contracts;[72] as well as swaps and options.[73] The trade in derivatives makes the vast majority of market activity on the EU ETS. In 2009, nearly 80 per cent of all trading activity was in derivative products.[74] These derivative instruments play a crucial role in the EU ETS market, as they provide companies with compliance obligations under the EU ETS with flexible ways through which they can manage their carbon price risks. Moreover, derivatives tend to be offered by financial intermediaries (banks and credit institutions) who are capable of taking on the price risks involved.

In comparison with the derivatives market, the spot market is considerably smaller and accounts for only 10–20 per cent of trading activity on

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[71] See the Jacques de Larosiere, *The High Level Group of Financial Supervision in the EU*, Brussels, 25 February, 2009; (the 'Larosiere Report') available 14 July 2016 at http://ec.europa.eu/finance/general-policy/docs/de_larosiere_report_en.pdf.

[72] Forwards and futures are contracts for the delivery of a set volume on a specified future delivery date. In turn, a swap is a contract through which one asset is substituted for another. For example futures contracts with different delivery dates can be 'swapped', or EUA and CER allowances can be swapped. *See* Ellerman et al., *Pricing Carbon: the European Emissions Trading Scheme* (Cambridge, UK: Cambridge University Press, 2010) 135–7.

[73] Options are contracts through which the buyer is granted the right (but not the obligation) to purchase a certain volume at a specified date for a set price. See *ibid*.

[74] European Commission (2010), '*Towards an enhanced market oversight framework for the EU Emissions Trading Scheme*', Communication from the Commission to the European Parliament and the Council, Brussels, Belgium.

the EU-ETS.[75] Spot contracts account for trades that are delivered between 24 and 48 hours after they are negotiated, and so present an important way for entities with compliance obligations under the EU ETS to quickly sell or acquire emissions allowances.[76]

While the EU ETS derivatives market was not subject to any significant instances of fraud (allowances theft or VAT-fraud) – which, as will be discussed below, reflects the higher levels of regulatory oversight applicable to the derivatives markets – it was the trade in emissions allowances in the spot market which was subject to significant fraudulent activities in 2009–2011. This could be attributed to the fact that the EU ETS spot market (like any other spot commodities markets) was unregulated at the EU level; and so no EU-wide legislation existed for spot transactions. Therefore, it was the EU Member States' national laws that applied to spot trading in emissions allowances. However, only a few Member States had introduced additional market oversight regulation to their domestic spot emissions trading markets.[77] This meant that no obligatory licencing, supervision of activities, or reporting requirements applied to spot market participants in many of the EU Member States. Despite the fact that most commodity spot markets are similarly unregulated, the completion of spot transactions of emissions allowances are near immediate and lack the requirement for delivery of any physical product, making their trade more vulnerable to fraud than is the case of other commodities traded in spot markets. Moreover, as was discussed above, the 'dematerialised' nature of emissions allowances makes them particularly vulnerable to fraud as compared to other commodity spot markets.

Given these additional vulnerabilities and the evident focus of allowance thefts and VAT fraud on the spot market, there has been concern that the level of market oversight was inappropriately low, leaving this market open to misuse.[78]

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75 *Ibid*. There were concerns that this proportion could be further decreased following the 2010–11 allowance thefts.

76 Ellerman (2010), above.

77 In August 2010 France passed legislation to allow for the extension of regulated markets rules to the spot carbon market; Germany already regulates commodity spot trades that take place through exchanges; Romania defines emissions allowances as financial instruments. See European Commission (2010), '*Towards an enhanced market oversight framework for the EU Emissions Trading Scheme'*, Communication from the Commission to the European Parliament and the Council, Brussels, Belgium.

78 *See also*, Nield and Pereira, above, n. 40.

5.2 Market Oversight of EU ETS – MiFID I and II

The Markets in Financial Instruments Directive (MiFID)[79] represents the main body of EU financial regulation applicable to the derivatives markets.[80] MiFID aims to protect investors (particularly retail investors) by regulating the financial intermediaries that provide derivative products.[81]

The extent to which the original version of MiFID covered trade in emissions allowances was not clear. Although MiFID has established a legal framework applicable to derivative products by defining them as ‘financial instruments’[82] (and thus applied to emissions allowance *derivatives*), the Directive did not clarify whether emissions allowances themselves fell under the MiFID definition of financial instruments. This led to distortions in the sense that allowances traded as derivatives products were subject the requirements under MiFID, while the spot market for emissions allowances was not regulated under MiFID. In light of the lack of guidelines from the EU institutions on this issue, the EU Member States were left to decide for themselves how allowances were treated under their own jurisdiction, leading to heterogeneous approaches across the EU Member States.[83] As was suggested by the International Emissions Trading Association (IETA) during the MiFID public consultations that

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[79] Directive 2004/39/EC of the European Parliament and of the Council of 21 April 2004 on markets in financial instruments amending Council Directives 85/611/EEC and 93/6/EEC and Directive 2000/12/EC of the European Parliament and of the Council and repealing Council Directive 93/22/EEC.

[80] See also, the European Market Infrastructure Regulation (EMIR) adopted in 2012 (in force from 15 March 2013), which extends clearing requirements to over-the-counter derivatives trading. See Regulation (EU) No 648/2012 of the European Parliament and of the Council of 4 July 2012 on OTC derivatives, central counterparties and trade repositories.

[81] The Directive stipulates that intermediaries require authorisation before they can offer these types of trading products. Once licenced, their activities are then closely supervised by the Member State’s financial regulator or national central bank to ensure that they abide by a number of operational and reporting requirements aimed at ensuring transparency and investor protection.

[82] Directive 2004/39/EC, Annex I, Section C.

[83] Romania was one of the few Member States that had already defined independently emissions allowances as financial instruments. Yet some level of legal harmonisation in the way emissions allowance transactions are treated had been achieved by the reliance on standardised trading contracts, such as that drawn up by IETA. See IETA, ‘Emissions trading master agreement for the EU emissions trading scheme, Version 3.0’ (2008), IETA website, available 3 August 2016 at: http://www.ieta.org/resources/Resources/Trading%20Documents/uk-1597905-v1-ieta_etma_v3_0_-_master_agreement_and_sched.pdf.

took place in 2010–2011, those divergent approaches decreased market confidence (in particular because of the associated lack of clarity surrounding ownership and liability rules) and detracted potential investors.[84]

In light of these and other concerns that were raised during the MiFID consultations (linked to the regulatory failures in connection with the global financial crisis in 2007–2009),[85] the EU Commission adopted a legislative proposal in October 2011 aimed at reforming MiFID.[86] This proposal aims to take account of technological innovations and the need for new safeguards for algorithmic and high frequency trading activities (which have increased drastically the speed of trading and pose systemic risks); and has introduced a new Organised Trading Facilities (OTF) category, which aims to increase transparency and competition in the EU capital markets trading activities.

Pursuant to the reforms under the revised MiFID II adopted in May 2014,[87] emission allowances will gain the status of 'financial instrument'.[88] So one of the main achievements of MiFID II is that it extends the EU financial markets rules to the spot markets in emissions allowances. This is expected to ensure better consistency and diminish the divergent approaches that were implemented by the EU Member States when defining emissions allowances. Moreover, these reforms are expected to make trading in emissions allowances less vulnerable to fraud given the increased

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[84] Particularly those from outside the EU, for whom there was particular uncertainty surrounding the application of EU market oversight regulations to the transaction. See ETA, 'IETA response to MiFID consultation', [2011] International Emissions Trading Association website, available 3 August 2016 at: https://circabc.europa.eu/webdav/CircaBC/FISMA/markt_consultations/Library/financial_services/MIFID%20-%20Review%20of%20the%20Markets%20in%20Financial%20Instruments%20Directive%20(2011)/registered_organisation/IETA%20-%20International%20Emissions%20Trading%20Association.pdf

[85] See European Commission, *Public Consultation, Review on the Markets in Financial Instruments Directive* (2010), Directorate General Internal Markets and Services; MiFID was open to Stakeholder consultation from 8 December 2010 to 2 February 2011.

[86] Proposal for a Directive of the European Parliament and of the Council on markets in financial instruments repealing Directive 2004/39/EC of the European Parliament and of the Council (Recast) (MiFID II).

[87] Directive 2014/65/EU of the European Parliament and of the Council of 15 May 2014 on markets in financial instruments and amending Directive 2002/92/EC and Directive 2011/61/EU (recast)' and published in the Official Journal of European Union on 12 June 2014 (OJ L 173, pp. 349–496).

[88] MiFID II extends Section C of Annex 1 to MiFID by adding item 11 with a new category of financial instruments—'emission allowances consisting of any units recognised for compliance with the requirements of Directive 2003/87/EC (Emissions Trading Scheme).

financial markets regulatory and supervisory oversight that will ensue from the broader application of MiFID to emissions trading. However, it should be noted that MiFID II does not specify whether emission allowances are to fall under any of the existing categories of financial instruments (such as derivatives or securities), or whether it should represent a separate category on its own right.[89] This suggests that emission allowances may be regarded as a separate category of financial instruments distinct from securities representing title to capital in a corporate entity or title to debentures.[90]

5.3 Analysis of the MiFID II Reforms

By classifying *all* emissions allowances as 'financial instruments' under Annex I, c) para. 11 of MiFID, the MiFID II reforms have extended the scope of EU financial markets regulation to both the derivatives and spot markets. This means that the current operators of trading venues on which emission allowances are traded will be obliged to obtain a MiFID authorisation and conduct their activities in one of the trading platforms recognised under MiFID: exchanges, multilateral trading facilities or organised trading facilities. The Commission has decided to implement these reforms despite the fact that, in a stakeholder consultation launched by the Commission in 2010 aimed to explore potential reforms of the EU carbon market's oversight framework, stakeholders appeared to be divided on this issue.[91]

One of the advantages of extending EU market oversight regulations to the spot market is that it would act to increase investor protection and transparency, as well as provide protection from possible future threats to the

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[89] *See* Krzysztof Gorzelak, 'The Legal Nature of Emission Allowances Following the Creation of a Union Registry and adoption of MiFID II—Are They Transferable Securities Now?' (2014) 9(4) *Capital Markets Law Journal* 373–387.

[90] *Ibid*.

[91] See in particular the responses to Question 66 of the MiFID consultation document (available at: http://ec.europa.eu/internal_market/consultations/docs/2010/mifid/consultation_paper_en.pdf). All consultation responses available 3 August 2016 at: https://circabc.europa.eu/faces/jsp/extension/wai/navigation/container.jsp. Under this public consultation, stakeholders were asked: 'What is your opinion on whether to classify emissions allowances as financial instruments?' See also, European Commission, 'Towards an enhanced market oversight framework for the EU Emissions Trading Scheme' (2010), Communication from the Commission to the European Parliament and the Council, Brussels, Belgium.

EU ETS, such as market abuse and money laundering.[92] During the MiFID review, some of the consultation respondents argued that this would be essential to support the integrity of the ETS system as a whole.[93] It was suggested that MiFID (and associated market oversight Directives) constitute a well-tried regime that has proved effective in other markets. It was further argued that applying the same regulatory framework across the market would provide a simpler regulatory landscape and a harmonised legal definition that would be less confusing for new investors.[94] Fundamentally, as was discussed above, one of the key advantages of the extension of MiFID to emissions allowances (beyond the market in derivatives) is that it would provide a welcome market-wide solution to the problem of VAT fraud on the EU ETS, as domestic trade in financial instruments is *exempt from VAT*, subject to the fulfillment of certain conditions.[95]

On the other hand, there were concerns that the application of extensive EU financial market regulation to all trade in emissions allowances will introduce a large administrative and financial burden on firms with compliance obligations under the EU ETS. In particular, the additional need for licensing, tighter contractual agreements, higher trade-related reporting and supervisory requirements under MiFID (and associated EU financial regulations)[96] could significantly increase the cost of trading

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[92] See the Money Laundering Directive 2005/60/EC. This directive introduces know-your-customer (KYC) requirements for credit institutions and investment firms to check the identity of their clients, as well as the nature of their trading activities. See also, Proposal for a Directive on the prevention of the use of the financial system for the purpose of money laundering and terrorist financing COM/2013/045 final – 2013/0025 (COD).

[93] *See* NASDAQ OMX, 'European Commission Public Consultation on Review of Markets in Financial Instruments Directive (MiFID), Reply from NASDAQ OMX' (2011) available 3 August 2016 at http://www.nasdaqomx.com/digitalAssets/80/80004_mifidreview_february2011.pdf; NASDAQ OMX run the Nordpool exchange, one of the largest EU emissions trading exchanges

[94] *Ibid.*

[95] See Article 135, Council Directive 2006/116 of 28 November 2010 on the common system for value added tax.

[96] The Market Abuse Regulation and Criminal Sanctions for Market Abuse Directive (discussed below); Anti Money Laundering Directive; Settlement Finality Directive will apply to emissions trading. At the same time, emission allowances trade will fall outside the scope of the following EU financial market legislation: the Prospectus Directive; Transparency Directive, and the Undertakings for Collective Investment in Transferable Securities (UCITS) Directive. *See* 'Review of the Markets in Financial Instruments Directive (MiFID) and Proposals for a Regulation on Market Abuse and for a Directive on Criminal Sanctions for Market Abuse: Frequently Asked Questions on Emission Allowances', MEMO/11/719 Brussels, 20 October 2011.

for those firms that only participate in the spot market.These costs would be especially significant to small compliance firms who have no previous experience of trading in financial instruments and have fewer resources with which to absorb the increased cost burden.[97]

An alternative option for carbon markets oversight reform that was originally suggested by the Commission would be the design and application of a specific market oversight framework for trade in emissions allowances. This could either be achieved by establishing an entirely new set of rules specifically tailored to the carbon market, or possibly by separately extending the application of *specific* existing MiFID rules (and associated EU financial regulations) to emissions trading.[98] One of the advantages of such a bespoke regime is that it could allow for a more appropriate mechanism to be developed in line with the specific nature of emissions allowances and the risks present within that market. This could avoid unnecessary regulatory burdens and costs falling on market participants that do not pose a significant risk to the market. For these reasons some commentators have argued that this constitutes a more proportionate approach than the definition of emissions allowances as financial instruments,[99] thus allowing room for the application of a legal definition more fitting to emissions allowances.This view was shared by some government departments and industry representative bodies during the MiFID consultation responses.[100] However, this proposal did not prevail during the MiFID negotiations process, and – for the reasons that will be elaborated below – the Commission decided to extend EU financial markets to spot trade in emissions allowances.

In an earlier work published in 2011, the authors of this chapter have

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[97] Economic studies have shown high trading costs within pollution permit markets can lead to a decreased willingness to trade among compliance firms, as the inclusion of these costs within firms' micro-rational trading/abatement decisions may decrease the perceived benefit of trading. See e.g. Stavins, R., 'Transaction Costs and Tradable Permits' (1995) 29 *Journal of Environmental Economics and Management* 133–48; and Gangadharan, L., 'Transaction Costs in Pollution Markets: An Empirical Study', (2000) 76(4) *Land Economics*, 601–14.

[98] *See* European Commission, 'Discussion paper in view of a European Climate Change Programme (ECCP) stakeholder meeting on carbon market oversight organised by the Commission services' (2011), Brussels, Belgium.

[99] See e.g. Prada, M., 'The Regulation of CO_2 markets: Assignment Report by Michel Prada, Emeritus General Inspector of Finance' (2010), Paris, France. On the costs of compliance with regulatory standards generally, see Stavins, above n. 97; and Gangadharan above n. 97.

[100] For example: HM Treasury, French Ministry of Finance and IETA all expressed support for a bespoke regime within their MiFID consultation responses.

argued generally against the extension of EU financial regulations to the carbon markets.[101] We have argued that this extension could lead to increased costs that small compliance firms might occur under such framework.[102] These levels of regulatory burden and supervisory review were only likely to increase after the legal reforms adopted by the EU in response to the global financial crisis in 2007–2009.[103] Fundamentally, we have argued that an extension of the full scope of EU financial market oversight regulation could be regarded as a disproportionate response to the impacts of fraud on the carbon market (in particular in light of the adoption of the EU Registry Regulation and the other reforms discussed above).[104] Moreover, MiFID provisions mainly act to protect investors and to ensure market transparency, with many of the associated safeguards focusing on the protection of uninformed *retail* customers. In contrast, the EU ETS spot market does not tend to attract investment from the general public.

Those concerns were to a large extent addressed with the introduction of exceptions under MiFID II for ETS compliance entities. Indeed, individual ETS compliance buyers buying and/or selling emission allowances on their own account are exempt from authorisation and compliance duties under the MiFID II.[105] Moreover, persons dealing on their own account and entities which provide investment services in emission allowances are exempted as long as this activity will be ancillary to their main business activity; and they are not part of a financial group.[106] Therefore, most ETS

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[101] See Nield and Pereira, above, n. 40, 273–9.

[102] Although both small compliance firms and the trading arms of larger operators would come under exemptions under MiFID, these exemptions were subject to revision in the on-going review process. So it is possible that as a result of this revision, compliance buyers and trading subsidiaries could become subject to burdensome regulations. See *ibid.*

[103] These reforms to EU financial markets supervision and regulation largely implement the recommendation of the 'Larosiere Report'. See Jacques de Larosiere, *The High Level Group of Financial Supervision in the EU*, Brussels, 25 February, 2009.

[104] *Ibid.*

[105] See Article 2(e) Directive 2014/65/EU, which states that 'operators with compliance obligations under Directive 2003/87/EC who, when dealing in emission allowances, do not execute client orders and who do not provide any investment services or perform any investment activities other than dealing on own account, provided that those persons do not apply a high-frequency algorithmic trading technique.'

[106] '(. . .) and that main business is not the provision of investment services within the meaning of this Directive or banking activities under Directive 2013/36/EU' See Article 2(J) *ibid.*

compliance buyers which have limited trading activity which is ancillary to their main business will be exempt from MiFID.[107]

In addition, the Commission has raised subsequently other strong (and generally convincing) arguments in support of its decision to extend the application of MiFID to the emissions spot market, and therefore deciding against creating a tailor-made regime for trading in emissions allowances. According to the Commission – and as acknowledged by stakeholders during the MiFID consultation – a bespoken regime would have to reproduce the overall approach and most of the technical solutions which were already applicable under MiFID (and associated EU financial regulations).[108] Therefore, an emerging regime for the spot carbon market would need to be fully coherent in any event with the regulation of financial markets, in particular as the greatest share of the carbon market consists of derivatives products and is therefore covered by financial markets rules.[109] Moreover, as the spot segment is currently a very small share of the overall carbon market activity, the Commission doubts that this would justify the development of a fully separate regime, which might be inconsistent with the rules that already govern the largest part of the market. The Commission has also raised broader concerns regarding consistency between the spot and derivatives markets, suggesting that placing the spot carbon trade under a potentially less stringent set of rules than is the case for carbon derivatives trade could eventually be detrimental to the spot segment's prospects. Furthermore, the Commission notes that the regulatory framework for the auctioning of emission allowances as of 2013 is closely aligned in key respects with the rules applicable to the secondary market in financial instruments.[110] Therefore, by defining emissions allowances as financial instruments, the reforms adopted under MiFID II could be regarded overall as a positive development, as it will allow for better consistency between the spot and derivatives markets, whilst simultaneously exempting smaller compliance entities from the MiFID regulatory requirements.

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[107] *See* 'Review of the Markets in Financial Instruments Directive (MiFID) and Proposals for a Regulation on Market Abuse and for a Directive on Criminal Sanctions for Market Abuse: Frequently Asked Questions on Emission Allowances', MEMO/11/719 Brussels, 20 October 2011.

[108] *Ibid*.

[109] *Ibid*.

[110] *Ibid*. The Commission has discarded the option of applying the Regulation on Energy Markets Integrity and Transparency (REMIT) as an alternative to extending the EU financial regulations to the carbon markets. See *ibid*.

5.4 Market Abuse

Products defined as financial instruments under MiFID are also subject to other cross-sector EU economic regulation such as the Market Abuse Directive (MAD)[111] and the anti-money laundering Directive.[112] The MAD aims to prevent insider dealing and market manipulation via the imposition of measures to detect and sanction abuse.[113] Market manipulation occurs when a market player acts to control the rest of the market's perception of the state of supply and demand, and then takes a position to exploit the resulting effect on price. For example, it could happen by a market participant 'squeezing' the market by buying off and retaining a large amount of allowances to give a false impression of scarcity, waiting for prices to rise as a result, and selling them on at this inflated price.[114] Another type of market abuse is insider dealing, which happens when a trader makes trading decisions or deals based on 'inside information'. 'Inside information' includes information that is not publicly available but is likely to have an effect on price.[115] In recent years, policymakers have recognised the importance of controlling insider dealing and market manipulation not only to protect shareholders against the misuse of proprietary information belonging to the company and others to whom a fiduciary duty is owed but also to promote a more efficient functioning of the capital markets by fostering minimum standards of fair dealing and best practices.[116] Indeed, the financial crisis in 2007–2009 has demonstrated how quickly markets react to price-sensitive information and how this can undermine investor confidence and financial stability. Moreover, high-frequency trading practices can easily engage in market

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[111] Directive 2003/6/EC of the European Parliament and of the Council of 28 January 2003 on insider dealing and market manipulation (market abuse).

[112] Directive 2005/60/EC of the European Parliament and of the Council of 26 October 2005 on the prevention of the use of the financial system for the purpose of money laundering and terrorist financing. See also, Proposal for a Directive on the prevention of the use of the financial system for the purpose of money laundering and terrorist financing COM/2013/045 final – 2013/0025 (COD).

[113] See also, European Commission Communication on 'Reinforcing sanctioning regimes in the financial services sector' of 8 December 2010 (see IP/10/1678).

[114] Prada (2010), above, n. 103.

[115] *Ibid.*

[116] K. Alexander, 'Market Structures and Market Abuse', in Gerard Caprio (ed.) *Handbook of Safeguarding Global Financial Stability: Political, Social, Cultural and Economic Theories and Models* (London: Academic Press, 2013).

manipulation that can destabilise the market and possibly lead to a type of 'flash crash' in the capital markets.[117]

In October 2011 the Commission decided to propose reforms to the EU market abuse legislation. The scope of the original Directive 2003/6/EC was limited as it focused on financial instruments admitted to trading on a regulated market or for which a request for admission to trading on such a market has been made. The EU Market Abuse Regulation (MAR, 2014)[118] aims to address the fact that in recent years financial instruments have been increasingly traded on multilateral trading facilities (MTFs). There are also financial instruments which are traded only on other types of organised trading facilities (OTFs) or only over-the-counter (OTC). Thus, the scope of Market Abuse Regulation (MAR) is to include any financial instrument traded on a regulated market, an MTF or an OTF, and any other conduct or action which can have an effect on such a financial instrument irrespective of whether it takes place on a trading venue. In line with the MiFID II reforms, the new market abuse regime includes several carbon markets-specific requirements. For example, MAR contains a specific definition of inside information, a tailored inside information disclosure duty, and a complete coverage of the primary market (auctioning).[119]

In addition, the EU institutions adopted a Directive on criminal sanctions for insider dealing and market manipulation (2014).[120] Under this directive, it is recognised that the imposition of criminal sanctions for market abuse will have increased deterrent effect on potential offenders. It requires the EU Member States to criminalise market manipulation,[121] including in the context of trade in emissions allowances.[122] The market

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[117] *Ibid*. See also, Di Noia, C. (2012) 'Reviewing the EU's market abuse rules'. ECMI Policy Brief No. 19, 27 February 2012 (Policy Paper).

[118] Regulation (EU) No 596/2014 of the European Parliament and of the Council of 16 April 2014 on market abuse ('market abuse regulation' or 'MAR').

[119] See e.g. Article 17(2), MAR ('public disclosure of insider information'). *See* 'Review of the Markets in Financial Instruments Directive (MiFID) and Proposals for a Regulation on Market Abuse and for a Directive on Criminal Sanctions for Market Abuse: Frequently Asked Questions on Emission Allowances', MEMO/11/719 Brussels, 20 October 2011.

[120] Directive 2014/57/EU of the European Parliament and of the Council of 16 April 2014 on criminal sanctions for market abuse ('Market Abuse Directive').

[121] Article 5(1) ibid.

[122] See Article 1(2): 'This Directive also applies to behaviour or transactions, including bids, relating to the auctioning on an auction platform authorised as a regulated market of emission allowances or other auctioned products based thereon, including when auctioned products are not financial instruments, pursuant to Commission Regulation (EU) No 1031/2010' *Ibid*.

abuse directive (MAD II) defines market manipulation as giving false or misleading signals as to the supply of, demand for, or price of, a financial instrument; securing the price of one or several financial instruments or related spot commodity contracts at an abnormal or artificial level.[123] Moreover, it requires Member States to criminalise 'insider dealing' as well as the 'unlawful disclosure of inside information'.[124] This Directive further requires Member States to make the most serious market abuse offences punishable by a maximum term of imprisonment of at least four years.[125]

Importantly, unlike as is the case with MiFID II, neither MAR nor MAD II contains provisions which exempt EU ETS compliance entities from market abuse rules. However, in the case of MAR, it foresees an exemption 'for those emission allowance market participants the activity of which (expressed in terms of annual emissions or thermal input or a combination thereof) would be below a certain minimum threshold'.[126] This threshold will be determined by the Commission 'by means of a delegated act'.[127] The implementation of this exemption is commendable given that, in practice, only information concerning activities of the largest emitters in the EU ETS – which typically belong to the EU power sector – can be expected to have a significant impact on the carbon price formation.[128]

Market abuse does not appear to be currently a significant threat to the EU ETS. Unlike in the electricity market, no operators in the EU ETS market find themselves in a dominant position from which to easily manipulate the market. This is partly due to the involvement of financiers in the market which greatly increases the number of market players.[129] As regards the offence of insider dealing, what would actually constitute inside information on the EU carbon market is unclear, as there is no one with enough market power to manipulate the market.[130]

Despite this, one cannot underestimate the potential and real risks of market abuse in the EU carbon markets. The fact that the EU ETS has not been subject to date to instances of market abuse does not mean

123 See Article 5(2)(a) *ibid.*
124 See Article 3 and Article 4(2) *ibid.*
125 Article 7, *ibid.*
126 Preamble, Recital 51, MAR.
127 *Ibid.*
128 See also, Review of the Markets in Financial Instruments Directive (MiFID) and Proposals for a Regulation on Market Abuse and for a Directive on Criminal Sanctions for Market Abuse: Frequently Asked Questions on Emission Allowances,' MEMO/11/719 Brussels, 20 October 2011.
129 See Nield and Pereira, above, n. 40.
130 *Ibid.*

that the risk of this happening in future is not real. As the market grows in value, and attracts larger financial players such as hedge funds and pension funds, this increases the risk of these market participants being able to gain dominance which would enable them to manipulate the carbon markets.[131] Therefore, although it is difficult to assess the likely future impacts of market abuse in the EU carbon markets, the recently adopted Market Abuse Regulation (MAR) and MAD II may act as an appropriate preventative measure against these future risks.

6. CONCLUSIONS

Although neither VAT fraud nor emissions allowance thefts have directly affected the environmental integrity of the EU ETS, both these forms of fraud have had negative financial, market function and public confidence implications which ultimately impact on the operation and effectiveness of the EU ETS. Due to imperfect implementation of the reverse-charge VAT treatment of emissions allowances across Member States, parts of the market are still vulnerable to VAT fraud. However, with the definition under MiFID II of EU ETS allowances as financial instruments, VAT-fraud is likely to become less of a threat to the European carbon markets as in general no VAT is paid in domestic sales of emissions allowances.

Although a central registry under Phase III of the EU ETS is likely to strengthen the security in trading in allowances, the Union registry will not necessarily itself be immune to cyber attacks. Being larger than individual national registries, it is possible that it may even become a more attractive target for criminals. Therefore, the security improvements implemented under the 2013 Registry Regulation are a positive preventative step against thefts in emissions allowances. Yet it should be noted that the Union Registry system remains essentially an open-access regime given the still relatively low standards of minimum documentation required for opening a Registry account, suggesting that the Registry system will not be immune to fraud in future.

In general, by defining emissions allowances as financial instruments, the reforms adopted under MiFID II can be regarded as a positive development, as it will allow for better consistency between the spot and derivatives markets, whilst simultaneously exempting smaller compliance entities from the MiFID regulatory requirements. And although market

131 *Ibid*.

abuse has not yet posed a major threat to the EU ETS, the reforms under MAD II and MAR could be regarded as an important preventative step against future threats of market manipulation and insider dealing that may arise in the EU carbon markets.

10. Implementation challenges for emission trading schemes: the role of litigation

*Josephine van Zeben**

1. INTRODUCTION

Within economic theory, emission trading schemes have long been advocated for their relative simplicity and their allocative and dynamic efficiency.[1] Despite these apparent strengths,[2] regulators have only recently started to meaningfully incorporate emission trading into their toolbox. The United States' Acid Rain program was the first large scale application of emission trading to the regulation of environmental 'bads', and its consequent success signalled a watershed moment for the use of emission trading schemes.[3] However, after the initial excitement of the 1970s, the use of ETS within environmental regulation remained limited.[4] The definite turning point did not arise until the late 1990s, with the European Union's

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[1] On cost-efficiency, see R Lane, 'The Promiscuous History of Market Efficiency: The Development of Early Emissions Trading Systems' (2012) 21(4) *Environmental Politics*, 583–603.

[2] For an in-depth discussion of the economic foundations of ETS see Chapter 2 by Dan Cole in this volume. For a recent critique of the legal, economic and political science literature on emissions trading, see B Stephan and R Lane, 'Zombie Markets or Zombie Analyses? Revivifying the Politics of Carbon Markets', in B Stephan and R Lane (eds) *The Politics of Carbon Markets* (Routledge 2015). See also S Bogojević, *Emissions Trading Schemes: Markets, States and Law* (Hart Publishing, 2013) for a critique of the legal literature.

[3] See e.g. AD Ellerman et al, *Markets for Clean Air: the US Acid Rain Program* (Cambridge/New York, Cambridge University Press, 2000); TH Tietenberg, *Emissions Trading, An Exercise in Reforming Pollution Policy* (2nd edn, Washington DC: Resources for the Future 2006).

[4] A Simons and J-P Voss, 'Politics by Other Means: The Making of the Emissions Trading Instrument as a 'Pre-history' of Carbon Trading', in Stephan and Lane (n 2), suggest that continuous lobbying took place since the 1960s through 'constituency formation', even if 'real' carbon trading did not start until the 2000s (at 63). See also RG Noll, 'Economic Perspectives on the Politics of Regulation', in R Schmalensee and R Willig (eds), *Handbook of Industrial Organization* (vol 2, New York: North-Holland 1989), 1275; RN Stavins, 'What

commitment to emissions trading as its main tool for Kyoto Protocol (KP) compliance.[5] The subsequent creation of the European Union's Emissions Trading Scheme (EU ETS) ensured a place for ETS within environmental regulation.[6]

The EU ETS surpasses the scale and scope of any previous applications of emission trading and has provided many additional insights regarding the legal complexity of ETS regulation.[7] These legal complexities cover a broad range of implementation processes including monitoring and enforcement,[8] market oversight,[9] and the strategic behaviour between regulators within the (EU) ETS.[10] Many of these complications have been explicitly addressed during the planned reform processes of the EU ETS, and led to changes in the EU ETS' design and implementation.[11]

This chapter focuses on an implementation challenge that does not necessarily present itself as such; litigation. Experiences with the EU ETS have made the potential impact of litigation on ETS development an increasingly salient issue for regulators. Nonetheless, the implications of

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Can We Learn from the Grand Policy Experiment? Lessons from So2 Allowances Trading' (1998) 12(3) *Journal of Economic Perspectives*, 69–88.

[5] The EU had initially been very skeptical of emissions trading and did not advocate this course of action until the 1997 Kyoto Conference of the Parties, see generally AD Ellerman and BK Buchner, 'The European Emissions Trading Scheme: Origins, Allocation, and Early Results' (2007) 1(1) *Review of Environmental Economics and Policy*, 66–88. As noted by Stephan and Lane (n 2), the United States itself notably did not support ETS as a mechanism for emission reductions despite its earlier successes under the Clean Air Act (at 10).

[6] Directive (EC) 2003/87 of 13 October 2003 establishing a scheme for greenhouse gas emission allowance trading within the Community and amending Council Directive 96/61/EC, [2003] OJ L 275/32.

[7] The EU ETS covers more than 11,000 power stations and manufacturing plants in the 28 EU Member States as well as Iceland, Liechtenstein and Norway. Aviation operators flying within and between most of these countries are also covered. In total, around 45% of total EU emissions are limited by the EU ETS. It is the world's biggest emissions trading market, accounting for over three-quarters of international carbon trading. DG Climate Action, The European Union Emission Trading Scheme (EU ETS), available 14 July 2016 at http://ec.europa.eu/clima/publications/docs/factsheet_ets_en.pdf.

[8] M Peeters, 'Inspection and Market-based Regulation through Emissions Trading: The Striking Reliance on Self-monitoring, Self-reporting and Verification' (2006) 2(1) *Utrecht Law Review* 177–95.

[9] See e.g. M Lederer, 'Market Making via Regulation: The Role of the State in Carbon Markets' (2012) 6 *Regulation & Governance* 1–21.

[10] For an overview of the academic literature, see J van Zeben, *Regulatory Competence Allocation in the European Union Emissions Trading Scheme* (Cambridge University Press 2014); see also Stephan and Lane (n. 2).

[11] See Section 3.B.

litigation for the design and functioning of an ETS are less clear than some of the before mentioned implementation challenges. This chapter provides an analytical overview of the types of litigation that ETS are exposed to and the ways in which these different categories of litigation can, and have, affect(ed) ETS design and development.

2. (E)VALUATING LITIGATION IN EMISSIONS TRADING SCHEMES

The state of continuous change in which society, and the legal system regulating it, finds itself requires a degree of flexibility within the law that causes it to be inherently incomplete. Yet, in order for the law to successfully regulate individuals' behaviour, a minimum level of predictability or 'legal certainty' is needed. If too much uncertainty exists, individuals are not able to tailor their expectations and behaviour without significant costs for themselves and/or others.[12] Litigation plays a key role in 'completing' legal rules and diminishing uncertainty,[13] but produces its own social and private costs. Moreover, as with most activities, individuals often fail to internalize the full costs and benefits of litigation. In practice, this divergence between social and private interests in litigation

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[12] For the effects of legal uncertainty on actors within the EU ETS, see G Dari-Mattiacci and J van Zeben, 'Legal and Market Uncertainty in Market-Based Instruments: The Case of the EU ETS' (2012) 19(2) *NYU Environmental Law Journal* 101–39. See also M Peeters and S Weishaar, 'Exploring Uncertainties in the EU ETS: "Learning by Doing" continues beyond 2012' (2009) 1 *Carbon & Climate Law Review* 88–101.

