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# The high cost of cost efficiency: A critique of carbon trading


Tese de doutoramento em Governação, Conhecimento e Inovação,
orientada por Prof. Doutor José Maria Castro Caldas e
apresentada à Faculdade de Economia da Universidade de Coimbra

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# The high cost of cost-efficiency

## A critique of carbon trading

Tese de Doutoramento em Economia (Programa de Doutoramento em Governação, Conhecimento e Inovação), apresentada à Faculdade de Economia da Universidade de Coimbra para obtenção do grau de Doutor

Orientador: Prof. Doutor José Maria Castro Caldas

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*Para a Noa, nascida no meio da aventura que foi esta tese.*

*Cover photo: A.T. Biopower station, accredited as a biomass energy project of the Clean Development Mechanism. Photo by Tamra Gilbertson, published by Carbon Trade Watch. Used with permission.*

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## Resumo

O comércio de carbono, enquanto uma política climática de mercado que permite aos poluidores cumprir com compromissos de redução de emissões recorrendo a direitos de poluição transacionáveis, é apresentado pelos seus proponentes como a alternativa mais eficiente para a mitigação das alterações climáticas, enquanto oponentes contrapõem que o argumento baseado na custo-eficiência negligencia os prejuízos que resultam da mercantilização do carbono. Esta tese contribui para este debate, que é fundamental para o futuro das políticas ambientais, expondo os custos sociais do comércio de carbono e posicionando-se contra a inclusão do comércio de carbono no leque de políticas climáticas. A argumentação aqui desenvolvida é baseada nas contribuições teóricas sobre os custos sociais de atividades privadas e conflitos de valores, assim como perspetivas críticas sobre a neoliberalização da natureza e os limites do mercado.

O comércio de emissões foi primeiramente proposto como uma alternativa às taxas ambientais pigouvianas maximizadoras da eficiência. Baseado na perspetiva sobre custos sociais assente em direitos de propriedade, o comércio de emissões permitiria ao regulador escapar à impossível tarefa de calcular um nível ótimo de poluição e providenciaria em alternativa uma forma custo-eficiente de atingir um nível de poluição determinado exogenamente. Esta transição teórica permitiria à Economia centrar-se na discussão dos melhores meios par atingir fins dados e esquivar-se à discussão dos fins. A dicotomia fins-meios, no entanto, não se aplica fora da teoria económica, tal como a descrição do comércio de emissões como uma alternativa simples e eficiente à regulação direta. Como a experiência dos EUA com o comércio de emissões demonstra, criar mercados para direitos de poluição transacionáveis requer investimento governamental num aparato regulatório que não é menos complexo do que é requerido pela regulação direta ou pela taxação. Esta experiência também ilustra o quanto a alegada eficiência dos mercados de emissões é resultado do seu fraco desempenho ambiental e da sua desconsideração pela justiça social e pela participação democrática.

Os mercados de carbono criados ao abrigo do Protocolo de Quioto suscitam problemas adicionais. Comparados com os esquemas de "limitação e comércio" baseados num único poluente e um número restrito de fontes, esquemas como o Sistema Europeu de

Comércio de Licenças de Emissão são mais complexos e requerem maior intervenção governamental. Para mais, instrumentos flexíveis como o Mecanismo de Desenvolvimento Limpo permitem aos países industrializados poluir além dos seus compromissos de emissões e suscitam preocupações com a integridade disputável de metodologias que contabilizam reduções de emissões de projetos de compensação em relação a um cenário de referência arbitrário. O fraco desempenho ambiental destes esquemas é ilustrado pela sua incapacidade de incentivar a descarbonização, enquanto distribuem rendas aos poluidores e criam novas fontes de corrupção.

Estas questões não são redutíveis a discussões sobre procedimentos contabilísticos e outras tecnicalidades. Abrindo a "caixa negra" da quantificação e comensuração do carbono, é revelado que os seus cálculos marginalizam incertezas relevantes e assumem um grau de precisão que o conhecimento científico e a tecnologia não podem providenciar no presente. No entanto, dado que contabilizar aumentos e reduções de emissões requer decisões políticas sobre o que deve ser contabilizado, qual a métrica relevante e o que é um grau de incerteza aceitável, avanços científicos e tecnológicos não são condição suficiente para que seja possível produzir os números inequívocos que o comércio de carbono requer.

Indo mais longe na discussão sobre as implicações da comensuração e abstração de carbono, esta tese apresenta um argumento contra a inclusão do comércio de carbono no leque de políticas climáticas, baseado em quatro críticas normativas. Com o apoio da literatura crítica, é defendido que o comércio de carbono é ineficaz, antidemocrático, injusto e antiético e que, por estas razões, só pode ser considerado como uma política custo-eficiente quando os seus custos sociais são ignorados. Um argumento contra o reformismo do comércio de carbono é então apresentado mostrando como tentar contrariar os efeitos negativos dos mercados de carbono através de restrições ao comércio conduz à erosão destes mercados. Uma melhor alternativa é o apoio a políticas climáticas que fomentam uma pluralidade de valores e providenciam benefícios sociais.

A tese conclui defendendo uma mudança no debate sobre política climática no sentido da discussão dos valores que são fomentados ou prejudicados por cada política. Um enquadramento geral é proposto que respeita o pluralismo de valores e reconhece conflitos entre valores incomensuráveis, o que não é compatível com políticas de mercado.

Palavras chave: Comércio de carbono, mercados de emissões, créditos de carbono, política climática, Protocolo de Quioto

**Abstract**

Carbon trading, as a market-based climate policy that allows polluters to comply with emissions reductions commitments with tradable pollution rights, is presented by its proponents as the most cost-efficient alternative for climate change mitigation, while critics counter that the cost-efficiency argument ignores the harms that result from commodifying carbon. This thesis contributes to this debate, which is fundamental for the future of environmental policies, by exposing the social costs of carbon trading and making the case against its inclusion in the climate policy-mix. The argument developed here draws from theoretical contributions on the social costs of private activities and on value conflicts, as well as critical perspectives on the neoliberalization of nature and the limits of the market.

Emissions trading was firstly proposed as an alternative to efficiency-maximizing or pigouvian environmental taxation. Based on the property rights approach to social costs, emissions trading would allow regulators to escape the impossible task of calculating the optimal level of pollution and offer instead a cost-efficient way to achieve an exogenously determined level of pollution. This theoretical shift would allow economics to be centred on discussing the best means to achieve given ends and relived it of discussing ends. The ends-means dichotomy, however, does not hold outside textbook economics, as well as the description of emissions trading as a simple and efficient alternative to direct regulation. As the US experience with emissions trading shows, creating markets for tradable pollution rights requires government investment in a regulatory apparatus that is no less complex than what is required for direct regulation or taxation. This experience also illustrates how the purported efficiency of emissions trading systems is a flip side of their weak environmental performance and their disregard for social justice and democratic participation.

Carbon trading schemes created under the Kyoto Protocol raise additional problems. Compared to “cap and trade” schemes based on a single pollutant and a restricted number of sources, schemes like the EU Emissions Trading System are more complex and require further government intervention. Furthermore, flexibility instruments like the Clean Development Mechanism allow industrialized countries to pollute beyond their emissions commitments and raise issues with the disputable integrity of methodologies that account for emissions reductions from offset projects relative to an arbitrary baseline. The dismal performance of these schemes is illustrated by their inability to provide an incentive to decarbonization, while distributing rents to polluters and creating new sources of corruption.

These issues are not reducible to discussions on accounting procedures and other technicalities. Opening the “black box” of carbon quantification and commensuration reveals that its calculations sideline relevant uncertainties and assume a degree of accuracy that scientific knowledge and technology cannot deliver in the present. Yet, since accounting for emissions increases or reductions requires political decisions on what is to be accounted for, what is the relevant metric and what is an acceptable degree of uncertainty, further scientific and technological developments are not enough to make it possible to produce the unambiguous numbers that carbon trading requires.

Going further on the discussion of the implications of carbon commensuration and abstraction, this thesis presents an argument against the inclusion of carbon trading in the climate policy-mix based on four normative critiques. With the support of critical literature, it is argued that carbon trading is ineffective, undemocratic, unjust and unethical and that, for these reasons, it can only be considered as a cost-effective policy when its social costs are ignored. An argument against carbon trading reformism is then presented by illustrating how trying to mitigate the negative effects of carbon markets by imposing restrictions on trading leads to the erosion of these markets. A better alternative is claimed to be supporting climate policies that foster a plurality of values and deliver social benefits.

The thesis concludes by advocating a shift in the climate policy debate to a discussion on the values that are fostered or hindered by each policy. A general framework is proposed that respects value pluralism and acknowledges conflicts between incommensurable values, which is not compatible with market-based policies.