[13] Some scholars argue that legal certainty is caused by gaps in the law, whereas others claim that these gaps do not exist since court decisions will fill any alleged gap, see M Weber, *On Law in Economy and Society* (M Rheinstein ed, Edward Shils trs, Harvard University Press 1954), 31–33; J Carbonnier, *Flexible Droit: Textes pour une Sociologie du Droit sans Rigueur* (6th edn, Paris: Librairie Générale de Droit et de Jurisprudence 1988) and C Perelman (ed.), 'Le problème des lacunes en droit, essai de sinthése' in *Le problème des lacunes en droit* (Bruxelles, Bruylant, Travaux du Centre National de Recherches Logiques, 1968), 537; N Bobbio, *Teoria generale del diritto* (Torino: Giappichelli, 1993), respectively. Dari-Mattiacci *et al* reconcile these two approaches by pointing at the temporal dimension of the problem: ex ante it may be difficult to predict court decisions if a problem has not yet been decided or the law is unclear (a 'gap'), however, ex post, no such uncertainty will exist, see Giuseppe Dari-Mattiacci et al., 'The Dynamics of the Legal System' (2011) 79(1–2) *Journal of Economic Behaviour and Organization*, 95–107.

means that the amount of litigation in any legal system is seldom socially optimal.[14]

Features of the legal regime itself can further augment the likelihood of 'excessive' litigation,[15] particularly the inadequacy of the legal rules (for example, overly complex or obscure rules), and institutional features of the legal system (for example, the availability of legal aid and (lack of) legal standing).[16] The presence or absence of either of these factors can aggravate the natural divergence between social and private incentives for litigation, which in turn makes excessive litigation more likely. Emission trading schemes are contextualized by the regulatory and legal framework of the jurisdiction in which they have developed, which means that the institutional causes of litigation vary depending on the legal system within which the respective ETS operates.[17] Before considering these features in more detail, a further distinction must be made between purpose and consequences of the *presence* of litigation and the *outcome* of litigation.

In terms of social costs and benefits, some argue that the possibility to bring a case has value regardless of the outcome. The notoriously costly adversarial system of the United States is considered intrinsically valuable as the individual's active role in the legal process recognizes the 'autonomy of the individual' and increases the acceptability of judgments.[18] This suggests a signalling function of the *process* of litigation, which is not necessary linked to the judgment and whatever certainty that brings and constitutes a separate social benefit (positive externality). The interplay between outcome and process can highlight potentially problematic legal rules or institutional features of an ETS. However, it can also point at tensions between different interest groups involved in the ETS,[19] rather than any legal or institutional failings.[20] This alternative meaning of litigation encourages caution when interpreting the presence of persistent litigation,

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[14] See S Shavell, 'The Fundamental Divergence between the Private and the Social Motive to Use the Legal System', (1997) 26 (S2) *The Journal of Legal Studies*, 575–612.

[15] 'Excessive' is defined here as more than socially optimal.

[16] S Shavell (n 14), 577.

[17] Section 3 will discuss these features, where relevant, with respect to specific ETSs.

[18] SR Gross, 'The American Advantage: The Value of Inefficient Litigation', (1987) 85(4) *Michigan Law Review* 734–57, at 744–7.

[19] See generally Stephan and Lane (n 2).

[20] See Section 3.B on the persistent litigation of industry groups before the European Courts regarding the EU ETS despite their confirmed lack of standing.

as it may not be a sign of imperfect implementation but of interest-based protest against regulation more generally.[21]

Depending on the type of challenge brought by the parties, the *outcome* of litigation can also constitute an external shock to ETS implementation. One example is the negative influence on the ETS market (price) when regulators are forced to release additional emission allowances into the market or to specific actors.[22] Similarly, the designation of sectors as (in)eligible for grandfathering and the application of national or regional ETS extraterritorially can have serious consequences for the functioning and viability of an ETS as it moves from design to implementation.[23] Litigation is thus an endogenous choice within ETS design as well a potential exogenous shock to an existing ETS.

3. ETS LITIGATION AND IMPLEMENTATION: OBSTACLE OR ACCELERATOR?

The diversity in design and institutional context makes it difficult to identify generic characteristics of 'ETS implementation'[24] and the challenges that litigation poses for this process. In broad terms, a distinction can be drawn between the setting of emission reduction goals at the national, federal or regional level and the distribution of these goals between individual installations and/or industries in the form of emission allowances or permits.[25] The latter may be seen as the implementation of the former and the most salient implementation feature of ETS systems generally. However, in order to establish a fully functioning ETS, far more is needed

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[21] This political interpretation of the role of litigation in ETS development and implementation has been recognized by the literature but is not yet fully understood. See for instance, J Pinske, 'Corporate Intentions to Participate in Emission Trading', (2007) 16(1) *Business Strategy and the Environment*, 12–25; and J Pinske and A Kolk, 'Multinational Corporations and Emissions Trading: Strategic Responses to New Institutional Constraints' (2007) 25(6) *European Management Journal* 441–452.

[22] See also G Dari-Mattiacci and J van Zeben (n. 12). On the economic causes of price volatility and possible solutions, see E Woerdman and A Nentjes, 'Emissions Trading Hybrids: The Case of the EU ETS' (2014) University of Groningen, Faculty of Law, Working Paper Series in Law and Economics.

[23] See e.g. Case C-366/10 *Air Transport Association of America and Others v Secretary of State for Energy and Climate Change* ECLI:EU:C:2011:864.

[24] See also J van Zeben, 'Subsidiarity in European Environmental Law: A Competence Allocation Approach', (2014) 38(2) *Harvard Environmental Law Review* 415–64.

[25] Ibid.

than the distribution of allowances; a market needs to be established and regulated, monitoring and verification of emissions put in place, and penalties for non-compliance need to be imposed. The diverse nature of implementation is reflected by ETS litigation, which speaks to most of these issues. In addition, a separate body of litigation has developed that aims to prevent the development of an ETS and questions its suitability and legitimacy as a regulatory instrument. In order to ascertain the actual effects of litigation on ETS implementation, a close inspection of the existing ETS systems is needed.

With the exception of the United States' Acid Rain Program, most existing and envisaged emissions trading schemes have been developed as a regulatory response to global climate change and international efforts to combat its causes, particularly greenhouse gas emissions.[26] Of these schemes, the EU ETS is by far the most established and has become a lightning rod for academic, and policy, research on emission trading. Any discussion regarding ETS litigation must look beyond this, admittedly very influential, system in order to incorporate the various legal cultures in which ETSs have been developed.[27] Even as the United States has foregone its leadership position within international environmental and climate regulation, American (in)activity since the Acid Rain Program continues to be of vital interest in this field. In light of federal reluctance to take action on, inter alia, climate change related emissions such as CO_2, litigation by states and private parties has become particularly important as a catalyst for regulatory action (section 3.1).[28] This reluctance to act is in stark contrast with the European Union's course of action, which has been committed to ambitious (often unilateral) climate goals since the late 1990s (section 3.2).[29] A third category of actors involved in ETS-based litigation includes other established

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[26] United Nations Framework Convention on Climate Change (adopted on 9 May 1992, entered into force on 21 March 1994) 1771 UNTS 107 (UNFCCC); Kyoto Protocol to the United Nations Framework Convention on Climate Change (adopted on 11 December 1997, entered into force on 16 February 2005) 2303 UNTS 162 (KPUNFCCC).

[27] On the legal culture of the EU ETS, see e.g. S Bogojević, 'EU Climate Change Litigation, the Role of the European Courts and the Importance of Legal Culture' (2013) 35(3) *Law & Policy* 184–207.

[28] See generally, J Peel and H M Osofsky, 'Sue to Adapt?' (2015) *Minnesota Law Review* (forthcoming).

[29] Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions: A Roadmap for moving to a competitive low carbon economy in 2050 COM (2011) 0112 final.

greenhouse gas contributors, including Australia, Canada, and New Zealand, as well as 'developing economies' such as China that are starting to consider ETS as a potential tool to combat national environmental problems – rather than a path to fulfilling international obligations (section 3.3). Regardless of the state of development of these ETS, litigation has evolved regarding each of them.

3.1 United States: From First to Last in Class

Regulatory context

The United States Acid Rain program (ARP), as contained within Title IV of the 1990 Clear Air Act Amendments (CAA),[30] was the 'first large-scale, long term U.S. environmental program to rely on tradable emissions permits'.[31] The federal program addressed SO_2 emissions, a precursor of acid rain, specifically from coal fired electric utility plants. The CAA amendments were partly based on the US Emissions Trading Program that had been piloted in the 1980s,[32] and politically, and economically, viable due to the 1975 growth ban.[33] The Acid Rain program was considered one of the most successful environmental regulatory strategies of its kind, achieving emission reductions (and near perfect compliance)[34] at very low costs.[35] At the federal level, the ARP's success has led to programs such as the NO_x Budget Program.[36] The trading element of the CAA has been replaced by four separate trading schemes under the Clean Air Interstate Rule (CAIR) in 2011. The 'Cross-State Air Pollution Rule' (CSAPR) replaced CAIR's SO2 and NOx program on 1 January 2015.[37] CAIR

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[30] The Clean Air Act Amendments 1990, Public Law 101–549.

[31] Ellerman *et al* (n. 3), 3.

[32] See A Simons and J-P Voss, Politics By Other Means: The Making of the Emissions Trading Instrument as a 'Pre-history'of Carbon Trading', in Stephan and Lane (n. 2), 58–9.

[33] See R Lane, 'Resources for the Future, Resources for Growth: The Making of the 1975 Growth Ban' in Stephan and Lane (n 2).

[34] Phase I of the program (1995–1997) experiences perfect compliance of all affected installations and no exemptions, exceptions or waivers. However, perfect compliance does not necessarily indicate sufficient reductions, see Ellerman et al. (n. 3), 109 onwards.

[35] These low cost reductions have since been attributed to external economic factors such as lower than expected abatement costs and low rail hire fees, see Lane (n. 33), 61. See also Ellerman et al. (n. 3).

[36] Information on which is available via https://www.epa.gov/airmarkets/nox-budget-trading-program (NO_x) (accessed 26 July 2016).

[37] Information on which is available via https://www3.epa.gov/crossstaterule/ (CAIR) (accessed on 26 July 2016).

combines the regulation of SO_2 and NO_x emissions and the achievement of ambient air standards.[38]

Since the ratification of the United Nations Framework Convention on Climate Change (UNFCCC),[39] and the adoption of the Kyoto Protocol (KP),[40] these initial trailblazing successes for emissions trading in the United States have been heavily discounted by the United States failure to implement comparable measures to regulate the emission of CO_2 and other greenhouse gasses. Despite being a signatory to the UNFCCC, and historically the largest emitter of greenhouse gasses, the United States has not ratified the Kyoto Protocol. In addition, there has been no federal legislation establishing emission reductions from stationary sources,[41] or other significant emitters.[42] This inaction has given rise to numerous state-based initiatives for climate change mitigation,[43] including several regional ETS, for example, the Regional Greenhouse Gas Initiative (RGGI),[44] the Western Climate Initiative (WCI),[45] and the California Emissions Trading System (CA ETS).[46] These regional schemes operate on a voluntary basis and are relatively young (for example, the first auction under the CA ETS took place in November 2012.

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38 This change has met with considerable objections, many of which fought through the courts, see Section 3.A. *Litigation*, below.

39 United Nations Framework Convention on Climate Change (n. 26).

40 Kyoto Protocol to the United Nations Framework Convention on Climate Change (n. 26).

41 The Environmental Protection Agency (EPA) has issued a Tailoring Rule that implements a permitting program that subjects the largest stationary emitters of greenhouse gases to permitting requirements under the Prevention of Significant Deterioration (PSD) and Title V sections of the Clean Air Act. Action to Ensure Authority to Implement Title V Permitting Programs Under the Greenhouse Gas Tailoring Rule, 75 Fed. Reg. 82,254 (Dec. 30, 2010) (to be codified at 40 C.F.R. pts. 52 & 70).

42 There has been some progress in terms of the regulation of vehicle emissions, see Light-Duty Vehicle Greenhouse Gas Emission Standards and Corporate Average Fuel Economy Standards, 75 Fed. Reg. 25,324 (7 May 2010) (to be codified at 40 C.F.R. pts. 85, 86 & 600 and 49 C.F.R. pts. 531, 533, 536, 537 & 538).

43 J DeShazo and J Freeman, ‘Timing and Form of Federal Regulation: The Case of Climate Change’ (2007) 155 *University of Pennsylvania Law Review* 1500–61 (on state action as a possible catalyst for federal action), see also K Engel, ‘Mitigating Global Climate Change in the United States: A Regional Approach’ (2005) 14 *NYU Environmental Law Journal* 55. For an overview of the literature and state initiatives other than ETS, see Engel, *infra*, at 57, notes 6 and 7.

44 See http://www.rggi.org/ (last visited 14 July 2016).

45 See http://www.wci-inc.org/ (last visited 14 July 2016).

46 See http://www.arb.ca.gov/cc/capandtrade/capandtrade.htm (last visited 14 July 2016).

Litigation

The Acid Rain Program had managed to align certain regulatory and industry interest, which, together with a phased-in reduction approach, resulted in very few legal challenges.[47] Most challenges that did occur concerned the interpretation of overly complex or unclear statutory provisions, and none of these challenges resulted in implementation delays.[48] Over the past two decades, the development of emission trading schemes has become increasingly linked to climate change mitigation. This shift away from broader environmental issues towards a focus on greenhouse gas emissions has also had an effect on the nature of ETS litigation. Within the United States, a rich body of jurisprudence has developed regarding the (lack of) federal regulation of GHG emissions. *Massachussetts v EPA* represents a turning point in this body of jurisprudence, as the Supreme Court established the Environmental Protection Agency's (EPA) responsibility to regulated carbon dioxide and other greenhouse gasses under the Clean Air Act by categorizing them as 'pollutants'.[49] Later attempts to hold companies responsible for the emission of GHGs under federal common law were unsuccessful, as the Supreme Court confirmed that the responsibility for the regulation of GHGs has now shifted entirely to the EPA.[50] Conversely, litigation has also been used to delay federal action,[51]

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[47] See cf. *Clean Air Markets Group v Pataki*, No. 00-CV-1738 (194 F. Supp. 2d 147) (United States District Court for the Northern District of New York, 9 April 2002) (where the New York Air Pollution Mitigation Law was held to be pre-empted by the Clean Air Act (CAA) and in violation of the U.S. Commerce Clause).

[48] S Napolitano et al, 'The U.S. Acid Rain Program: Key Insights from the Design, Operation, and Assessment of a Cap-and-Trade Program' (2007) 20(7) *The Electricity Journal* 47–58, 50.

[49] 549 U.S. 497 (2007).

[50] *American Electric Power v Connecticut*, 131 S. Ct. 2527 (2011). For a discussion of this case and related litigation, see H Osofsky, 'Litigation's Role in the Path of U.S. Federal Climate Change Regulation: Implications of AEP v Connecticut', (2012) 46 *Valparaiso University Law Review* 447 (stressing the willingness of the Supreme Court to engage with this type of litigation as long as the focus is statutory). See also J Flynn, 'Climate of Confusion: Climate Change Litigation in the Wake of *American Electric Power v Connecticut*', (2013) 29(3) *Georgia State University Law Review* 823 onwards, and J Hessler, 'A Temporary Solution to Climate Change: The Federal Common Law to the Rescue?' (2011) 38 *Hastings Constitutional Law Quarterly* 407.

[51] See *Utility Air Regulatory Group v EPA* 134 S. Ct. 2427 (2014) (holding that specific EPA regulations that were adopted after *Massachusetts v EPA* were partially unlawful due to ultra vires application of EPA's authority to stationary sources emitting less than 250 tons of air pollutant per year).

question the interpretation of federal ETS legislation by states,[52] and the scientific causes of climate change.[53]

Federal litigation surrounding the functioning of emission trading schemes developed since the US Acid Rain Program has been rather limited, with the exception of several cases regarding the Clean Air Interstate Rule (CAIR). CAIR – a direct descendant of the Acid Rain Program – was developed to capture SO_2 and NO_x pollution caused by power plants that was likely to 'drift' from one state to another.[54] CAIR regulated states rather than individual installations. Implementation of the scheme – by requiring power plants to participate in an EPA-administered interstate cap-and-trade system or by meeting an individual state emissions budget through measures of the state's choosing – is in the hands of the states. Since its foundation in March 2005, litigation has been brought as to the methods of state implementation, for instance in *Mirant Potomac Rover LLC v EPA*,[55] where Virginia's decision to designate nonattainment areas under its state implementation plan, within which emissions trading would not be allowed was questioned by affected companies, and upheld by the court.

More fundamental questions regarding the integrity of the program were raised in *North Carolina v EPA*.[56] The DC Circuit initially vacated CAIR in its entirety, pointing at flaws within the system that would have to be remedied by new legislation by EPA.[57] The Court of Appeal later remanded this decision without vacatur, giving the EPA the opportunity to draft new legislation to replace the current rules while continuing to implement the existing system.[58] In August 2011, the EPA issued a replacement

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[52] See *Mirant Potomac Rover LLC v EPA* (4th Cir. 12 August 2009). See also *Coalition for a Safe Environment v California Air Resources Board* (EPA, filed 8 June 2012), where environmental justice advocates alleged that California's economy wide cap-and-trade system violates Title VI of the Civil Rights Act of 1964 because of its (alleged) adverse impact on low-income and minority neighborhoods. The design of the system would not lead to reductions in emissions in (low-income) neighborhoods located close to affected industries and therefore would not lead to any improvements for these groups).

[53] See e.g. N Oreskes and EM Conway, *Merchants of Doubt: How a Handful of Scientists Obscured the Truth on Issues from Tobacco Smoke to Global Warming* (New York: Bloomsbury Press 2010), 5–9; and J Hoggan and R Littlemore, *Climate Cover-Up: The Crusade to Deny Global Warming* (Greystone Books 2009).

[54] See https://archive.epa.gov/airmarkets/programs/cair/web/html/index.html (accessed 26 July 2016).

[55] 12 August 2009, 4th Cir.

[56] 11 July 2008, judgment, D.C. Circuit, No. 05-1244.

[57] 11 July 2008, judgment, D.C. Circuit, No. 05-1244.

[58] 23 December 2008 decision, D.C. Court of Appeal, No. 05-1244.

rule known as CSAPR, which creates an elaborate SO_2 and NO_x emissions trading scheme.[59] In August 2012, the Court of Appeals for the District of Columbia vacated CSAPR citing a lack of competence of the EPA under the Clean Air Act.[60] The EPA successfully appealed this decision with the Supreme Court overturning the judgment in April 2014.[61] The first phase of CSAPR will be implemented in 2015.[62]

Another significant challenge was made against EPA's federal hydrofluorocarbon (HCFC) cap-and-trade program,[63] founded in order to fulfil Montreal Protocol obligations.[64] Two manufacturing companies – Honeywell and DuPont – challenged the transfer of allowances to two competitors since these additional allowances could affect their baseline, as the latter is based on historical usage. This in turn would negatively affect Honeywell and DuPont's market share in allowances. The Court upheld its earlier decision in favour of the EPA's decision to honour the transfers.[65]

In the body of state and regional litigation, California's efforts to mitigate greenhouse gas emissions feature most prominently.[66] Most challenges concern California's authority to implement a (mandatory) cap-and-trade system on the basis of procedural defects, for example, insufficient consideration of alternative forms of regulation,[67] and in terms of broader legislative powers. With respect to the latter, the auctioning of emission allowances has been particularly controversial. In November 2012, the California Chamber of Commerce (CCC) initiated proceedings against the California Air Resources Board (CARB) because it introduced auctions under its cap-and-trade system.[68] Under the Global Warming Solutions

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59 CSAPR was challenged by a coalition of industry, state and utility interests.

60 *EME Homer City Generation v EPA*, 12 August 2012 decision, D.C. Court of Appeal, No. 11-1302.

61 *EPA v EME Homer City Generation*, 29 April 2014 decision U.S. Supreme Court, No. 12–1182.

62 For information on implementation, see http://www.epa.gov/crossstaterule/ (available 14 July 2016).

63 See 42 U.S.C. §§ 7671d(c), 7671e(b).

64 The Montreal Protocol on Substances that Deplete the Ozone Layer (adopted on 16 September 1987, entered into force on 1 January 1989) 1522 UNTS 3.

65 *Honeywell International v EPA* (D.C. Cir. 22 January 2013) affirming *Arkema Inc. v EPA* (618 F.3d 1, 6-9 (D.C. Cir. 2010).

66 See also n. 52 supra.

67 See e.g. *Association of Irritated Residents v California Air Resources Board* (42 ELR 20127 No. A132165 (Cal. App. 1st Dist. 19 June 2012).

68 *California Chamber of Commerce v California Air Resources Board* (Cal. Ct. App.)

Act (AB 32),[69] CARB was tasked to reduce greenhouse gas emissions to 1990 levels by 2020. The CCC posited that AB 32 did not give CARB the statutory authority to raise revenue from auctions within that emission trading scheme. In addition, the CCC argued that the auctions amounted to a tax. The Court ruled that CARB did in fact possess such authority.[70] In February 2014, the CCC launched a (pending)[71] appeal with support of the National Association of Manufacturers (NAM), which was given leave to intervene.[72] The CCC and NAM maintain that the expected revenue of the auctions (expected between $12 and $70 billion) bears no relation to the funds needed to administer the program and as such beyond the reasonable regulatory burden on the affected industries and constitute a tax.[73] Conversely, the state of New Jersey (through the New Jersey Department of Environmental Protection (NJDEP)) was successfully sued for withdrawing from the RGGI without following formal repeal procedures for its RGGI regulations.[74]

Other, voluntary, state-based and regional schemes have also met with opposition in the courts, particularly with respect to their operation rather than their creation. The Southern California Regional Clean Air Incentives Market (RECLAIM) was one of the first regional ETSs and was a mixed success.[75] RECLAIM has delivered several lessons for federal ETS programs, which is also reflected in its litigation; in 2009, two members of Congress requested the court to unseal pleadings in a criminal case concerning alleged fraud related to RECLAIM. The information would aid the design of a federal cap-and-trade system by showing how fraud may be prevented.[76] Fraud was also a source for litigation under the Chicago

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[69] AB 32, 2006.

[70] Ruling available 26 July 2016 at http://lawcenter.nam.org/Results.aspx?P=California%20Chamber%20of%20Commerce.

[71] Notice of appeal. http://www.calchamber.com/GovernmentRelations/Documents/02202014-AB32-Notice-of-Appeal.pdf (available14 July 2016). At time of writing, the case is pending.

[72] *California Chamber of Commerce v California Air Resources Board,* 3rd Appellate District 20 October 2014, Nos. 34-2012-80001313,34-2012-80001464, available 14 July 2016 at vhttp://www.nam.org/Advocacy/The-Center-for-Legal-Action/Briefs-Online/2014/NAM-Opening-Brief-in-California-Chamber-of-Commerce-v-California-Air-Resources-Board-(Cal-Ct-App)/.

[73] Ibid.

[74] In *re Regional Greenhouse Gas Initiative* (RGGI), No. A-4878- 11T4 (N.J. Super. Ct. App. Div. 25 March 2014).

[75] See for an evaluation http://www.epa.gov/Region09/air/reclaim/index.html (available 14 July 2016).

[76] *United States v Sholtz* (C.D. Cal. December 15, 2009)

Climate Futures Exchange.[77] In this case, the allegations were market based rather than ETS specific – a reminder as to the intrinsically financial nature of the emissions market despite its regulatory foundation.

3.2 The EU ETS: Not all that Glitters is Green

Regulatory background

After a very condensed legislative period, trading under the European Union Emissions Trading Scheme (EU ETS) started in 2005.[78] The EU ETS is a regional market, comprised of all the European Union Member States and, since 2008, the EEA-EFTA states, Iceland, Liechtenstein and Norway. During the first two trading phases (Phase I running between 2005 and 2007, Phase II between 2008 and 2012), the EU ETS exclusively covered CO_2 emissions from installations performing specific activities, including energy production, oil refinery, iron and steel manufacturing and cement, glass, lime, bricks, paper and board production.[79] From 2013 – the beginning of the third trading phase – additional greenhouse gasses and industries are included.[80] The capture, transport and storage of greenhouse

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[77] *Barnett v Chicago Climate Futures Exchange*, LLC (Cook Co. Dist. Ct, filed 16 December 2011).

[78] Commission Communication 'Climate Change – Towards an EU post-Kyoto Strategy' COM (98) 353 final; Commission Communication, 'Energy for the Future: Renewable Sources of Energy. White Paper for a Community Strategy and Action Plan' COM (97) 599 final; Commission Green Paper, 'A European Strategy for Sustainable, Competitive and Secure Energy' SEC (2006) 317; Proposal for a Directive of the European Parliament and of the Council establishing a scheme for greenhouse gas emission allowance trading within the Community and amending Council Directive 96/61/EC COM (2001) 581 final; Directive (EC) 2003/87 of the European Parliament and of the Council of 13 October 2003 establishing a scheme for greenhouse gas emission allowance trading within the Community and amending Council Directive 96/61/EC [2003] OJ L 275. See also: European Commission, *EU Action against Climate Change: EU Emission Trading Scheme-An Open Scheme Promoting Global Innovation, The EU Brochure* (September 2005), found at http://ec.europa.eu/environment/climat/pdf/emission_trading3_en.pdf (available 16 July 2016).

[79] The Netherlands ran a national NO_x ETS alongside the EU ETS between 1 January 2005 until 1 January 2014.

[80] CO_2 emissions from petrochemicals, ammonia and aluminium will be covered, as well as N_2O emissions from nitric, adipic and glyocalic acid production, see Directive (EC) 2009/29 of the European Parliament and of the Council of 23 April 2009 amending Directive 2003/87/EC so as to improve and extend the greenhouse gas emission allowance trading scheme of the Community [2009] OJ L 140/63, Annex I.

gas emissions will now also come under the EU ETS, as well as aviation emissions from airplanes arriving in or departing from the EU.[81]

The EU ETS is the largest ETS currently in operation, both in terms of trade volume and installations covered.[82] Its development and implementation have not been without criticism.[83] The (lack of) cap stringency and related price fluctuations have been an enduring source of dissatisfaction,[84] while environmental advocates continue to question the appropriateness of emissions trading as a regulatory tool to mitigate the causes of climate

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[81] Directive (EC) 2008/101 of the European Parliament and of the Council of 19 November 2008 amending Directive 2003/87/EC so as to include aviation activities in the scheme for greenhouse gas emission allowance trading within the Community [2009] OJ L8/3. NB: the application of the EU ETS to aviation has been temporarily suspended, see also n. 115 below.

[82] By means of comparison, the global market share of the Regional Greenhouse Gas Initiative (RGGI) in the United States dropped from 9% to less than 1% in 2010 due to the lack of prospects for a federal cap-and-trade scheme in the United States. The projected value of the world's carbon markets in 2020 is €1.7 trillion, provided that other markets are implemented by e.g. Japan and Australia.

[83] For general writings on its development, see e.g. M Faure and M Peeters (eds), *Climate Change and European Emissions Trading: Lessons for Theory and Practice* (Edward Elgar, 2008); S Bogojević (n. 2); S Öberthur and M Pallemarts (eds), *The New Climate Policies of the European Union: Internal Legislation and Climate Diplomacy* (VUB Press, 2010), specifically chapter 2; J-B Skjærseth and J Wettestad, 'The EU Emissions Trading System Revised (Directive 2009/29/EC)', 65–93; J van Zeben (n. 10); A Epiney, 'Climate Protection in the European Union – Emergence of a New Regulatory System' (2012) 9(1) *Journal of European Environmental and Planning Law* 5–33; A Vlachou, 'The European Union's Emissions Trading Scheme' [2014] 28 *Cambridge Journal of Economics*, 127–52; B Perez de las Heras, 'Beyond Kyoto: The EU's Contribution to a More Sustainable World Economy' (2013) 19(4) *European Law Journal*, 577–93.

[84] See inter alia, AD Ellerman and B Buchner, 'Over-allocation or Abatement? A Preliminary Analysis of the EU Emissions Trading Scheme based on the 2006 Emissions Data' (2008) 41 *Environmental and Resource Economics* 457; AD Ellerman and B Buchner, 'The European Union Emissions Trading Scheme: Origins, Allocation, and Early Results' (2007) 1 *Review of Environmental Economics and Policy* 66; S Weishaar and E Woerdman, 'Auctioning EU ETS Allowances: An Assessment of Market Manipulation from the Perspective of Law and Economics' (2012) 3 *Climate Law* 247–63; E Woerdman, A Arcuri and S Cló, 'Emissions Trading and the Polluter-Pays Principle: Do Polluters Pay under Grandfathering?' (2008) 25 *International Review of Law and Economics* 565; JB Skjærseth and J Wettestad 'Implementing EU emissions trading: success or failure?' (2008) 8 *International Environmental Agreements* 275–90; D Perez Rodriguez, 'Absorbing EU ETS Windfall Profits and the Principle of Free Allowances: *Iberdrola and others*' (2014) 51 *Common Market Law Review* 679–96; D Böhler, 'The EU Emissions Trading Scheme – Fixing a Broken Promise' (2013) 15 *Environmental Law Review*, 95–103.

change.[85] The environmental results of the EU ETS have been disappointing compared to the EU's ambitions, especially as it is difficult to ascertain the effects of the EU ETS on investment as compared to the effects of, for example, the economic crisis. Nevertheless, the EU's commitment to the EU ETS remains and the EU ETS continues to be the cornerstone of its internal and external climate change agenda.[86]

In order to contextualize the EU ETS litigation, the key features of its design must be briefly set out. During the first two trading phases (2005–2012), the EU ETS was a 'decentralized' system.[87] Member States determined the distribution of emission allowances between industries and to specific installations and summarized this distribution in 'National Allocation Plans' (NAPs).[88] These NAPs were subject to approval by the Commission,[89] and if approved, were implemented through installation-specific national allocation decisions.[90] The EU ETS' third trading phase is marked by increased centralization of allowance distribution through the abolition of the NAPs and the introduction of a central cap.[91] There have been several changes to monitoring and enforcement practices in the EU ETS as well, some of which have also been centralized, but most of which remain in the hands of the Member States and private verifiers.[92]

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[85] See e.g. BD Solomon and R Lee, 'Emissions Trading Systems and Environmental Justice' (2000) 42(8) *Environmental Justice* 32–45; LN Chinn, 'Can the Market be Fair and Efficient? An Environmental Justice Critique of Emissions Trading' (1999) 26(1) *Ecology Law Quarterly* 80–125.

[86] See Communication 'A Policy Framework for Climate and Energy in the Period from 2020 and 2030', COM (2014) 15 final, and Commission 'A Roadmap for moving to a competitive low carbon economy in 2050' COM (2011) 112 final.

[87] At the same time, the Member States had agreed to achieve their Kyoto commitments collectively, UNFCCC (n. 26), Article 4.

[88] Directive 2003/87/EC (n 78), Article 9.

[89] Ibid, Annex II.

[90] Directive 2003/87/EC (n 78), Article 11.

[91] Directive 2009/29/EC (n 80), Article 9.

[92] Monitoring and enforcement is subject to a complex distribution of competences between the EU and the Member States. As neither has been subject of significant litigation before the European courts, an in-depth discussion of these elements will be foregone. The exception is a case brought in Wales – Alphasteel (ENDS 2008, 401, 64–65) where the imposed civil penalties were challenged as being criminal rather than civil in nature. For additional insights, see J. van Zeben (n. 10), 159–62; see also P Mendes de Leon, 'Enforcement of the EU ETS: the EU's Convulsive Efforts to Export its Environmental Values' (2012) 37 *Air and Space Law*, 287–306 (for a discussion of enforcement of airline emissions under Directive 2008/101/EC).

Litigation

In parallel to the legal and political forces that continue to shape the EU ETS, litigation has been a prominent feature in its development. Many cases were brought before the European Courts;[93] to date, there have been over 70 decisions by the European Courts regarding Directives 2003/87/EC, 2009/29/EC and 2008/101/EC.[94] More than half of these cases were brought during the first four years of the EU ETS' existence.[95] There have been few challenges to the creation of the EU ETS, none of which were successful.[96] Most European cases concern the implementation of the trading scheme through the NAPs in Phase I and II of the EU ETS. Since the start of the third trading phase, the relevance of these cases is limited, as the National Allocation Plans have been abolished. At the same time, these changes show the importance of this body of case law in terms of the effect that litigation can have on implementation.[97] There were several

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[93] In addition, a small body of jurisprudence has developed within the Member States themselves. See e.g. for The Netherlands: Raad van State, 11-06-2014, 201311081/1/A4; Raad van State, 19-05-2009, 200809472/1/M1 (both concerning the NAP of the Netherlands); United Kingdom: *INEOS Manufacturing Scotland Limited v Grangemouth CHP Limited & FORTUM O&M (UK) Limited* [2011] EWHC 163 (Comm), and *Armstrong v Winnington Networks Ltd* [2012] EWHC 10 (Ch) (concerning the transfer of emission allowances between companies); Ireland: *Viridian Power Limited and Huntstown Power Company Limited v Commission for Energy Regulation and the Attorney General*, [2012] IESC 13 (on windfall profits).

[94] These numbers are up-to-date as to the 15 January 2015 and include judgments and orders of both the CFI and the CJEU as may be found in the CURIA database (http://www.curia.eu) when using 'Directive 2003/87/EC', 'Directive 2009/29/EC' (combined 71 cases), 'Directive 2008/101' (two cases) as search keys. In addition there are 14 pending cases concerning Directives 2003/87/EC and 2009/29/EC.

[95] See J van Zeben, 'The European Emissions Trading Scheme Case Law' (2009) 18 *Review of European Community & International Environmental Law* 119.

[96] This is true both for the European courts and the national courts. Before the European courts, see Case C-127/07, *Société Arcelor Atlantique et Lorraine and Others v Premier Ministre, Ministre de l'Économie, des Finances et de l'Industrie, Ministre de l'Écologie et du Développement durable* ECLI:EU:C:2008:728 *('Arcelor Atlantique and Others')*. In this case, Arcelor challenged the legality of the EU ETS on the basis that the Directive was discriminatory since some industries had been included whereas others had not. The CJEU held that while differential treatment had been applied, the Commission's step-by-step approach was warranted due to the complexity of the system, *infra*, para. 49. See also M Peeters, 'The EU ETS and the Role of the Courts: Emerging Contours in the Case of Arcelor' (2011) 2 *Climate Law*, 19–36. For the national courts, see Belgium: N. 2012 – 2420 [2012/203555] Uittreksel uit arrest nr. 76/2012 van 14 juni 2012 (on the legality of the creation of a 'Walloon Kyoto Fund' as part of the implementation of Directive 2003/87/EC).

[97] See Section 3 in more detail.

shared themes among the 'Member State challenges to Commission decisions on NAPs' cases: A first set of cases questioned Commission decisions regarding the national process by which the respective National Allocation Plans had come into being,[98] and regarding certain design choices within the NAPs, such as the inclusion of a new entrants reserve.[99]

The second, more substantial, number of cases concerned challenges to Commission decisions, which had rejected NAPs on the basis of suspected over-allocation of allowances to ETS industries.[100] Both the Court of First Instance (CFI) and the Court of Justice of the European Union (CJEU) persistently held that the Commission's powers to reject an NAP were confined to the grounds listed in Annex III of the Directive. In the

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[98] Case T-178/05, *United Kingdom v European Commission* ECLI:EU:T:2005:412; here the CFI had to decide whether the Commission was entitled to reject amendments to a National Allocation Plan, if these amendments had not previously been included in the provisional NAP that was submitted by a Member State earlier. The Court examined the roles and powers of the Commission and the Member States under the Directive, and found that with respect to amendments to NAPs, the Commission cannot restrict a Member States' right to propose amendments but that any proposed amendment must be adopted by the Commission in order to become effective.

[99] *Germany v Commission* (Case T-374/04, *Germany v European Commission* ECLI:EU:T:2007:332), in which the Commission questioned Germany's decision to include an ex post adjustment mechanism in its NAP, which would allow the German government to take back allowances from installations under five different scenarios and to place them in the new entrants reserve. According to the Court of First Instance ('CFI', now General Court), the Commission did not prove that the German ex post adjustment mechanism was incompatible with criteria 5 and 10 of Annex III to Directive 2003/87/EC. Specifically, the CFI held that the arguments of the Commission were neither 'factually substantiated nor legally well founded'. See Case T-374/04, *Germany v European Commission* ECLI:EU:T:2007:332, at paras 151–64. See also S Weishaar, 'Germany v Commission: The CFI on ex-post adjustments under the EU ETS' (2008) 17(1) *Review of European Community & International Environmental Law*, 126–9 and S Weishaar, 'Ex-Post Korrektur im Europaeischen CO2-Emissionshandel: Auswirkungen der Rechtsprechung fuer Deutschland' (2008) 3 *Zeitschrift fuer Europaisches Umwelt-und Planungsrecht*, 148–51. The mere fact that 'the practice of ex-post adjustments are liable to deter operators from reducing their production volume and, therefore, their emission rates is not sufficient to call into question the adjustments' legality in light of the directive's objectives as a whole' (Case T-374/04, *Germany v European Commission* ECLI:EU:T:2007:332, at para. 148). For a national challenge regarding the use of the new entrants reserve, see *Rechtbank Rotterdam*, 02-11-2010, AWB 10/3978 VWOB T2.

[100] See inter alia Case T-499/07, *Bulgaria v Commission* [2008] OJ C 246/50; Case T-500/07, *Bulgaria v Commission* [2008] OJ C 209/51; Case T-483/07, *Romania v Commission* [2008] OJ C 273/22; Case T-484/07, *Romania v Commission* [2008] OJ C 258/27; Case T-369/07, *Latvia v European Commission* ECLI:EU:T:2011:103.

landmark cases *Poland v Commission*[101] and *Estonia v Commission*,[102] the CFI confirmed that the Commission acted ultra vires by replacing data and calculations used by the Member States with their own.[103]

The contested allowances amounted to 26.7 and 47.8 per cent of the total allowances of Poland and Estonia's respective NAPs. After the CFI's annulments of the Commission decisions, revised NAPs were submitted by both Member States, which were rejected again, on different legal grounds, revised once more and ultimately accepted. The approved NAPs were not put in place until April 2010, two years after the start of the second trading phase. The CJEU dismissed the Commission's appeal, which ran in parallel to these renegotiations, emphasizing that the method of allowance calculation was part of the discretion of the Member States, and that this need not be harmonized by the Commission in order to ensure equal treatment between the Member States.[104] In short, the Member States have successfully resisted attempts by the Commission to indirectly tighten the caps for their national ETS sectors during the first two trading phases. However, this has come at the costs of increased uncertainty regarding the number of allowances on the market and thus a more unstable carbon price.[105]

Companies and industry groups also brought numerous cases before the European Courts regarding allocations under National Allocation Plans but have been unsuccessful due to their lack of standing. Since installations depend on the national allocation decision taken by the Member States once the NAP has been approved, and not the actual NAP which impacts on the companies rights, these companies are not considered 'individually

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[101] Case T-183/07, *Poland v European Commission* ECLI:EU:T:2009:350.

[102] Case T-263/07, *Estonia v European Commission* ECLI:EU:T:2009:351.

[103] Case T-183/07, *Poland v European Commission* (101) §163, and Case T-263/07, *Estonia v European Commission* (n. 102) §114.

[104] Cases lodged under number Case C-504/09P, *European Commission v Poland/European Commission v Estonia* [2011] OJ (C.370) and C-505/09P (2009); Judgment (Poland), §65–66; Case T-183/07, *Poland v European Commission* (101) §67, and Case T-263/07, *Estonia v European Commission* (n 102) §68. See further J van Zeben, 'Cases C-504/09 P, Commission v Poland, and C-505/09 P, Commission v Estonia, Judgment of the European Court of Justice (Second Chamber) of 29 March 2012 (annotation)' (2013) 50 *Common Market Law Review* 231–46.

[105] See G Dari-Mattiacci and J van Zeben (n. 12). The Commission also recognized that the legal and political disagreement regarding NAPs created elements of uncertainty and lack of predictability negatively affecting the market price and its stability, see Impact assessment attached to the Proposal for a Directive of the European Parliament and of the Council amending Directive 2003/87/EC so as to improve and extend the greenhouse gas emission allowance trading system of the Community, COM(2008) 16 final, 91.

concerned' with respect to the Commission Decisions.[106] Despite the fact that these cases were invariably, and expectedly, dismissed due to lack of standing of the applicants, applications continued. One explanation for the continued stream of cases may be their signalling function as a sign of persistent protest of industries to the introduction of the EU ETS.[107]

The most recent challenge to the EU ETS has been an 'external' one.[108] The inclusion of aviation into the EU ETS, specifically the emissions from *all* airplanes landing in or departing from a European airport,[109] including those from non-EU airlines, has met with significant resistance from third-party countries. In part, this resistance has taken the form of a diplomatic row between the EU and some of its main trading partners.[110] In addition, a legal battle was brought before the European courts by a coalition of the Air Transport Association of America, American Airlines,

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[106] See for instance Case T-387/04, *EnBW Energie Baden-Württemberg v Commission* ECLI:EU:T:2007:117 and Case T-27/07, *U.S. Steel Kosice v European Commission* ECLI:EU:T:2007:302; CFI 25 June 2007, Case T-130/06 *Drax Power and others v Commission*, [ECLI:EU:T:2007:188.

[107] This political interpretation of the role of litigation in ETS development and implementation has been recognized by the literature but is not yet fully understood. See for instance, J Pinske, 'Corporate intentions to participate in emission trading', (2007) 16(1) *Business Strategy and the Environment* 12–25 and J Pinkse and A Kolk, 'Multinational Corporations and Emissions Trading: Strategic Responses to New Institutional Constraints' (2007) 25(6) *European Management Journal* 441–52.

[108] For a broader discussion of the EU's efforts to engage with and shape international environmental law: J Scott, 'Extraterritoriality and Territorial Extension of EU Law' (2014) 62 *American Journal of Comparative Law* 87; J Scott and L Rajamani, 'EU Climate Change Unilateralism' (2012) 23 *European Journal of International Law* 469; E Morgera and K Kulovesi, 'The Role of the EU in Promoting International Standards in the Area of Climate Change' in I Govaere and S Poli (eds), *EU Management of Global Emergencies: Legal Framework for Combating Threats and Crises* (Brill 2014).

[109] Directive (EC) 2008/101 of the European Parliament and of the Council of 19 November 2008 amending Directive 2003/87/EC so as to include aviation activities in the scheme for greenhouse gas emission allowance trading within the Community [2008] OJ L 8/3. See also A Epiney, 'Climate Protection Law in the European Union – Emergence of a New Regulatory System' [2012] 9 *Journal of European Environmental and Planning Law* 5–33.

[110] See J Hartmann, 'A Battle for the Skies: Applying the European Emissions Trading System to International Aviation' [2013] 82 *Nordic Journal of International Law* 187–220; A Lykotrafiti, 'EU Innovation Policy: Lessons Learned from the Inclusion of Aviation in the EU Emissions Trading Scheme' [2013] 20 *Legal Issues of Economic Integration* 339–362; M Staniland, 'Regulating Aircraft Emissions: Leadership and Market Power' [2012] 19 *Journal of European Public Policy* 1006–25.

Continental Airlines, and United Airlines, who challenged the implementation of Directive 2008/101 by the United Kingdom.[111] The Court validated the inclusion of non-EU airlines in the EU ETS, stating that EU was not bound by the Chicago Convention on Civil Aviation,[112] and had the competence under customary international law to adopt the provisions of Directive 2008/101/EC. Finally, the Court also held that Directive 2008/101/EC does not conflict with the bilateral 'Open Sky' Agreement between the EU and the US (the home state of the affected airlines that brought the case).[113] The CJEU's judgment has been widely criticized,[114] and the application of the EU ETS to non-ETS airlines has been temporarily suspended until a policy solution is reached.[115]

The reform of the EU ETS for the third trading phase has done away with the litigation-sensitive NAPs but other elements are likely to create new issues for the courts to decide. The gradual shift to auctioning has already raised questions regarding state-aid and continued windfall profits,[116] and the Commission's plan to 'backload' some of

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[111] Case C-366/10, *Air Transport Association of America and Others v Secretary of State for Energy and Climate Change* ECLI:EU:C:2011:864.

[112] Chicago Convention on Civil Aviation (adopted on 7 December 1944, entered into force on 4 April 1947) Doc 7300/9.

[113] See also B Mayer, 'Case C-366/10, *Air Transport Association of America and Others v Secretary of State for Energy and Climate Change*, Judgment of the Court of Justice (Grand Chamber) of 21 December 2011', (2012) 49 *Common Market Law Review* 1113–40, and S Bogojević, 'Legalising Environmental Leadership: A Comment on the CJEU's Ruling in C-366/10 on the Inclusion of Aviation in the EU Emissions Trading Scheme', (2012) 24 *Journal of Environmental Law* 345–56.

[114] See B Havel and J Mulligan, 'The Triumph of Politics: Reflections on the Judgment of the Court of Justice of the European Union Validating the Inclusion of Non-EU Airlines in the Emissions Trading Scheme', (2012) 37 *Air and Space Law* 3–33; G De Baere and C Ryngaert, 'The ECJ's Judgment in Air Transport Association of America and the International Legal Context of the EU's Climate Change Policy', (2013) 18 *European Foreign Affairs Review* 389–410.

[115] Decision No 377/2013/EU of the European Parliament and of the Council of 24 April 2013 derogating temporarily from Directive 2003/87/EC establishing a scheme for greenhouse gas emission allowance trading within the Community, [2013] OJ L 113/1, available at http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=uriserv:OJ.L_.2013.113.01.0001.01.ENG. See also P Manzini and A Masutti, 'The Application of the EU ETS System to the Aviation Sector: From Legal Disputes to International Retaliations?' (2012) 37 *Air and Space Law*, 307–24.

[116] See E Gawel and S Strunz, 'State Aid Disputes on Germany's Support for Renewables', [2014] 11 *Journal of European Environmental and Planning Law* 137–50 and S Weishaar and E Woerdman, 'Does Auctioning Emission Rights Avoid State Aid? Empirical Evidence from Germany', [2014] 2 *Carbon and Climate Law Review* (2012) 114–27.

the emission allowances in order to create more scarcity in the market has been controversial.[117] Reconciling the presence of, seemingly unavoidable, litigation with the continuity that is needed to create a more stable market price will be one of the challenges over the next trading phases.

3.3 The Next Generation of ETS

The categorization of countries such as China, Brazil and India as developing economies and their role regarding climate change mitigation has been increasingly challenged.[118] However, for the purposes of international environmental law, and particularly the UN Framework Convention on Climate Change, they continue to be 'non-Annex I' countries.[119] As such, they are subject to a different set of climate change responsibilities than Annex I countries.[120] Rather than being subject to mitigation goals, the non-Annex I countries are on the receiving end of technology transfers and adaptation aid to help them cope with the effects of climate change. The emphasis on 'historic emissions' and the desire for economic progress makes it difficult to envisage an international consensus on the status of these countries within the context of the UNFCCC and related KP commitments in the near future. Other countries, such as Australia and Canada,[121] have not made meaningful progress towards their international mitigation obligations under the UNFCCC and KP. However, over the past few years, both groups of countries have started to consider emission trading as a path towards emission reduction, often under the pressure of their national electorate.

Australia started its ETS on 1 July 2014 with a view to 'meet its international emissions reduction targets under the second commitment period

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[117] J van Zeben (n. 10), 215–18.

[118] See e.g. http://www.bbc.co.uk/news/science-environment-29239194 (available 14 July 2016).

[119] Non-Annex I parties Parties are mostly developing countries. Certain groups of developing countries are recognized by the Convention as being especially vulnerable to the adverse impacts of climate change.

[120] Annex I Parties include the industrialized countries that were members of the OECD (Organisation for Economic Co-operation and Development) in 1992, plus countries with economies in transition (the EIT Parties), including the Russian Federation, the Baltic States, and several Central and Eastern European States.

[121] Canada has withdrawn from the Kyoto Protocol in December 2011; a decision that was litigated in front of the Canadian federal court in *Turp v Canada (Attorney General) Federal Court of Canada* [2012] FC 893, T-110-12.

of the Kyoto Protocol (2013 to 2020)'.[122] However, after a lengthy political battle, the Australian Senate scrapped the plan on 17 July 2014.[123] This has been a substantial blow to efforts to price carbon emissions in Australia. Thus far, litigation in the area has primarily focussed on climate change related issues.[124] One of the few successful challenges to CO_2 emissions regulation has been the case of *Gray v Macquarie Generation*.[125] In this case, the environmental impact assessment and consequently awarded permit regarding mining activity related to the Anvil Hill coalmine were set aside as the Court agreed that the foreseen CO_2 emission amounted to the 'disposal of waste' which was not incorporated into the permit.[126]

In contrast, the *New Zealand* ETS has been in continuous operation since 2008.[127] The system is linked to other UNFCCC related ETS and covers little over half of its emissions. In 2015, agriculture will be added to the scheme, which accounts for the remaining 47 per cent of emissions.[128] *Canada* has yet to complete its emissions trading agenda, despite the fact that legislative steps have been developing since 2006.[129] Canada's failure to act despite its obligations under the Kyoto Protocol has been the subject of (unsuccessful) litigation in front of the Canadian federal courts.[130] *China* plans to launch its national ETS in 2016, which would immediately become the world's largest trading scheme.[131] Reports on ETS-related litigation are

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[122] Australian Government, Starting Emissions trading on 1 July 2014, Policy Summary, July 2013.

[123] http://www.reuters.com/article/2014/07/18/us-australia-carbon-vote-idUSKBN0FM04J20140718 (available 14 July 2016).

[124] See BJ Preston, 'The Influence of Climate Change Litigation on Governments and the Private Sector' (2011) 4(2) *Climate Law* 485–513.

[125] See generally on Australia's role in litigation: H Osofosky and J Peel, 'The Role of Litigation in Multilevel Climate Change Governance: Possibilities for a Lower Carbon Future?' (2013) 30 *Environmental and Planning Law Journal* 303.

[126] [2010] NSWLEC 34.

[127] See http://www.climatechange.govt.nz/emissions-trading-scheme/about/basics.html (available 14 July 2016).

[128] See The Climate Change Response (Moderated Emissions Trading) Amendment Act 2009 and Climate Change Response (Emissions Trading and Other Matters) Amendment Act 2012.

[129] For step-by-step overview, see http://www.aph.gov.au/About_Parliament/Parliamentary_Departments/Parliamentary_Library/Browse_by_Topic/ClimateChangeold/governance/foreign/canadian (available 14 July 2016).

[130] Friends of the Earth v Governor in Council et al., [2009] 3 F.C.R. 201.

[131] http://www.scientificamerican.com/article/china-will-start-the-world-s-largest-carbon-trading-market/ (accessed 26 July 2016), see also C. Marlone, 'Business leaders call for stability reserve in EU Emissions Trading Scheme' (14 January 2015), available 14 July 2016 at http://blueandgreentomorrow.com/2015/01/14/business-leaders-call-for-stability-reserve-in-eu-emissions-trading-scheme/.

yet to be released but their eventual occurrence, or lack thereof, must be explained with a view on China's distinct legal culture.[132]

4. LITIGATION LESSONS FOR ETS IMPLEMENTATION

This chapter set out to provide an analytical overview of the types of litigation that ETS are exposed to and the ways in which these different categories of litigation can, and have, affect(ed) ETS design and development. In doing so, the different functions of the process and outcome of litigation were considered – highlighting the signalling function of both successful and unsuccessful legal challenges to emission trading systems. The latter can be illustrated by experiences in the context of the EU ETS, where appeals against the NAPs persisted despite the lack of standing of the plaintiffs.[133] In combination with the successful challenges of the Member States, NAP litigation in the EU ETS can be said to have had a concrete effect on the implementation of the EU ETS. The direct effect has been one of delay and added uncertainty regarding the state of the EU ETS market due to the large number of allowances that were subject to the litigation.[134] The abolition of the NAPs in the third trading phase may be considered an indirect effect of this litigation on the implementation and design of the EU ETS.[135]

The pre-existing legal system and its rules regarding, inter alia, standing, legal aid, and the reviewability of legislative or administrative decisions continues to be an important parameter for the potential influence of litigation on implementation. Similarly, the institutional features of the legal system, for example, being a federal or multi-level governance system, affects litigation and implementation as different challenges may

[132] See e.g. M Faure and L Jing, 'Compensation for Environmental Damage in China: Theory and Practice' (2013) 31 *Environmental Damage Compensation* 240–321 and M Faure and L Jing, 'Compensating Nuclear Damage in China' (2013) 11 *Washington University Global Studies Law Review* 781–816 (on the role of the courts in environmental litigation more generally). See also Chapter 6 in this volume by M Peeters and H Chen.

[133] See Section 3.B.

[134] See also Dari-Mattiacci, van Zeben, (n. 12).

[135] There are also several other potential explanations for this change, which may be considered cumulative rather than alternative. For detailed discussion see van Zeben, a political economy explanation for competence allocation in the EU ETS, in van Zeben (n. 10).

be brought at different levels of governance.[136] This will impact implementation on different levels to varying degrees and in potentially unexpected ways. ETS specific rules and institutions may create a final level of complexity if they are empowered to review or implement ETS related decisions. These instrument specific provisions can create additional and/or cut off existing paths to litigation. Aside from the contextual drivers for litigation, the maturity of the relevant ETS can be seen to influence the nature, presence and frequency of litigation. In addition, the political environment has proven to significantly drive litigation; both pro- and anti-regulatory parties use the courts to gain influence that they may not be able to achieve through the political system.[137]

Due to the varying stages of global ETS development, it is likely that systems will go through comparable stages of litigation depending on the ETS' maturity. Some of the litigation effects may be mitigated through alternative implementation strategies. An example of such a strategy would be the creation of a litigation reserve that would reduce market effects of litigation concerning the concrete allocation of allowances.[138] Some litigation may be prevented completely, for instance by clarifying the nature of the property right in an emission allowance during the ETS' development.[139] Similarly, future systems may learn from implementation problems related to VAT fraud.[140]

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[136] See cf S Bogojević, 'Climate Change Litigation: All Quiet on the Luxembourgian front?' in G Van Calster, W Vandenberghe and L Reins (eds), *Research Handbook on Climate Mitigation Law* (Cheltenham, UK and Northampton, MA, USA: Edward Elgar, 2014) and S. Bogojević, 'EU Climate Change Litigation, the Role of the European Courts, and the Importance of Legal Culture' (2013) 35(3) *Law and Policy* 184–207 (claiming that litigation under the EU ETS should be considered to illustrate issues of EU constitutional law, e.g. regarding subsidiarity, rather than issues of EU climate change law – the current author respectfully disagrees with this conclusion as the mere presence of competence-based arguments does not preclude the presence of ETS based litigation).

[137] See e.g. CJ Hilson, 'Climate Change Litigation: An Explanatory Approach (or Bringing Grievance Back In)', in F Fracchia and M Occhiena (eds), *Climate change: la riposta del diritto* (Editoriale Scientifica, 2010), 421–36.

[138] The design of such a feature could resemble that of a stability reserve, as currently debated under the EU ETS. See Marlone (n. 131).

[139] Within the EU ETS, this problem is yet to be resolved due to the heterogeneous approach of all the Member States to this issue. See e.g. L Chambers and C Buckingham, 'Intangible Property and Proprietary Restitution in the High Court' (2013) *Lloyd's Maritime and Commercial Law Quarterly* 296–304.

[140] P Efstratios, 'Halting the Horses: EU Policy on the VAT Carousel Fraud in the EU Emissions Trading Scheme' [2012] *EC Tax Review* 39–51.

The question that remains is whether litigation regarding market-based instruments poses a challenge for implementation that is different from that in non-market based regulatory instruments? The dichotomy between emissions trading, as a market-based instrument, and 'traditional' regulation is often exaggerated.[141] Nevertheless, the effects of litigation on the implementation of market-based instruments can be distinguished from that of 'traditional' regulation. The key difference is the potential effect of litigation on the market that has been created by an ETS. The effects of litigation are typically limited to those parties that are included in it. Within an ETS, litigation that touches upon the regulatory fabric of the market potentially affects all parties subject to the scheme. Command-and-control regulation may also be challenged in the courts and judgments may bring about changes in regulation that affect a broader audience. However, within an ETS, the freedom for actors to determine their own mitigation strategy and to capitalize on this strategy on the market is a fundamental strength of the system. If legal uncertainties become too pervasive, these endogenous challenges may undermine the market element of a system such as emissions trading.

[141] See e.g. D. Driesen, 'Is Emissions Trading an Economic Incentive Program?: Replacing the Command and Control/Economic Incentive Dichotomy' (1998) 55 *Washington and Lee Law Review* 289–350.

11. Emissions trading systems and international liability of single major emissions sources

Elena Kosolapova[1]

1. INTRODUCTION

Largely based on mitigation and adaptation, today's international climate policy aims to prevent dangerous anthropogenic interference with the climate system.[2] Although mitigation and adaptation could significantly reduce the risks of climate change, they cannot eliminate all climate change impacts, and some degree of harm is unavoidable. Yet, the international climate regime does not address the injurious consequences of climate change. Proposals to include provisions to that effect in the Kyoto Protocol have been rejected by industrialized nations. A proposal to give the Kyoto Protocol Compliance Committee 'the power to require a state to pay for the restoration of damage to the environment' was not accepted.[3] Attempts to include the polluter-pays principle were likewise rejected.[4] Instead, the international climate process seeks to address climate change risks that are unavoidable even with mitigation and adaptation measures in place, through discussions on loss and damage. The issue has been under negotiation for many years, with the initial call for the need to address unavoidable impacts of climate change dating back to the early 1990s.[5] Originally introduced by small island

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[2] UN Framework Convention on Climate Change 1992, Art. 2.

[3] R Lefeber, 'Climate Change and State Responsibility' in R Rayfuse and S Scott (eds), *International Law in the Era of Climate Change* (Cheltenham, UK and Northampton, MA, USA: Edward Elgar 2012), 328.

[4] TN Slade, 'Climate Change: The Human Rights Implications for Small Island Developing Countries' (2007) 37 *Environmental Policy and Law* 215, 218.

[5] JP Hoffmaister, M Talakai, P Damptey and A Soares Barbosa, 'Warsaw International Mechanism for loss and damage: Moving from Polarizing

developing States (SIDS), the concept of loss and damage has gradually gained support from other vulnerable countries, including the African Group, the like-minded developing countries (LMDCs) and the least developed countries (LDCs).[6]

In 2010, the Conference of the Parties (COP) to the United Nations Framework Convention on Climate Change (UNFCCC) established a work programme on approaches to address loss and damage associated with climate change impacts in developing countries that are particularly vulnerable to the adverse effects of climate change.[7] In 2012, parties to the Convention agreed to establish institutional arrangements, such as an international mechanism, to address loss and damage.[8] With a break-through in the negotiations that came in 2013, COP 19 established the Warsaw Mechanism for Loss and Damage under the Cancun Adaptation Framework to address loss and damage associated with extreme weather and slow onset events in developing countries that are particularly vulnerable to the adverse effects of climate change.[9]

The COP also decided to review, in 2016, the Warsaw Mechanism for Loss and Damage, including 'its structure, mandate and effectiveness'. The interim Executive Committee of the Warsaw Mechanism for Loss and Damage began its work in 2014, adopting, among others, an initial two-year workplan, which COP 20 approved in Lima in December 2014. While liability has been an important element of the loss and damage discussions in the climate process, the workplan contains no mention of it. This reflects an altered framing of the issue in the negotiations, which places a greater emphasis on, inter alia, risk management; challenges

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Discussions towards Addressing the Emerging Challenges Faced by Developing Countries' (2014) Loss and Damage in Vulnerable Countries Initiative, available 15 July 2016 at http://www.lossanddamage.net/4950; see also E Kosolapova, 'Loss and Damage: the Road to Paris and Beyond: Policy update #19' (2015) IISD Climate Change Policy and Practice, available 15 July 2016 at http://climate-l.iisd.org/policy-updates/loss-and-damage-the-road-to-paris-and-beyond/.

[6] Unless otherwise stated, country groupings used in this chapter correspond to negotiating groups under the UNFCCC.

[7] *Cancun Agreements: Outcome of the work of the* Ad Hoc *Working Group on Long-term Cooperative Action under the Convention*, UNFCCC Decision 1/CP.16 (2010), para. 26.

[8] *Approaches to address loss and damage associated with climate change impacts in developing countries that are particularly vulnerable to the adverse effects of climate change to enhance adaptive capacity*, Decision 3/CP.18 (2012), para. 9.

[9] *Warsaw international mechanism for loss and damage associated with climate change impacts*, UNFCCC Decision 2/CP.19 (2013), para. 1.

associated with losses of ecosystems, livelihoods and non-economic losses; and migration, displacement and human mobility.[10]

With no liability provisions contained in the UNFCCC and Kyoto Protocol, and discussions on loss and damage moving away from liability issues, climate liability has to be assessed outside the international political process.

The lack of political guidance poses considerable challenges to establishing liability for climate change-related damage at the international, regional and national levels. Those challenges are also palpable in the context of emissions trading systems (ETS). Along with other climate policy instruments, including carbon pricing mechanisms, such as carbon taxes, ETS provide domestic or regional mitigation incentives.[11] As a market-based mitigation strategy, ETS cannot guarantee reductions in greenhouse gases (GHGs),[12] and cap and trade schemes, particularly the European Union Emission Trading Scheme (EU ETS), have been repeatedly criticized for their 'disappointing environmental results'.[13] Furthermore, as Peeters observes, even in cases of compliance with ETS, climate change-related damage may be caused, and the question of how this damage should be rectified remains largely unregulated.[14] It has been proposed that a climate change liability and compensation regime be developed to address this problem within EU ETS,[15] the largest ETS currently in operation, and that the notion of climate change liability be

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[10] *Report of the Executive Committee of the Warsaw International Mechanism for Loss and Damage associated with Climate Change Impacts*, FCCC/SB/2014/4 (2014), Annex II.

[11] International emissions trading allows Parties with emissions limitation and/or reduction commitments under the Kyoto Protocol to trade emissions units on the international carbon market to meet their international obligations.

[12] AD Ellerman, FJ Convery and Christian de Perthuis, *Pricing Carbon. The European Union Emissions Trading Scheme* (Cambridge: Cambridge University Press 2010), 158.

[13] See J van Zeben's Chapter 10 in this volume on the role of litigation; see also BD Solomon and R Lee, 'Emissions Trading Systems and Environmental Justice' (2000) 42(8) *Environmental Justice* 32; LN Chinn, 'Can the Market be Fair and Efficient? An Environmental Justice Critique of Emissions Trading' (1999) 26(1) *Ecology Law Quarterly* 80.

[14] See M Peeters, 'The Regulatory Approach of the EU in View of Liability for Climate Change Damage' in M Faure and M Peeters (eds), *Climate Change Liability* (Cheltenham, UK and Northampton, MA, USA: Edward Elgar 2011), 90–123.

[15] Peeters argues for a combination of individual liability of liability of GHG emitting entities and an international or European compensation fund to supplement EU ETS.

'loosened' from compliance with ETS requirements, yet none of these suggestions have been implemented.

The present chapter deals with the problem of liability for climate change-related damage, particularly against the background of ETS, addressing the question of how and under what circumstances GHG emitting installations that comply with ETS regulations can still be held liable for climate change-related harm. It first evaluates prospects for a state liability regime that could potentially accommodate such situations. Second, it provides a brief overview of climate liability cases in national jurisdictions, highlighting obstacles to successful litigation. Finally, the chapter identifies pathways towards liability under the law of state responsibility on the basis of breach of the international customary obligation to prevent significant transboundary harm, which could be extended to particular GHG emitting sources.

2. STATE LIABILITY AND CLIMATE CHANGE

As a transboundary environmental problem climate change requires a global solution, also in the field of liability. One way to disengage the issue of liability for climate change-related damage from compliance with ETS rules could be through an international liability regime – either providing for state liability for climate change-related damage or channeling liability towards operators.

Interstate liability can be established under primary (state liability) and secondary norms of international law (state responsibility).[16] In the law

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[16] On the distinction between primary and secondary rules of international law, see, e.g. D Bodansky, JR Crook and J Crawford, 'The ILC's Articles on Responsibility of States for Internationally Wrongful Acts: a Retrospect' (2002) 96 AMJIL 874; M Fitzmaurice, 'International Responsibility and Liability' in D Bodansky, J Brunnée and E Hey (eds), *The Oxford Handbook of International Environmental Law* (New York: Oxford University Press 2007). See also, on the distinction between state responsibility and state liability: C Hoss, 'State Responsibility, Liability and Environmental Protection' in R Wolfrum, C Langenfeld and P Minnerop (eds), *Environmental Liability in International Law: Towards a Coherent Conception* (Berlin: Erich Schmidt Verlag 2005); A Kiss and D Shelton, 'Strict Liability in International Environmental Law' in TM Nndiaye and R Wolfrum (eds), *Law of the Sea, Environmental Law and Settlement of Disputes: Liber Amicorum Judge Thomas A. Mensah* (Leiden: Martinus Nijhoff Publishers 2007); MA Drumbl, 'Trail Smelter and the International Law Commission's Work on State Responsibility for Internationally Wrongful Acts and State Liability' in RM Bratspies and RA Miller (eds), *Transboundary Harm in International Law* (New York: Cambridge University Press 2006).

of state responsibility, international obligations regulating the conduct of international actors in a specific sector of interstate relations are referred to as primary rules of international law. The determination of legal consequences of a State's failure to fulfil those obligations is often governed by the secondary rules of international law rather than by the primary rules of a particular sector. Thus, state *liability* arises from *primary* norms of international law. It is triggered when lawful acts of a State lead to harm in another State's territory. In contrast, state *responsibility* is engaged under *secondary* rules of international law.[17] It is distinct from state liability in that it is predicated on the existence of an internationally wrongful act.[18] It is with this distinction in mind that the discussion of state liability and state responsibility is taken forward.

In international law, four main conceptual approaches to state liability can be distinguished. Those approaches are based on the nature of international obligations the relevant primary norms create under various multilateral environmental agreements (MEAs). Depending on the approach adopted by a particular legal regime, state liability may give rise to: (1) the obligation to pay compensation; (2) the obligation to negotiate a redress settlement; (3) the obligation to ensure prompt, adequate and effective compensation; or (4) the obligation to take response action.[19] Below, the four liability models are examined individually, and an assessment of their suitability to address climate change-related damage is provided.

The first approach to state liability involves the obligation to pay compensation. It dates back to the 1960s when the proliferation of space and nuclear activities brought with it new risks associated with the administration of those ultra-hazardous activities. Thus, many environmental legal scholars, including Birnie, Boyle, Kiss, Shelton and Lefeber, associate strict liability of States with activities that are considered 'especially new or dangerous'.[20] The concept of strict liability of States has been

[17] Secondary norms of international law are codified by the International Law Commission in its Articles on Responsibility of States for Internationally Wrongful Acts.

[18] See e.g. M Fitzmaurice, 'International Responsibility and Liability' in D Bodansky, J Brunnée and E Hey (eds), *The Oxford Handbook of International Environmental Law* (New York: Oxford University Press 2007).

[19] On various approaches to state liability in environmental law, see R Lefeber, *Transboundary Environmental Interference and the Origin of State Liability* (The Hague: Kluwer Law International, 1996). On approaches to state liability vis-à-vis climate change, see E Kosolapova, *Interstate Liability for Climate Change-Related Damage* (The Hague: Eleven International Publishing 2013), 35–59.