Keywords: Carbon trading, cap and trade, offsets, climate policy, Kyoto Protocol

## Contents

**List of acronyms and symbols:**

AAU – Assigned Amount Unit

AIJ – Activities Implemented Jointly

AQMP – Air Quality Management Plan

ARP – Acid Rain Program

ASC – Area Source Credit

BACT – Best Available Control Technology

BARCT – Best Available Retrofit Control Technology

BDI – Federation of German Industries

Btu – British thermal unit

CAA – Clean Air Act

CAIR – Clean Air Interstate Rule

CAN–E – Climate Action Network – Europe

CCL – Climate Change Levy

CDM – Clean Development Mechanism

CEMS – Continuous Emission Monitoring System

CER – Certified Emission Reduction

$CH_4$ – methane

CITL – Community Independent Transaction Log

CO – carbon monoxide

$CO_2$ – carbon dioxide

$CO_2eq$ – carbon dioxide equivalent

COP – Conference of Parties

CSAPR – Cross–State Air Pollution Rule

DNA – Designated National Authority

DOE – Designated Operational Entity

EB – Executive Board

EC – European Commission

ECCP – European Climate Change Programme

EDI – Economic Damage Index

EE – Environmental Economics

EEA – European Environment Agency

EPA – Environmental Protection Agency

ERC – Emissions Reduction Credit

ERU – Emission Reduction Unit

ETG – Emissions Trading Group

ETP – Emissions Trading Program

EU ETS – European Union's Emissions Trading System

EUA – European Union Allowance

EUAA – European Union Aviation Allowance

EUTL – European Union Transaction Log

GAO – Government Accountability Office

GCC – Global Climate Coalition

GETS – Greenhouse Gas and Energy Trading Simulations

GHG – greenhouse gas

gpg – grams per gallon

gplg – grams per leaded gallon

GTP – Global Temperature Change Potential

GWP – Global Warming Potential

HC – hydrocarbons

HFCs – hydrofluorocarbons

IATA – International Air Transport Association

ICAO – International Civil Aviation Organization

ICCP – International Climate Change Partnership

IPCC – Intergovernmental Panel on Climate Change

JI – Joint Implementation

KP – Kyoto Protocol

LAER – Lowest Achievable Emissions Rate

LDC – Least Developed Country

LULUCF – Land Use, Land Use Change and Forestry

MSERC – Mobile Source Emission Reduction Credit

MSR – Market Stability Reserve

$N_2O$ – nitrous oxide

$N_2O$ – nitrous oxide

NAAQS – National Ambient Air Quality Standard

NAMA – Nationally Appropriate Mitigation Actions

NAP – National Allocation Plans

NAPAP – National Acid Precipitation Assessment Program

NBP – $NO_x$ Budget Program

NER – New Entrants Reserve

NGO – Non–governmental organization

NMM – New Market Mechanism

$NO_x$ – nitrogen oxides

NSPS – New Source Performance Standard

$O_3$ – ozone

ODP – Ozone Depletion Potential

OECD – Organisation for Economic Co–operation and Development

OTC – Ozone Transport Commission

PDD – Project Design Document

PFC – Progressive Flow Control

PFCs – perfluorocarbons

$PM_x$ – particulate matter

PSD – Prevention of Significant Deterioration

RACT – Reasonably Available Control Technology

RECLAIM – Regional Clean Air Incentives Market

REDD+ – Reducing Emissions from Deforestation and Forest Degradation

RTC – RECLAIM Trading Credit

SC – Supervisory Committee

SCAQMD – South Coast Air Quality Management District

$SF_6$ – sulphur hexafluoride

SIP – State Implementation Plan

$SO_2$ – sulphur dioxide

$SO_x$ – sulphur oxides

UK ETS – United Kingdom's Emissions Trading System

UNFCCC – United Nations Framework Convention on Climate Change

UNICE – Union of Industrial and Employers' Confederation of Europe

VOC – Volatile Organic Compound

WCB – Water Control Board

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# 1 Introduction

It is no exaggeration to present climate change as one of the most important, if not the most important, crisis humanity faces. According to the scientific evidence compiled by the Intergovernmental Panel on Climate Change, global warming is unequivocal. The result of anthropogenic greenhouse gas (GHG) emissions, mostly from burning fossil fuels, climate change is a broad term that encompasses changes in precipitation, temperature and occurrence of extreme weather events, like cyclones, floods and heat waves, as well the sea level rise that follows diminishing snow and ice cover. Climate change negatively impacts agricultural productivity, human health, biodiversity and freshwater availability. The growing intensity and frequency of extreme weather events and the sea level rise compromise the survival of a growing number of people, affecting in particular those who cannot easily adapt due to poverty or discrimination and those that live in the most vulnerable areas of the planet (namely low-lying islands) (IPCC 2014). In parallel, the increasing uptake of carbon dioxide ($CO_2$) emissions by oceans leads to their acidification, which negatively impacts shell-forming marine organisms, including species of molluscs, corals and plankton (Doney *et al*., 2009).

Considering the dire and growing impacts of climate change, many of which will be irreversible for centuries, substantial and sustained reductions in greenhouse gas emissions are needed, as well as investment in adaptation measures that limit climate risks. Avoiding global warming is no longer possible but it is feasible to limit it to a mean temperature increase under 2ºC and avoid some of the worst impacts of climate change. Achieving this goal requires reducing GHG emissions by 40 to 70 percent by 2050, relative to 2010, targeting a long-run objective of zero emissions by 2100 (IPCC 2014). Reducing $CO_2$ emissions, in turn, requires reducing fossil fuel energy use in all productive sectors, including transportation, as well as reducing deforestation and agricultural practices that compromise land-based carbon sinks. Simultaneously, it is necessary to reduce emissions from other GHGs, namely methane ($CH_4$) emissions from landfills and livestock, nitrous oxide ($N_2O$) emissions from agriculture and other trace gas emissions from specific industrial processes.

The relevance of the climate crisis stems not only from the impacts of climate change but also from the major challenge that reducing GHG emissions represents, in particular considering that industrial societies are highly fossil fuel dependent and, therefore, that the production of goods and services inevitably leads to increased emissions. Overcoming fossil fuel dependence implies major changes in production and consumption, which are not reducible to technological innovations. Considering that these changes conflict with fossil fuel corporations' profitability, it is to be expected that these corporations form lobbying coalitions, to prevent climate action altogether or to condition it to their interests. Such coalitions can benefit from greater political clout compared to communities and social movements that advocate for leaving fossil fuels underground, in particular considering that those who are most affected by climate change are also among the poorest and most vulnerable and, therefore, among the most politically disenfranchised.

It is in this context that carbon trading emerged and became the main climate policy in industrialized countries. Carbon trading allows polluters, defined as firms, countries or even individuals, to replace reducing emissions for paying others to reduce their emissions. This is achieved by “cap and trade” and “baseline and credit” schemes. In the former case, the government distributes tradable pollution allowances according to a pre-determined emissions cap, relieving the regulator of the task of determining emissions limits for each polluter while burdening the regulator with new tasks related to carbon markets creation and regulation. In the latter case, polluters can increase their emissions by buying credits from offset projects that reduce emissions beyond a business as usual baseline. The Kyoto Protocol, as the international treaty that determines how climate change should be addressed, incorporates both types of schemes, allowing industrialized countries with emissions commitments to trade carbon allowances and to buy carbon credits generated in other countries, including countries without emissions commitments.

After more than a decade of Kyoto's carbon trading, the market-based climate policy has been touted by many as a cost-efficient alternative to other forms of regulation, while others point out at the rising GHG emissions and the many problems created by carbon trading as evidence supporting the case for its abandonment. This debate is fundamental not only for its implications to climate policy but also for its intersections with debates on other market-based environmental policies, namely renewable energy credits (Berry, 2002), energy efficiency credits (known as white certificates) (Giraudet *et al*., 2012), and markets in ecosystem services, namely in biodiversity offsets and water quality (Robertson, 2007; Walker *et al*., 2009). The present thesis aims to contribute to this debate by presenting a radical critique of carbon trading, which supports arguments for not including it on the climate policy-mix. The argument developed here draws from theoretical contributions on the social costs of private activities and on value conflicts, as well as critical perspectives on the neoliberalization of nature and the limits of the market.

To this end, a review of emissions trading theory will be presented, which shows how it was based mostly on a shift in economic theory away from taxation aimed at achieving an estimated optimal pollution level to property rights arrangements that minimize the cost of achieving an exogenously determined pollution level. This is followed by a review of emissions trading practice, starting with the early experiences in the US and proceeding with international carbon trading, which is contrasted with the theory to illustrate how the move from "blackboard economics" to "real world economics" is far from trivial. An analysis of the commensuration processes that establish equivalences across carbon emissions is then presented to expose the divergence between the accuracy and accountability in measurement activities that carbon trading demands and what science and technology can provide. Finally, a normative critique of carbon trading is developed, based on the social values that carbon markets disregard, and it is explained why this critique cannot be reduced to debates on market reforms.

In the first chapter, the intellectual roots of emissions trading are reviewed through the lens of the Coasean critique of the Pigouvian approach to social costs. The Pigouvian approach conceptualizes social costs as externalities, that is, external effects of productive activities for which no market price exists. It follows from this approach, applied to environmental problems, that externalities can be corrected through the taxation of pollution, which would bring it to its optimal level and thus maximize efficiency. The Coasean critique points out that, while pollution causes harm, the taxation of productive activities also does, and that the question that should be raised *a priori* is whether or not any intervention in the market is warranted. Underlining the role of property rights and transaction costs, the Coasean approach to social costs was one important stepping stone towards devising environmental policies based on tradable pollution rights. The first proposals for emissions trading in air and water pollutants could, then, be presented as a cost-effective alternative to direct regulation and taxation, given that decisions on pollution distribution and pricing would be transferred from governments to markets. Furthermore, emissions trading was purportedly simpler and more flexible than previous regulatory policies.

In the second chapter, empirical evidence on the performance of early emissions trading schemes is given. Before the Kyoto Protocol was signed, both emissions trading theory and practice was entirely US-based, so this chapter focuses on the US experience with emissions trading. The first attempts to introduce some flexibility in compliance with pollution limits are analysed, showing how nascent emissions trading schemes presented regulators with new problems and challenges. The chapter then proceeds to an empirical review of the Acid Rain Program, the first emissions trading scheme in the world, and of two regional emissions trading programs, the Regional Clean Air Incentives Market and the $NO_x$ Budget trading. After presenting data on their environmental performance and the history of their evolution, this review of emissions trading in the US finalizes with some lessons for environmental policy.