[20] P Birnie, A Boyle and C Redgwell, *International Law & the Environment* (New York: Oxford University Press 2009), 218; A Kiss and D Shelton, *Guide to*

developed to facilitate the recovery of compensation for harm caused by ultra-hazardous activities of a nuclear and space exploration nature.[21] The 1972 Space Liability Convention[22] is an example of an international agreement using liability based on the obligation to pay compensation.[23] The Convention provides a legal framework for the settlement of claims associated with harm arising out of space activities. Under it, compensation is available to the injured State under an absolute liability scheme in cases of damage caused by space objects on the surface of the Earth or to aircraft in flight.[24] In cases of damage to a space object of one launching State by a space object of another launching State elsewhere than on the surface of the Earth, liability of the latter State is fault-based.[25]

While this conceptual approach to liability might be an attractive option for countries suffering from the injurious consequences of climate change, state liability for climate change-related damage based on the obligation to pay compensation must be rejected. Under international law, this approach is limited to space activities associated with ultrahazardous risks. In the case of climate change-related damage, including that caused by GHG emissions from single sources, also those in compliance with ETS requirements, this liability model would be politically unacceptable as States are generally reluctant to accept absolute liability for the conduct of private actors, which emit most of the GHGs that contribute to global warming. From a legal point of view, too, causation would render the absolute liability approach to climate change-related damage unsuitable as it would

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International Environmental Law (Leiden: Martinus Nijhoff Publishers, 2007), 24; A Kiss and D Shelton, 'Strict Liability in International Environmental Law' in TM Nndiaye and R Wolfrum (eds), *Law of the Sea, Environmental Law and Settlement of Disputes*: *Liber Amicorum Judge Thomas A. Mensah* (Leiden: Martinus Nijhoff Publishers 2007), 1135; R Lefeber, *Transboundary Environmental Interference and the Origin of State Liability* (The Hague: Kluwer Law International 1996), 150.

21 R Lefeber, *Transboundary Environmental Interference and the Origin of State Liability* (The Hague: Kluwer Law International 1996), 159.

22 Convention on International Liability for Damage Caused by Space Objects, United Nations 1972, Treaties and Principles on Outer Space, Text of treaties and principles governing the activities of States in the exploration and use of outer space, adopted by the United Nations General Assembly, ST/SPACE/11, United Nations, New York, 2002, p. 13.

23 The only other agreement using strict liability based on the obligation to pay compensation is the Council for Mutual Economic Assistance (CMEA) Convention on Liability for Damage Caused by Radiological Accidents in International Carriage of Irradiated Nuclear Fuel from Nuclear Power Plants 1987 (now considered obsolete).

24 Space Liability Convention 1972, Art. II.

25 Space Liability Convention 1972, Art. III.

be impossible to trace damage in one country to GHG emissions originating in another.[26] Also, since, in environmental law, the absolute liability standard is usually associated with activities carrying significant risk, it is not a good fit for the emission of GHGs – an activity that, in and of itself, cannot be considered either risky or dangerous, particularly when regulatory requirements under ETS are met.

The second approach to state liability is based on the obligation to negotiate a redress settlement and is supported by few sources of international law.[27] The 1997 Convention on the Law of the Non-navigational Uses of International Watercourses is one of those rare examples.[28] Under the Convention, the obligation to negotiate a redress settlement is aimed at the reparation of transboundary harm when such harm has occurred despite the source State's diligent conduct.

State liability based on the obligation to negotiate a redress settlement has not taken root in international law, making this approach unlikely also in the context of climate change-related damage. In any event, this liability model would hardly be able provide victims with any redress at all and is generally unsuitable due to the fact that climate change is compounded by multiple activities worldwide.

While providing some possibilities for state liability, the third approach to liability imposes on the source State the obligation to ensure that victims of transboundary damage receive prompt, adequate and effective compensation, and is chiefly adopted by international civil liability regimes. The obligation to ensure prompt, adequate and effective compensation

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[26] So far only the Maldives has announced its intention of becoming carbon-neutral by 2020; pledges of carbon neutrality have also been made by Bangladesh, Barbados, Bhutan, Ghana, Kenya, Kiribati, Nepal, Rwanda, Tanzania and Vietnam, see Declaration of the Climate Vulnerable Forum, Malé, Maldives (10 November 2009), available 15 July 2016 at http://daraint.org/wp-content/uploads/2010/12/Declaration-of-the-CVF-FINAL2.pdf.

[27] This approach to liability was considered by the ILC as part of its work on international liability for injurious consequences arising out of acts not prohibited by international law but ultimately rejected in favour of an approach based on the obligation to ensure prompt, adequate and effective compensation. See E Kosolapova, *Interstate Liability for Climate Change-Related Damage* (The Hague: Eleven International Publishing 2013), pp. 43–44.

[28] Convention of the Law of the Non-navigational Uses of International Watercourses 1997, Doc. A/51/869 (11 April 1997). Other examples include Council of Europe Draft Convention on the Protection of Fresh Water against Pollution 1969, CECA Doc. 2561 (1969) and bilateral Exchange of Notes Between the United States of America and Canada Constituting an Agreement Relating to Liability for Loss and Damage from Certain Rocket Launches 1974, UNTS, vol. 992, p. 97 (1975).

performs a reparative function in cases when the State has complied with its due diligence obligation to prevent transboundary environmental harm under customary international law[29] but damage has nonetheless been caused. It has been suggested that the obligation to ensure prompt, adequate and effective compensation be extended to all transboundary environmental interference, irrespective of its nature, in order to provide victims with financial guarantees against harm caused by hazardous and non-hazardous activities alike.[30] At present, however, this obligation can only be discerned in respect of harm arising out of hazardous activities, which is reflected in the Principles on Allocation of Loss in the Case of Transboundary Harm Arising Out of Hazardous Activities 2006 of the International Law Commission (ILC).[31] Notably, as Lefeber points out, it has been the practice adopted by international agreements and municipal laws to limit the applicability of special civil liability regimes on the basis of the nature of the activity.[32]

In principle, state liability based on the obligation to ensure prompt, adequate and effective compensation, which places primary liability with the operator as opposed to the State and thus directly enables victims to seek compensation for the harm incurred, would be of use to climate change victims. However, it can only be invoked with regard to hazardous activities and, as such, is conceptually unfit to address climate change-related damage. There is nothing in the nature of the emission of GHGs *per se* to suggest that this activity can be qualified as hazardous under international law; GHGs are emitted by virtually every human activity, like farming or driving, for instance. GHGs only become dangerous when they reach high levels of concentration in the Earth's atmosphere. Additionally, the liability model based on the obligation to ensure prompt, adequate and effective compensation would be unsuitable for legal reasons. Channeling liability to operators could result in unreasonably wide coverage, and causation problems would be difficult to overcome due to the multiplicity of GHG emitting sources.

Along with the liability model based on the obligation to ensure prompt, adequate and effective compensation, the fourth approach involving the obligation to take response action, which applies to situations arising

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[29] On the obligation to prevent significant transboundary harm see section 3.2.

[30] R Lefeber, *Transboundary Environmental Interference and the Origin of State Liability* (The Hague: Kluwer Law International 1996), 233.

[31] *Allocation of loss in the case of transboundary harm arising out of hazardous activities*, GA Res. 61/36 (4 December 2006).

[32] See R Lefeber, *Transboundary Environmental Interference and the Origin of State Liability* (The Hague: Kluwer Law International 1996), 239–54.

after the occurrence of transboundary damage, too, is reflected in the ILC Principles on Allocation of Loss in the Case of Transboundary Harm Arising Out of Hazardous Activities 2006. In accordance with the Principles, upon the occurrence of an incident involving a hazardous activity, the State of origin is duty-bound to obtain from the operator the necessary information and promptly notify all (likely to be) affected States.[33] The source State is then expected to ensure that appropriate response measures are taken using best available technology (BAT).[34] It should also consult and cooperate with the affected States to mitigate and, if possible, eliminate the effects of transboundary damage.[35] Once notified, all affected States shall take all feasible measures to mitigate and eliminate the effects of transboundary damage.[36]

The principal examples of international agreements that rely on this approach are Annex VI to the Protocol on Environmental Protection to the Antarctic Treaty regarding liability arising from environmental emergencies (Liability Annex)[37] and the Nagoya – Kuala Lumpur Supplementary Protocol on Liability to the Convention on Biological Diversity (Supplementary Protocol).[38] In utilising the same approach to

[33] Principles on the Allocation of Loss in the Case if Transboundary Harm Arising out of Hazardous Activities 2006, Principle 5(a).

[34] Principles on the Allocation of Loss in the Case if Transboundary Harm Arising out of Hazardous Activities 2006, Principle 5(b).

[35] Principles on the Allocation of Loss in the Case if Transboundary Harm Arising out of Hazardous Activities 2006, Principle 5(c).

[36] Principles on the Allocation of Loss in the Case if Transboundary Harm Arising out of Hazardous Activities 2006, Principle 5(d).

[37] Annex VI to the Protocol on Environmental Protection to the Antarctic Treaty (Liability Arising from Environmental Emergencies) 2005, available 15 July 2016 at www.ats.aq/documents/recatt/Att249_e.pdf.

[38] Nagoya – Kuala Lumpur Supplementary Protocol on Liability and Redress to the Cartagena Protocol on Biosafety 2010, UNEP/CBD/BS/COP-MOP/5/17 (15 October 2010). This approach has also been incorporated into the 2010 UNEP Guidelines for the Development of Domestic Legislation on Liability, Response Action and Compensation for Damage Caused by Activities Dangerous to the Environment, available 15 July 2016 at http://www.unep.org/environmentalgovernance/Portals/8/Guidelinesdomesticlegislation-FINAL.pdf. On the Supplementary Protocol, see generally: R Lefeber, 'The Legal Significance of the Nagoya-Kuala Lumpur Supplementary Protocol: The Result of a Paradigm Evolution' (2012) Amsterdam Law School Legal Studies Research Paper No. 2012-87, Centre for Environmental Law and Sustainability Research Paper No. 2012-02; R Lefeber and J Nieto Carrasco, 'Negotiating the Supplementary Protocol: the Co-Chairs' Perspective' in A Shibata (ed.), *International Liability Regime for Biodiversity Damage: The Nagoya-Kuala Lumpur Supplementary Protocol* (Routledge 2014).

liability, Annex VI and the Supplementary Protocol take on somewhat different perspectives. Essentially, as explained by Shibata, the Liability Annex provides for operators' liability for environmental emergencies in the Antarctic, with the State being liable only when it itself is the operator.[39] The Supplementary Protocol applies to damage resulting from intentional, unintentional and illegal transboundary movements of living modified organisms (LMOs) and obligates States to require operators to inform the competent authority, evaluate damage and take appropriate response measures.[40] In turn, the competent authority shall identify the operator which has caused the damage, evaluate the damage and determine which response measures the operator has to take.[41] The crucial difference of the perspective adopted by the Supplementary Protocol from the one used by the Liability Annex lies in the former's extended definition of response measures, which encompasses prevention and minimization of harm in the event of damage caused by an incident involving transboundary movement of LMOs. The Supplementary Protocol also takes operators' liability a step further and provides for restoration measures to redress the damage and to eliminate, as far as possible, its consequences by bringing biodiversity to its original state or its nearest equivalent and, in case of loss of biodiversity, its replacement with other components thereof.[42] In accordance with the Supplementary Protocol, the operator shall be required to take response measures if damage has occurred or there is a 'sufficient likelihood' that damage will occur.[43]

Unlike international agreements based on compensation-related approaches to state liability (obligation to pay compensation and obligation

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[39] Annex VI to the Protocol on Environmental Protection to the Antarctic Treaty (Liability Arising from Environmental Emergencies) 2005, Art. 5(1); see also A Shibata, 'How to Design an International Liability Regime for Public Spaces: the Case of the Antarctic Environment' in T Komori and K Wellens (eds), *Public Interest Rules of International Law: towards Effective Implementation* (Farnham: Ashgate 2009), 352.

[40] Nagoya – Kuala Lumpur Supplementary Protocol on Liability and Redress 2010, Arts 3, 5(1). On the Supplementary Protocol, see: R Lefeber, 'The Legal Significance of the Nagoya-Kuala Lumpur Supplementary Protocol: the Result of a Paradigm Evolution' (2012) Amsterdam Law School Legal Studies Research Paper No. 2012-87, Centre for Environmental Law and Sustainability Research Paper No. 2012-02.

[41] Nagoya – Kuala Lumpur Supplementary Protocol on Liability and Redress 2010, Art. 5(2).

[42] Nagoya – Kuala Lumpur Supplementary Protocol on Liability and Redress 2010, Art. 2(2)(d).

[43] Nagoya – Kuala Lumpur Supplementary Protocol on Liability and Redress 2010, Art. 5(3).

to ensure prompt, adequate and effective compensation), these instruments do not focus on compensating the injured party (State or non-State), but rather require States to ensure that operators take response action aimed at the avoidance of damage following an environmental emergency, which may include clean-up (as in the Liability Annex), or going as far as to necessitate restoration measures (as in the Supplementary Protocol). However, state liability based on the obligation to take response action cannot accommodate climate change for conceptual reasons as this approach is directed at responding to an incident, whereas climate change damage results from a series of complex processes involving numerous actors across time and space. Further, liability regimes relying on the duty to take response measures deal with environmental damage whereas, at the end of the day, a large proportion of climate change-related harms amount to other kinds of damage, including damage to property.

The above evaluation of international state liability regimes that may be relevant to climate change suggests that none of the conceptual approaches to liability are entirely appropriate to accommodate climate change issues, and most of these approaches cannot be applied to instances when single GHG emitting sources are concerned. A liability mechanism imposing on States the duty to pay compensation would be an attractive option for countries suffering from the injurious consequences of climate change. However, under international law, this approach is limited to space activities associated with ultrahazardous risks, and, in the case of climate change-related damage, would be politically unacceptable. In providing for liability of States, this liability model cannot be used to deal with GHG emitting installations.

Given its nature, the approach based on the obligation to negotiate a redress settlement would not be able to provide victims with redress and is generally unsuitable due to the fact that multiple activities worldwide contribute to climate change.

The obligation to ensure prompt, adequate and effective compensation would be of use to climate change victims, such as members of coastal communities or inhabitants of low-lying islands, and, in principle, could be extended to GHG emitting installations. However, it can only be invoked with regard to hazardous activities, and, as such, is conceptually unfit to address climate change-related damage.

Finally, while liability regimes relying on the duty to take response measures do provide for operators' as well as state liability, they deal with environmental damage whereas, at the end of the day, much of climate change-associated harm amounts to property damage. Thus, the approach based on the obligation to take response action is conceptually incompatible with climate change-related damage.

3. LEGAL CHALLENGES TO CLIMATE LIABILITY IN DOMESTIC COURTS[44]

Domestically, too, the question of liability for climate change-related damage remains far from solved, with the past decade marking a significant rise in climate change litigation worldwide.[45] Hundreds of climate change lawsuits have been launched in the US, Australia, New Zealand and Canada, to name but a few.[46] Plaintiffs, ranging from environmental groups to federal states and private individuals, have brought actions inculpating corporations, government agencies, oil refineries, motor vehicle manufacturers, power plants and other public and private entities. Actions have been brought in tort (for example, public nuisance, negligence, civil conspiracy, misrepresentation), under administrative law (including merits review and judicial review) and constitutional law, among others. One goal of climate change litigation has been 'to impose legal liability upon a party that is somehow responsible for the emission of greenhouse gases that contribute to climate change',[47] albeit, in practice, climate change lawsuits have targeted a broader range of issues, such as forcing municipal or federal governments to act or challenging the approval of particular GHG-intensive projects.[48]

Until today, no GHG emitter has been found liable for climate change-related damage by any domestic court.[49] Although numerous legal avenues have been tested, climate litigation remains fraught with difficulties. General obstacles to liability for climate change-related damage, including standing and causation, have been exacerbated by country-specific

[44] More specifically, on climate litigation and ETS, see J van Zeben's chapter on the role of litigation, Chapter 10 in this volume.

[45] For an overview of climate change litigation in domestic courts, see E Kosolapova, *Interstate Liability for Climate Change-Related Damage* (The Hague: Eleven International Publishing 2013), 85–127; see also: Columbia Center for Climate Change Law, available 15 July 2016 at www.climatecasechart.com/.

[46] Despite the fact that in many countries, adequate legal instruments for adjudicating global warming claims may be lacking, some authors anticipate that climate change litigation will keep spreading to other jurisdictions. See, e.g., MG Faure, A Nollkaemper and Amsterdam International Law Clinic, *Climate Change Litigation Cases* (Milieudefensie: Amsterdam 2007), 59.

[47] S-L Hsu, 'A Realistic Evaluation of Climate Change Litigation Through the Lens of a Hypothetical Lawsuit' (2008) 79 U. Colo. L. Rev. 701, 702.

[48] For an exhaustive classification of climate lawsuits by the type of action, see Columbia Center for Climate Change Law, available 15 July 2016 at www.climatecasechart.com/.

[49] See E Kosolapova, *Interstate Liability for Climate Change-Related Damage* (The Hague: Eleven International Publishing 2013), 85.

hurdles, such as the political question doctrine in the United States (US) and Canada.[50] Yet, overall, procedural injury cases[51] have enjoyed a greater degree of success than claims for compensation/damages[52] and injunctive relief cases.[53] For instance, the requirement of causation as an element of procedural standing in US courts is relaxed and has not posed any significant challenges to plaintiffs.[54] Having been extensively considered by Australian courts and, to some extent, by courts in New Zealand,

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[50] The political question doctrine is based on the notion that the judiciary must not intervene in policy issues that are to be decided by the government. See, generally BC Mank, 'Standing and Global Warming: Is Injury to All Injury to None?' (2005) 35 Envtl. L. 1; P Weinberg, '"Political Questions": An Invasive Species Infecting the Courts' (2008) 19 Duke Envtl. L. & Pol'y F. 155; P Daly, 'Justiciability and the "Political Question" Doctrine,' (2010) P.L. 2010 Jan 160. On the political question doctrine in climate lawsuits, see E Kosolapova, *Interstate Liability for Climate Change-Related Damage* (The Hague: Eleven International Publishing 2013), 91–127.

[51] See, e.g., in the US: *Friends of the Earth, Inc. v. Mosbacher*, 488 F.Supp. 2d 889 (N.D.Cal. 2007); *Natural Resources Defense Council v. Kempthorne*, 506 F.Supp. 2d 322 (E.D.Cal. 2007); *Center for Biological Diversity v. United States Department of the Interior*, 563 F. 3d 466 (D.C.Circ. 2009); in Australia: *Australian Conservation Foundation v. Minister for Planning* [2004] VCAT 2029; *Gray v. Minister for Planning and Ors* [2006] NSWLEC 720; *Wildlife Preservation Society of Queensland Proserpine/Whitsunday Branch Inc. v. Minister for the Environment & Heritage & Ors* [2006] FCA 736; *Re Xstrata Coal Queensland Pty Ltd & Ors* [2007] QLRT 33; *Gippsland Coastal Board v. South Gippsland SC & Ors (No 2) (includes Summary) (Red Dot)* [2008] VCAT 1545;); in New Zealand: *Environmental Defence Society (Inc) v. Auckland Regional Council* [2002] NZRMA 492; *Greenpeace v. Northland Regional Council & Mighty River Power Ltd* [2006] NZHC 1212.

[52] See, e.g., in the US: *California v. General Motors Corp.*, 2007 WL 2726871 (N.D.Cal.); *Native Village of Kivalina v. ExxonMobil Corp.*, 663 F.Supp.2d 863 (N.D.Cal. 2009) & *Native Village of Kivalina v. ExxonMobil Corp.*, 2012 WL 4215921 (9th Cir.(Cal.)) (21 September 2012) (NO. 09-17490); *Comer v. Nationwide Mutual Insurance Co*., WL 1066645 (S.D.Miss.), 23 February 2006, *Comer v. Murphy Oil USA, Inc*., 585 F.3d 855, C.A.5 (Miss.), 2009 and *Comer v. Murphy Oil USA, Inc*. 718 F.3d 460, C.A.5 (Miss.), 14 May 2013.

[53] See, e.g., in the US: *Connecticut v. American Electric Power Co., Inc.*, 406 F.Supp. 2d 265 (S.D.N.Y. 2005), 35 Envtl. L. Rep. 20, 186 & *Connecticut v. American Electric Power Co., Inc.*, 582 F.3d 309, C.A.2 (N.Y.), 2009; *Korsinsky v. EPA*, 2005 WL 2414744 (S.D.N.Y.); *Northwest Environmental Defense Center v. Owens Corning Corp.*, 434 F.Supp. 2d 957 (D.Or. 2006); *Massachusetts v. EPA*, 127 S.Ct. 1438, U.S., 2007; in Canada: *Friends of the Earth – Les Ami(e)s de la Terre v. The Governor in Council and The Minister of the Environment*, 2008 FC 1183, 20 October 2008.

[54] For procedural standing requirements, see *Lujan v. Defenders of Wildlife*, 112 S.Ct. 2130 (U.S.Minn. 1992).

causation – general as well as specific – has been recognized in a number of procedural injury claims.[55] This is indicative of a lower standard of proof involved in procedural cases due to the fact that no actual injury is at stake. It is significant that demonstrating that climate change must be taken into account by the relevant authority in approving a particular project or carrying out an environmental impact assessment (EIA) does not require the plaintiffs to meet the rigours of the *causa proxima* or but-for tests.

Actions for injunctive and/or declaratory relief have been somewhat less successful than claims related to procedural injury. While the political question doctrine has presented some challenges, most difficulties have been associated with demonstrating standing and, especially, causation as one of the requirements for standing. It is significant that the singular success of *Massachusetts v EPA* appears to be rooted in the procedural character of the injury alleged by the plaintiffs. In practice, the US Supreme Court's ruling that the Environmental Protection Agency (EPA) did have the authority to regulate GHGs amounted to an injunction requiring the EPA to take regulatory action. While few pronouncements on the merits have been made, the courts have indicated that, on the merits, the standard of proof for demonstrating causation between the action complained of and the potential injury is necessarily higher than that for establishing standing.

Claims for compensation constitute the least successful category of climate change-related claims. To a large extent, compensation claims have been hindered by the political question doctrine, which the courts have relied on to avoid making determinations of a political nature in accordance with the principles of the separation of powers. Standing and causation as part of the standing inquiry have presented insurmountable challenges to plaintiffs. It appears that courts have been more cautious in their approach to claims for compensation for actual damage as opposed to claims seeking injunctive relief to redress potential harm. Should a compensation claim be decided on the merits, enormous evidentiary challenges to establishing a causal link between the defendant's GHG emissions and the actual harm suffered by the plaintiff must be expected.

In the ETS context, the success of any kind of domestic liability

[55] Op. cit. 51; on general and specific causation, see R Verheyen, *Climate Change Damage and International Law – Prevention Duties and State Responsibility* (Leiden: Martinus Nijhoff Publishers 2005), 257. General causation requires proof that anthropogenic GHG emissions cause changes in radiative forcing and the global climate. Specific causation requires proof that a particular injury is attributable to (particular) anthropogenic emissions or to the global warming caused by them.

litigation involving climate change-related damage would necessarily depend on national regulatory frameworks. It would also require that the notion of liability be divorced from compliance with ETS requirements. Given the fact that, as of today, no climate change claim has been marked by any significant degree of success, save for procedural injury cases, it stands to reason that challenging an ETS-compliant installation for climate change-related harm in a national court would be difficult. Standing, causation, various country-specific hurdles (for example, the political question doctrine), as well as legal challenges that courts have not yet explicitly addressed (for example, retroactivity, attribution), would present additional obstacles to liability. Therefore, alternative pathways to liability options must be explored, including under international law.

4. OPPORTUNITIES UNDER THE LAW OF STATE RESPONSIBILITY

As the international climate regime has no liability mechanism of its own, there are no primary rules governing liability for climate change-related damage. The only available legal framework for addressing internationally wrongful conduct and obtaining reparation for an international wrong is the law of state responsibility, which can also provide interstate litigants with the necessary remedial mechanisms.[56] In other words, a State can only be held to account having committed an internationally wrongful act. In order to determine wrongfulness of a State's conduct, it must be established whether the act in question is attributable to that State and whether that act constitutes a breach of the relevant primary obligation.[57]

Thus, first and foremost, it is important to identify the primary obligations of States under the global legal framework for international climate policy and customary international law. The legal consequences of breach

[56] See Articles on Responsibility of States for Internationally Wrongful Acts 2001, ILC Report on the work of its 53rd session, A/56/10, YILC, vol. II, Part Two, Art. 1 (Responsibility of a State for its internationally wrongful acts), Part Two (Content of the international responsibility of a State). In focusing on 'the general conditions under international law for the State to be considered responsible for wrongful actions or omissions, and the legal consequences which flow therefrom,' the Articles apply to all areas of international law: Articles on Responsibility of States for Internationally Wrongful Acts 2001, ILC Report on the work of its 53rd session, A/56/10, YILC, vol. II, Part Two, general commentary, p. 59, para. 1.

[57] Articles on Responsibility of States for Internationally Wrongful Acts 2001, Art. 2 (Elements of an internationally wrongful act of a State).

of those obligations could then be determined under the secondary norms of state responsibility.

Under the international climate regime, States have obligations on climate change mitigation and adaptation. As the present chapter deals with the question of *liability* for climate change-related *damage*, States' obligations related to mitigation must be considered more closely.[58]

4.1 International Obligations on Climate Change Mitigation

It is crucial to note here that the international climate regime is based on the distinction between industrialized and developing countries reflected in the UNFCCC principle of common but differentiated responsibilities and respective capabilities (CBDRRC).[59] This differentiation is predicated on the fact that the largest share of historical and current global emissions of GHGs has originated in developed countries, that *per capita* emissions in developing States are still relatively low, and that the developing countries' share of global emissions will continue to grow to meet their development needs.[60] Albeit this interpretation of the CBDRRC principle has been repeatedly challenged by States and academics alike[61] and is currently under consideration in Ad Hoc Working Group on the Durban Platform

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[58] While the UNFCCC and the Kyoto Protocol require developed States to make available to developing countries financial resources for the development of adaptation policies and for the transfer of technology, industrialized States are under no obligation to finance the actual costs of adaptation measures in the developing world. See UNFCCC 1992, Art. 4(3) and 4(4) and Kyoto Protocol 1997, Art. 11(2). A potential legal basis for holding developed States liable to finance adaptation measures in developing countries could be found in the polluter-pays principle: see R Lefeber, *An Inconvenient Responsibility* (Utrecht: Eleven International Publishing 2009), p. 13; R Lefeber, 'Climate Change and State Responsibility' in R Rayfuse and S Scott (eds), *International Law in the Era of Climate Change* (Cheltenham, UK and Northampton, MA, USA: Edward Elgar 2012), 326.

[59] UNFCCC 1992, Art. 3(1).

[60] See the UNFCCC 1992, preamble.

[61] In UNFCCC negotiations, several countries (e.g. the EU, Switzerland, the US) have repeatedly stressed the need to reflect, in the new agreement, evolving capabilities and responsibilities: see B Antonich, E Kosolapova et al., 'Summary of the Lima Climate Change Conference: 1–14 December 2014' (2014) *Earth Negotiations Bulletin*, Vol. 12, No. 619. See also J Allan, E Kosolapova et al., 'Summary of the Warsaw Climate Change Conference: 11–23 November 2013' (2013) *Earth Negotiations Bulletin*, Vol. 12, No. 594, 30; E Kosolapova, 'A Responsibility to Mitigate' (2014) *The Environmental Forum*, Vol. 31, Issue 5, September/October 2014, 30; and, on categorization on the basis of levels of prosperity, E Kosolapova, *Interstate Liability for Climate Change-Related Damage* (The Hague: Eleven International Publishing 2013), 172–3.

for Enhanced Action (ADP) negotiations, essentially, it is accepted that the entire international community has a responsibility to mitigate climate change, with different countries bearing various degrees of responsibility, which is determined by their historic and contemporary contributions as well as their implementation capacity.[62]

It is significant that, under the UNFCCC and Kyoto Protocol, only industrialized countries are legally obligated to adopt national policies and take corresponding measures on the mitigation of climate change.[63] Developing States are only bound by the general obligation to formulate national or regional programmes containing measures to mitigate climate change, whereas the obligation to implement such programmes is not binding and is further subject to the provision of technological, financial and capacity-building assistance to developing countries by industrialized nations.[64] Therefore, as far as the UNFCCC and the Kyoto Protocol are concerned, only developed States can potentially be held responsible for failure to adopt national policies and take measures on climate change mitigation. Once attributed to a particular industrialized State, such an omission can potentially constitute a breach of an international obligation, thereby giving rise to an internationally wrongful act, which, in turn, would entail the international responsibility of that State. Additionally, the Kyoto Protocol imposes, on developed States only, quantified emissions reduction or limitation targets, and industrialized countries could incur state responsibility for failure to meet those targets.[65] However, given that not all industrialized countries have undertaken commitments under the Kyoto Protocol second commitment period[66] and that developing nations have no binding GHG limitation or reduction commitments, the Protocol only covers approximately 15% of global emissions. Therefore, it is essential to further explore and utilize the potential of the customary obligation

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[62] See E Kosolapova, *Interstate Liability for Climate Change-Related Damage* (The Hague: Eleven International Publishing 2013), 135.

[63] UNFCCC 1992, Art. 4(2)(a); see also Kyoto Protocol 1997, Art. 2(1)(a).

[64] UNFCCC 1992, Arts 4(1)(b) and 4(7); see also Kyoto Protocol 1997, Art. 10(b).

[65] Kyoto Protocol 1997, Art. 3(1) and Art. 3(1)*bis* of the 2012 Doha amendment to the Kyoto Protocol, available 15 July 2016 at http://unfccc.int/files/kyoto_protocol/application/pdf/kp_doha_amendment_english.pdf.

[66] In addition to the US, Canada, Japan, New Zealand and the Russian Federation have refused to take on emissions limitation or reduction commitments under the Kyoto Protocol's second commitment period. See *Outcome of the work of the Ad Hoc Working Group on Further Commitments for Annex I Parties under the Kyoto Protocol at its sixteenth session*, UNFCCC Decision 1/CMP.7 (2011), Ann. 1.

to prevent significant transboundary harm to ensure that *all* countries take adequate mitigation measures.[67]

4.2 Obligation to Prevent Significant Transboundary Harm

According to many international environmental law researchers, including Schwarte, Verheyen and Voigt, the customary international law obligation to prevent significant transboundary harm[68] can be extended to damage from GHG emissions,[69] and its potential to compel States to mitigate climate change cannot be overstated.[70] Breach of this obligation can

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[67] E Kosolapova, 'A Responsibility to Mitigate' (2014) *The Environmental Forum*, Vol. 31, Issue 5, September/October 2014, 30, 32, emphasis added. The obligation not to cause transboundary damage originated in the *Trail Smelter* arbitration and is widely considered a reflection of the international custom. See *Trail Smelter case* (*United States v. Canada*), 16 April 1938 and 11 March 1941, UNRIAA, vol. III 1905; on the arbitration, see, e.g.: G Handl, 'Trail Smelter in Contemporary International Environmental Law: Its Relevance in the Nuclear Energy Context' in RM Bratspies and RA Miller (eds), *Transboundary Harm in International Law* (New York: Cambridge University Press 2006); M Fitzmaurice, 'International Responsibility and Liability' in D Bodansky, J Brunnée and E Hey (eds), *The Oxford Handbook of International Environmental Law* (New York: Oxford University Press 2007); N de Sadeleer, 'The Principles of Prevention and Precaution in International Law: Two Heads of the Same Coin?' in M Fitzmaurice, DM Ong and P Merkouris (eds), *Research Handbook on International Environmental Law* (Cheltenham, UK and Northampton, MA, USA: Edward Elgar 2010).

[68] The customary law duty originated in the *Trail Smelter* arbitration: 'under the principles of international law [. . .] no State has the right to use or permit the use of its territory in such a manner as to cause injury by fumes in or to the territory of another or the properties or persons therein, when the case is of serious consequence and the injury is established by clear and convincing evidence.' See *Trail Smelter case* (*United States v. Canada*), 16 April 1938 and 11 March 1941, UNRIAA, vol. III 1905, p. 1965.

[69] See C Schwarte and R Byrne, 'International Climate Change Litigation and the Negotiation Process, Foundation for International Environmental Law and Development' working paper (2010), available 15 July 2016 at http://www.caneurope.org/resources/publications/member-publications/mitigation/1942-field-international-climate-change-litigation-and-the-negotiation-process-oct-2010/file, pp 6–7; R Tol and R Verheyen, 'Liability and Compensation for Climate Change Damages – a Legal and Economic Assessment' working paper FNU-9 (2001) Research unit Sustainability and Global Change, Hamburg University, available 15 July 2016 at www.fnu.zmaw.de/fileadmin/fnu-files/publication/working-papers/adapcap.pdf, p. 12; C Voigt, 'State Responsibility for Climate Change Damages' (2008) 77 *Nordic Journal of International Law* 1, 7–9.

[70] See E Kosolapova, *Interstate Liability for Climate Change-Related Damage* (The Hague: Eleven International Publishing 2013), 147.

provide a much broader legal basis for state responsibility than treaty-based obligations: under it, all States, industrialized as well as developing, are duty-bound to take adequate measures to ensure that activities within their jurisdiction or control do not cause transboundary harm.

The obligation to prevent significant transboundary harm is one of due diligence in that it does not impose an absolute obligation to prevent harm;[71] it is an obligation of conduct 'to deploy adequate means, to exercise best possible efforts, to do the utmost, to obtain this result'.[72] Thus, in order to achieve compliance with their customary duty to prevent significant transboundary harm, States must take regulatory and enforcement measures preventing, limiting or reducing GHG emissions with the aim of preventing transboundary damage. As the present author detailed elsewhere, such mitigation action may or may not result in prevention of significant transboundary harm; the obligation incumbent upon States is to show due diligence in their efforts to avoid it.[73]

Like industrialized countries, developing States, too, must conduct themselves with due diligence in their endeavours to prevent significant transboundary harm. Developing countries would need to take some sort of mitigation action despite the fact that they are not specifically required to do so under the international climate regime. Formulating a national programme containing measures to mitigate climate change, in line with their treaty obligations, cannot be considered sufficiently diligent unless some of those measures are actually implemented.[74]

The degree of due diligence to be exercised by industrialized countries must be different from that expected of developing nations in line with the CBDRRC principle. As suggested by Lefeber, and by the present author elsewhere, the degree of due diligence to be deployed by States needs to correspond to an objective international standard for countries with

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[71] See, e.g., *Case Concerning Pulp Mills on the River Uruguay (Argentina v Uruguay),* Judgment of 20 April 2010, 2010 ICJ Rep. 14, p. 79, para. 197; *Responsibilities and Obligations of States with Respect to Activities in the Area*, Advisory Opinion of 1 February 2011, 2011 ITLOS Rep. 10, paras 110–111; see also A Kiss and D Shelton, *Guide to International Environmental Law* (Leiden: Martinus Nijhoff Publishers 2007), p. 91.

[72] *Responsibilities and Obligations of States with Respect to Activities in the Area*, Advisory Opinion of 1 February 2011, 2011 ITLOS Rep. 10, para. 110.

[73] E Kosolapova, *Interstate Liability for Climate Change-Related Damage* (The Hague: Eleven International Publishing 2013), 148.

[74] See E Kosolapova, *Interstate Liability for Climate Change-Related Damage* (The Hague: Eleven International Publishing 2013), 172; E Kosolapova, 'A Responsibility to Mitigate' (2014) *The Environmental Forum*, Vol. 31, Issue 5, September/October 2014, 30, 33.

an equivalent level of prosperity.[75] This is why the level of due diligence expected of developing countries with a higher level of prosperity cannot be the same as the level of due diligence expected of developing nations with a lower level of prosperity.[76]

Also, due diligence is not a constant and 'may change over time as measures considered sufficiently diligent at a certain moment may become not diligent enough in light [. . .] of new scientific or technological knowledge'.[77] With the release of the Fifth Assessment Report (AR5) by the Intergovernmental Panel on Climate Change (IPCC) in 2013–2014, it has become clear that the level of due diligence required of States to avoid significant transboundary harm associated with GHG emissions is now higher than ever.[78]

Another important aspect of the obligation to prevent significant transboundary harm is that, structurally, it is a composite one. Compliance with the obligation to prevent involves carrying out a variety of (1) procedural and (2) substantive duties.[79] Due diligence requires that a State regulate activities within its jurisdiction or control through various measures, including general ones and those pertaining to a particular activity. It is significant that GHG emissions from a particular source, regardless of whether the installation participates in/complies with an ETS, contribute to global climate change that can affect the entire international community, including the source State.

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[75] See R Lefeber, 'Climate Change and State Responsibility' in R Rayfuse and S Scott (eds), *International Law in the Era of Climate Change* (Cheltenham, UK and Northampton, MA, USA: Edward Elgar 2012), 335; E Kosolapova, *Interstate Liability for Climate Change-Related Damage* (The Hague: Eleven International Publishing 2013), 149; E Kosolapova, 'A Responsibility to Mitigate' (2014) *The Environmental Forum*, Vol. 31, Issue 5, September/October 2014, 30, 33.

[76] Cf: China, India, the Republic of Korea, Brazil, South Africa, OPEC countries on the one hand and, e.g., SIDS and LDCs on the other.

[77] *Responsibilities and Obligations of States with Respect to Activities in the Area*, Advisory Opinion of 1 February 2011, 2011 ITLOS Rep. 10, para. 117.

[78] IPCC AR5, consisting of three Working Group reports and a Synthesis Report, available 15 July 2016 at https://www.ipcc.ch/.

[79] See generally A Kiss and D Shelton, *Guide to International Environmental Law* (Leiden: Martinus Nijhoff Publishers 2007), 91. Procedural duties may include: assessment of the transboundary impact of a particular measure; notification of potentially affected States; exchange of information with the States involved; consultations and negotiations with those States; and the monitoring of transboundary environmental impacts throughout the implementation stage of the relevant measure: see R Lefeber, 'Climate Change and State Responsibility' in R Rayfuse and S Scott (eds), *International Law in the Era of Climate Change* (Cheltenham, UK and Northampton, MA, USA: Edward Elgar 2012), 335.

Therefore, States that are likely to be affected by climate change-related damage have the right to demand that other States comply with their implementation duties as part of meeting their obligation to prevent significant transboundary harm. In 2010, for example, the Federated States of Micronesia (Micronesia) took advantage of this right by submitting a claim through diplomatic channels.

In a letter to the Ministry of the Environment of the Czech Republic dated 4 January 2010, Micronesia's Government expressed concern over the plan for the modernization of the lignite-fired power plant Prunéřov II.[80] In the letter, Micronesia observed that while an EIA of the proposed modernization plan had been made, it did not take into account the transboundary impact of the project, requesting that the Czech Ministry of the Environment issue a negative final statement on the EIA. Micronesia noted that Prunéřov, one of the largest single GHG emitting sources in the world and the largest one in the Czech Republic, made a significant contribution to climate change and that its serious environmental impacts could affect the territory of the Micronesian State. Micronesia emphasized its vulnerability to the dangerous impacts of climate change, stating that Prunéřov could dangerously affect its environment by contributing to the accelerated sea-level rise. On 1 February 2010, the Czech Minister of the Environment decided to submit the project for independent international assessment. The Minister's decision contained no reference to Micronesia's request.[81] The subsequent EIA of the project accepted by the Minister on 29 April 2010 still did not take into account the transboundary impacts of the plan.[82] However, as explained above, the requirement to carry out

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[80] Viewpoint of the Federated States of Micronesia on the complex renovation of Prunéřov II power plant 3x250 MWe plan, letter of 4 January 2010 from the Director of the Office of Environment and Emergency Management of the Federated States of Micronesia to the Ministry of the Environment of the Czech Republic, available 15 July 2016 at http://aktualne.centrum.cz/domaci/fotogalerie/2010/01/06/dokument-dopis-mikronesie-k-planu-na-rekonstrukci-/foto/287153/?cid=657469.

[81] Ministry of the Environment of the Czech Republic press release of 1 February 2010, available 15 July 2016 at www.mzp.cz/en/news_100126.

[82] Ministry of the Environment of the Czech Republic press release of 30 April 2010, available 15 July 2016 at www.mzp.cz/en/news_100430_statement_Prunerov. This happened after the customary status of the requirement to make an EIA 'where there is a risk that the proposed industrial activity may have a significant adverse impact in a transboundary context' was confirmed by the International Court of Justice (ICJ) in *Case Concerning Pulp Mills on the River Uruguay (Argentina v Uruguay)*, Judgment of 20 April 2010, 2010 ICJ Rep. 14, p. 82, para. 204. The same finding was made by the International Tribunal for the Law of the Sea (ITLOS) a year later in *Responsibilities and Obligations of States*

an EIA in a transboundary context – and, in the case of GHG emitting sources, there is always a transboundary dimension – is part of customary international law. Without it, the due diligence duty to prevent significant transboundary harm cannot be considered fulfilled.

Substantive duties forming part of the obligation to prevent significant transboundary harm would depend, to a large extent, on the source of damage.[83] In order to prevent significant transboundary harm related to climate change, States must exercise due diligence by taking mitigation action by, first and foremost, reducing their aggregate GHG emissions. While limiting GHG emissions may involve the use of cap-and-trade, ETS alone may or may not result in actual emissions reductions.[84] Therefore, the use of the BAT standard by major single emissions sources must be viewed as a requirement additional to their compliance with ETS regulations.

While, in its challenge of the Prunéřov modernization plan, Micronesia did not dispute the project *per se*, it demanded that the BAT standard be used for the plant's modernization in accordance with EU and Czech laws on net energy efficiency of new power plants.[85] Micronesia questioned whether the renovation of Prunéřov was not in fact a new construction, which would be subject to higher limits of efficiency than reconstruction projects, expressing concern over a 4 percent discrepancy.[86] Ultimately, the EIA approval was subject to a condition aimed at compensating for the use of technology with lower effectiveness than BAT.[87] The compensatory measures that were eventually accepted 'will not only attain, but even

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with Respect to Activities in the Area, Advisory Opinion of 1 February 2011, 2011 ITLOS Rep. 10, para. 145.

[83] See, e.g., R Lefeber, *Transboundary Environmental Interference and the Origin of State Liability* (The Hague: Kluwer Law International 1996), 336.

[84] See *infra* 12.

[85] The modernization plan EIA proposed a 38% net energy efficiency in contrast with a minimum of 42% net energy efficiency for a new power plant required by law.

[86] Viewpoint of the Federated States of Micronesia on the complex renovation of Prunéřov II power plant 3x250 MWe plan, letter of 4 January 2010 from the Director of the Office of Environment and Emergency Management of the Federated States of Micronesia to the Ministry of the Environment of the Czech Republic referencing Directive 2008/1/EC of the European Parliament and of the Council of 15 January 2008 concerning integrated pollution prevention and control, OJ 2008 L 24/8, and Reference Document on Best Available Techniques for Large Combustion Plants.

[87] Ministry of the Environment of the Czech Republic press release of 20 October 2010, available 15 July 2016 at www.mzp.cz/en/news_101020_Prunerov.

exceed by 84%'[88] the required GHG savings.[89] The Minister's decision to subject the Prunéřov modernization project to the requirement to take compensatory measures to counterbalance the use of inferior technology is indicative of the implicit intention of the Czech Republic to exercise due diligence in its efforts to meet the obligation to prevent significant transboundary harm.

Additionally, the obligation to prevent transboundary harm is subject to the requirement of significance; it only requires that States prevent actual or potential transboundary harm that is significant.[90] In the climate change context, significance of harm can be determined on the basis of the internationally agreed global temperature rise benchmark. States have agreed that the global temperature increase of not more than 2°C above preindustrial levels would be tolerable, at the same time recognizing the need to revise the long-term temperature goal, including in relation to 1.5°C.[91] The potential of the obligation to prevent significant transboundary harm could be used to challenge a particular State's failure to take adequate mitigation measures as well as in relation to single major emissions sources in those States' territory or in areas of their jurisdiction or control. Micronesia's challenge to the Prunéřov modernization plan has demonstrated that a single major source of GHG emissions could be challenged for damage it could potentially cause in another State's territory, if only by contribution. Micronesia agreed that Prunéřov's share of the global GHG emissions was 'only' 0.0161 percent and that the plant did not 'directly cause sea-level rise, change weather patterns and increase storms'.[92] It

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[88] Op. cit. 87: Ministry of the Environment of the Czech Republic, press release of 20 October 2010.

[89] Note also that, while the use of technology with lower effectiveness than the BAT in combination with certain compensatory measures may have been justified by the Minister of the Environment with respect to the modernization of an existing plant, the Czech Environmental Ministry did not contest the fact that the approval of new plants construction proposals must be determined by the BAT standard.

[90] See Principles on the Allocation of Loss in the Case of Transboundary Harm Arising out of Hazardous Activities 2006, ILC Report on the work of its 58th session, A/61/10, YILC, vol. II, Part Two, commentary to Principle 2, para. 2.

[91] *Cancun Agreements*: *Outcome of the Work of the Ad Hoc Working Group on Long-Term Cooperative Action under the Convention*, UNFCCC Decision 1/CP.16 (2010), para. 4.

[92] Viewpoint of the Federated States of Micronesia on the complex renovation of Prunéřov II power plant 3x250 MWe plan, letter of 4 January 2010 from the Director of the Office of Environment and Emergency Management of the Federated States of Micronesia to the Ministry of the Environment of the Czech Republic.

emphasized, however, that there were 5,000 lignite-fired power plants worldwide, and each of them accelerated climate change by contributing to total global CO_2 emissions.

Finally, since the obligation to prevent significant transboundary damage is contingent on the occurrence of actual or potential harm, it requires that there be a causal link between GHG emissions and climate change damage.[93] The standard of proof would be different depending on whether the harm alleged is actual or potential as well as on the type of the remedy sought. As far as challenges to single major GHG emitting sources are concerned, a State (potentially) affected by the injurious effects of climate change could allege a breach by the respondent State of certain procedural duties stemming from the obligation to prevent transboundary harm, such as the duty to prepare an EIA of a particular project, the duty to inform the States likely to be affected or the duty to consult and negotiate with the affected States. Micronesia's challenge of Prunéřov's modernization plan has shown that, although a single GHG emissions source cannot be considered to cause sea level rise or increase storms directly, its contribution to the global emissions is sufficient to necessitate a transboundary EIA, without which due diligence cannot be considered to have been exercised. Thus, in order to substantiate breaches of procedural duties, it is sufficient to show that the emissions from a particular source are capable, by way of contribution, of causing significant harm in another State's territory. This conclusion is also supported by numerous decisions from domestic jurisdictions concerning procedural claims: in procedural injury cases, the causal link, both general and specific, has not been a major challenge for plaintiffs.[94]

It must also be pointed out that the fact that Micronesia's claim was submitted through diplomatic channels may be indicative of difficulties associated with launching a claim in a permanent court or an *ad hoc* arbitral tribunal established for the settlement of a particular dispute. In 2002, the island nation of Tuvalu considered suing the US and Australia for their contribution to global warming but later abandoned the idea having weighed the legal and political difficulties associated with pursuing such

[93] R Lefeber, 'Climate Change and State Responsibility' in R Rayfuse and S Scott (eds), *International Law in the Era of Climate Change* (Cheltenham, UK and Northampton, MA, USA: Edward Elgar 2012), 338; R Lefeber, *Transboundary Environmental Interference and the Origin of State Liability* (The Hague: Kluwer Law International, 1996), p. 89; E Kosolapova, *Interstate Liability for Climate Change-Related Damage* (The Hague: Eleven International Publishing 2013), 157.

[94] See section 2 of the present chapter.

a claim.[95] Because of its binding character, international adjudication/arbitration is subject to acceptance of the competent court's jurisdiction by all parties to a dispute.[96] Therefore, in practice, due to the voluntary nature of interstate dispute resolution, future claims are more likely to go through diplomatic channels, which, as demonstrated by Micronesia's success, may be preferable to litigation, which States often view as confrontational.

5. CONCLUSION

While the notion of (interstate) liability for climate change-related damage is not new, the law of state responsibility casts new light on the potential liability of GHG emitting installations that comply with ETS regulations. The present chapter has demonstrated that the protective scope of the obligation to prevent significant transboundary harm can be extended to single major emitting sources, whether or not those are compliant with the requirements of a given ETS. In the transboundary context, compliance with ETS regulations does not free States from their obligation to prevent significant transboundary harm.

It has been shown that, in ETS, the issue of liability is overshadowed by compliance. At the same time, the international climate regime contains no liability provisions. Existing approaches to state liability based on the obligation to pay compensation, the obligation to negotiate a redress settlement, the obligation to ensure prompt, adequate and effective

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95 'Tuvalu threat,' ABC Local Radio, Australia, AM Archive, transcript from 4 March 2002, available 16 July 2016 at www.abc.net.au/am/stories/s495507.htm. See also E Kosolapova, *Interstate Liability for Climate Change-Related Damage* (The Hague: Eleven International Publishing 2013), 2, 129, 180.

96 Next to diplomatic methods of settling a dispute, UNFCCC 1992, Art. 14 and Kyoto Protocol 1997, Art. 19 envisage mandatory recourse to non-binding conciliation and optional recourse to the ICJ and/or arbitration, which is subject to optional declarations to be submitted by Parties to the depositary. Thus far, only Solomon Islands and Tuvalu have submitted, upon ratification, declarations opting for compulsory arbitration and, in 2010, the Kingdom of the Netherlands made a declaration accepting both means of dispute settlement. Parties to a dispute may also recognize the compulsory jurisdiction of the ICJ under Article 36 of its Statute or under any other instrument relating to amicable settlement of disputes; however, only about one-third of all States have accepted the ICJ's jurisdiction under the Statute of the Court. Similarly, the claimant State and the respondent State could choose to submit their dispute to an arbitration by an *ad hoc* tribunal or to a permanent arbitral body, such as the Permanent Court of Arbitration (PCA) in The Hague. Recognition of the arbitral tribunal's jurisdiction by all parties to the dispute is a prerequisite due to the binding nature of its decisions.

compensation, and the obligation to take response action are incapable of accommodating climate change-related damage. It has also been established that obstacles in the pathways to domestic liability are numerous. Given the fact that domestic liability is largely dependent on national regulations, the law of state responsibility has been analysed as a global legal framework for potential accountability for climate change-related harm.[97]

The various aspects of the obligation to prevent significant transboundary harm have been considered, including its scope (applying to developed as well as developing countries) and character (a composite obligation of due diligence subject to the requirement of significance). It has been concluded that breach of the obligation to prevent significant transboundary harm, and the procedural and substantive duties stemming from it in particular, can provide the legal basis for challenging single emissions sources. Admittedly, under the law of state responsibility, such challenges would involve State claimants. These claims may be possible, as has been demonstrated by Micronesia in its landmark challenge, but it remains to be seen whether more States would be willing to go down that avenue.

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[97] For a detailed analysis of the origins, content and implementation of state responsibility of industrialized and developing nations, see E Kosolapova, *Interstate Liability for Climate Change-Related Damage* (The Hague: Eleven International Publishing 2013), 161–85.

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12. Allowance 'surplus' and governance implications

Christian de Perthuis and Raphael Trotignon

1. INTRODUCTION

The case of the European Union Emission Trading Scheme (EU ETS) provides to date the most complete experience of carbon pricing through a quantitative tool, a cap-and-trade program. The launching of this instrument and its functioning during the first trading period (2005–2007) has been analyzed by Ellerman et al., 2010, who considered this experience as a major innovation in the field of climate policies, that could inspire the development of other schemes in the world.

Many carbon pricing initiatives are also taking place out of Europe, especially in Asia with the South Korean project scheduled to start in 2015 and the Chinese pilot cap-and-trade schemes that are being progressively launched since 2013 (see Park and Hong, 2014 and Quemin and Wang, 2014), and in Northern America with California's and Quebec's cap-and-trade schemes (see California Air Resources Board, 2013 and Québec, 2014), as well as the Regional Greenhouse Gas Initiative abbreviated as RGGI (see RGGI, 2013).

The lessons from the European experience could benefit those other schemes. Indeed, since the publication of its first *ex post* evaluation, the EU-ETS has faced new challenges: the unexpected economic recession strongly affected the industries under the cap and contributed to the reduction in their CO_2 emissions; the market was subjected to interactions with other climate-energy policies also reducing CO_2 emissions independently of the permit price; and the large possibility of using offset. Those three factors contributed to reduce the severity of the cap defined for the second trading period which ended with a carbon price collapse and a large number of unused permits. At the current price, between €5 and €10 per ton of CO_2, most observers consider that the EU ETS does not provide the right incentives to reduce emissions both in the short and the long term. This raises the issue of the rules that should govern the market and could provide important lessons for schemes developing in the rest of the world.

Since the end of 2011, the EU ETS is subject to this debate, and many observers attribute the current price levels to the existence of a large allowance 'surplus'. In July 2012, The European Commission made a proposal dedicated to reducing the supply of allowances in the market between 2013 and 2015 (European Commission, 2012a). This so called '*backloading*' proposal took time to be agreed upon because of opposition from some Member States and the European Parliament, and began to be implemented in 2014, two years after the discussions started. In addition, the European Commission published in December 2012 a report on the state of the European carbon market, which outlines options for a more profound structural reform of the EU ETS, beyond the short-term '*backloading*' measure (European Commission, 2012b). Following the publication of this report, the Commission made a formal proposal for the establishment of a 'Market Stability Reserve', a non-discretionary and rule-based system for bringing supply flexibility to the market (see European Commission, 2014b).

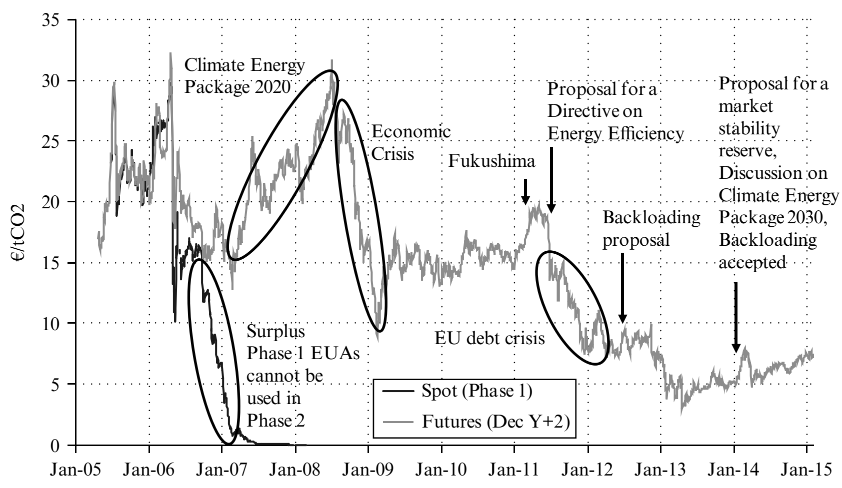
This chapter is an attempt to explain, analyze and contribute to this debate on surplus control, and on the ways supply-flexibility should be brought into the market. By nature, the EU ETS aims at minimizing the cost of reaching a certain predefined emission target. The carbon price has a major role to play, in influencing the decisions of economic players both in the short-term management of their existing assets, and in the longer-term direction of their investments. The economic efficiency of the policy is thus dependent on the EU ETS capacity to establish rules that will modify the short-term behavior of agents as well as their investment decisions, which requires changing their medium- to long-term anticipations. In the current EU ETS framework, the major implication of the choice of quantitative instruments is that the price associated with carbon emissions will not be explicitly fixed by the public authorities but will be revealed by the market. It will reflect the current and anticipated scarcity of emission allowances, so that the economic efficiency relies not on a normative desirable price level but on actors' anticipations of the medium- to long-term emission constraint, and especially how these expectations evolve over time. The notion of an allowance surplus driving down the price does not seem to be entirely satisfactory by itself, because it refers to the idea of a static stock of worthless allowances, when the right question is that of the dynamic value of this stock in a context of uncertainties and imperfect anticipations. This chapter is thus an opportunity to explore measures such as the '*backloading*' or the 'structural reform' proposed by the European Commission, but also other options not yet discussed such as setting up an Independent Carbon Market Authority (ICMA) to reanimate the European carbon market.

In section 2 we identify the three major causes of the current EU ETS' weaknesses, and distinguish among these the economic influences (energy prices, economic growth and so on) from the effects of other structural settings (interactions with other policies, changes in market rules and so on). In the third section, we analyze the key role of anticipations in a cap-and-trade program by comparing past expectations to actual EU ETS developments. In section 4 we examine the options for structural reform made by the European Commission. Section 5 tries to build on these lessons and explores the possibility of improving the current governance framework with the creation of an ICMA, whose mandate could allow participants to build sounder expectations over time. The last section concludes in trying to outline the general lessons that can be drawn from the EU-ETS case which can be useful for the design of any cap-and-trade program.

2. EU ETS: THE THREE CAUSES OF CURRENT MARKET WEAKNESSES

Observers often attribute the current weakness of the EU ETS to the economic crisis that strongly affected industrial output and induced a 'surplus' of allowances. We will demonstrate that this rationale is incomplete and does not allow the drawing of the correct lessons from the functioning of the market and thus the proposal of adequate recommendations.

There are three main causes for the current malfunctioning of the market (see the observed EU allowance (EUA) price on Figure 12.1 below). The



Source: authors from ICE ECX and BlueNext

Figure 12.1 Observed EUA price since 2005 first is effectively the unforeseen decline in industrial activity since the 2008 crisis, and future economic growth perceived as unfavorable. The second is the high use of carbon offsets over a short period of time resulting from the unforeseen evolution of the international Kyoto system (namely that the EU ETS would end up being the only important source of demand for offsets and that the supply of such credits would surpass the demand from ETS installations) in conjunction with the time-flexibility left to participants for using offsets. The third reason is the interaction between the EU ETS and other energy and climate policies, mainly renewable energy and energy efficiency policies that can drive emissions down independently of the EUA price. Even though the 2020 renewable and ETS targets were jointly decided and assessed ex ante, their practical ex post effects on one another have to be accounted for by market participants.

2.1 The Influence of Economic Conditions

The first cause of market disequilibrium is macro-economic conditions, which had a strong influence on the change of expectation occurring over Phase 2, in the short term (production decrease) as well as in the longer term (degraded growth outlooks). Between 2008 and 2009, the production levels of the covered sectors dropped on average by 10 percent (Eurostat, 2012), with stronger decreases in industrial sectors like cement and steel. But ultimately those influences of economic conditions on the price are desirable. Part of the economic efficiency of a cap-and-trade scheme comes from this flexibility that makes the price lower if economic conditions degrade, the cap remaining unchanged.

2.2 Uncontrolled Interactions with Other Climate Energy Policies

On top of this desirable influence, the system suffered from undesirable weaknesses that came for structural reasons. The effects on the market of other climate and energy policies (energy efficiency, renewable energy) and the unforeseen use of carbon credits over time, resulted in a strongly decreased demand for EUAs in the market in the short term, as well as blurred anticipations in the long term.

Weigt et al., 2012 evaluate the effect of renewable energy support in Germany to be responsible for a reduction of 10 to 16 percent in the German electricity sector's emissions. In the same way, energy efficiency policies can reduce the demand for electricity generated by EU ETS covered sectors, thus implying emission reductions independently of the carbon price. If those structural weaknesses are not controlled in some way, this process of increasing interaction will automatically lead to the marginalization of the ETS, because the emission base of the system will be eroded by other policies. The fact that both the environmental and economic effectiveness of cap-and-trade programs can be significantly compromised by interactions with other regulations is crucial, and has been pointed out as a key element for the implementation of cap-and-trade programs (for example, by Goulder, 2013). Even if, as is the case, both renewable and GHG targets were jointly formulated and assessed, it is much harder for participants to make sound expectations for the future in a context of uncontrolled policy super-imposition, because no adjustment would take place in the carbon market if one complementary policy over-achieves or under-achieves, or if consequences initially unaccounted for in the impact assessment change the emission constraint associated with the cap in the EU ETS.

Of course, a cap-and-trade scheme alone cannot do everything by itself, and other targeted policies are probably needed to support specific goals, which will have an impact on EU ETS emissions, for example, low carbon innovation in general and the promotion of advanced renewable technologies in particular. As a consequence, there will be policy interactions between the EU ETS and other policies, not just European climate and energy policies, but also unilateral national policies. The United Kingdom's tax on electricity sectors emissions is a good example (see United Kingdom's HM Revenue and Customs, 2013). If such measures are taken individually by Member States, the economic efficiency of the EU ETS will suffer from it, because the advantage of having a uniform CO_2 price falls when individual countries or sectors 'force' a carbon price that is higher than the market price.

2.3 Unexpected Evolution of the Carbon Offsets' Market

The third market weakness is related to the use of offsets. There is only one cap that matters in the end, which is the total domestic cap plus the allowed offsets over the period. The rules for using offsets in the EU ETS fixed the amount that could be used over 2008–2012 to approximately 1,400 Mt. This limit was then extended to around 1,600 Mt over 2008–2020 when the Climate Energy Package was voted on (see European Parliament and the Council of the EU, 2009). This provision leaves most participants free to decide the timing at which offsets will be used (the right to use offsets can be transferred to later years[1]). Between 2008 and 2009, around 80 million

[1] Although the limit of 1,450 Mt of emissions is set over the relevant phase, Member States can individually decide to establish annual limits of use. Limits can

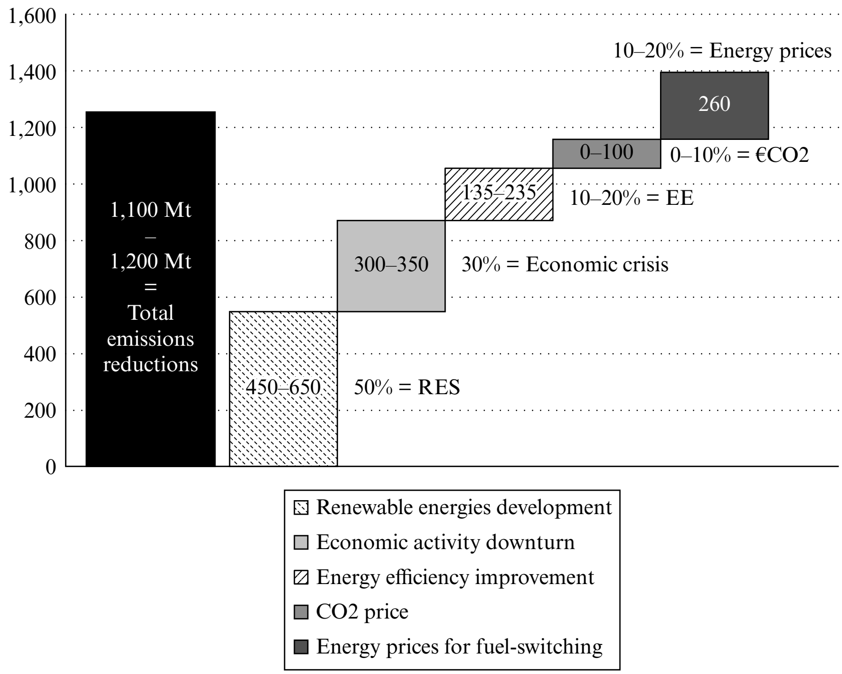
offsets per year have been used in the EU ETS (see Trotignon, 2012a). But in 2010, the European Commission announced qualitative restrictions on certain offset types that represented the majority of existing offsets, stating that the restriction would apply only from 2013 onward (see Hedegaard, 2010). As a consequence and by anticipation of the future restriction, the use of those offsets surged over the rest of Phase 2 to represent a cumulated amount of around 900 Mt over five years. The price of those largely available offsets dropped to less than €1/t, allowing participants to comply with the ETS constraint at very low cost, because of the unforeseen evolution of the Kyoto trading system. It was first anticipated that Europe would not be the only buyer of offsets, but no other large scale source of demand eventually emerged.

The lesson is that if the domestic cap is unchanged but the authorized use of offsets over time is changed, this is strictly equivalent to changing the cap. If the public authority leaves too much flexibility for using offsets, then the anticipations of the future constraint over time can be blurred, and the public authority can lose part of its sovereignty in deciding the reduction effort that will be effective domestically over a certain period. In proposals for cap-and-trade programs outside of Europe, this uncertainty has been accounted for by measures such as conversion rates between offsets and allowances, or price threshold above which more offsets become allowed in the system (for example this option was implemented in RGGI but has never been triggered because of the low price level; it was replaced in 2014 by a cost control mechanism deemed more transparent: a cost control reserve (see RGGI, 2013)).

2.4 Consequences on CO_2 Emissions and Price

The three factors analyzed above are reducing the demand for carbon allowances, reductions that to a large extent are not caused by the carbon price. Quantifying the emission reductions related to the economic crisis and the interactions with other policies is a difficult task, because it requires constructing a counterfactual scenario in which (a) the economic crisis did not happen and (b) policy interactions did not occur. Only then can we calculate the difference between the observed emissions in the presence of the economic crisis and policy interactions and what the emissions

also differ across sectors for each country. For example, although the limit in the UK is set annually, UK installations may bank any unused limit for the next year; moreover, the percentage allowed for large electricity producers is slightly higher than other sectors. The reader can refer to individual National Allocation Plans for Phase 2 for more details, and to Trotignon, 2012a.



Source: Gloaguen and Alberola (2013)

Figure 12.2 The source of emission reductions compared to counterfactual over 2005–2012

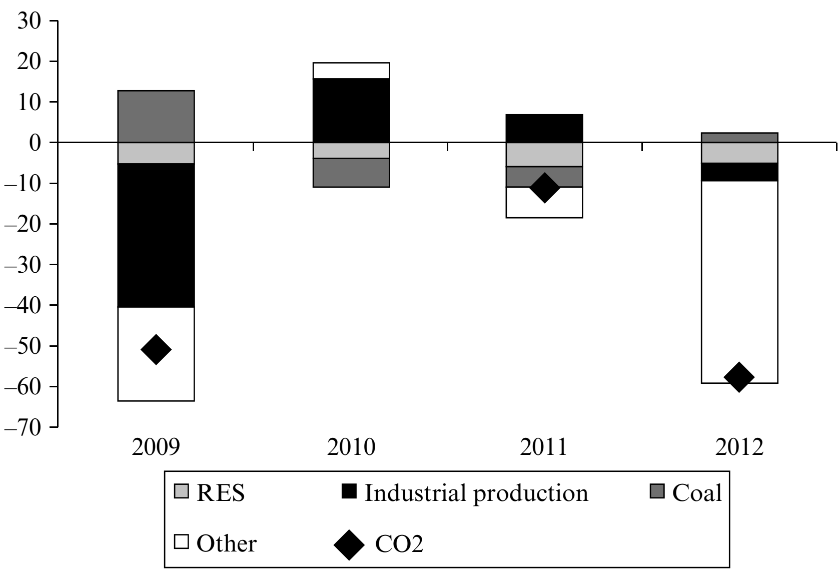
would have been in their absence. Many uncertainties are involved, nevertheless some econometric studies have been done to try and quantify those reductions, such as that of Gloaguen and Alberola (2013).

Figure 12.2 represents a decomposition by source of emission reductions in the EU ETS, over the period from 2005 to 2012, compared to a 'business as usual' counterfactual scenario. Those results suggest that other climate and energy policies are the largest cause of emission reductions over that period, with 60 percent of total reductions. The economic crisis, often presented as the main source of emission reductions over this period, comes second and represents 30 percent of total reductions. According to the calculations of the authors, the carbon price was only responsible for around 10 percent of the reductions observed, which shows that the causes identified above are not negligible at all, and tend to substitute the carbon price instead of supplementing it.

Over the same period, more than 1 $GtCO_2$ of carbon offsets have been used, which correspond to emission reductions outside of the scheme, but should be counted as additional supply in the EU ETS. As explained earlier, those carbon offsets (also called credits) are available on top of EU allowances to fulfill the industries' compliance needs. We thus see that the supply-demand equilibrium in the EU ETS has been impacted significantly by the three causes described in this section. Because of this reduced demand and due to the intense use of carbon offsets, around 2.5 billion carbon allowances still remained unused in 2014.

Other studies have focused on the impact of those factors on the carbon price (again using econometric tools). The most thorough assessment is that of the European Commission in its Energy Economic Development in Europe report (see European Commission, 2014a). The main results are presented in Figure 12.3.