In the third chapter, the history and performance of Kyoto's carbon trading is reviewed. After presenting a history of how carbon trading was introduced in international climate negotiations and how it evolved, the chapter proceeds to present an empirical review of the two main carbon markets created following Kyoto: the Clean Development Mechanism (CDM) and the EU Emissions Trading System (ETS). The former allows industrialized countries to buy carbon credits generated from offset projects located in developing countries, which are supposed to be additional relative to business as usual and contribute towards sustainable development. The latter allows large polluters in the EU to trade emissions allowances, which have been mostly given away for free, as well as buy carbon credits from the CDM. Both markets were created following the successful efforts by some major polluters to prevent the implementation of new regulations or taxes and, unsurprisingly, have led to polluters being subsidized, instead of penalized, for their pollution, as well as created loopholes for polluters to evade emissions reductions commitments.

In the fourth chapter, the “black box” of carbon commensuration is opened and analysed, to highlight the kind of abstractions and calculations needed to create equivalences across different carbon emissions and inquire into the accuracy and accountability of the calculations. As with all emissions trading systems, a carbon trading system requires that carbon emissions can be measured accurately, to prevent that polluters, traders and other market actors can profit from measurement errors, as well as reduce ambiguities in evaluating compliance. The accuracy of carbon emission measurements is evaluated in this chapter by presenting an empirical review of measurement uncertainties, issues with the use of Global Warming Potentials (GWPs) to convert different GHGs into a common metric, the role of co-pollutants and the uncertainties, inaccuracies and indeterminacies inherent to the inclusion of land-use emissions in carbon inventories. The review shows how carbon trading requires the use of complex calculations, which hide inherent uncertainties that reflect not only lack of scientific knowledge but also the impossibility of predicting social futures and are supported by unjustified political assumptions.

In the sixth chapter, a critique of carbon trading is presented. This critique is based on a taxonomy of normative critiques constructed from the critical literature. It upholds that carbon trading is *ineffective*, since it does not promote the kind of changes needed to decarbonize production, *unjust*, since it aggravates existing intragenerational and intergenerational inequities, *undemocratic*, since it transfers the power to make decisions on carbon emissions reductions and the accountability of climate policy over to polluters and other carbon market actors, and *unethical*, since it turns polluting the common environment from a wrong into a right. Proceeding into a discussion on the possibilities of addressing these critiques through changes in carbon markets design, the prospects of finding an idealized carbon trading system are contrasted both with the impossibility of addressing some critiques through reforms and the trade-off between carbon trading's cost-efficiency and the restrictions to trading that are inherent to design changes.

The thesis concludes by arguing that carbon trading has failed in all of its purported objectives, while being successful at blocking the necessary transition to a low-carbon development path. These considerations, along with the incompatibility between carbon trading and other climate policies and its social costs, justify dropping carbon trading from the policy-mix. Further considering that climate policy decisions are just as much about what alternatives are chosen as they are about what alternatives are on the menu, the question that climate policy theory has to answer is what policies are worthy of being pursued and what policies are not. The critique of carbon trading presented here can, then, be used as a basis for a new policy framework that respects value pluralism, instead of taking efficiency as the overriding societal goal to which all others must cede.

# 2 The intellectual roots of emissions trading: Revisiting the Coase vs. Pigou debate on social costs

"Unless I am very much mistaken, markets *can* be used to implement any anti-pollution policy that you or I can dream up. The only real question is how much you or I are willing to pay in order to reduce pollution" (Dales, 1968b: 100)

The adoption of emissions trading as a means of compliance with the Kyoto Protocol and the growing pressure to create new markets in environmental services based on carbon markets shows how an idea that was somehow esoteric in the 1960s became very popular in the political arena. Trading rights to pollute, as proposed by Crocker (1966) and Dales (1968) not only is on the menu of instruments to address the social costs of production but is further the preferred instrument of many economists and policy-makers.

Emissions trading theory questioned the Environmental Economics tradition of framing pollution as an externality, a market failure that can be corrected through the efficient taxation of pollution. This tradition of understanding social costs as a case of missing prices drew its inspiration from Pigou (1932), who argued for government intervention to increase social welfare and opposed the *laissez faire* ideals of free-market libertarianism. Against such elegant solution for the problem of social cost, Coase (1960) famously argued that its efficiency claims are unwarranted, and proposed instead a cost-benefit analysis of different institutions to deal with social costs.

The Coasean critique of Pigou's Welfare Economics became the cornerstone of social cost theories that emphasized the role of property rights and/or supported an anti-regulatory agenda. Even if some of these theories are based on a somewhat misguided reading of both Coase and Pigou, this does not diminish the relevance of the Coasean critique to debates on instrument choice for environmental policy.

This chapter reconstructs the intellectual history of emissions trading using the Coasean critique as a starting point. Alternative critiques, such as Kapp's (1978) institutional analysis of social costs as pervasive unpaid costs of production that are borne by the population, will not be addressed here. While such theories had their role in fostering critical perspectives on social costs, namely in the Ecological Economics tradition, their contribution to emissions trading theory is negligible.

We begin by presenting Pigou's approach to social costs, based on his contribution to Welfare Economics, which is contrasted with the modern theory of externalities and the Environmental Economics concept of Pigouvian taxes. We then proceed to present Coase's approach to social costs and the inherent critique of Pigouvian taxes. The Coasean critique is itself critiqued, by arguing that it cannot be applied to Pigou's Welfare Economics. The Coase Theorem and its use to defend a deregulatory agenda by the Coaseans is also subjected to critical analysis. We follow by presenting the early contributions to the theory of emissions trading and tracing their intellectual sources, exposing the importance of the Coasean critique. The chapter concludes by illustrating the connections between the Coasean and Pigouvian views on property rights and pollution and the first proposals for a market for pollution rights.

## 2.1 The Pigouvian approach to social costs: A case of missing prices

### Pigou's Welfare Economics

We can trace the origin of the concept of externalities to Alfred Marshall (1890). In his *Principles of Political Economy*, the Cambridge University professor discusses gains from spatial concentration of firms in industrial districts that market prices fail to capture. In particular, technological spillovers, due to imitation by competitors, undermine the a firm's capacity to fully appropriate the gains from its innovation efforts. As a result, firms will have a sub-optimal level of innovation and the market equilibrium will not maximize economic efficiency.

Marshall's successor as chair of Political Economy at Cambridge University, Arthur Pigou, extended the analysis of external effects of production to consider different sources of divergence between social and private marginal costs. Pigou's (1932 [1920])

*The Economics of Welfare* became the basis for the analysis of external costs and benefits of production not reflected on market prices, or, to use the modern term, externalities.[1]

Pigou starts by defining an indicator which he called the National Dividend, which roughly translates to the Gross National Product in the current terminology, calculated as the flow of goods and services produced or consumed in a country. The National Dividend is an indicator of economic welfare or wealth, i.e., the measurable dimension (in monetary terms) of social welfare. Even though an increase in economic welfare should lead to an increase in overall welfare, Pigou mentions two exceptions to this rule.

The first is the decrease of non-measurable dimensions of welfare, in spite of an increase in wealth. Since it isn't possible to draw a boundary between the measurable and non-measurable dimensions of welfare, Pigou assumed that the two should be correlated. In some exceptional cases, however, he admitted that this might not happen, as it is the case when workers are coerced to work more than they wish to, and the dissatisfaction brought about by an increase in work done exceeds the satisfaction provided by the consumption of its product.

The second exception stems from equity concerns. If an increase in the size of the National Dividend is achieved at the expense of a decrease in the share of the divided distributed to the poor, welfare will not increase. On the one hand, there is an inverse relationship between income and share of income dedicated to consumption. Since economic welfare depends on income consumed and not income earned, and since poor people spend relatively more than rich people, the law of diminishing utility leads to the conclusion that economic welfare will, in general, increase if the share of income distributed to the poor increases and the National Dividend does not decrease. On the other hand, even though relative income contributes more to rich people's welfare than absolute income, as being *richer* is more important than being *rich*, welfare will not diminish when the share of income earned by the poor increases. This is because the decrease in welfare borne by the rich when the distribution of income becomes more egalitarian is substantially inferior to the increase in welfare accrued by the poor from the increase in

1 This book expands and builds upon Pigou's first major book, *Wealth and Welfare* (Pigou 1912).

consumption.

Pigou then proceeds to expand on the Marshallian theory to explain how self-interest will not maximize welfare, measured by the National Dividend. The implicit distinction between wealth and welfare followed the moral philosophy of another Cambridge professor, Henry Sidgwick (1887), whose utilitarianism opened the way for a critique of *laissez faire* by allowing for a sub-optimal equilibrium to emerge from the sum of optimizing individual actions (Backhouse 2006). To this endeavour, Pigou distinguishes between the marginal social and private net products. While the former is "the total net product of physical things or objective services due to the marginal increment of resources in any given use or place, no matter to whom any part of this product may accrue," the latter only accounts for the net product accruing to the individual firm or person (Pigou, 1932: 132).

Pigou considered that, while the marginal private product, net of costs of resource reallocation, will tend to be equalized across different investments by self-interest, the marginal social product will tend to differ from the marginal private product, preventing the maximization of the National Dividend. In particular, the existence of social costs of production leads to a divergence between social and private products:

> Here the essence of the matter is that one person A, in the course of rendering some service, for which payment is made, to a second person B, incidentally also renders services or disservices to other persons (not producers of like services), of such a sort that payment cannot be exacted from the benefited parties or compensation enforced on behalf of the injured parties. (*ibid.*: 183)

Pigou mentions, following Marshall, scientific research as one example of a situation where the marginal private product is inferior to the marginal social product and adds other examples, like the creation of public parks or the installation of lamps on houses that illuminate the streets. As for examples of situations where the marginal private product is superior to the marginal social product, Pigou mentions the construction of a polluting factory in a residential zone, the use of automobiles (which wear out roads) and women work in the perinatal period (which negatively affects babies' health).

Since self-interest alone cannot maximize the national dividend, Pigou (*ibid.*: 192) concludes that divergences between private and social products often call for "extraordinary encouragements" or "extraordinary restraints", the most obvious forms of which are "bounties and taxes". But direct regulation, in the form of prohibitions, is also considered when the excess of private over social product is large enough.

In the case of women's work during "the periods immediately preceding and succeeding" childbirth, Pigou (1932: 190) considers that the serious harm to the health of their children is motive enough to advocate the prohibition of such work. Since poverty is the cause for women's work in the perinatal period, and since the harm caused by extra poverty can be superior to the harm caused by such work, Pigou then concludes that this prohibition must be accompanied by a relief to the families that become impoverished by it.

The pervasive character of social costs and other market failures led Pigou to advocate for state intervention to increase the national dividend:

> In any industry, where there is reason to believe that the free play of self-interest will cause an amount of resources to be invested different from the amount that is required in the best interest of the national dividend, there is a *prima facie* case for public intervention. (*ibid.*: 331)

The case for intervention, however, could not be more than a *prima facie* one, since it is necessary to consider the possibility of government failure:

> It is not sufficient to contrast the imperfect adjustments of unfettered private enterprise with the best adjustment that economists in their studies can imagine. For we cannot expect that any public authority will attain, or will even whole-heartedly seek, that ideal. Such authorities are liable alike to ignorance, to sectional pressure and to personal corruption by private interest. A loud-voiced part of their constituents, if organised for votes, may easily outweigh the whole. (*ibid.*: 236)

We can find in this analysis an important inconsistency in Pigou's welfare economics. Even though Pigou distinguishes wealth from welfare, his analysis of social costs implicitly dismisses this distinction and focuses on maximizing the value of production. This focus on efficiency and disregard for equity and non-measurable elements of welfare later became the norm in the economic orthodoxy.

### Pigouvian taxation and externalities

Within the framework of Environmental Economics (EE), which developed in the 1960s, Pigou's analysis was subsumed under the concept of Pigouvian taxes, which rearrange market equilibria to internalize the cost of externalities (e.g. Cropper and Oates, 1992; Pearce, 2002; see also Aslanbeigui and Medema, 1998). In the neo-classical mathematical models, this implies that pollution and production levels must be adjusted to reach their optimum, which corresponds to the point where the marginal benefit of production is equal to the marginal cost of pollution.

This formalization of eco-taxes exposed EE to criticisms regarding the practicability of its prescription for maximizing efficiency.

The difficulties were identified early on by Mishan (1967), which noted that the quantification of externalities, if possible, is a complex task given the data dispersion and the non-separability of externalities that emerge from complementary economic activities. Consequently, it might be impossible or, at least, very difficult, to estimate the optimal level of pollution, assuming that such a level exists.

A possible solution for this problem was presented by Baumol and Oates (1971), who proposed that environmental taxes be set at a level consistent with a pre-defined standard for an acceptable environment, though a trial-and-error process. This standard-and-taxes approach restricts the role of economics to finding the least-cost instrument to achieve an acceptable level of pollution. While not maximizing efficiency, the solution purportedly avoids the problems with information gathering and quantification that permeate Pigouvian taxes. Still, an alternative based on an iterative process is arguably as impracticable as the efficiency-maximizing tax, considering the high bureaucratic costs and the time it takes to change a tax rate, not to speak of the uncertainty associated with entrepreneurial projects.

The impracticability of Pigouvian taxes is a consequence of EE failing to follow Pigou's welfare economics to concentrate on maximizing welfare and not efficiency. While Pigou also failed to follow this maxim in his analysis of social costs, EE's Pigouvian taxes went further and not only ignored the role of equity and non-measurable dimensions of welfare but also the possibility of government failure. The Pigouvian tax is, therefore, more than a *prima facie* case for intervention.

Another difficulty of Pigouvian taxes is that its defence relies on framing pollution as a textbook example of a negative externality. It is possible, however, to turn this argument upside down, by framing pollution prevention as an example of an investment with positive externalities. In fact, following Pigou's analysis of the prevention of smoke from factory chimneys (Pigou, 1932: 139), investing resources in smoke prevention renders services to the population for which it is difficult to exact payment, which implies that the affected population should compensate the polluter for the costs incurred in reducing its emissions. Regarding the important choice to make on who is to pay the cost of reducing pollution, EE's preference for taxes indicates a political agenda covered by efficiency arguments, an agenda that had already been contested by the Coasean approach to social costs.

## 2.2 The Coasean approach to social costs: A case of missing property rights

### Coase's comparative institutional analysis

The main challenge to the Pigouvian approach to social costs came from the Chicago School economist and co-founder of the Law and Economics field Ronald Coase (1960), with a contribution that can be seen as the basis of the property rights approach to social costs of production.

Coase starts by exposing the object of his paper as “those actions of business firms which have harmful effects on others” (*ibid*.: 1). His starting point for a critique of the Pigouvian approach to the external effects of production is the reciprocal nature of social costs:

> The traditional approach has tended to obscure the nature of the choice that has to be made. The question is commonly thought of as one in which A inflicts harm on B and what has to be decided is: how should we restrain A? But this is wrong. We are dealing with a problem of a reciprocal nature. To avoid the harm to B would inflict harm on A. The real question that has to be decided is: should A be allowed to harm B or should B be allowed to harm A? The problem is to avoid the more serious harm. (*ibid.*: 2)

The question of who is to be allowed to harm who is to be answered by comparative institutional design, by which different arrangements of property rights are ranked according to the total social product (*ibid.: 34*). Using a classical example of social costs of production, Coase argued that if a factory emits smoke that harms people who live nearby, the most important question, which must be answered by the law, is whether the factory has the right to pollute or the people affected by the smoke have the right to live in a smoke-free area. In the former case, the people affected by the smoke must either live with the smoke or bribe the firm that owns the factory to reduce its emissions, in the latter case the firm must either reduce its emissions or bribe the people affected by the smoke to compensate them for the harm caused.

According to Coase, "the ultimate result (which maximises the value of production) is independent of the legal position if the pricing system is assumed to work without cost." (Coase, 1960: 8). This is a point that was already present in his analysis of the allocation of broadcasting rights (Coase, 1959: 27). The Coase Theorem, as it later on became known, states thus that, assuming that transaction costs are zero, as standard neoclassical theory does, any property rights arrangement will lead, through bargains, to the maximization of the value of production. But Coase noted that this is a highly unrealistic assumption:

> In order to carry out a market transaction it is necessary to discover who it is that one wishes to deal with, to inform people that one wishes to deal and on what terms, to conduct negotiations leading up to a bargain, to draw up the contract, to undertake the inspection needed to make sure that the terms of the contract are being observed, and so on. (Coase, 1960: 15).

In the real world, where such transaction costs exist and are non-negligible, Coase outlined four possible alternative solutions for the problem of social cost: property rights transactions between affected parties, integration of the relevant parties in a firm, regulation and doing nothing.

The first solution, bargaining, occurs whenever transaction costs are lower than the increase in social production that results from the negotiation between the affected parties, taking into account the pre-existing property rights arrangement. As Coase noted, a tax on the firm that causes harm on others is equivalent to a bargaining solution where the property rights are awarded to the victims, whenever the tax revenue is used to compensate victims. Conversely, we might add that a Coasean bargain is equivalent to a subsidy given to the firm to reduce the harm done whenever the cost of the subsidy is supported by the victims.

The second solution, creating or expanding a firm, applies in the special case when a productive activity causes harm in another productive activity. Coase here follows his theory of the firm (Coase, 1937), which states that a firm is an alternative to the market for the organization of production and the choice between market transactions and intra-firm transactions depends on their relative costs. The concentration of the activity that causes harm and the activity that is harmed in one firm, therefore, is a viable solution whenever the firm's administrative costs are inferior to the transaction costs from bargaining.

The third solution, regulation, is relegated to cases where transaction costs are high, namely because the number of parties involved is high. But even in this case it is necessary to take into account how the government acts. If the costs of government action are higher than transaction costs and/or if the government fails to act in a way that maximizes the value of production, the fourth solution, doing nothing, is preferable.

Coase concludes, then, with the defence of a comparative institutional analysis approach, by which the status quo is compared to the effects of a proposed policy change, taking the value of production as the metric to compare the two, in order to decide whether or not to approve the said policy: "In devising and choosing between social arrangements we should have regard for the total effect. This, above all, is the change in approach which I am advocating." (Coase, 1960: 44).

Taking into account that in many real world situations, transaction costs are high, the Coasean approach often leaves us with the choice between regulation (which, presumably, includes taxation) and doing nothing. Further taking into account the high costs of gathering all the relevant information and the possibility of government failure, the cost-benefit approach to regulation defended by Coase can be used to support arguments against regulation.