This decomposition analysis also sheds light on the impact of the economic recession on the price change especially in 2009. The renewables seem to have had less of an impact in terms of price. The most striking information in this graph is the large share of unidentified causes of price changes (lightest grey areas). This seems to show that the currently low price levels are not directly attributable to the surplus allowances freed up because of the economic crisis and renewable policy interactions. Other important factors must be taken into account, and saying that the root issue



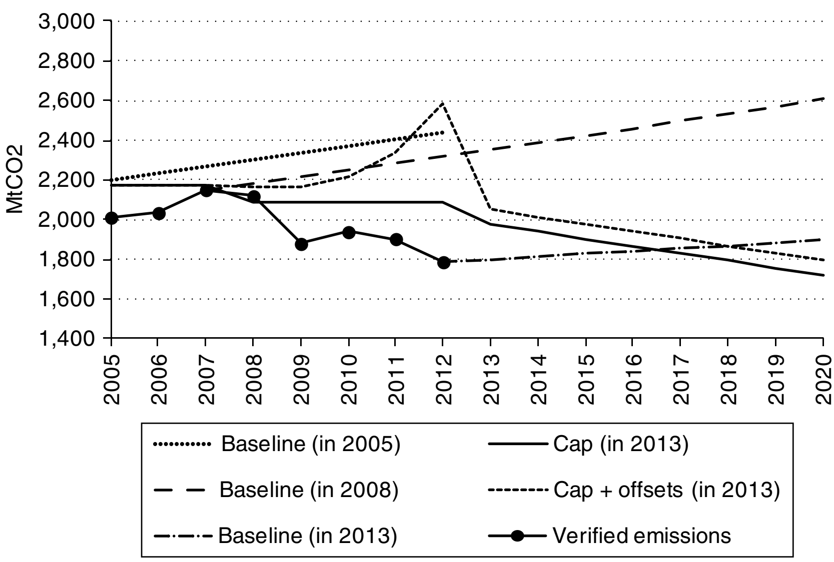
Source: European Commission, 2014a

Figure 12.3 Decomposition of carbon price changes over 2008–2012 in the EU ETS is the existence of a large allowance surplus seems to be largely incomplete. In the next section, we will explain why this is the case.

3. THE IMPORTANCE OF EXPECTATIONS IN A CONTEXT OF FUTURE UNCERTAINTY

In a cap-and-trade scheme with unlimited banking (participants can hold unused allowances for a later use) the allowance price depends on current allowance scarcity, but also on the anticipated scarcity of emission allowances. This anticipated constraint is very important for determining the price level. As a matter of fact, because allowances can be kept during decades without degrading and at almost no cost-of-carry, their current value is determined by their future probable value, which can be very high if the future scarcity is expected to be important, and most importantly, even if the quantity of currently unused allowances is large.

This perceived constraint of the cap can be measured ex ante as the difference between business-as-usual emissions of covered sectors and the allowance cap itself, over the same period. Figure 12.4 represents,



Source: authors and CITL, 2013

Figure 12.4 EU ETS ex-ante anticipations compared to ex-post observations in the case of the EU ETS, how those expectations evolved between the beginning of Phase 1 (in 2005), the beginning of Phase 2 (2008), and the beginning of Phase 3 (in 2013). For these calculations, we assume an elasticity of baseline emissions to growth of 0.5, and GDP scenarios of 3%/yr (in 2005 and 2008) and 1.5%/yr in 2013 (Trotignon, 2012b).

The EU ETS commenced in 2005, Phase 1 (2005–2007) being a trial period. At that time, very little information was available on the emissions of covered installations, as well as on the probable scarcity of allowances, and it was not yet clear if banking into Phase 2 was going to be allowed in some Member States or not. Accounting for a sustained economic growth and positing emission levels around the cap in 2005, a certain scarcity of EUAs was anticipated.

It eventually turned out that verified emissions for the year 2005 were lower than initially expected, and the market price immediately integrated this new information at the time of publication in April 2006. The carbon price progressively dropped to zero in 2007 as it became clear that the quantity of allowances was sufficient to cover verified emissions over the period, and that banking between Phase 1 and Phase 2 had been definitely forbidden (that is, Phase 1 allowances could not be carried over and used in Phase 2 of the scheme).

As the first year of Phase 2, 2008 is marked by the vote of the European Climate Energy Package, a set of directives and regulations aiming to reach the 2020 objectives (targets for greenhouse gas emissions, renewable energy, and energy efficiency). In particular, the rules for the third period of the EU ETS (2013–2020) were determined by taking into account the emission reduction target announced in the conclusions of Brussels European Council 8/9 March 2007 and in Directive 2009/29/EC of the European Parliament and of the Council of 23 April 2009 amending Directive 2003/87/EC. A calculation accounting for sustained economic growth and an equally spread use of offsets over time would show a large anticipated deficit of allowances until 2020. Most price forecasts at the time were counting on an EUA price in 2012 of around €35/tCO_2 (see Trotignon, 2012b).

Again, expectations did not materialize and in 2012 the carbon price had dropped below €10/tCO_2. The unforeseen financial crisis and the degraded growth outlooks are indeed responsible for part of this change in anticipations, but there are, as we shall see in the next section, other reasons which are at least as important.

Figure 12.4 also incorporates anticipations to 2020 as they were at the beginning of Phase 3 in 2013. The ability to bank unused allowances, more than 2,000 Mt according to European Commission (2012b), appears to allow for very little reduction effort to 2020, in the current context of low economic growth.

The first lesson we draw from this analysis is that the anticipations made in 2005 and in 2008 have turned out to be wrong. One can observe a strong tendency for participants and public authorities to overestimate the emissions constraint ex ante. One important aspect is to sufficiently account for the effects of other instruments of public policy that are put into place on top of emissions trading. As a matter of fact, this is a lesson that is not specific to the EU ETS. It has also been the case for the US SO_2 trading program, where unanticipated cost savings have been obtained due to, for example, the deregulation of railroad rates, allowing for more low-sulphur coal substitution than expected (see Schmalensee and Stavins, 2013). In the Regional Greenhouse Gas Initiative (RGGI), emissions were lower than previously anticipated due to low natural gas prices prompting a conversion to the lower-emitting fuel, and to a lesser degree energy conservation policies and the economic downturn. This led to a revision of the system with a view to tightening the cap (see RGGI, 2013). This phenomenon also took place in the Kyoto Protocol's government to government emissions trading system, which turned out to be much less constraining than initially anticipated (see for example Aldy and Stavins, 2010).

These behavioral lessons highlight that there seems to exist a general tendency for any authority implementing an emission trading system (and market actors participating to this system) to overestimate the constraint ex ante. The interesting challenge arising is that of the capacity for the public authority to establish a coherence between the short-term and the longer-term constraint that will be robust over time, in this context of uncertainty.

4. THE COMMISSION'S PROPOSALS: OPTIONS FOR EU ETS REFORM

The European Commission took two parallel tracks to try to solve the current weaknesses. The first is a short-term action, called *backloading* (see European Commission, 2012a). This measure consists of delaying the auctioning of 900Mt taken from the 2014–2016 allowance cap, which would be injected back in the market through the 2019–2020 auctions. In that way the overall cap over Phase 3 would not be changed but the timing of auctions would shift volumes towards the end of the period. After a complicated legislative process, *backloading* was finally voted in and began to be implemented in 2014.

The second measure was to launch discussions on the 'structural reform' of the EU ETS, following the publication of the Commission's report on the state of the European carbon market (European Commission, 2012b).

This report proposed six different options for extending or strengthening the system, listed hereafter:

- Option a: Increasing the EU reduction target to 30 percent in 2020
- Option b: Retiring a number of allowances in phase 3
- Option c: Early revision of the annual linear reduction factor
- Option d: Extension of the scope of the EU ETS to other sectors
- Option e: Use access to international credits
- Option f: Discretionary price management mechanisms

Following the consultation of stakeholders, the Commission decided to add another option to the list, and a few months later made a formal proposal for the establishment of a Market Stability Reserve (MSR) as a structural provision in the EU ETS rules. The aim of the Commission was to propose a non-discretionary (that is, automatic) and quantity based (without explicit price triggers) system that would allow the progressive removal of some allowances from auctions to put them in a reserve. In case of future tensions in terms of supply on the market, the reserve could then be emptied. The mechanics of this proposal are relatively complex (see European Commission, 2014b for more information). One can remember that this market stability reserve is a sort of automatic *back-loading* machine, getting filled when the number of unused allowances is higher than a certain threshold, and getting emptied when it is below another.

However, in the light of the issues discussed in the previous sections, none of the routes proposed by the Commission seems completely satisfactory, because the question of market governance remains a taboo that is not explicitly addressed. The market stability reserve could also induce further risks of perturbing the orderly functioning of the market, mainly because of its automatic nature, its reaction delay, and the rigidity of its parameters (see Trotignon et al., 2014).

Even in the event of the adoption of a clearer long-term reduction target, retaining the current governance would, however, leave a quite rigid system unable to adapt to shocks which are unpredictable today but are certain to occur between now and 2030. In particular, the Market Stability Reserve does not allow the authority to quickly react to changes in market dynamics because of its two-year reaction delay (by construction the figures for the total number of allowances in circulation, which serves as a basis for intervention by the reserve, are only known with a delay of one and a half years (see Trotignon et al., 2014)). It is hard to imagine how to properly regulate a market without being able to react to short-term change in dynamics. In the next section, we propose exploring an alternative route to the options currently on the table, in which an independent carbon market authority would be established.

5. THE CASE FOR AN INDEPENDENT CARBON MARKET AUTHORITY (ICMA)

A cap-and-trade program is fundamentally an instrument of public policy, consequently it will not be revived unless there is a strong political involvement, especially in determining its long-term emission reduction target. The negotiation of a Climate Energy Package for 2030 was concluded in 2014 and led to the adoption of a −40% reduction target in 2030 compared to 1990, see the conclusions of the European Council of 23/24 October 2014. This decision on a longer-term reduction target is an important prerequisite to the propositions of governance improvements detailed hereafter.

The experience from eight years of EU ETS market history previously analyzed shows that the current governance framework does not enable participants to shape sound expectations over time. Over the long term, the most inconvenient influences are not those of economic conditions but those induced by structural weaknesses linked to climate-energy policies overlaps and to potential unforeseen consequences of international linking (offsets, linking with other cap-and-trade systems). Dealing with those two uncertainties requires a more flexible intervention framework than the one available today. It would be inefficient for Europe to engage in years-long debates such as the *backloading* negotiations every time something unexpected happens.

The recovery of the market calls for strong political support at a European level (not only the setting up of a longer-term reduction target, which is now agreed upon, but also a broader involvement in supporting EU ETS' position as the European cornerstone for low-carbon policies) and a commitment to reform its governance, involving the establishment of a predictable and dedicated intervention framework. This mandate could be entrusted to an ICMA, which would ensure the consistency and credibility of the allowances system in the short- to long-term through the dynamic management of the supply of allowances. This framework is inspired by the example of monetary policies, with which emission trading has many similarities, as shown by Whitesell, 2012. In particular, Whitesell underlines that in both systems the public authority tends to be naturally subjugated by short-term market conditions and is less inclined to ensure the credibility of the long-term target over time. A decision 'on paper' for a longer-term target is necessary but not sufficient: a political process, economic and social policies, evaluation and actions must accompany and give credibility to these commitments on a daily basis.

5.1 Possible Mandate for the Independent Carbon Market Authority

In our proposal, the role of the political authority remains unchanged: namely, to define detailed policy objectives for emission reduction at a European and national level; and to select a range of public policy instruments to achieve these objectives.ICMA's mandate (detailed in Table 12.1), however, is to maintain the credibility and political ambition of that policy over time by a dynamic management of allowances supply, from the short term to the long term.

Table 12.1 Mandate of the Independent Carbon Market Authority

Function	Associated action
Regular monitoring and transparency of information	Collecting, analyzing and sharing information on: ● Transactions on the ETS market ● Emission trajectories ● Compliance behaviour ● Low-carbon investment ● Effects on competitiveness Motivating and justifying its decisions.
Liquidity and good functioning of the market in the short term	Primary market: time management of allowances auctions. No need for intervention in the secondary market.
Credibility over time of the medium-to-long-term constraint	The public authority determines the detailed emissions reduction objectives and the policy instruments to achieve these objectives. The independent carbon market authority implements this policy objective in the sectors covered and can dynamically adjust the allowances cap in two cases: ● To maintain consistency with other climate and energy policy instruments ● To control interactions with carbon credits and international allowances. No need for a price corridor or cost control reserve (see sub-section 5.2).
Reporting and compliance with the mandate	Periodic hearings by the European Parliament and the European Council. Frequent public reporting.

Source: Authors.

In the short term, it would be a matter of being able to adjust the timing of the auctions so as to ensure proper functioning and liquidity in the carbon market. In the medium and long term, it would be a matter of being able to adjust the emissions cap in order to control interactions with other climate and energy policies and with international carbon credits.

To motivate and justify its actions, the independent authority should implement a fair and transparent monitoring of the system (monitoring of transactions, compliance behavior, low-carbon investment, emission trajectories, effects on competitiveness). It should also report regularly and publicly on its actions to the Council and the European Parliament.

At an institutional level, the mandate of this authority could either be assigned to a new agency, or the powers of the existing national energy markets authorities or of the Agency for the Cooperation of Energy Regulator (ACER) could be extended.

In practical terms, it may be wondered how such an authority would have reacted to the recent market malfunctioning. In the short term, the question of *backloading* would no longer arise because of the mandate given by the European Parliament and the Council to the Independent Carbon Market Authority for the dynamic management of auctions. Faced with the three previously identified causes for the fall in the market price, the independent carbon market authority would probably not have made any changes to the cap following the economic recession (in view of the normal and desirable adjustment of the equilibrium price after an economic shock and in spite of the fact that many market participants advocate that adjustment mechanisms should respond to change in output). It would, however, have investigated the impact of changes in the functioning of the international Kyoto credit market and the impact of other Climate and Energy Package directives, with a view to tightening the cap. This tightening would involve returning to the constraint level initially assigned by the public authority to the sectors covered.

This proposal of an ICMA has many similarities with other innovative governance frameworks established in other schemes. For example, the UK has a statutory body called the Committee on Climate Change, established under the Climate Change Act 2008 (see UK, 2008), which makes advice and recommendations on the level of the 2050 target and the connection with UK carbon budgets, reports on the progress of carbon policies, and provides advice or other assistance on request. Australia has a statutory body called the Climate Change Authority, established under the Climate Change Authority Act, 2011 and Clean Energy Act, 2011. The Climate Change Authority was set up to provide independent advice on climate change mitigation initiatives (the operation of Australia's carbon pricing policies, emission reduction targets, caps and trajectories and so on). Under Australian legislation, the responsible Minister, when setting caps on emissions under Australia's emissions trading scheme, must consider a review into caps and targets by the Climate Change Authority.

One difference between the ICMA proposal and other frameworks such as the Climate Change Authority is that the ICMA engages in monitoring of transactions and compliance behavior; while in Australia, these activities are performed by the Clean Energy Regulator (which administers the Carbon Pricing Mechanism) and the Australian Securities and Investments Commission (which oversees financial markets). The second and most important feature is that, neither in Australia nor in the UK, the independent authority has a mandate to adapt the cap directly and quickly; they can only advise and recommend that a correction be made.

5.2 Is There a Need for a Price Floor or a Price Collar?

Cost control has emerged as a major point of contention in many countries where carbon pricing schemes have been considered. The basic concepts and policy options for cost management are very well described in the literature; see for example Tatsutani and Pizer, 2008, Murray, Newell and Pizer, 2009 or Fankhauser, Hepburn and Park, 2010. The different sources and temporal dimensions of cost uncertainty can be explored, along with possible mechanisms for addressing short- and long-term cost concerns, including banking and borrowing, emissions offsets, a price cap (or safety valve), quantity-limited allowance reserve, and the concept of an oversight entity for GHG allowance markets similar to our proposal of ICMA. Recognizing that the inherent trade-off between environmental certainty (the emissions cap) and cost certainty (the emissions price) has no perfect solution, the literature nonetheless indicates that numerous options exist for striking a reasonable and politically viable balance between these two objectives. As such, 'hybrid' schemes combining a market based approach as well as a price floor and price ceilings (price collars or dynamic reserves) are attractive and used particularly in Northern America (see California Air Resources Board, 2013, Québec, 2014 and RGGI, 2013). This attractiveness is due to the effects of such limits, namely a minimum return on emission reduction investments due to the price floor, and a control of the risk of price spikes in the case of price ceilings; and despite the fact that they may undermine the overall efficiency of the policy. But in Europe the question is less straightforward because discussing such explicit price targets or triggers at the European level is politically highly contentious.

In our view, ICMA's means of action should be based on quantitative instruments, and there is no explicit need to introduce a long-term price floor or a price collar as it is the case for example in California's cap-and-trade program. But the governments could decide to increase the visibility of the carbon price signal by introducing such price targets. Would these decisions solve the problem and make the creation of an ICMA unnecessary?

If governments want to give a long-term signal with explicit price target trajectories in the medium- and long-term, the only practical way to implement it would be to introduce a dynamic supply management of allowances to adapt the quantitative parameters of the market (diminishing allowance supply as long as the price is below the price floor, or raising supply as long as the price is above the ceiling price). This requires a change in the way the market is managed today. In other words, an ICMA does not require explicit price targets. But if these price targets have to be introduced, the only way to manage the co-existence of price targets with a quantitative system like the EU ETS would be to establish an ICMA and to add to its mandate these additional provisions related to price levels.

6. CONCLUSION

The historical development of cap-and-trade programs reveals a strong tendency for public authorities and market participants to over-estimate the emissions constraint ex ante, fearing high allowance prices, which leads to the implementation of flexibility measures and additional policies aiming at containing the costs of the constraint. What is observed *ex post* is very different from initial expectations, with prices generally much lower than expected. The notion of an allowance surplus driving down the price does not seem to be entirely satisfactory by itself because it refers to the idea of a static stock of worthless allowances, when the right question is that of the future value of this stock in a context of uncertainties and imperfect anticipations. The key point is that the public authority and market participants will never know and anticipate perfectly in advance the future developments that will determine the actual emissions constraint.

It is thus very hard for the public authority to ensure the predictability of the constraint in a context which is very uncertain by nature. This awkward situation requires a governance framework that can express very clearly the medium- to long-term targets of the policy, and at the same time has the capacity to act and react in the short term to unanticipated situations.

One of the ways to reconcile both requirements is to have the public authority determine the long-term goals and the policy mix allowing for reaching these goals, while entrusting an independent authority with the means to maintain this constraint over time as a function of the uncertainties. The job of the ICMA would be to give credibility and robustness over time to the reduction constraint set by the public authority. There are three pillars for such a framework to be effective: the existence of a precise mandate that determines the independence of the ICMA, the level of expertise of the ICMA, and the reporting and accountability rules of the ICMA.

In the short term, complicated and time-consuming processes such as that of *backloading* would not be necessary anymore because of ICMA's mandate on the timing of auctions. In the longer term, the ICMA would also have the mandate to adapt the ETS cap, not in reaction to a change in economic conditions, but when unexpected events such as policy instruments overlap[2] would require an intervention to maintain the credibility of the scheme to reach both short- and long-term goals of greenhouse gas emission reductions.

This chapter provides important lessons for schemes developing outside of Europe, which can be derived from the current European climate and energy policy framework. Indeed, there is a high chance that the carbon pricing initiatives considered in South Korea and developing in China will display a high degree of interaction with other policy instruments as well, such as energy efficiency and renewable energy policies, and probably also with domestic or international offsets. The governance of the carbon market and the way that price/quantity flexibility is given, while preserving the incentive to reduce emissions from the short to the long term, is a crucial point that should be taken into account in the design of those climate policies.

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[2] Instruments may be designed to interact with each other, in which case they are accompanied by a regulatory impact analysis. This type of work can highlight ex ante the risks associated to interactions, but it does not allow one tool or the other to be adapted ex post to these interactions, which may be far different from the ones accounted for in the impact assessment.

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PART III

INTERNATIONAL DIMENSION

13. Linking emission trading schemes: concepts, experiences and outlook

Andreas Tuerk and Andrej F. Gubina

1. INTRODUCTION

The European Commission successfully implemented the first phase of the European Emissions Trading Scheme (EU ETS) from 2005–2008 with the objective for the scheme to serve as a possible nucleus of a global carbon market that would complement a possible new global climate regime in a bottom-up manner (EC 2009). Through its activities within the Organisation for Economic Co-operation and Development (OECD), the main priority of the European Commission was to establish a transatlantic link between the EU ETS and a federal US scheme as basis of a OECD-wide trading system envisioned for 2015 (EC 2009). A combined EU-US market would have covered a major part of OECD emissions, and could have thus constituted the backbone for the future international emissions trading regime (Tuerk et al., 2009). The introduction of an Australian ETS, a mandatory federal Japanese ETS and a federal US scheme have, however, failed so far, and it remains to be seen how fast the US Clean Power Plan presented in July 2015 (US EPA, 2015a) that paves the way for a federal emissions trading scheme will be implemented. While the vision of an OECD-wide carbon market remains distant, regional initiatives in the US and in Japan have emerged in recent years. At the same time a strong dynamic to establish emissions trading schemes is becoming visible in emerging countries, in particular in Asia, with new trading systems under discussion or already evolving in China, South Korea, Thailand and Vietnam. In Central and Latin America, the discussions on introducing carbon trading are just beginning, such as in Mexico or Brazil.

The aim of this chapter is to give an overview of the motivations to link trading schemes, the forms of linking, the challenges to link trading schemes and an outlook on the role, options and likelihood for linking trading schemes in the upcoming years.

2. RATIONALES FOR LINKING OF ETS

There are several main motivations to link emissions trading schemes; they are of economic and also of political nature. From an economic viewpoint the countries aim to link emissions trading schemes in order to increase the cost efficiency of meeting their greenhouse gas emissions targets by achieving greater heterogeneity of abatement costs, while also reducing competitiveness distortions and the threat of carbon leakage arising from different carbon price levels (Tuerk et al., 2009; Anger et al., 2006; Jaffe and Stavins, 2007, Jaffe et al., 2009, Tiche et al., 2014). Smaller schemes may aim to increase liquidity by linking, and a joint carbon market is expected to reduce the overall chances of market abuse by the dominant players (Jaffe and Stavins, 2007). Besides economic arguments, linking is often motivated by political reasons, and represents an instrument of international cooperation to mitigate climate change (Flachsland et al., 2009a). In the USA for example the establishment and linking of the regional emission trading schemes is hoped to ultimately lead to the emergence of a national trading scheme. The expansion of the EU ETS to Switzerland and possibly to other EU neighbour countries is also motivated by the aim of the EU to align its energy and climate policy towards its neighbour countries and to expand a system that it considers to be a success and wants to maintain. A linking agreement between emission trading systems can also make the ETS politically more sustainable, locking-in particular targets which in turn could render the policy target more credible (Flachsland et al., 2009b, Brunner et al., 2012).

But the importance the EU gave to linking national emissions trading schemes was also motivated by the uncertainty regarding the nature of a new global climate agreement and to what extent it would follow the Kyoto top-down style approach. A *top-down* approach would involve global coordination, be centred on the pursuit of a common objective and on comparable efforts. It would be implemented through targets and timetables based on commonly agreed rules, e.g. on offsets, with a strong measuring, reporting and verification (MRV) system and a compliance mechanism (Hare et al., 2010). A common international standard for trading units would facilitate international linking of ETS (Tuerk et al, 2009). A top-down approach could include governmental emission trading as was the case under the Kyoto Protocol. In order to increase cost efficiency, the countries with emissions targets (Annex-I countries) could trade governmental emission reduction units of the Kyoto Protocol, the so-called Assigned Amount Units (AAUs), representing one ton of greenhouse gas emissions reduced as verified under the rules of the United Nations Framework Convention on Climate Change (UNFCCC). A *bottom-up* approach would focus first upon the domestic development of company-level cap-and-trade systems, and then consider whether and how such systems in different jurisdictions may be linked (Carbon Trust, 2009). A network of linkages between emissions trading schemes coupled with unilateral emission reduction commitments could be fostering mutual trust and constitute a stepping stone towards an international climate change alliance (Tiche et al., 2014). A linked bottom-up carbon market could complement a global top down climate agreement, at the same time serving as a second-best solution in case no global top-down agreement should emerge. In the context of the new international climate agreement that was agreed upon in Paris in December 2015, the concept of 'Intended Nationally Determined Contributions' (INDCs), an approach that was introduced into the negotiation process at COP19 in Warsaw in 2013 reflected the bottom up nature of the new climate agreement (see UK HC, 2015).

3. FORMS AND IMPLICATIONS OF LINKING OF TRADING SCHEMES

3.1 Forms of Linking

Conceptually, a link between two emission schemes can be either direct or indirect, with direct linking based on an explicit decision by at least one of the linked jurisdictions to accept the emission reduction units from another jurisdiction (Tuerk et al., 2009; Jaffe and Stavins, 2007). Direct links allow trade between different schemes and can be distinguished by whether they allow trading in one or more directions. Under a unilateral link, entities in system A can purchase and use allowances from system B for compliance, but not vice versa (Sterk et al., 2006). Norway, for example, accepted Phase I EU allowances for compliance purposes, but the EU ETS did not accept Norwegian allowances (Tuerk et al., 2009). A unilateral link between trading schemes can be established through a simple legislative amendment specifying the conditions for recognition of foreign allowances (Mehling, 2007). An example is the EU Linking Directive (European Council, 2004), linking the EU ETS to the Kyoto offset mechanism Clean Development Mechanism (CDM). Under a bilateral linking emission allowance units are mutually recognized by the other jurisdictions and units can be traded across the schemes, however the amount of credit flow could be limited by the participating jurisdictions.

Indirect linkage occurs when two systems do not accept allowances from each other, but both accept allowances (or offset credits) from a common third party (Ranson and Stavins, 2013). For example, cap-and-trade systems in two jurisdictions might allow companies from both jurisdictions to comply using offsets purchased from an emission reduction credit system as has happened with the offset mechanisms of the Kyoto Protocol, credits from the flexible mechanisms, the Joint Implementation (JI) and the CDM. The EU ETS was also indirectly linked to the AAU market. Even if AAUs were excluded from the system Annex-1 governments and Japanese companies could use both CDM and AAUs for compliance. Japanese companies for example bought large quantities of certified emission reduction units (CERs) for domestic compliance at the time that CERs were the least expensive option. Later, as the AAU price collapsed in 2010, Japanese companies purchased additional AAUs and sold CERs to the EU ETS-bound companies that could use them for compliance up to a certain limit. This may have contributed to a decrease of the price of European Emission Allowances (EUAs) (Tuerk et al., 2013a). Even though there is no sufficient evidence of the extent to which this happened, this example shows that depending on the amount of offset credits allowed in the directly linked systems, indirect links could lead to the same effects as direct links.

3.2 Implications and Challenges of Linking

Direct linking can result in a convergence of the allowance prices (Tuerk et al., 2009; Jaffe and Stavins, 2007). When two emissions trading schemes are linked, market prices will rise for allowances in one scheme, and fall in the other scheme, until full or partial convergence is achieved depending on possible trade limits. The greater the initial price difference of the scheme, the greater the potential gain in economic efficiency (Tuerk et al., 2009). A trading link also creates a larger, more liquid carbon market where volatility can be reduced or increased, depending on the size of the scheme (Carbon Trust, 2009).

Some of the impacts of linking also entail considerable challenges, most notably with regard to distributional issues, and environmental performance and may pose barriers to linking of schemes (Tuerk et al., 2009). As design features in one scheme will propagate into the linked scheme, an agreement on important design features such as cost containment measures, type and stringency of targets, banking and borrowing provisions, sectoral coverage or offset provisions, new entrants rules or ex-post adjustment mechanisms needs to be made in advance. Any decision that may impact the carbon price or the environmental performance in one scheme after linking will also affect the linked scheme and as a consequence such decisions would need to be coordinated among the linked schemes. If the second system for example decides to link with a third system, allows domestic offsets, or expands the scope of allowed international offsets, for example, all these will impact the first system, and may influence its price and hence effect distributional impacts. From a legal perspective if linking partners aim for high uniformity and intensive linking (comparable to a free trade agreement), differences in legal rules can be quite important, for example, in tax rules or in liability rules (Weishaar, 2015).

A linking agreement entails the surrender of regulatory freedom and an ETS after linking is unable to directly determine the domestic carbon price. Similarly, the loss of autonomy entails that a linking agreement may prolong the existence of early suboptimal ETS designs and prevent further improvements (Green et al., (2014)). Trade restrictions could be imposed by governments that would limit the price convergence, the propagation of the design features of one scheme into the other and the loss of regulatory control (Tuerk et al., 2009). In contrary to many other environmental targets, the reduction of Greenhouse gases and corresponding policy instruments, including a CO_2 price, may affect large parts of the economy and have to be considered as part of the economic and technology policy framework of a country. As can be seen so far in practice (see Section 4), emission trading systems are therefore likely to establish full links only to systems that are part of the same or a similar climate policy framework with similar stringent reduction targets, and have a history of policy coordination, trade relationships or are even part of the same economic area.

4. EXISTING AND EMERGING SCHEMES, PLANS FOR LINKING AND FIRST EXPERIENCES

4.1 Overview of Emission Trading Schemes

In 2015, there were 17 emissions trading systems in force across four continents, covering 35 countries, 12 states or provinces, and seven cities, as shown in Table 13.1 (ICAP, 2015a). The collective Chinese pilot systems form the world's second largest carbon market after the EU ETS, covering the equivalent of 1159 Mt of CO_2 emissions compared to the EU ETS with 2007.8 Mt in 2015 (Afrion and Swartz, 2015). A possible national scheme that China aims to establish by late 2017 could be up to twice as large as the EU-ETS (Zhong, 2015). The third largest carbon market is the new South Korean Scheme with 573 Mt covered (ICAP, 2015a).

Both in the USA and Canada national schemes were planned but so far not implemented, however regional schemes are emerging (ICAP, 2015a). The EU ETS is the largest emissions trading market, followed by smaller ETS such as the Korean ETS or the US Regional Greenhouse

Table 13.1 Overview of emerging worldwide emission trading schemes

Measure	Carbon Tax		ETS	
Country/Region	Implemented/ Scheduled	Under consideration	Implemented/ Scheduled	Under consideration
Canada				
Alberta			x	
British Columbia	X			x
Manitoba		X		x
Ontario		X		x
Québec			x	
RGGI			x	
USA				
Washington		X		x
Oregon		X		x
California			x	
Latin America				
Mexico	X			x
Brazil		X		x
Chile		X		x
Europe				
European Union			x	
Denmark	X		x	
Finland	X		x	
France	X		x	
Rep. of Ireland	X		x	
Sweden	X		x	
UK	X		x	
Iceland	X		x	
Norway	X		x	
Switzerland	X		x	
Ukraine		X		x
Asia				
China		X		x
Beijing			x	
Chongqing			x	
Guangdong			x	
Hubei			x	
Shanghai			x	
Shenzhen			x	
Tianjin			x	
Rep. of Korea			x	
Japan	X			x

Table 13.1 (continued)

Measure	Carbon Tax		ETS	
Country/Region	Implemented/ Scheduled	Under consideration	Implemented/ Scheduled	Under consideration
Kyoto			x	
Saitama			x	
Tokyo			x	
Kazakhstan			x	
Thailand		X		x
Turkey		X		x
Australia and Oceania				
Australia			x	
New Zealand			x	
Africa				
South Africa	X			

Source: ICAP, 2015a.

Gas Initiative (RGGI). While a strong dynamic to implement ETS can be observed in Asia, at the same time Australia has abolished its plans to introduce a federal ETS.

4.2 Selected Trading Schemes: Scheme Design and Linking

This chapter describes existing and major emerging emission trading schemes. While the EU-ETS is covered in other parts of this book and Norway, Iceland, and Liechtenstein have been linked with the EU ETS since 2007 through incorporation of the Directive in the EEA Agreement (Meadows, 2015), we briefly present the New Zealand ETS, the Swiss ETS, the US Regional Greenhouse Gas Initiative, the Western Climate Initiative as well as the federal US ETS activities, the California – Québec initiative, the Tokyo Metropolitan Government ETS, the South Korean ETS, and the Chinese pilot schemes.

The *New Zealand Emissions Trading Scheme* (NZ ETS) scheme introduced in 2008 brought in all sectors of the economy over a period of several years (Harrison, 2015). Forestry was the first sector to enter the scheme. The New Zealand emissions trading scheme thereby was the first worldwide to include emissions from deforestation under the cap, rather than via offsets. New Zealand showed strong interest in linking with Australia several years ago, in particular to overcome possible liquidity constraints that could hamper its relatively small scheme (Jotzo and Betz, 2009), but as the Australian ETS has been abandoned linking is not on the policy agenda anymore. As the possibility of unlimited use of Kyoto credits led to a low carbon price in the NZ ETS, the use of Kyoto units was restricted from surrender after 31 May 2015 (New Zealand Government, 2013).

The *Swiss emissions trading scheme* started in 2008 with a five-year voluntary phase as an alternative option to the CO_2 levy on fossil fuels. Revised regulations entered into force on 1 January 2013. The system subsequently became mandatory for large, energy intensive industries (ICAP, 2015c). It now covers about 10 per cent of the country's total Greenhouse Gas (GHG) emissions. In the 2013–2020 mandatory phase, participants in the ETS are exempt from the CO_2 levy (ICAP, 2015c). Switzerland aims to link to the EU ETS and has been in negotiations with the European Commission since 2010 (Meadows, 2015). Switzerland's motivations to linking include the aim to increase the size and the liquidity of the market with stable prices, and to enable Swiss companies to operate in the same emissions market as their business partners in the EU (Swiss Government, 2015).

The *Regional Greenhouse Gas Initiative (RGGI)* is a state-level emissions trading system in the North East US that started trading on 1 January 2009 (ICAP, 2015a). It covers CO_2 emissions from power generation. When it took effect in 2009, RGGI became the first mandatory CO_2 cap-and-trade program in the United States, (Leff, 2014). RGGI covers fossil-fuelled electric power plants greater than 25 MW located in any of the nine participating states (Leff, 2014). CO_2 emissions in the RGGI region accounted for 4 per cent of the total emissions from the electric power sector in the United States in 2012 (Leff, 2014). A 2012 program review by the participating states resulted in regulatory changes that took effect on 1 January 2014, including a revised cap on the emissions from the power sector of 50 per cent below 2005 levels by 2020 (New and Johnson, 2015).