### Critiquing the Coasean critique

The Coasean critique can be used to expose an internal inconsistency of neoclassical solutions for externalities. In a zero transaction cost world, Coase argued, an efficiency-maximizing solution to the problem of social cost, which arises from the harm caused on others by business firms through their productive activities, can be achieved through bargains involving the affected parties. This means that the case for intervention made by neoclassical economists, namely those in EE, is unwarranted, since they already assume the inexistence of transaction costs.

The critique does not extend easily into Pigou's welfare economics, though, which is what Coase was targeting. There is nothing in Pigou's work that indicates that he was assuming a zero transaction cost world. On the contrary, Pigou analysed how obstacles to movement of resources, information asymmetries and other transaction costs prevent the equalization of rates of return across different investments that should emerge from self-interest.

Another important aspect in which Pigou's welfare economics diverges from the target of Coase's critique is in the variable that is to be maximized. For Pigou, the priority was to maximize welfare, which combined efficiency, equity and ethical considerations. Even if Pigou incoherently deviated from this norm in his analysis of social costs, focusing merely on maximizing the value of production, his welfare economics was still grounded on moral considerations that were lost with in Coase's representation.

Taking the maximization of welfare, instead of the value of production, as the social objective, Coase's reciprocity argument does not hold. A policy change that increases the value of production while decreasing the share of this value that accrues to the poor would not pass Pigou's welfare maximization criterion. The same can be said of a policy that allows those responsible for significantly harmful behaviour to buy the right to cause harm, as the example of women labor given in the previous section shows. It is possible, therefore, to use Pigou's approach to social costs to argue against redistributing income from the victims of pollution to the polluter or to frame the right to cause harm as a factor of production.

The reciprocity argument is complemented by what became known as the Coase Theorem. This theorem was never formulated explicitly by Coase, but rather by his colleague George Stigler as “under perfect competition, private and social costs will be equal” (Stigler, 1966: 113). A more descriptive formulation was given by Calabresi: “if one assumes rationality, no transaction costs, and no legal impediments to bargaining, all misallocations of resources would be fully cured in the market by bargains" (Calabresi, 1968, p. 68).

Many free-market economists used the theorem to support a general case against regulation and, occasionally, for market solutions to the problem of social costs (e.g. Demsetz 1972, Buchanan 1973, Anderson and Leal 2001). The anti-regulation argument, however, is unwarranted, since the theorem can only be seen as a rhetorical device to mobilize against neoclassical economics and expose the importance of transaction costs, as Coase explained in Nobel Prize in Economics acceptance speech:

> I tend to regard the Coase Theorem as a stepping stone on the way to an analysis of an economy with positive transaction costs. The significance to me of the Coase Theorem is that it undermines the pigouvian system. Since standard economic theory assumes transaction costs to be zero, the Coase Theorem demonstrates that the pigouvian solutions are unnecessary in these circumstances. [...] My conclusion; let us study the world of positive transaction costs. (Coase, 1992: 117)

Using the Coase Theorem as a basis for a theory of deregulation exposed Coasean approaches of social costs to critiques of the theorem, which question its validity. Apart from critiques to the assumption of utility maximizing behaviour by individuals, the theorem has been criticized extensively for disregarding obstacles to efficient trading in property rights.

One such obstacle was pointed out by Mishan (1967: 91-94), when discussing "external diseconomies" and property rights. Following Mishan, efficiency can be maximized either by having a polluting factory compensate the affected people or having the inhabitants bribe the firm to reduce its output or install pollution control equipment. In this situation, it is possible to estimate the optimum product using market prices by, for instance, calculating the costs of cleaning clothes and houses borne by the affected people.

But Mishan also considers situations in which market prices cannot be used to value the harm caused. In such situations, the efficient solution that emerges from a barter in property rights among the involved parties will be determined by their relative wealth. Concretely, the probability that the outcome of the barter will be the desired outcome of one party is positively correlated with its or her wealth. The distribution of property rights that follows from the assignment of liability is no longer irrelevant, contrary to what was postulated in the Coase Theorem, since it affects wealth distribution and, therefore, the value of production.

A practical consequence of wealth and income effects in the presence of social costs is that people will give different answers to the questions “how much are you willing to pay to avoid harm?” and “how much are you willing to accept to tolerate harm?”. Another reason for this divergence lies on the endowment effect, described by Thaler (1980) as what happens when people assign a greater value to a good when they own it. The endowment effect leads economic agents to pay more to avoid the loss of something they own than to obtain something they don't own. Kahneman *et al.* (1990) argue, based on experimental results, that this effect invalidates the Coase Theorem, even when transaction costs and income effects are absent, since the allocation of resources will depend on the distribution of property rights.

A second obstacle to efficient trading in property rights is the existence of strategic behaviour. Bribes and charges are asymmetric, as the former leads to a higher output in the long run, due to strategic behaviour by firms to secure a larger bribe and to the entry of new firms in the industry (Bramhall and Mills, 1966; Kamien *et al.*, 1966). When dealing with the social costs of pollution, therefore, distributing property rights to emitters or receptors has different effects on the long run optimal output.

Another way of looking at the problem of strategic behaviour is by thinking of transactions in property rights not as market transactions but rather as bilateral bargains. Applying this perspective, Farrel (1987) noted that these Coasean bargains will only be efficient when all parties involved have complete information. In real world situations, where this assumption does not hold, parties face an incentive to misrepresent their costs and secure a larger gain from the negotiations.

Hahnel and Sheeran (2009) argue that this problem is even worse when the number of parties involved is large, as is the case of pollution. In these cases, separate negotiations with individual victims will be inefficient, regardless of liability attribution. If the polluter is liable for the damage, an individual victim has an incentive to hold out, by demanding a compensation higher than the costs borne and threatening to veto the negotiations if her conditions are not met. If the victims are liable, an individual victim has an incentive to free ride, by not participating in the negotiations, in the expectation that others will pay the bribe the polluter is demanding. The creation of a coalition of victims becomes necessary, but it does not eliminate strategic behaviour, as it is still necessary to know who are the victims and what is the extension of damage borne by each victim, and victims have an incentive to lie on both of these questions.

A third obstacle to efficient trading in property rights is rent-seeking behaviour. Jung *et al*. (1995) noted that the distribution of property rights is a zero-sum game, with effects on wealth and income distribution. This creates an incentive for rent-seeking, prior to the trading of property rights, to affect the property rights distribution. Since allocating resources to rent-seeking is inefficient, even in the zero transaction cost world of the Coase Theorem efficiency will not be maximized.

These obstacles to efficient trading translate into arguments on the invalidity of the Coase Theorem. It still might be objected, though, that these obstacles amount in practice to transaction costs. As Wohar (1988) noted, different versions of the Coase Theorem arise partly from different concepts of transaction costs. In fact, Coase never defined precisely what counts and what does not count as a transaction cost, not excluding income effects, the prevention of strategic or rent-seeking behaviour, as well as information costs and other impediments to an efficient trading in property rights from the definition. This, however, leaves us with a dilemma, as the theorem is only correct if transaction costs are defined broadly as “anything that might prevent the achievement of a Pareto solution” (Swan, 1975: 70). In other words, either the theorem is inconsistent or it is a tautology.

Despite its theoretical shortcomings, Coase's contribution to the analysis of social costs sparked an interest in solutions to the problem of social cost based on tradable property rights. A market of permits to pollute emerged, then, as a possible solution both to implement such rights and to minimize transaction costs.

## 2.3 Emissions trading as an alternative to environmental taxation

### Early proposals for tradable pollution rights

In the 1960s, when the field of EE was in its infancy, Coase's critique of Pigou was largely either ignored or considered irrelevant by the first environmental economists, which viewed pollution as a market failure, to be corrected through taxation (e.g. Kneese, 1962). One exception was Tom Crocker, a PhD candidate at the University of Wisconsin-Milwaukee, who, following a grant from the U.S. Public Health Service to study the economics of air pollution, proposed pricing pollution through a market on emissions rights.

In his contribution to a symposium on the economics of air pollution, held in Washington D.C., Crocker (1966) frames pollution as a market failure. The problem, according to Crocker, is that the polluter pays nothing for using the air to dispose its waste, while the receptors of pollution pay nothing for their use of the life and property-supporting capacity of the air. It follows, then, that the objective of an atmospheric pollution control program should be to narrow the difference between the cost savings that accrue to emitters from not having to reduce their emissions and the damage costs that accrue to receptors from the pollution. This can be achieved through a market on rights to use the air:

> […] a market pricing system which defines emissions rights, open to emitters and receptors on a periodic competitive basis can drastically increase the likelihood of achieving optimal intertemporal and interspatial allocation of the air's two value dimensions. The informational requirements to achieve the optimum will be minimal, as the regulator will not have to know the emitter and receptor preference functions. (Crocker, 1966: 81)

While the maximization of two value dimensions in time and space is a manifest impossibility, the approach purportedly reduces the information gathering requirements for the regulator, relative to the optimal taxation of pollution. What is needed for this market to work is an institutional design that defines property rights over the air and makes it possible for polluters and receptors to trade these rights. Receptors have to be organized in groups, in order to prevent the free rider problem, as some receptors can choose not to pay for clean air and expect other receptors to pay. How such a market would work in practice was, however, a question left unanswered by the author.

A more elaborated argument for emissions trading was made by John H. Dales, an economist from the University of Toronto, though not an environmental economist. In his book on pollution and property rights, Dales (1968b) criticized the EE tradition of presenting “simple solutions for simple problems,” arguing that the measurements necessary to find optimal levels of pollution by comparing costs and benefits are to be distrusted. Actual problems, he stated, are characterized by a lack of information, because some data is difficult to obtain (namely data on emission control costs) and some is unobtainable and can only, at best, be approximated (namely data on valuation of environmental amenities).

Dales then proceeds to propose a solution for social costs that is less demanding on information gathering than environmental taxes based on an optimal level of pollution. After arguing that “treating air and water as unrestricted common property is socially indefensible” (*ibid.*: 75), since it leads to excessive environmental degradation, the author sets up a blueprint for an emissions trading scheme in water pollution rights. In this scheme, a politically independent Water Control Board (WCB) defines a cap for aggregate waste and accordingly issues tradable emissions rights, which are auctioned to waste dischargers. Individual firms can then buy and sell emissions rights through the WCB, which can also intervene in the market to implement both a floor and a ceiling price. Environmental groups can also participate in the market to buy emissions rights and set them aside, thus buying a cleaner environment. This scheme, that became known as “cap and trade”, differs from Crocker's proposal by having the level of environmental quality defined exogenously, rather than having it somehow determined by the emissions rights market.

The criticism of the neoclassical orthodoxy present in EE allows Dales to reconfigure the economist's task, since “[...] economic analysis, which is all but useless in helping us to decide on a policy, is all but indispensable in helping us to decide on the very best way of implementing a policy once it has been chosen” (*ibid.*: 99). Economics does not have to define what the optimal or acceptable level of pollution is but rather faces the more modest task of finding the least costly way of enforcing a level of pollution that is set politically.

While Crocker's emissions trading system aimed to the impossible task of optimizing two value dimensions in time and space, Dales' system doesn't rely on any claims of optimization. In this framework, emissions trading is preferable to other regulatory instruments, whenever it can be used, not because it allows an optimal allocation of resources but rather because it minimizes the cost of meeting a given environmental objective. Administrative costs, in particular, are lower than with emissions charges:

> Such markets would automatically set the correct level of the pollution charge (instead of it having to be set by some committee, after long and learned discussion) and would also automatically, and continuously, adjust the level of the charge to take account of economic growth. A simple market that can be operated by three or four people and a small staff of stenographers to register purchases and sales is very much cheaper, and just as efficient, as a large bureaucracy replete with computers to give answers to complicated pricing problems. If it is feasible to establish a market to implement a policy, no policy-maker can afford to do without one. (*ibid.*: 100)

The argument for emissions trading is repeated in a paper published in the same year (Dales, 1968a), in which the relationship between prices and property rights is explored. For Dales, the inexistence of property rights in water led to the inexistence of a mechanism to transform technological externalities into prices. Using a framework similar to Crocker (1966), the author distinguishes between two uses for water, waste disposal and other uses or amenities, which are conflicting. Unlike Crocker, though, Dales argues that it is impossible to calculate the value of a marginal change in amenity use (or receptor damage costs, to use Crocker's terminology). Consequently, the optimum level of waste disposal cannot be calculated.

Dales (1968a) elaborates on how the level of pollution is to be determined by considering that decisions on the maximum level of emissions are political in nature, representing an equilibrium between complaints about the cost of anti-pollution measures and the complaints about pollution, rather than an equilibrium between the costs and benefits of anti-pollution measures. Implicitly, Dales assumes that advances in scientific knowledge on the effects of pollution have only an indirect effect on the computation of the equilibrium level of pollution. The question, then, becomes how to choose the best way of enforcing the cap on emissions, taking into account the costs of different options, rather than trying to find a property rights distribution that maximizes the social product.

There are six different options considered by Dales: individual quotas on emissions, emissions standards, individual subsidies to polluters, subsidies per ton of emissions, individual charges and charges per ton of emissions. Leaving aside the subsidies, which generate excess profits, across-the-board regulations, which lead to a non-optimal distribution of the costs, and individual solutions, which have very high administrative costs, we are left with charges per ton of emissions as the only acceptable option. In order to have a unique charge on emissions and minimize administrative and other transaction costs, a market on pollution rights can be created.

This solution, according to Dales, minimizes costs because the regulator no longer has to estimate an optimal level of pollution nor resort to a trial-and-error procedure to find a charge that induces this level of pollution. The market mechanism will not lead to an optimal distribution of the costs but it will likely lead to savings in administrative costs that outweigh the losses measured against a theoretical optimum that assumes zero administrative and other transaction costs.

Even though Dales (1968a, 1968b) and Crocker (1966) elaborated their policy proposals in parallel, the two proposals for a market in emissions rights follow a strikingly similar framework. Both authors think of pollution as the result of a zero price on emissions but refrain from automatically concluding that taxing emissions is the best policy by considering that it is also possible to price emissions by setting tradable property rights. In both cases, environmental taxation, even if feasible, is a more costly and information demanding instrument to achieve a desired level of pollution than a market on emissions rights, in which emitters and receptors can participate.

The two authors also diverged from the emerging EE in explicitly criticizing the view of pollution as a problem to be fixed, rather than as a natural fact of life. Thus, Crocker (1966) rejects the idea that we should have clean air no matter what the cost, and Dales (1968b) named the first chapter of his book “To live is to pollute”. For these authors, pollution was not an issue serious enough to justify tough regulation to protect public health, with Dales (*ibid.*: 103) going as far as to claim that “The only wholly tenable argument against pollution at the present time is the aesthetic one”.

The discussion on social costs present in early emissions trading theory shows many similarities with Coase's analysis. This is reinforced by the rejection of the concept of externalities by the authors in other publications. Crocker (2008: 10) dismisses this concept as “awful jargon” that distracts economists from analysing the connections between institutions, behaviour and the environment. Dales (1975: 496-7) also recommends that economists dispense both the language and concept of externalities, arguing that it reflects an arbitrary selection of activities not constrained by prices but relevant to welfare, and advocates following Coase's suggestion of comparing the efficiency of different institutional restrictions to behaviour.

## Tracing the sources of emissions trading theory

We can trace the intellectual underpinnings of emissions trading theory by comparing the sources used by the two authors. Crocker (1966) did not quote any of his sources in the economics literature, so we shall resort to a personal communication (Crocker, 2008), in which the author recalls the process of writing his original contribution. Dales (1968b) quoted his sources indirectly, by bundling them in an appendix to his book.

Despite being written in parallel, the two seminal contributions to emissions trading theory are both rooted the property rights approach to social costs, popularized by Coase's critique of Pigou. The starting point is the critique of EE and its Pigouvian taxation, which Crocker traces back to Pigou (1920), while Dales quotes instead the influential Kneese and Bower (1968) book on the economics of water pollution. This is contrasted with the property rights approach, which both authors identify with Coase (1960), while Dales also adds Reich's (1964) analysis of the property rights created by regulations and court decisions as a source.

The case for establishing property rights on environmental goods is also justified, in the authors view, by Gordon's (1954) argument against open access. The argument, that was later popularized as the “tragedy of the commons” by Hardin (1968), states that common-property natural resources will be over-exploited to the point of depletion, since individuals treat them as free goods, unless property rights are established over their use and access.

Crocker further adds Samuelson (1954) as basis for claiming that non-rival and open access goods, like clean air, will be undersupplied due to free-riding behaviour. In his proposal for a market on emissions rights, this concern leads him to advocate the aggregation of individual receptors in organizations, to assure that all individuals contribute to the purchase of clean air.

Mishan's (1967) discussion of external diseconomies and property rights, which was published a year after Crocker's contribution, was also quoted by Dales as an inspiration. Supposing that different individuals will have different preferences regarding pollution and going so far as to admit the possibility that some individuals might be so used to polluted air that they will dislike being in an unpolluted environment, Mishan argues that the enforcement of amenity rights by law would cause market forces to create a separate facilities solution for the problem of external diseconomies. Concretely, polluting businesses would be located in places where the local inhabitants are willing to accept higher levels of pollution, while people who prefer cleaner environments would choose to live in less polluted areas. As Dales (1968b) notes, granting fewer rights to pollute leads to the same outcome as granting more rights to a cleaner environment, so his emissions trading proposal differs from Mishan's separate facilities proposal as it conceives not just the creation of environmental property rights but also a market to trade them.

A final influence in emissions trading theory is Hayek's (1945) argument against centralized planning and for free markets. Following Hayek, since information is dispersed across individuals and firms and cannot be fully centralized, planning is impracticable. The market provides a more efficient alternative to planning, because market prices already convey all the relevant information on production and consumption, therefore eliminating the need to centralize information. Crocker quotes Hayek's theory as an evidence that prices not only transmit information but also that rationality is institutionally constructed through people's interactions with the market, that is, people learn how to be rational by participating in market transactions. Dales, on the other hand, does not quote Hayek, but his discussion on the role of emissions rights markets in setting a price for emissions rights that removes from the regulator's shoulders the impossible task of centralizing all the information on pollution control costs is remarkably Hayekian.

In short, both Crocker and Dales' proposals for emissions trading to be considered as a cost-effective instrument to be added to the policy-mix regarding pollution control were inspired by the discussion regarding the connections between prices and property rights, which in turn was influenced by Coase's critique of Pigou. Analysing the theoretical underpinnings of emissions trading theory allows us to put the importance of the Coasean critique in perspective and conclude that, while not negligible, it is clearly not decisive enough to make Coase the "grandfather of emissions trading".