The *Western Climate Initiative (WCI)* is an initiative of US states and Canadian provinces to develop emissions trading systems (ICAP, 2015a). Currently only California and Québec have implemented trading systems, and trading formally started on 1 January 2013 (Laing and Mehling, 2013).

The basic structure of the WCI is a decentralized cap-and-trade program in which jurisdictions cooperate to design individual systems that can be linked to create a single market (Québec Government, 2015). Each jurisdiction is responsible for setting its own cap in light of the regional aim of a 15 per cent reduction of 2005 GHG levels by 2020. While some general guidelines for establishing jurisdiction-specific caps were agreed upon, these guidelines were extremely broad in the hopes that flexibility in this regard would facilitate greater participation (Tuerk et al., 2013b).

Table 13.2 Summary of WCI jurisdictions' current positions on cap-and-trade regulation

Cap-and-trade regulations adopted	Expressed interest but no regulations	Will not be implementing cap-and-trade
California Québec	Ontario British Columbia Manitoba Washington Oregon	Montana Utah New Mexico Arizona

Source: ICAP, 2015a.

However, prior to linking one jurisdiction's system to another, each would have the opportunity to review the other jurisdiction's program to assess its consistency with the program design (Tuerk et al., 2013b). Within the jurisdiction in which it is adopted, the coverage of the cap-and-trade scheme is very high. In the initial phase of development, the majority of large-emitting installations in all industrial and power sectors would be included, and transportation and commercial sectors are due for inclusion in the second phase (Tuerk et al., 2013b). In the participating jurisdictions, all installations emitting over 25,000 tCO_2e per year would be included in the scheme. The basic guidelines of the WCI allow for some cost containment measures, including allowance reserves, limited borrowing, and auction floor prices, but exclude hard price caps and unlimited borrowing as contained in some of the earlier proposals for a federal US cap-and-trade system (Hoffmann et al., 2015).

Table 13.2 shows the current positions of the 11 WCI partners (ICAP 2015a). The figures in brackets denote auctions. Depending on the final participants in the scheme the WCI could account for about 800 megatons of CO_2e per year, over half of which is comprised of emissions from California.

Québec formally linked its system with that of California on 1 January 2014, (Québec Government, 2015). Under the linked California – Québec scheme both allowances and offsets approved under either scheme can be transferred between the schemes. The effectiveness of the Québec-California ETS, however, is safeguarded through the implementation of a price corridor consisting of an auction floor price and soft price ceiling (Hoffmann et al., 2015). The first joint Québec-California carbon market auction was held on 25 November 2014 (California Environmental Protection Agency, 2014). While the conceptual foundations for the two systems are similar as they are part of the WCI there were still differences that constituted potential barriers to linking and negotiations took two years. Common purchase and holding limits that protect against market manipulation, a common allowance registry, the recognition of existing voluntary offset programmes and other provisions are not identical (ICAP 2015a). Both schemes have to produce similar outcomes such as monitoring of reporting, so that a ton of GHG emitted and verified in a partner jurisdiction equals a ton of GHG emitted and verified everywhere in the partnership.

The *Tokyo Metropolitan Government Emissions Trading System* (TMG ETS) was established in 2010 (Kimura, 2015), starting from 2011 with an emissions trading scheme targeting energy related CO_2 and requiring 1,300 of Tokyo's most energy and carbon intensive organizations to meet legally binding emission targets (Tuerk et al., 2013a). The covered entities comprise 1,100 commercial and institutional buildings and 200 industrial facilities with annual energy consumption of 1,500 kilolitre (kl) or more (crude oil equivalent) representing 40 per cent of commercial and industrial sectors' emissions (Chiba, 2011). The ETS, with its target, forms the main pillar of the TMG's comprehensive effort to achieve emissions reductions of 25 per cent between 2000 and 2020. During the first phase of the scheme, which ran up to 2014, participating organizations had to cut their carbon emissions by 6–8 per cent and in the second period from 2015 to 2019 by 15–17 %, depending on the type of entity (Kimura, 2015). Allowances are allocated based on historic emissions (Tuerk et al., 2013a).

The *South Korean ETS* started in January 2015, aiming to reduce its GHG emissions by 30 per cent against business-as-usual (BAU) by 2020 (Reklev et al., 2015). With a cap of 573 MtCO2e in 2015 and covering two-thirds of the country's total emissions, it is the second-largest ETS worldwide after the EU ETS (ICAP 2015b). The South Korean ETS includes 23 sub-sectors from steel, cement, petro-chemistry, refinery, power, buildings, waste sectors and aviation (ICAP 2015b), covering around 500 entities. Most sectors will receive their free allowances based on the average GHG emissions of the base year (2011–2013) (ICAP 2015b). The amount of free allocations will be reduced from 100 per cent in the first phase (2015–2017) to 97 per cent in the second phase (2018–2012) and 90 per cent in the third phase (2021–2025) (ICAP 2015b). The emissions trading legislation provides for linking to other schemes such as the EU ETS, with details to be set by a national government's decrees.

The Chinese pilot schemes are a part of an effort to create a national emission trading scheme that China plans to implement by 2017 (Jun Dong et al. 2016). China's strategy has been to mandate creation of several pilot trading systems with different designs, allowing it to compare experiences prior to deciding on an approach for a future nationwide system

Table 13.3 Emerging emissions trading schemes in China

China	Emissions reduction target (cf. 2010 levels)
Beijing	18 %
Chongqing	17 %
Guangdong	19.5 %
Hubei	17 %
Shanghai	19 %
Shenzhen	21 %
Tianjin	19 %

Source: Carbon Brief 2014.

(Qian and Yu, 2015). The National Development and Reform Commission announced its plan to develop seven official ETS pilot programs (Beijing, Shanghai, Tianjin, Chongqing, Guangdong, Hubei and Shenzhen) in 2011 (IAMC, 2014). The implementation of this plan began in 2013. By October 2014, six of the seven pilot schemes started operation and the remaining one – Chongqing – started on 19 June 2014 (ICAP, 2015d). Before discussing specific systems, we present some main design options of the emission trading schemes.

The seven pilot emissions trading schemes vary in terms of caps and targeted sectors across cities and regions in order to provide a solid basis for implementing a unique and national wide emission trading scheme (see Table 13.3). An important feature of the Chinese schemes is that some of them include the *building sector* (for example, Beijing) or in some cases also the *transport sector* (Schleicher, 2015). Beijing is the only pilot ETS that requires annual absolute emission reductions for existing facilities in the manufacturing and service sectors. The other Chinese pilot ETSs do not require absolute reductions, but a reduction of carbon intensity per unit of industrial added value (Schleicher, 2015). Shenzhen and Tianjin allow individual investors and entities that are not covered in the ETS, such as financial institutions, to participate in the emission trading, resulting in higher trading frequency and potentially larger price fluctuations (Schleicher, 2015). Focusing only on carbon dioxide, the pilots cover roughly 40–60 per cent of a city or province's total emissions, and apply to power and other heavy manufacturing sectors such as steel, cement, and petrochemicals (Song, Lei, 2014). Table 13.4 presents an overview of the emissions markets of the seven Chinese pilots (Zhong, 2014).

The standard method for distributing emission allowances in China is *grandfathering* based on historical emissions data in the past few years, with consideration of sector characteristics or mitigation costs (Calderon

Table 13.4 Overview of the emissions markets of the seven Chinese pilots

Pilots	Covered Emitters	Annual Cap Million Ton	Trade Volume Million Ton	Trade Amount Million Yen
Guangdong	242	388	1.29 (11.1)	70.6 (667)
Hubei	138	324	5.28	125
Shanghai	191	160	1.55	60.9
Tianjin	114	160	1.06	21.9
Chongqing	240	130	0.145	4.46
Beijing	490	78	2.03	100
Shenzhen	832	30	1.66	113

Source: Zhong, 2014.

et al., 2013). In most cases the cap is an *emissions intensity cap*. For the purpose of price management and cost containment, the local governments of Tianjin, Shanghai and Hubei may reserve some allowances (Calderon et al., 2013). At the same time, auctioning may be used as a complementary method in Beijing, Tianjin, Shanghai, Shenzhen, and Guangdong pilots for a small portion of allowances (Calderon et al., 2013). Most Chinese schemes consider to set aside permits to regulate the market, for example, to buy/sell allowances in case of market fluctuation (Schleicher, 2015).

The Chinese pilot ETSs also require large electricity users to submit emissions permits to the government, thus also indirect emissions from energy use are covered by most Chinese ETS providing incentives for demand side management (Song, Lei, 2014). In China the linking of regional trading schemes has been recently proposed but may prove to be a big challenge given the differences among pilot designs across China. While these differences provide useful experimental results, their linkage in the short-term could weaken the diversity and therefore reduce the value of the experiment. In contrary to the WCI that provides for a similar scheme design for the participating states, a linked Chinese Carbon market cannot be expected any time soon.

In the USA, a federal US ETS was discussed for several years, with ETS bills introduced in the Senate but no legislation has ever been adopted. In August 2015 the US Environmental Protection Agency (EPA) finalized its proposed Carbon Pollution Standards for Existing Power Plants, known as *the Clean Power Plan* (EPA, 2015a). The Clean Power Plan establishes different target emission rates for each state due to regional variations in generation mix and electricity consumption. Overall, it is projected to achieve a 32 per cent reduction in power sector emissions by 2030 from 2005 levels (C2ES, 2015). State plans can include market-based mechanisms. In addition, the EPA plans to develop a federal emissions trading scheme that would be implemented in those states that do not have a fully approved state plan by 2016 as required under the final Clean Power Plan (EPA, 2015b).

4.3 Comparing Design Features and an Outlook for Linking

As seen in Section 4.2, the existing and the emerging trading schemes differ significantly regarding design features such as scope of the systems or allocation methods as they respond to specific greenhouse gas reduction challenges of different countries or regions or try to experiment with new approaches compared to the EU ETS.

While the EU ETS focuses on industry and large energy producers, ETS schemes in the US and Canada may also include the transport sector in the long term. Some of the emerging schemes in Asia try different approaches. They often involve smaller companies, buildings, or include indirect emissions from energy consumption. While schemes in the US, Canada or Europe have absolute caps, the Chinese pilot ETS have relative caps, sometimes complemented by absolute caps (Schleicher et al., 2015). Only few ETS worldwide have a significant share of auctioning from the beginning. Similar to the EU ETS where grandfathering was used in the beginning, the Chinese ETS pilots allocate most allowances for free. Most ETS except the EU ETS provide market stabilization measures to manage price fluctuations or provide for price corridors providing increased price certainty to the included companies.

The international developments show that emissions trading schemes worldwide may evolve very diversely making short term direct linkages difficult. At the same time, direct linkages between schemes with close political and trade relationships may continue to be established. The EU will continue to try to expand its scheme to neighbour regions and a set of Asia-Pacific interlinkages could emerge more easily than these systems linking with the EU ETS.

Given the ETS development dynamics there, Asia and the Pacific could become the centre of global carbon market if the implemented and emerging carbon trading schemes would eventually be linked, possibly including emissions of more than 7,000–8,000 MtCO2e (Lingshui, 2015).

Indirect linkages via common offset as emerged during the Kyoto period may not play an important role in the future as a new international climate agreement may not set comparable standards. Also several ETS, such as the EU ETS or the NZ ETS have reduced the limits to which the international offsets can be used compared to the Kyoto Protocol.

5. CONCLUSIONS

While the EU vision of an OECD-wide carbon market by 2015 has not come true, linking of emissions trading schemes is still on the international policy agenda. At the same time, the realization of the trading links is proving to be far slower than expected a few years ago. Several developments can be attributed to this observation. First, within OECD countries on the one hand, in the US a federal scheme has not so far been implemented and it remains to be seen in what timeframe the new Clean Power Plan will be able to trigger the development of such a scheme; while on the other hand the Australian ETS with which the EU-ETS had planned to link was abandoned. Second, the emerging new international climate agreement is characterized by bottom-up elements more strongly than by top-down elements. As a result, linking agreements may take longer to be concluded in the absence of comparable stringent emission reduction units, common offset provisions and reduction efforts. Third, some of the ETS in Asia are trying different approaches compared to the EU ETS, often involving sectors not covered by the EU ETS such as buildings, but being at the same time less compatible with EU ETS. Experiences with (planned) linking between countries in similar economic areas or climate policy frameworks such as California and Québec or the EU and Switzerland show that reaching linking agreements can take a lot of time.

While carbon trading as a climate policy instrument is gaining attraction as this chapter has shown, the carbon market will be characterized in the short term by a patchwork of different approaches. Overall linking of trading schemes will remain an important part of the international climate policy design, but a significantly linked global market will most probably not emerge any time soon.

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14. Emissions trading and WTO law

Kateryna Holzer

1. INTRODUCTION

The multilateral trading system, originally set up by the 1947 General Agreement on Tariffs and Trade (GATT) and further developed by the 1995 WTO Agreement (Marrakech Agreement), is aimed at liberalizing trade, promoting competition and at facilitating economic growth and development.[1] WTO rules impose constraints on policy space that is available for taking climate change mitigation and adaptation measures to the extent that domestic climate policy measures may have negative impacts on trade. This is particularly the case regarding emissions trading schemes (ETSs), which increase prices of traded goods and services and affect conditions of international trade.

The interaction of emissions trading and WTO rules usually begins with an introduction of regulatory measures or design features of an ETS that are aimed at preventing carbon leakage. Carbon leakage is the situation where the total amount of global emissions increases due to the expansion of emissions-intensive production elsewhere.[2] Under the 'pollution haven' hypothesis, faced with increased production costs resulting from participation in an ETS, domestic emissions-intensive industries move their production facilities to countries without emissions trading systems or lose their market shares to imported products.[3] The problem of carbon leakage under an ETS can be addressed through different instruments, including free allocation of emissions allowances, income-supporting recycling of ETS revenues, use of border adjustment measures and so on. None of these instruments is

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[1] See the Preamble to the Marrakesh Agreement Establishing the World Trade Organization.

[2] P Wooders et al. (2009), 'Border Carbon Adjustment and Free Allowances: Responding to Competitiveness and Leakage Concerns', at 8–11. See also H van Asselt et al. (2009), 'Addressing Leakage Competitiveness in US Climate Policy: Issues Concerning Border Adjustment Measures', at 9; R Ismer (2010), 'Mitigating Climate Change through Price Instruments: An Overview of the Legal Issues in a World of Unequal Carbon Prices', at 211–12.

[3] MS Taylor (2004), 'Unbundling the Pollution Haven Hypothesis', at 3.

ideal.[4] All of them have their shortcomings in terms of effectiveness, implementation feasibility, costs for state budget and compliance with WTO rules.

Research has been done to map out areas of possible interactions between emissions trading and WTO law.[5] There are also studies that focus on particular issues of emissions trading,[6] and scientific papers that examine the compatibility of the emissions trading scheme of the European Union (EU ETS) with WTO law.[7] Building on existing research, this chapter looks more closely at the design elements of an ETS and possible flanking support schemes that are most vulnerable to a WTO challenge.

Before analyzing the ETS design features and possible flanking support schemes, a close examination of the legal status of an emissions allowance is required so as to clarify the applicability of WTO rules to ETSs (section 2). Regarding design features and support schemes, free allocation of emissions allowances is examined first (section 3), followed by revenue recycling (section 4), and, finally, border tax adjustments such as import taxes and cost rebates for emissions allowances for domestic producers on exportation are reviewed (section 5). Section 6 discusses emissions trading between countries, taking linking of ETSs as well as flexibility mechanisms under the Kyoto Protocol or other potential international climate agreements into account.

2. LEGAL NATURE OF EMISSIONS ALLOWANCES AND THE SCOPE OF APPLICATION OF WTO RULES

WTO rules are relevant each time a government introduces a measure that influences the competitive relationship between domestic and imported

4 K Holzer (2014), *Carbon-Related Border Adjustment and WTO Law*, at 45–50.

5 See e.g. J Werksman and J Lefevere (1999), 'WTO Issues Raised by the Design of an EC Emissions Trading System'; C Voigt (2008), 'WTO Law and International Emissions Trading: Is There Potential for Conflict?'; J de Cendra de Larragan (2012) 'Emission trading schemes and WTO law: A typology of interactions' and more recently F Deane (2015), *Emissions Trading and WTO Law*.

6 See e.g. J Pauwelyn (2007), 'U.S. Federal Climate Policy and Competitiveness Concerns: the Limits and Options of International Trade Law'; K Holzer (2010) 'Proposals On Carbon-Related Border Adjustments: Prospects for WTO Compliance'; I Jegou and L Rubini (2011), 'The Allocation of Emission Allowances Free of Charge: Legal and Economic Considerations'.

7 See e.g. L Bartels (2011), 'The Inclusion of Aviation in the EU ETS: WTO Law Considerations'.

products, domestic and foreign providers of services or affects international trade in any other way.[8] Market access and non-discrimination are two key benchmarks against which a measure is tested on WTO compliance in the majority of cases. At the same time, WTO rules are not absolute. They provide for derogations if measures are taken to protect the environment.[9]

Emissions trading schemes (ETS) are introduced to put a price on greenhouse gas emissions to incentivize climate change mitigation measures. ETSs often set a quota on total emissions that are permissible (commonly referred to as a cap-and-trade system) and provides emitters falling within the scope of the ETS the possibility to purchase and sell scarce emissions allowances on a secondary market. Moreover, an ETS can include additional elements, such as the participation of foreign producers, linking arrangements with foreign ETSs, international offsets and so on. Thus, an ETS is not a single measure but constitutes a complex system which is implemented through a multitude of measures, including flanking support schemes such as border tax adjustments schemes, which must be assessed under WTO rules. For the purpose of this chapter the conventionalization of an ETS is therefore constructed broadly to include such features.

Because there is (or at least, should be) a lesser supply of emissions allowances than is demanded by covered entities, emission allowances constitute a scarce resource with a positive market price that raises the total costs of production.[10] Consequently, companies participating in an ETS have a less advantageous position vis-à-vis competitors that are participating in an ETS. Cost increases relating to ETS 'indirectly' raise issues under WTO law if a higher costs burden is placed on foreign producers. In such situations a 'direct impact' of an ETS on trade is less obvious because trading in emission allowances does not constitute trade in its traditional sense. Traded are not conventional goods and services, but emissions allowances, which essentially represent rights to certain amounts of emissions.[11] Whether the WTO Agreement covers trade in emissions

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[8] S Charnovitz (2002), 'The Law of Environmental "PPMs" in the WTO: Debunking the Myth of Illegality'.

[9] For instance, paragraph (g) of GATT Article XX provides an exception for measures taken with the purpose to conserve exhaustible natural resources.

[10] T Epps and A Green (2010) *Reconciling Trade and Climate Change: How the WTO Can Help Address Climate Change*, at 65.

[11] It should also be noted that the market for emissions allowances (the carbon market) is artificially created by the legislator.

allowances is debated in the literature. Some argue that 'allowances cannot be described as either products or services under the WTO, and thus rules governing the transfer and mutual recognition of allowances are not covered by WTO disciplines'.[12] Others compare emissions allowances to commodities: like commodities, emissions allowances are traded on the market and have a price.[13] If emissions allowances were considered to be commodities, they would fall within the regulatory ambit of the General Agreement on Tariffs and Trade (GATT). However, the prevailing view is that emissions allowances resemble services and fall within the regulatory ambit of the General Agreement on Trade in Services (GATS).[14] It can be argued that emissions allowances have similarities with negotiable instruments and therefore should be regulated by the GATS rules on financial services.[15] It is particularly true for transactions carried out in the carbon market that are similar to transactions with derivatives (hedging, speculation, arbitraging and so on).

While the legal status of emissions allowances is disputable and will remain so until it is clarified in a future WTO dispute, one element appears certain: irrespective of whether emissions allowances fall within the scope of the WTO Agreement or not, restrictions on the eligibility of emissions allowances for compliance with an ETS requirement and also other design features of emissions trading and flanking support schemes have the potential to affect international trade indirectly, through an impact on trade in goods and services, and as such come into conflict with the rules of the GATT and/or the GATS.[16]

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[12] J Werksman and J Lefevere (1999) 'WTO Issues Raised by the Design of an EC Emissions Trading System', at 3.

[13] J Button (2008), 'Carbon: Commodity or Currency? The Case for an International Carbon Market Based on the Currency Model', at 575–7. It should be noted that the WTO's Appellate Body (AB) defines a good as an item, which is tangible and capable of being possessed. It also notes that the meaning of a good under different provisions of the WTO Agreement is not necessarily the same. See *US-Softwood Lumber IV*, AB report, para. 59.

[14] R Howse (2009), 'World Trade Law and Renewable Energy: The Case of Non-Tariff Barriers', at 15–16. See also J Werksman and J Lefevere (1999) 'WTO Issues Raised by the Design of an EC Emissions Trading System', at 8.

[15] Ibid.

[16] A measure can affect trade in goods and trade in services at the same time and fall under both the GATT and the GATS. See *EC-Bananas*, AB report, paras. 221–22.

3. FREE ALLOCATION OF EMISSIONS ALLOWANCES

The initial stage of emissions trading is the allocation of emissions allowances, whereby a government distributes emissions allowances to firms participating in an ETS. There are several means for the distribution of emissions allowances; most prominent are free allocation or auctioning.[17] Free allocation reduces the financial burden of an ETS for domestic firms and thus is widely used as an approach to addressing competitiveness and carbon leakage concerns under existing national cap-and-trade systems.[18]

While the free allocation of emissions allowances may help prevent carbon leakage,[19] it negatively affects the state budget[20] and reduces incentives for firms to reduce emissions.[21] Free allocation also creates the risk of windfall profits.[22] Faced with these problems, the EU Member States are moving from free allocation of emissions allowances towards auctioning of an increasing part of them in the third phase of the EU ETS.[23] Concerns have been raised in the literature about the consistency of free allocation

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17 I Jegou and L Rubini (2011) 'The Allocation of Emission Allowances Free of Charge: Legal and Economic Considerations', at 3 and 6.

18 S Dröge (2009), 'Tackling Leakage in a World of Unequal Carbon Prices', at 46.

19 There is no conclusive evidence though. See e.g. I Jegou and L Rubini (2011), 'The Allocation of Emission Allowances Free of Charge: Legal and Economic Considerations', at 21.

20 It is estimated that the foregone revenues from the free allocation of emissions allowances in the third phase of the EU ETS (2012–2020) will constitute €100 billion. See ibid.

21 J Hoerner and F Muller (1996), 'Carbon Taxes for Climate Protection in a Competitive World', at 46. Yet, it could be argued that an emissions allowance, even if a firm received it for free, would still have an opportunity cost. It could be sold on the carbon market instead of being used for compliance with ETS and thus it stimulates its holder to reduce emissions.

22 Windfall profits are addressed in Chapter 5 by Beat Hintermann in this Research Handbook.

23 In the third phase of EU ETS (2012–2020), roughly half of the allowances are being auctioned. For instance, all electricity generators (with some exceptions) are obliged to buy emissions allowances at an auction. Yet, the free allocation is still available for carbon-intensive sectors with a significant risk of carbon leakage. The free allocation is based on the benchmark of the 10% most efficient EU producers in the sector. See I Jegou and L Rubini (2011), 'The Allocation of Emission Allowances Free of Charge: Legal and Economic Considerations', at 6.

of emissions with the WTO subsidy (section 3.1) and anti-dumping rules (section 3.2).[24] These are reviewed below.

3.1 Issues Arising under the WTO Rules on Subsidies

GATT Article XVI and the provisions of the WTO Agreement on Subsidies and Countervailing Measures (ASCM) regulate the use of state support measures, or subsidies.[25] The use of subsidies comes under legal scrutiny if subsidies (1) promote exports or support import substitution[26] and (2) adversely affect the interests of other WTO members (including injury to their industries, impairment of their benefits under tariff concessions and serious prejudice to their interests).[27]

The first category of subsidies is prohibited, whereas the second category is challengeable under the WTO dispute settlement system. The latter is often called ‘actionable subsidies’. Subsidies with adverse effects for industries of other WTO members can also face countermeasures, particularly the unilateral imposition of countervailing duties (CVDs) on subsidized imports by affected WTO members.[28] Only those subsidies, which are specific, that is subsidies given to particularly firms or industries, are actionable. Non-specific subsidies, that is, those available across the board to all sectors of the economy, do not fall within the regulatory scope of the ASCM and cannot be challenged.

The starting point of the analysis of the free allocation of emissions allowances under WTO rules is the determination of whether it constitutes a subsidy within the meaning of the WTO Agreement. The WTO definition of a subsidy is contained in Article 1 of the ASCM. It consists of two components: (1) a measure must constitute a ‘financial contribution’ by a government or any form of ‘income or price support’; and (2) must confer a ‘benefit’. The financial contribution by a government can take three forms: (1) direct transfers of funds (for example, loan guarantees);

[24] See e.g. R Howse (2010), ‘WTO Subsidies Disciplines and Climate Change Mitigation Policies: Options for Reconciliation’, at 10–11; I Jegou and L Rubini (2011), ‘The Allocation of Emission Allowances Free of Charge: Legal and Economic Considerations’.

[25] The rules governing the use of subsidies are also contained in GATT Art. XVI. The GATT and ASCM provisions form an integrated set of rules governing the use of subsidies and countervailing duties. The ASCM does not however cover the use of subsidies in the agricultural sector, which are regulated by the provisions of the WTO Agreement on Agriculture.

[26] ASCM Art. 3.

[27] ASCM Art. 5.

[28] See Part V of ASCM.

(2) fiscal incentives (government revenue that is otherwise due is forgone); and (3) provision of goods or services apart from general infrastructure or purchase of goods.[29] In addition, it covers situations where a government entrusts a private body to provide a financial contribution in any of the three forms or provides financial support indirectly (for example, through a funding mechanism).[30]

Based on the fact that under the free allocation allowances are distributed for free instead of being exchanged for money, one could argue that the free allocation could be considered to be a financial contribution by a government in the form of the revenue foregone that would otherwise have been due.[31] Consequently, the free allocation meets the first part of the subsidy definition under Article 1.1(a)1(ii) of the ASCM. It could also be argued that free allocation confers a benefit[32] to firms so long as an emissions allowance, which was received for free, can always be sold in the market if a firm achieves emissions reduction and has no need in the allowance to comply with its emissions quota.[33] Thus, due to the receipt of free allowances, a firm gets a better financial position than before, which is a benefit.[34] The finding that the free allocation of emissions allowances

[29] ASCM Art. 1.1(a)(1)(i)–(iii).

[30] ASCM Art. 1.1(a)(1)(iv).

[31] Depending on whether an emissions allowance could qualify as a good or service, the free allocation of emissions allowances could also acquire the meaning of a provision of goods and services under Article 1.1(a)1(iii). I Jegou and L Rubini draw here a parallel to the stumpage arrangements, which provided rights to lumber (which was found by the AB to be a good) to Canadian lumber harvesters. See I Jegou and L Rubini (2011), 'The Allocation of Emission Allowances Free of Charge: Legal and Economic Considerations', at 30–1. Providing rights to a good was found by the AB to constitute a financial contribution by government in the form of a provision of goods and services. See *US-Softwood Lumber IV*, AB report, para. 75. Yet, the difference of stumpage arrangements to emissions allowances is that the latter provide rights to emissions and not to such natural resources or goods as lumber. Since the AB considers a good to be a tangible and possessable item, it is difficult to qualify emissions allowances (i.e. rights to pollute) as a good.

[32] In *Canada-Aircraft*, when interpreting the meaning of the benefit under this provision, the AB noted: 'the word "benefit", as used in Article 1.1(b), implies some kind of comparison. This must be so, for there can be no "benefit" to the recipient unless the "financial contribution" makes the recipient "better off" than it would otherwise have been, absent that contribution. In our view, the marketplace provides an appropriate basis for comparison in determining whether a "benefit" has been "conferred" . . .'. See *Canada-Aircraft*, AB report, para. 157.

[33] I Jegou and L Rubini (2011) 'The Allocation of Emission Allowances Free of Charge: Legal and Economic Considerations', at 22.

[34] Ibid., at 22. See also L Rubini (2013), 'Subsidies for Emissions Mitigation under WTO Law', at 575–6.

constitutes a state financial contribution that confers a benefit to a firm would be enough to render the free allocation a subsidy under WTO law. However, it would not be enough for the complaining party to win a dispute in the WTO or to serve as justification for the unilateral imposition of countervailing duties on subsidized imports. As long as emissions allowances are not provided for free specifically on exportation[35] or under the condition that a firm would use domestically produced components, the free allocation is unlikely to be viewed as a prohibited subsidy. However, since the free allowances are usually available only to certain firms or industries, namely to those under a significant risk of carbon leakage, the free allocation is likely to be viewed as a specific subsidy, and as such, could potentially be actionable.[36] This means that this measure could successfully be challenged in the WTO, and eventually forced to be withdrawn, or could be targeted by trading partners through CVDs, if the complaining party (or the CVD-imposing country) could claim adverse effects, including material injury to its domestic industry.[37]

In this respect, an analogy is drawn with the situation in the *US-Softwood Lumber IV* dispute, where the claim of subsidy was based on the fact that the companies did not pay 'adequate remuneration' to the government for the access to the natural resource (lumber).[38] The analogy makes sense if complaining countries themselves have emissions trading in place.[39] Such countries, especially those with ETSs based on auctioning, might be able to claim that their domestic industries buying emissions allowances in an auction are adversely impacted by imports from countries where emissions allowances are distributed for free.[40] Yet, currently there are very few countries, in which domestic producers bear emissions costs. For countries with no climate change legislation in place, it would be difficult to claim that the free allocation of emissions allowances causes adverse effects to their domestic industries, which bear no emissions costs at all.[41] Therefore,

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[35] It could be argued that the link to exports could be established based on the fact that the government gives allowances for free to those industries that are most carbon-intensive and trade-exposed. However, the trade exposure is also understood in terms of the dependency on imports. See *Inside U.S. Trade* (2009), at 2.

[36] A subsidy is deemed to be specific when it is not sufficiently broadly available throughout an economy. See *US-Upland Cotton*, Panel report, para. 7.1142.

[37] ASCM Art. 5.

[38] R Howse (2010) 'WTO Subsidies Disciplines and Climate Change Mitigation Policies: Options for Reconciliation', at 10–11.

[39] K Holzer (2014) *Carbon-Related Border Adjustment and WTO Law*, at 212–13.

[40] Ibid.

[41] Ibid., at 213.

the risk of disputes that could be brought under the ASCMagainst the free allocation of emissions allowances currently appears to be minimal.

3.2 Issues Arising under WTO Anti-Dumping Rules

The free allocation of emissions allowances can also trigger the initiation of anti-dumping procedures by trading partners and lead to the imposition of anti-dumping duties (ADDs) on imports from countries with the free allocation. It could be argued that the free allocation enabled producers to charge unusually low prices. Imports originating from such enterprises could therefore qualify as dumped imports. Under WTO anti-dumping rules, if dumped imports cause or threaten material injury to a domestic industry, they can be offset by anti-dumping duties charged on top of the ordinary import duties.[42]

The mere non-payment of emissions costs that results in the lower prices would not, however, suffice to make a case for the imposition of ADDs. Under the definition of dumping contained in Article VI:1 of the GATT,[43] dumping is a situation where the export price of the product is:

> (a) less than the comparable price, in the ordinary course of trade, for the like product when destined for consumption in the internal market of the exporting country, or,
> (b) in the absence of such domestic price, is less than either
> (i) the highest comparable price for the like product for export to any third country in the ordinary course of trade, or
> (ii) the cost of production of the product in the country of origin plus a reasonable addition for selling cost and profit.

In all these cases, the price of the imported product does not represent the normal value of the product.[44] Importantly, the comparison is always made with the price at which the like product is sold in the market of the exporting country or with the price, which is otherwise related to the exporting country. By contrast, when dumping is referred to the case of the free allocation of emissions allowances, the comparison is made with the price of the like product in the market of the importing country that distributes allowances through an auction. This does not reflect the WTO meaning of

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[42] See GATT Art. II:2 and GATT Art. VI. The imposition of anti-dumping duties is further regulated by the Agreement on Implementation of Article VI of the General Agreement on Tariffs and Trade 1994 (the Anti-Dumping Agreement).
[43] See also Art. 2 of the Anti-Dumping Agreement.
[44] The normal value is thus a benchmark, against which the export price is compared in dumping cases.

dumping and, hence, does not provide justification for the imposition of ADDs.[45]

Yet, the case of dumping could arguably be made under the provisions of Article 2.2 of the Anti-Dumping Agreement, which refers to the situation where the comparison with the price in the internal market of the exporting country is not possible 'because of the particular market situation'. In other words, it could be argued that the non-payment for emissions allowances results in the price of products not reflecting the normal value of the good, and therefore the export price cannot be compared to the price in the internal market.[46] In that situation, the comparison could be made with 'the cost of production in the country of origin plus a reasonable amount for administrative, selling and general costs and for profits' (ASCM Article 2.2) and 'costs shall normally be calculated on the basis of records kept by the exporter or producer under investigation, provided that such records . . . reasonably reflect the costs associated with the production and sale of the product under consideration' (ASCM Article 2.2.1.1).

It could thus be argued that the costs associated with the production of imported products are not reasonably reflected because of the distribution of emissions allowances for free. Peter Holmes, Thomas Reilly and John Rollo refer here to the anti-dumping procedure initiated against Ukrainian steel some time ago.[47] In that case, the dumping resulted from the fact that the price of gas in Ukraine was not reasonably reflected in the production costs of Ukrainian steel producers and hence in the price of Ukrainian steel.[48] Similarly, in the case of the free allocation of emissions allowances, the country with an ETS which is fully based on auctioning of emissions allowances, could argue that the price of exports from countries with the free allocation of emissions allowances does not reflect the normal value.[49] The benchmark for the normal value could be the price in the market of the importing country or a third country distributing emissions allowances only through auction. The difference between the 'normal value' and the

[45] J Pauwelyn (2013) 'Carbon Leakage Measures and Border Tax Adjustments under WTO Law', at 505.

[46] P Holmes et al. (2011) 'Border Carbon Adjustments and the Potential for Protectionism', at 889.

[47] See Council Regulation (EC) No 954/2006 of 27 June 2006 imposing definitive anti-dumping duty on imports of certain seamless pipes and tubes, of iron or steel originating in Croatia, Romania, Russia and Ukraine.