## 2.4 Conclusions

Pigou's (1920) treatise on economic welfare spurred a vast literature on external economies, or externalities, defined as positive or negative effects of a production or consumption activity that are not reflected in market prices. EE, in particular, was founded on the framing of pollution as a negative externality, to be corrected preferably through taxation of the polluting activity. Despite having departed from Pigou in assuming that social costs constitute more than a *prima facie* case for intervention, the orthodox approach to environmental issues dubbed its eco-taxes as Pigouvian.

This approach to social costs of production was criticized by Coase (1960), who argued that the case for imposing a cost on the polluter to reduce the harm caused to the polluted is unwarranted. Due to the reciprocal nature of social costs, the cost of pollution reduction might also be efficiently attributed to the polluted. Coase then concluded that, in a world with positive and non-negligible transaction costs, the questions of if and how to control pollution and who is to bear the cost of this control are to be answered by comparative institutional design, where different solutions are ranked according to their efficiency.

Coase's critique of Pigou was reduced by the Coaseans into a theorem that postulates the irrelevance of property rights distributions, assuming zero transaction costs. The Coase Theorem then became the basis of a more general critique of government intervention. As several contributions have shown, this theorem, even if considered relevant in the real-world, where transaction costs are positive, is either inconsistent or tautological.

Drawing from the Coasean critique of Pigou the idea that pollution problems can be thought as property rights allocation problems, Crocker (1966) and Dales (1968b) argued that it is possible and even desirable to put a price on pollution by creating a market on pollution rights. While pollution taxes were already seen in the economic orthodoxy as more efficient than direct regulation, this position wrongly assumed that the optimal level of pollution is calculable and ignored the significant difficulties in gathering the necessary information to determine the adequate level of pollution taxes. Tradable pollution rights, then, would relieve the regulator from gathering information about pollution control costs and allow a given level of pollution to be achieved at the least cost for regulated industries.

This approach implied a shift away from trying to estimate the optimal pollution level to trying to find out the best instrument to achieve a given pollution level. It also implied a shift in the framing of pollution, from a problem that should be corrected to a nuisance that should be managed. Accordingly, while the emerging EE field proposed using taxes as an efficient alternative to direct regulation in pollution control, emissions trading theorists proposed using tradable pollution rights as an efficient alternative to both taxes and direct regulation, an idea that slowly became a reality with the progressive liberalization of industrialized countries' economies in the following decades.

# 3 Born in the USA: The early experiences with emissions trading

"Market-based programs require significant planning, preparation, and management during development and throughout the life of the program" (EPA, 2002: 66)

Originally proposed by Dales (1968b) and formalized by Montgomery (1972), emissions trading, or "cap and trade", is a market-based regulatory instrument based on tradable emission allowances. The advantages of this instrument compared to direct regulation, the argument goes, stem from its superior cost-efficiency, simplicity and flexibility, as well as its inferior informational requirements. Since it was introduced in the Kyoto Protocol, emissions trading has been at the centre of the discussion over instrument choice for environmental protection. Before the late 1990s, however, emissions trading theory and practice which emerged from several amendments made to air quality regulations was circumscribed to the US.[2]

Air quality in the US is regulated by the Clean Air Act (CAA), a federal legislation approved in 1963 aimed at first to research into techniques for pollution monitoring and control. In 1970, the CAA was significantly revised to enact direct regulations on pollution. Emissions limits for some pollutants were set as National Ambient Air Quality Standards (NAAQS), defined for air regions and met through measures specified in State Implementation Plans (SIPs).[3] New Source Performance Standards (NSPS) were defined for new emissions sources (a category that also includes existing sources that underwent major modifications), setting limits on emissions based on

2 Even though emissions trading can also be applied to water quality regulation this variant has not been used extensively and, therefore, is excluded from this survey. Water quality trading was proposed in the academic literature (e.g. David *et al.*, 1980; Eheart *et al.*, 1980; Joeres and David, 1983; Maloney and Yandle, 1983; Brill et al. 1984) and even in an EPA publication (deLucia, 1974) but it is limited to small-scale local experiences, with only a reference to the failed Wisconsin Fox River market being made in the literature (Oates 1984, Hahn 1989a).

3 NAAQS were defined for 247 air quality regions and six pollutants: sulphur dioxide ($SO_2$), carbon monoxide (CO), nitrous oxide ($N_2O$), particulate matter ($PM_x$), hydrocarbons (HC), and photochemical oxidants. Some substances face two standards: primary (aimed to protect public health) and secondary (aimed to protect public welfare, i.e., vegetation, materials and visibility).

the best abatement technology. These revisions coincided with the creation of the US Environmental Protection Agency (EPA), in December 1970, to enforce environmental regulations and monitor pollution (EPA, 2013).

Following the emergence of a conflict between this new bureaucratic machinery and industrial growth in several states and of neoliberal governance within the US Presidency, mechanisms for trading pollution credits were gradually introduced by the EPA, granting polluters an added flexibility in compliance with emissions standards. These mechanisms gave way in the early 1990s to the first “cap and trade” schemes implemented in the world, applied at the national and regional scale. This chapter presents a critical review of both the US experience with emissions trading for air quality regulation and the literature based on this experience, aiming to present some relevant conclusions for contemporary debates on pollution markets.

The structure of the chapter is as follows. The first section surveys the first attempts at emissions trading in the US: the EPA Emissions Trading Program, a patchwork of different provisions that allowed trading in pollution credits, and the lead credits market, through which refineries could average their lead in gasoline content. The survey proceeds in the second section with the first “cap and trade” system, the Acid Rain Program, while the third section is dedicated to similar schemes applied at the regional level, in California and in the Northeast. The final section concludes by presenting some lessons that can be inferred from these experiences.

## 3.1 Early emissions trading programs

### The Emissions Trading Program

The 1970 CAA included regulations like the NSPS, which were applied on a source-by-source basis, so that a plant with several sources (corresponding to boilers or other equipment) had to comply with the regulations in each source. The first experiments with emissions trading can be traced to a reinterpretation of the word “source” by the EPA, to include a whole plant.

In 1972, the EPA approved a “netting” provision for smelters, allowing new or modified sources to avoid the mandatory emissions review procedures from the NSPS if a firm lowers its emissions in other sources enough to assure that plant-wide emissions remain constant. This provision was successfully challenged in court but was brought back in 1975, when the EPA restricted its revised definition of stationary source to existing sources undergoing modifications (Landau, 1980).

Flexibility in compliance with the CAA was further enhanced as it became clear that many areas in the US would not be able to comply with the 1975 deadline for meeting the NAAQS. This policy context was framed as a "growth ban" by the federal government, due to a clash between the CAA regulations and the industrial growth in non-attainment areas (Lane, 2015). The EPA reacted by proposing an "offset" provision, through which the emissions from new sources could be compensated by reducing emissions from existing sources (Yandle, 1978).

The Offset Policy, approved in 1976, allowed the installation of new sources in non-attainment areas provided that two conditions were met. First, emissions had to be more than offset, which is to say, emissions reductions from existing sources had to be greater than the emissions from the new source. Second, new sources had to comply with Lowest Achievable Emissions Rate (LAER) standards, set at a level at least as stringent as Best Available Control Technology (BACT) standards (Dudek and Palmisano, 1987).

In 1977, the CAA was amended by the US Congress, and a "banking" provision, that allowed firms to reserve credits from netting, bubbling or offsetting for future use or sale, was introduced. This provision faced several limitations, reinforced by the 1980 EPA decision that credits were not an absolute property right and, consequently, states could take away a part or all of the banked credits. Given the uncertainty of the long term status of banked credits, firms were wary of using this provision (Hahn, 1982).

The 1977 CAA Amendments also included a Prevention of Significant Deterioration (PSD) provision, which applied in areas that were in attainment of NAAQS, to maintain air quality in these areas and prevent pollution to rise up to the standard. A small emissions increase in attainment areas was allowed, as long as all major sources were subjected to BACT standards, which were designed to be at least as stringent as

NSPS (Landau, 1980). The steel industry, lagging in compliance with the CAA, reacted to the new regulations by lobbying for a "bubble" proposal, through which several sources within a plant could be grouped in an imaginary bubble and face a single emissions limit. In 1977, this proposal was presented to the EPA, facing opposition from environmental groups. The EPA reacted by approving the bubble policy, in 1979, subject to some restrictions. Bubbles could only group sources in an attainment area and could not lead to an increase in emissions. Polluters had to show that bubbles were equivalent to point-by-point regulation in terms of pollution reduction, enforceability and environmental impact. Bubbles could not be used by noncomplying sources neither be a means for extending deadlines for compliance (Liroff, 1986: 37-46).

In December 1979, the bubble policy was revised to allow for multiple plants to be grouped in a single bubble and for emissions increases in individual bubbles. In 1980-81, further revisions allowed bubbles to be formed in non-attainment areas for which SIPs had not yet been approved and extended deadlines for compliance. The revisions also partially removed the requirement that bubbles were treated as SIP revisions, a bureaucratic process that could take up to a year, by allowing states to define generic regulations (*ibid*.: 46-51).

These four provisions were unified under the Federal Emissions Trading Policy Statement, released in its interim version in 1982 and in its final version in 1987 (Ellis *et al.,* 1982; Brady and Morrison, 1984; Borowsky *et al.,* 1987). The Emissions Trading Program (ETP) thus created the Emissions Reduction Credit (ERC) as the common currency for netting, bubbling, offsetting and banking, defined as an emissions reduction that is not required by law, and is enforceable, quantifiable, and permanent.