[48] Until the mid-2000s Ukraine bought gas from Russia at prices which were considerably lower than the market price.

[49] P Holmes et al. (2011), 'Border Carbon Adjustments and the Potential for Protectionism', at 889.

actual export price would then be a margin of dumping to be offset with anti-dumping duties.[50]

Whether the 'reasonable costs' argument can establish a solid foundation for the imposition of anti-dumping duties in the case of non-payment of emissions costs under free allocation of emissions allowances in exporting countries is uncertain and can only be tested in a WTO dispute.[51] With this in mind, the use of ADDs, like the use of CVDs, for leveling the playing field between domestic and foreign producers in the world of different emissions costs cannot be excluded.

4. REVENUE RECYCLING

Another element relevant in the context of carbon leakage and WTO-law compliance is the mode of allocation of state revenues from emissions allowances. A government can use revenues it receives from the distribution of emissions allowances through auctioning in many ways.[52] It can use the revenues to fund various social and infrastructure programs (salaries, pensions, medical care, army and so on), as it normally does with all other tax revenues. It can further use them to enable the economy-wide tax reform through the reduction of other taxes, for instance, capital taxes (corporate taxes, personal income rates on interest, dividends, capital gains and so on) or labor taxes (payroll, personal income taxes and so on).[53] The revenues can also be earmarked, that is, they can be spent to fund climate change and other environmental projects. In this respect, it is argued that ETSs can be an important source for global and national action on climate change.[54] Finally, the revenues can also be recycled (that is, redistributed) via lump sum rebates to low-income households, which are most disadvantaged by increased emissions costs. Or a government may choose to recycle the emissions allowances revenues back to firms, particularly to those which are most vulnerable to competitiveness losses. In that case, while the revenue

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[50] Ibid., at 888–9.

[51] J Pauwelyn (2013), 'Carbon Leakage Measures and Border Tax Adjustments under WTO Law', at 505.

[52] A Baranzini et al. (2000), 'A Future for Carbon Taxes', at 400.

[53] J Carbone et al. (2014), 'Getting to an Efficient Carbon Tax – How the Revenue is Used Matters'.

[54] A Esch (2013), 'Using EU ETS Auctioning Revenues for Climate Action: What is the Appetite for Earmarking within Specific EU Member States?' at 6–7.

recycling can serve as a tool of preventing carbon leakage,[55] it will raise issues under the WTO rules on subsidies.

Like the WTO law analysis of free allocation of emissions allowances, the examination of ETS revenue recycling schemes under the WTO legal framework focuses on the question of whether a particular mode of revenues allocation subsidizes national producers to the detriment of foreign industries, and as such is an actionable subsidy. As already discussed, to qualify as an actionable subsidy, a measure must constitute a state financial contribution or any form of income or price support, must confer a benefit, be specific and create adverse effects on foreign industries.

Based on case law, it could be argued that the recycling of revenues from emissions allowances back to the most vulnerable producers may constitute a state financial contribution in the form of a direct transfer of funds (ASCM Article 1.1(a)(1)(i)) or foregone budget revenues (ASCM Article 1.1(a)(1)(ii)).[56] Yet, strong objections can be raised against this.[57] First, under this mode of revenue allocation, the same money that was collected would be recycled back to the contributing entities, and not the revenues from other sources that were collected before and put in the state budget or comprising state reserves. Hence, the recycling of emissions allowances revenues back to firms acquires characteristics of a redistribution of funds between private entities rather than a direct transfer of funds from the budget. Second, the allegation of foregone government revenues might be correct. Yet, one should remember that such a judgment cannot be taken out of context, namely that '[i]n a country where the status quo is not to tax emissions at all, which is the normal case, the institution of a charge and rebate system should not constitute foregoing government revenue otherwise due, but simply a means of taxation that limits the cost impacts of the measure on its industry'.[58] For similar reasons, it is difficult to argue that such mitigation of ETS impacts on industries confers a benefit to the firms, and it would be even more difficult to prove that ETS revenue rebates created adverse effects for foreign industries.[59]

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[55] N Shariff (2012), 'Enhancing Competitiveness and Addressing Carbon Leakage: A Value Added Based Approach to Emissions Pricing System Design', at 35.

[56] See e.g. *EC and certain Member States – Large Civil Aircraft*, Panel report, para. 7.1292; *US-Large Civil Aircraft (2nd complaint)*, AB report, paras. 617, 812, 815; *US-FSC (Art. 21.5-EC)*, AB report, para. 104.

[57] N Shariff (2012), 'Enhancing Competitiveness and Addressing Carbon Leakage: A Value Added Based Approach to Emissions Pricing System Design', at 48.

[58] Ibid.

[59] Ibid.

Nevertheless, certain design elements of a revenues recycling scheme need to be included in order to withstand allegations of WTO law-inconsistency. First, an ETS revenue recycling scheme needs to be administered so that it can demonstrate a clear connection between the revenue received from the allocation of emissions allowances and its recycling back to firms. This means that, instead of depositing the revenue in the budget account, the government should instantly redistribute it with only a small portion being used for the purposes of ETS administration.[60] This could help address the claim of a direct transfer of government funds. Second, if the government revenues under an ETS were used, even partially, to fund environmental and climate change projects, this could serve as evidence of the environmental rationale of an ETS and constitute an important indicator of the neutrality of the system.[61] It would also increase chances for justification of some elements of ETS under the environmental exception of GATT Article XX, as discussed below.[62] In contrast, the use of revenues from the distribution of emissions allowances solely to support the development of certain domestic industries may impair justification of WTO law violations stemming from the ETS under environmental exceptions provided for under the WTO Agreement. Finally, WTO law does not impose obstacles to redistributing revenues as part of a national tax reform.[63] A study shows that the revenues from carbon-related measures could legally be used to reduce the

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[60] Ibid., at 50.

[61] It is noteworthy that the EU ETS legislation recommends EU Member States to use at least 50% of revenues from auctioning emissions allowance for climate policy purposes, such as: '(a) to reduce greenhouse gas emissions, including by contributing to the Global Energy Efficiency and Renewable Energy Fund and to the Adaptation Fund. . .; (b) to develop renewable energies. . .; (c) measures to avoid deforestation. . .; (d) forestry sequestration in the Community; (e) the environmentally safe capture and geological storage of CO2, in particular from solid fossil fuel power stations; (f) to encourage a shift to low-emission and public forms of transport; (g) to finance research and development in energy efficiency and clean technologies in the sectors covered by this Directive; (h) measures intended to increase energy efficiency and insulation or to provide financial support. . .'. See Directive 2009/29/EC of 23 April 2009, Art. 10.3.

[62] K Holzer (2014), *Carbon-Related Border Adjustment and WTO Law*, at 237.

[63] In this respect it should be noted that the revenues from the UK Climate Change Levy on fossil fuels and electricity are partly recycled back to UK producers through a 0.3% reduction of the employer payment to the National Insurance Contributions. Part of revenues is also diverted to the Carbon Trust, an institution that fosters research and promotes energy efficiency and renewable energy. See R Martin et al. (2009), 'The Impacts of the Climate Change Levy on Business: Evidence from Microdata', at 4.

rates of mandatory social charges, such as health insurance premiums, or income taxes.[64]

5. BORDER CARBON ADJUSTMENTS

The competitive disadvantages of domestic producers bound by ETS obligations can also be tackled by border carbon adjustment (BCA) – a trade measure equalizing emissions costs of domestic and foreign producers. A BCA would put in effect the destination principle widely used in taxation of traded products, whereby the ETS requirement would apply equally with respect to domestic products and imports, on the one side, and emissions allowance rebates can be provided to domestic producers on exportation, on the other side.[65] BCAs are foreseen as one of the measures to support certain energy-intensive industries in the event of carbon leakage under the EU ETS.[66] The decision is to be taken based on the results of the assessment of carbon leakage risks across the EU ETS sectors and the developments at international climate negotiations.[67]

The EU was close to putting the idea of BCAs into practice with its decision to include international aviation in the EU ETS.[68] The EU and non-EU passenger and cargo airlines landing in or departing from EU airports would be required to surrender emissions allowances on their flights.[69] Yet, the EU plan of inclusion of international aviation in the EU ETS was frustrated by the opposition of other countries, which threatened the EU with retaliatory measures.[70] The case was litigated at the European Court of Justice (ECJ), which found the measure legal under the EU law and international law.[71] Today the EU applies the ETS requirement only to

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[64] T Cottier et al. (2014), 'Differential Taxation of Electricity', at 60–63.

[65] This is usual practice in the application of value-added taxes (VAT). See K Holzer (2014), *Carbon-Related Border Adjustment and WTO Law*, at 63–7.

[66] Art. 10b of the EU ETS Directive.

[67] Ibid.

[68] EU Directive 2008/101/EC.

[69] For the details of the EU regulation and examination of its WTO compliance, see L Bartels (2011) 'The Inclusion of Aviation in the EU ETS: WTO Law Considerations', and J Meltzer (2012) 'Climate Change and Trade – The EU Aviation Directive and the WTO'.

[70] See the joint declaration of the Moscow meeting on the inclusion of international civil aviation in the EU-ETS of 22 February 2012.

[71] ECJ Case C-366/10, *Air Transport Association of America and others v. Secretary of State for Energy and Climate Change* [2011] ECR I-1133, paras 142–144, 147.

internal flights, whereas the application to non-EU flights is suspended for the time that a multilateral solution to aviation emissions is negotiated in the International Civil Aviation Organisation (ICAO).[72]

In the US, BCAs become an issue each time a new bill on federal climate legislation is introduced in the Congress.[73] BCAs seem to be indispensable for reaching a political compromise over the imposition of emissions constraints on US producers. Imports from countries characterized by the low emissions intensity in the sector or that are parties to international agreements with binding national commitments at least as stringent as those of the US are expected to be exempt from the cap-and-trade obligation.[74] In this respect, BCAs are to be used as a leverage to get other countries to undertake mandatory emissions reduction commitments. This purpose of BCAs finds little support from countries not bound by international emissions reduction obligations, particularly developing countries. Referring to the UNFCCC principle of common but differentiated responsibilities and the historical responsibility of developed countries, developing countries argue that they are not obliged to share a burden of emissions costs with developed countries and cannot thus be punished by BCAs. The use of BCAs can thus trigger disputes in the WTO and lead to retaliatory measures and trade conflicts.[75] This can arguably be a major reason why BCAs have not been put into practice as yet.

Little clarity exists also regarding the WTO-compatibility of BCAs. On the one hand, it is because they have never been tested in a WTO dispute. On the other hand, it is due to the special nature of the measures, which belong to the category of non-product related process and production methods (npr-PPMs). BCAs are imposed in connection to emissions, which are intangible and cannot be traced in the final product. The matter becomes even more complex given the fact that emissions result from the production process happening abroad. In this respect, BCAs are measures with an extraterritorial reach, as they interfere with other

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72 See e.g. 'Promoting Mutual Supportiveness between Trade and Climate Mitigation Actions: Carbon-related Border Tax Adjustments', Communication from Singapore to the CTE, 30 March 2011, WT/CTE/W/248, 2.

73 See e.g. the Lieberman-Warner and Bingaman-Specter bills of 2007, and the Waxman-Markey (American Clean Energy and Security Act) and Kerry-Boxer bills of 2009. For an analysis of BCA schemes proposed in these bills, see K Holzer (2010) 'Proposals on Carbon-Related Border Adjustments: Prospects for WTO Compliance'.

74 See e.g. Part IV, Section 401 of the 2009 Waxman-Markey bill.

75 J de Melo and N Mathys (2010), 'Trade and Climate Change: The Challenges Ahead', at 36.

countries' jurisdiction to regulate environmental matters and the use of technologies.[76]

While the use of npr-PPMs remains a politically sensitive issue, a more tolerable approach to npr-PPMs is emerging in the judicial field. It seems that WTO case law is moving towards the position that admits that npr-PPMs are not illegal so long as they apply on a non-discriminatory basis.[77] If they are found to discriminate against imports, they may still be justified under the exceptions provided in GATT Article XX for measures taken with moral, health, environmental and other public policy objectives.[78]

5.1 The Inclusion of Imports in an ETS

Notwithstanding the new approach seeking to accommodate npr-PPMs under WTO rules, the PPM-nature of BCAs raises many questions as to the ability of BCAs to meet the requirements for border adjustment.[79] An important question is whether the ETS requirement to surrender emissions allowances can be considered to be an indirect tax, since only indirect taxes (that is, taxes applied to products) can be adjusted at the border.[80] This question can further be split into two sub-questions: whether the requirement to surrender emissions allowances is a tax, and if so, whether it is an indirect tax. The first question is important because the rules on border adjustment vary with the type of the measure. If the ETS requirement is a tax (or a

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[76] See e.g. S Charnovitz (2002), 'The Law of Environmental "PPMs" in the WTO: Debunking the Myth of Illegality', at 62–3. Besides the coercive effect on policies of other countries, which enjoy sovereign rights and regulatory autonomy under international public law, PPMs inflict considerable costs on exporting countries. To meet the requirements of an importing country, exporting countries have to make investments in technological modernization and upgrading of their standards. It is therefore not surprising that PPMs with extraterritorial jurisdiction are opposed, especially by developing countries. See e.g. the Statement of Mexico, the complaining party in the *Tuna/Dolphin* dispute before the panel: *US-Tuna (Mexico)*, GATT Panel report (unadopted), para 3.31.

[77] K Holzer (2014), *Carbon-Related Border Adjustment and WTO Law*, at 96–8; N Bernasconi-Osterwalder et al. (2006), *Environment and Trade: A Guide to WTO Jurisprudence*, at 205–18.

[78] This approach was developed in the wake of the US-Shrimp dispute. See also J Frankel (2005), 'Climate and Trade: Links between the Kyoto Protocol and WTO'.

[79] For a more comprehensive overview of legal issues of BCAs, see K Holzer (2014), *Carbon-Related Border Adjustment and WTO Law*.

[80] It follows from the text of the legal provisions of GATT Art. II:2(a), Art. VI:4 and Ad Art. XVI. See also GATT, Report by the Working Party on Border Tax Adjustments, L73464, 2 December 1970, BISD 18S797, para. 14.

charge), when adjusted on importation, it falls under GATT Article III:2 and, accordingly, the tax burden for imports must be the same as for the like domestic products.[81] If it is a domestic regulation, when applied to imports, it falls under the provisions of GATT Article III:4 and, accordingly, the treatment of like imported products may sometimes be different but never less favourable.[82] The second question matters because only indirect taxes, that is, taxes levied on products and not on producers, are adjustable.[83]

There is no consensus on whether the ETS requirement to surrender emissions allowances can qualify as a tax. Referring to the OECD definition of a tax being 'an unrequited payment to the government' or 'a compulsory contribution imposed by the government for which taxpayers receive nothing identifiable in return', both Javier De Cendra and Joost Pauwelyn consider an emissions allowances requirement to be a tax eligible for adjustment at the border.[84] They submit that an emissions allowance requirement can qualify as a tax even if emissions allowances are distributed for free, given that allowances always have an opportunity cost.[85] This contrasts with Lorand Bartels' argument that the ETS requirement is a domestic regulation (that is, a non-fiscal measure), for it cannot be an unrequited payment so long as emissions allowances can be sold in the carbon market and bring revenue.[86] This echoes with the ECJ finding that the ETS requirement constitutes a market-based measure, rather than a tax.[87]

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[81] *Japan-Alcoholic Beverages II*, AB report, at 18 and 22.

[82] *Korea-Various Measures on Beef*, AB report, para. 137.

[83] This follows from the text of the provisions relevant for border adjustment, including GATT Art. II:2(a), Art. VI and Ad Art. XVI. See also GATT Working Party Report on Border Tax Adjustments, L/3464, 2 December 1970, para. 14.

[84] J de Cendra (2006), 'Can Emissions Trading Schemes be Coupled with Border Tax Adjustments? An Analysis vis-à-vis WTO Law', at 136, and J Pauwelyn (2007), 'U.S. Federal Climate Policy and Competitiveness Concerns: the Limits and Options of International Trade Law', at 21–2. Also supported by R Ismer and K Neuhoff (2008) 'International Cooperation to Limit the Use of Border Adjustment', at 8.

[85] J Pauwelyn (2007) 'U.S. Federal Climate Policy and Competitiveness Concerns: the Limits and Options of International Trade Law', at 22.

[86] L Bartels (2011), 'The Inclusion of Aviation in the EU ETS: WTO Law Considerations', at 4.

[87] The ECJ did not find the ETS requirement to be a tax for the following reasons. First, a tax has a fixed rate, whereas the costs of emissions allowances for a firm vary depending on the number of allowances initially allocated for free and the market price of an allowance if the firm purchased additional allowances to comply with its obligations under the ETS. Second, unlike a tax, the ETS requirement is not primarily intended to generate revenue in the budget. See ECJ Case C-366/10, *Air Transport Association of America and others v. Secretary of State for Energy and Climate Change* [2011] ECR I-1133, paras. 142–144, 147.

If the ETS requirement is a domestic regulation (that is, a non-fiscal measure), the discussion about direct and indirect taxes becomes no longer relevant[88] because the analysis shifts to the examination of the application of the measure. The thorny issue here is the likeness of carbon-intensive and low-carbon products, as the non-discrimination rules (the most favoured nation (MFN) and national treatment (NT) principles) governing the application of BCAs to imports are relevant only for like products.[89] Traditionally, when determining whether products are like, WTO adjudicative bodies look at whether the competitive relationship between products is strong enough, and whether products are characterized by the same physical characteristics, end-uses, consumer preferences and tariff codes.[90] Based on this traditional 'like product' test, it is difficult to refer to the imported steel produced in an open hearth process (with a higher GHG emission footprint) and the domestic steel produced with the electric arc technology (with a lower GHG emission footprint) as unlike products. Yet, the increasing consumer awareness of climate change and their growing preferences for products with a low carbon footprint may impact the competitive relationship and render carbon-intensive and low-carbon products unlike.[91] In that case, the application of BCAs would not trigger a violation of the MFN and NT provisions.

Under present circumstances, however, a more likely scenario is that the use of BCAs would need justification under the general exceptions of GATT Article XX.[92] BCAs fit in the scope of paragraph (b), which provides justification for measures 'necessary to protect human, animal or plant life or health', and paragraph (g), which exempts from compliance measures 'relating to the

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[88] Opinions vary as to whether the ETS requirement can be attributed to indirect taxes. For the arguments in favour, see J Hoerner and F Muller (1996), 'Carbon Taxes for Climate Protection in a Competitive World'; J Pauwelyn (2007), 'U.S. Federal Climate Policy and Competitiveness Concerns: the Limits and Options of International Trade Law'; P Wooders and A Cosbey (2010) 'Climate-Linked Tariffs and Subsidies: Economic Aspects (Competitiveness and Leakage)'. For the arguments against, see G Goh (2004), 'The World Trade Organization, Kyoto and Energy Tax Adjustments at the Border'; P Low et al. (2010), 'The Interface between the Trade and Climate Change Regimes: Scoping the Issue'.

[89] GATT Art. I and Art. III. If the ETS requirement is a regulation, it might also be subject to the provisions of the WTO's Agreement on Technical Barriers to Trade (TBT Agreement). See K Holzer (2014), *Carbon-Related Border Adjustment and WTO Law*, at 140–45.

[90] See e.g. *EC-Asbestos*, AB report, para. 101; *Philippines-Distilled Spirits*, AB report, paras 119 and 131.

[91] T Cottier et al. (2014), 'Differential Taxation of Electricity', at 32–3.

[92] See e.g. J Pauwelyn (2007), 'U.S. Federal Climate Policy and Competitiveness Concerns: the Limits and Options of International Trade Law'.

conservation of exhaustible natural resources . . .'. Yet, the major challenge for justification of BCAs is the conditions of the chapeau of Article XX. The chapeau requires that a measure does not constitute 'a means of arbitrary or unjustifiable discrimination between countries where the same conditions prevail, or a disguised restriction on international trade'. In simple terms it means that it would not tolerate any differences in the design and the implementation of a measure made in relation to countries, where conditions relevant for the policy objective pursued by the measure are the same.[93] BCAs should therefore be flexible enough to exclude imports from countries that pursue emissions reduction policies, no matter whether in the form of an ETS, a carbon tax or any other measure. No single recipe however exists for the application of BCAs in compliance with WTO rules. The WTO law-consistency of the inclusion of imports in an ETS will be decided by WTO adjudicative bodies on a case-by-case basis, and the outcome of each case will be predetermined by the concrete design of the scheme.

5.2 Export Rebates

In addition to the inclusion of imports in a national ETS (import-side BCA), the playing field between domestic and foreign producers could also be levelled through export rebates. This could be done through the reservation of some percentage of emissions allowances in the total allocation of emissions allowances and recycling them to firms on exportation. This approach was discussed in the framework of a BCA scheme proposal called 'The Foreign Allowance Import Requirement' ('FAIR').[94] From 2014 onwards, 2 percent of the total number of emissions allowances issued under phase III of the EU ETS would be set aside and then allocated as emissions allowance rebates to EU exporters. Alternatively, export rebates could be provided through the reimbursement of costs of emissions allowances.[95]

Unlike the inclusion of imports in an ETS, which mainly falls under

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[93] *Brazil-Retreaded Tyres*, AB report, para. 227; *EC-Seal Products*, AB report, paras. 5.299–5.300. See also R Quick (2000), 'The Community's Regulation on Leg-Hold Traps: Creative Unilateralism Made Compatible with WTO Law through Bilateral Negotiations?', at 254; J Pauwelyn (2007), 'U.S. Federal Climate Policy and Competitiveness Concerns: the Limits and Options of International Trade Law', at 43.

[94] See Art. 29:5 of the 2007 version of draft Proposal amending the EU ETS Directive.

[95] In that case, the calculation of adjustment level may present a problem, given that some emissions allowances were distributed for free and others were purchased on a secondary market at various prices.

the GATT non-discrimination rules (MFN and NT), the adjustment of the ETS requirement on exportation will primarily be regulated by WTO subsidy rules. Like the border adjustment on importation, the border adjustment on exportation is possible only for indirect taxes.[96] Rebates of direct taxes will be deemed to constitute a prohibited export subsidy.[97] As already discussed, it is uncertain whether the ETS requirement can qualify as an indirect tax or a tax at all. If the ETS requirement qualifies as a domestic regulation, its WTO-compliance will be assessed against general rules of the WTO's Agreement on Subsidies and Countervailing Measures (ASCM).

If the ETS requirement is found to be an indirect tax, it will be eligible for adjustment on exportation subject to the requirement that they are given 'not in excess' of surrendered allowances or incurred costs.[98] Exports rebates would fail to meet the 'not in excess' requirement under an ETS with the free allocation of emissions allowances. It also seems difficult not to provide exports rebates 'in excess' under an ETS with auctioning of allowances, given that allowances can also be acquired at various prices on a secondary market.[99]

The issue of likeness of carbon-intensive and low-carbon products can also be relevant. The Note to Article XVI of the GATT and footnote 1 to the ASCM do not consider the 'exemption of an exported product from duties or taxes borne by the *like* product when destined for domestic consumption . . .' an export subsidy (emphasis added). As carbon-intensive and low-carbon products may qualify as like, to avoid an allegation of export subsidy, export rebates need to be given at a rate that corresponds to the lowest level of emissions in the industry (for example, based on the benchmark of the best available technology[100]).

Moreover, the issue of a prohibited export subsidy may arise, if rebates on exportation are given selectively to certain sectors, rather than to all the sectors covered by an ETS. The coverage of sectors by export rebates should therefore correspond to the coverage of sectors by an ETS.[101]

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96 See e.g. GATT Ad Art. XVI.

97 *US-FSC*, Panel report, paras. 7.108 and 7.131.

98 Both Ad Art. XVI of the GATT and footnote 1 of the ASCM stipulate that '. . . the remission of such duties or taxes in amounts not in excess of those which have accrued, shall not be deemed to be a subsidy'.

99 M Genasci (2008), 'Border Tax Adjustments and Emissions Trading: The Implications of International Trade Law for Policy Design', at 39–41.

100 R Ismer and K Neuhoff (2008), 'International Cooperation to Limit the Use of Border Adjustment'.

101 This match is also important for the import-side adjustment. If the sectorial coverage for the inclusion of imports in an ETS does not correspond to the ETS

It is important to note that the WTO rules applicable to border adjustment do not require implementing a BCA scheme symmetrically on importation and exportation.[102] A country would be free to apply the adjustment only on exportation, combine export rebates with the inclusion of imports in an ETS, or limit a BCA scheme only to the application of the ETS requirement to imports.[103]

Besides the uncertainty about the consistency of export-side BCAs with the WTO rules on subsidies, both Gavin Goh and Julia Reinaud allude to the problem of environmental integrity of export rebates.[104] They argue that export rebates of emissions costs are not consistent with the 'polluter pays' principle and contrary to the climate policy objective of emissions reduction.[105] This may create a hurdle for justification of a BCA scheme consisting of both import-side and export-side border adjustment under the environmental exception clause of GATT Article XX.[106] At the same time, one could argue that the purpose of export rebates is to prevent carbon leakage and thus reduce global emissions.[107] In this sense, export rebates contribute to the environmental objectives.

In sum, the adjustment of ETS requirement on exportation, be it in the form of remission of emissions allowances or compensation of emissions allowances costs, is characterized by legal uncertainty. It raises the issue of a prohibited export subsidy and reduces the chances for a BCA scheme to be justified under the GATT exceptions as a measure taken for environmental purposes.

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coverage of domestic industries, it will entail a violation of the national treatment principle.

[102] K Holzer (2014), *Carbon-Related Border Adjustment and WTO Law*, at 78–80.

[103] Ibid.

[104] G Goh (2004), 'The World Trade Organization, Kyoto and Energy Tax Adjustments at the Border', at 405; J Reinaud (2009), 'Would Unilateral Border Adjustment Measures be Effective in Preventing Carbon Leakage?' at 74.

[105] Ibid. The reimbursement of emissions costs can encourage the expansion of carbon-intensive production for exports.

[106] G Hufbauer et al. (2009), *Global Warming and the World Trading System*, at 69.

[107] Ecoplan et al. (2013), 'Border Tax Adjustments: Can Energy and Carbon Taxes be Adjusted at the Border?' at 99.

6. INTERNATIONAL EMISSIONS TRADING

Having examined the WTO-consistency of ETS design features and flanking support schemes, we turn in this section to discuss WTO law issues that might arise from emissions trading taking place among countries. International emissions trading can emerge as a result of linking arrangements among different national ETSs or as a flexibility mechanism under existing or potential international climate agreements.

As ETSs are spreading among countries,[108] linkages among different ETSs through the acceptance of allowances from different jurisdictions could be established and a global carbon market could emerge.[109] The EU ETS legislation foresees linking of the EU ETS with ETSs of countries that undertook quantified emissions reduction commitments under the Kyoto Protocol (Annex B countries) through the mutual recognition of emissions allowances.[110] The EU ETS is already linked with the Norwegian ETS and preparations are being made to link it with the Swiss ETS.[111] Moreover, the EU ETS is linked to the emissions credits systems under the Kyoto Protocol. Credits earned by companies under Clean Development Mechanism (CDM) and Joint Implementation (JI) projects are accepted in a limited quantity for achieving compliance under the EU ETS.[112]

Linking of ETSs is crucial for achieving the maximum efficiency of emissions reductions and minimizing carbon leakage.[113] However it is not an easy task in light of differences between ETSs in terms of size, sectorial coverage, stringency of emissions reduction targets and other design features.[114] To preserve the environmental integrity of its ETS, a country would need to use certain conditions or criteria for linking. Countries

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[108] Besides the EU, ETSs have also been established in Switzerland, Norway, New Zealand, Australia and some US states and Canadian provinces. See A Tuerk et al. (2009), 'Linking Emissions Trading Schemes', at 7. See also Chapter 13 on linking of emissions trading systems contained in this volume.

[109] For more details on ETS linking arrangements, see Chapter 13 by Andreas Tuerk and Andrej F Gubina in this volume.

[110] See Art. 25.1 of Directive 2003/87/EC.

[111] I Jegou and L Rubini (2011), 'The Allocation of Emission Allowances Free of Charge: Legal and Economic Considerations', at 8. Initially the EU had a highly ambitious goal to create a common carbon market of countries that are members of the Organization for Economic Cooperation and Development (OECD) by 2015. See A Tuerk et al. (2009), 'Linking Emissions Trading Schemes', at 1.

[112] See Directive 2004/101/EC ('EU ETS Linking Directive').

[113] A Tuerk et al. (2009) 'Linking Emissions Trading Schemes', at 4–5.

[114] Some schemes are based on an absolute cap, while others use the benchmark of emissions intensity; some are based on the free allocation of emissions allowances, while others foresee allocation through an auction.

may also condition the admittance of emissions allowances on countries' commitments under the Kyoto Protocol or a post-Kyoto international climate agreement.[115] Such conditions could be established either unilaterally through the inclusion of the clause in the ETS legislation specifying the condition for acknowledging other countries' emissions allowances, or bilaterally/plurilaterally through the conclusion of a mutual recognition agreement (MRA) over the ETS-related issues with other countries.[116]

Restrictions on admittance of emissions allowances issued in other jurisdictions could potentially trigger violations of WTO non-discrimination rules, particularly the MFN principle. Depending on whether emissions allowances could qualify as commodities or services or not, violations would be direct or indirect. For instance, if an emissions allowance is a financial service (for example, a 'negotiable instrument'), restrictions on the eligibility of emissions allowances can be challenged under the market access provisions of GATS Article XVI:2(b) if a country imposing such a restriction undertook in this sector a specific commitment not to limit market access on the basis of 'the total value of service transactions or assets in the form of numerical quotas'.[117] Yet, even if emissions allowances do not fall within the scope of the WTO Agreement, origin-based restrictions on the admittance of allowances have the potential to hamper sales of products or services and thus may entail violations of the MFN or NT rules of GATT or GATS.

However, if restrictions on the eligibility of emissions allowances for compliance were based on some objective criteria that are fixed in MRAs, it is unlikely that they would raise issues under the MFN and NT rules.[118] Furthermore, restrictions based on objective criteria, such as stringency of emissions caps, can be justified under the general exceptions of GATT or GATS as measures taken for health or environmental purposes or with the objective to secure compliance with domestic laws that are not themselves inconsistent with WTO rules.[119]

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115 J Werksman and J Lefevere (1999), 'WTO Issues Raised by the Design of an EC Emissions Trading System', at 9.

116 A Tuerk et al. (2009), 'Linking Emissions Trading Schemes', at 2–3.

117 J Werksman and J Lefevere (1999), 'WTO Issues Raised by the Design of an EC Emissions Trading System', at 10.

118 I have to admit though that the compliance of MRAs themselves with the MFN principle can be a matter of discussion. See e.g. W Davey and J Pauwelyn (2000), 'MFN Unconditionality: A Legal Analysis of the Concept in View of its Evolution in the GATT/WTO Jurisprudence with Particular Reference to the Issue of "Like Product"'. However, entering into MRAs is widespread practice and no complaints have been made so far in the WTO.

119 J Werksman and J Lefevere (1999), 'WTO Issues Raised by the Design of an EC Emissions Trading System', at 17–19.

Moreover, an international emissions trading scheme can also be established under an international climate agreement.[120] It will imply a state-to-state transfer of units within countries' emissions caps (emissions reduction targets). Such an option was available for Annex B Parties of the Kyoto Protocol (that is, countries with emissions reduction commitments) under the first commitment period. Under Article 17 of the Kyoto Protocol, countries that have spare emission units (that is, emissions amounts permitted to them but not used) can sell their excess of emissions rights to countries that experience difficulties to meet their emissions reduction commitments.

As the final contours of the post-Kyoto international climate regime are not yet set and it is not clear whether an international emissions trading system will be established under an international agreement, it is difficult to draw definitive conclusions about the WTO-compatibility of this mechanism.[121] It is also uncertain whether a state-to-state emissions trading scheme will fall within the scope of the WTO Agreement. On the one hand, it can be argued that a state-to-state transfer of emissions units merely implies a re-allocation of sovereign obligations under an international treaty.[122] On the other hand, if state-to-state emissions trading affects the competitive relationship between domestic and foreign producers (for example, where it involves the exchange of credits among private legal entities, like the use of credits earned under the CDM and JI projects, or affects the price of allowances of a domestic ETS), it could become an issue of scrutiny under WTO rules.

7. CONCLUSIONS

WTO compliance is an important consideration in the debate regarding design elements and flanking support schemes of an ETS. Considerable research has been devoted to this area. However, no study can predict with confidence the outcome of scrutiny of an ETS under WTO law. There

[120] At the time of writing, an international climate agreement to replace the Kyoto Protocol is being negotiated by UNFCCC parties. It is expected to be signed at the 21st Conference of Parties (COP) in Paris in December 2015 and come into force in 2020.

[121] For the possible content of a post-2020 climate agreement to be signed in 2015 in Paris, see E Haites et al. (2014), 'Possible Elements of a 2015 Agreement to Address Climate Change'.

[122] J Werksman and J Lefevere (1999), 'WTO Issues Raised by the Design of an EC Emissions Trading System', at 6.

are a few reasons for that. First, an ETS does not have a fixed design and design elements significantly vary across ETS schemes. Second, ETS-related issues have never been raised in WTO disputes and have not been tested thus far. This adds uncertainty to the analysis of compliance of emissions trading with WTO rules.

The design measures of an ETS that seem most likely to be challenged under WTO law include the free allocation of emissions allowances, recycling of ETS revenues to domestic producers, the inclusion of imports in an ETS and emissions allowance rebates on exportation. They raise issues under the GATT non-discrimination rules and ASCM disciplines on subsidies. The availability of exceptions for justification of these measures is therefore of great importance.

WTO rules are also relevant in the context of international emissions trading, where national ETSs get linked to each other through the mutual recognition of emissions allowances, so that allowances issued in one jurisdiction are accepted for compliance under an ETS in another jurisdiction. Agreeing on the common design features of ETSs of different countries and bringing them into compatibility with each other presents the main challenge of ETS linking arrangements. WTO law would apply in this case to the terms of the mutual recognition of emissions allowances of different origin. It would ensure that conditions for the acceptance of emissions allowances do not negatively affect the competitive relationship between domestic and foreign producers or service suppliers.

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