As for the environmental effectiveness of bubbles, offsets, netting and banking, theoretically it should be null or slightly positive, in the sense that trading ERCs leads to emissions going down in one source by the same amount (or possibly more) than the emissions increase in another source. In practice, though, considering that some polluters were able to evade regulations through "paper trades" that represented accounting tricks rather than real emissions reductions and that credits were often were awarded to incidental reductions that would have happened anyway, the trading provisions often led to local emissions increases (Dudek and Palmisano, 1987; Driesen, 1998). On the other hand, EPA's refusal to award ERCs to the shutdown of old power plants, while justified by the distinction between an investment in emissions reductions and capital replacement, strengthened the incentive to prolong the life of old plants that was already present with NSPS regulations, which might have led to increased emissions (Maloney and Yandle, 1984; Maloney and Brady, 1988).

Even though in some cases, namely bubbles, polluting industries actively lobbied for emissions trading, in most cases industries were either indifferent or opposed to emissions trading, seeing it as a non-workable system and as another means for government intervention. Environmentalists, on the other hand, were generally opposed to any flexibility provision, seeing them as a way for polluters to evade regulations through accounting tricks, and some NGOs even tried to block them through the legal system (Liroff, 1986: 130-131). It seems, then, that the motives for the adoption of emissions trading provisions must be found within the regulatory system itself. The simple explanation that the EPA adopted emissions trading because of the potential cost savings associated to it is hardly convincing, though, as there were no *ex ante* estimates of the efficiency gains. On the other hand, the connection between theory and practice is not clear, as economist Robert Hahn, while working for the US Council of Economic Advisers, illustrated by simultaneously defending that the “patient” (the government) followed the “doctor's” (the economists) orders on “economic prescriptions for environmental problems” (Hahn, 1989a) and, conversely, that the “patient” did not exactly do what the “doctor” ordered (Hahn, 1989b).

Meidinger (1985) proposes that emissions trading was the result of the promotion of a new regulatory culture by policy entrepreneurs recently graduated from law and economics programs. This culture is based on four principles: (1) science is crucial to evaluate the impacts of regulation but cannot form the basis for choosing the social goals; (2) regulatory decisions are inevitably the result of compromises among contending interests; (3) the role of regulatory agencies is to develop a framework through which compromises can be achieved; (4) regulatory policies are based and legitimised, as much as possible, through compromises. This was the product of a historic period marked by economic recession, growing conservatism and the emergence of neoliberalism, when government agencies like the EPA were under pressure to adopt a more business-friendly approach and reconcile (even if just apparently) their objectives with economic and industrial growth.[4]

Meant to increase the economic efficiency of environmental regulation, the various flexibility provisions considerably differed from the "cap and trade" system that Dales (1968b) had proposed and were widely criticized in the economics literature as being too complex and bureaucratic to deliver significant cost savings (Noll, 1982; Brady, 1983; Oates, 1984; Levin, 1985; Hahn and Hester, 1987, 1989a, 1989b; Tietenberg, 2006). As Lane (2012: 586) puts it, the ETP "produced crude chimerical forms, merely ad hoc programmes" superimposed on pre-existing regulations and based on the narrative of the superior efficiency of economic incentives relative to "command and control" regulation that had been established in the Environmental Economics discipline to support the flexibilization of CAA targets (e.g. Baumol and Oates, 1975; Kneese and Schultze, 1975).

**Lead Averaging and Banking**

In the 1920s, refineries started adding lead to gasoline as an inexpensive way to increase octane levels. Higher octane gasoline improves the engine performance but the use of lead in gasoline corrodes engines and exhaust systems, increasing maintenance costs and fuel consumption. Furthermore, lead is a toxic substance that does not break down over time and its use in gasoline is associated with increased blood pressure and hypertension rates in adults and reduced cognitive performance in children, among other health issues. Still, lead phase-out was successfully resisted by refineries due to the higher cost of unleaded gasoline, involving either additional processing or replacement of lead for more expensive additives (Nichols, 1997).

The first step to reduce lead in gasoline was given in the early 1970s, with the

4 For a historical review of this period and the concomitant rise of free-market economics, see Backhouse (2005).

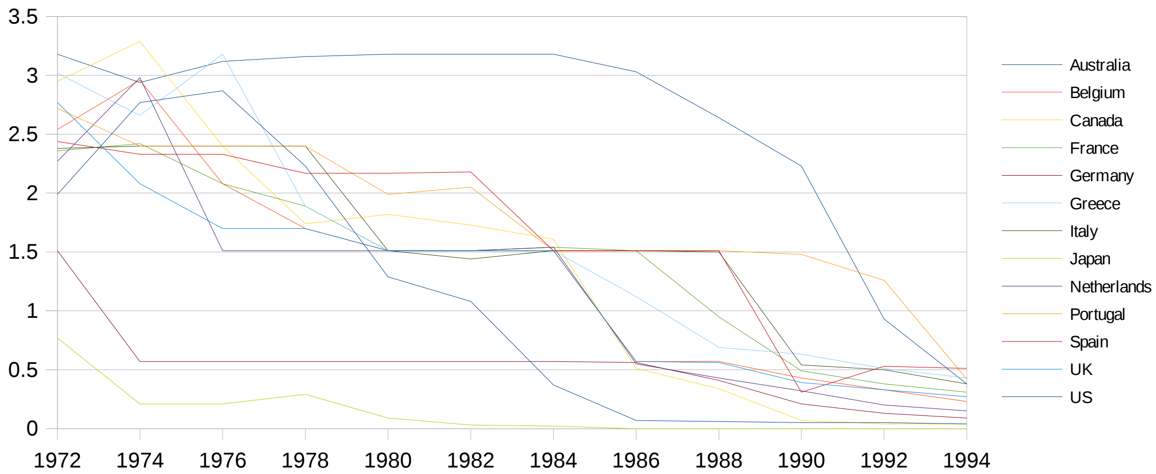
introduction of catalytic converters in cars, used to reduce emissions of hydrocarbons (HC), nitrogen oxides ($NO_x$), and carbon monoxide (CO). Since the use of leaded gasoline is incompatible with catalytic converters, which became mandatory in new cars from 1975, in 1974 new regulations were approved to spread the diffusion of unleaded gasoline and prevent owners of cars with catalytic converters from using leaded gasoline. This was achieved by forcing retailers to offer unleaded gasoline in each gas station and install different sized nozzles for leaded and unleaded gasoline. At the same time, incentives for old vehicle substitution were approved, to accelerate the introduction of cleaner engines (Kerr and Newell, 2003).

The introduction of regulations for catalytic converters in 1975 was accompanied by the introduction of a standard of 0.5 grams per gallon (gpg) for lead in gasoline by the EPA. The standard, which only came to effect in 1979, contrasted with the average lead content of 2.4 gpg that was registered in the early 1970s, when unleaded gasoline did not exist. Since the standard was calculated on the basis of total gasoline production, refineries could comply with it by expanding their unleaded gasoline production share. Small refineries faced less strict standards, ranging from 0.8 gpg to 2.65 gpg, according to gasoline production in the previous year, since they lacked the equipment necessary to produce unleaded gasoline (Nichols, 1997).

In 1982 the EPA, following a recommendation from the Task Force on Regulatory Relief, considered relaxing the lead standard for large refineries and deferring the deadline for small refineries to face the same standard as large refineries did. Opposed by environmentalists, public health officials and even members of the EPA staff, this recommendation was shelved. Instead, the EPA narrowed the definition of small refineries, determined that special provisions for small refineries would be phased out by mid-1983 and reviewed the standard so that lead limits were calculated as an average of leaded gasoline only. The new standard was set at a quarterly average of 1.1 grams per leaded gallon (gplg), which was equivalent to the previous standard taking into account the leaded gasoline market share. Following a cost-benefit analysis on reducing lead in gasoline, published in 1984, the standard was lowered to 0.5 gplg in mid-1985 and further reduced to 0.1 gplg in 1986 (Nichols, 1997).

The 1982 review of regulatory policy also allowed lead averaging. This was achieved through a “baseline and credit” program, through which refineries producing gasoline with less lead than the standard could sell lead credits to refineries that were above the standard. During the first year, trading was restricted to refineries of similar size, with a segmented market that allowed averaging among large refineries and among small refineries but not among large and small refineries. From 1985, the averaging program was supplemented with a banking provision that lasted until the program ended, in the end of 1987. Lead was definitively eliminated as fuel additive in 1996 (Sovaccool, 2011).

***Figure 1: Lead in gasoline in OECD countries, 1972-1994 (gpg)***




*Source: Hilton (2006)*

The environmental results of the regulations to phase out lead were encouraging. By 1980, the US already had the third lowest lead content in gasoline in OECD countries, after Germany and Japan. After the trading period, in the end of 1986, the US had a much lower lead in gasoline content than most of OECD countries, being surpassed only by Japan, which had already eliminated lead. This contrasts with the EU, which delayed lead phase out until 1987 due to trading restrictions that imposed a minimum content of lead in gasoline and opposition from some Member States to regulations mandating the instalment of catalytic converters in new vehicles (Hammar and Löfgren, 2004).

As for the economic performance of the lead credits market, both trading and banking levels were high and above the initial EPA estimates, indicating low transaction costs (Ellerman *et al*., 2003). The administrative costs of this system were quite low, as there was no need to certify tradable credits, so a refinery which managed to produce gasoline with less lead content than the average received credits automatically. Yet, these low administrative and transaction costs were achieved at the expense of opening a backdoor to fraudulent activities.

Since monitoring gasoline production and lead incorporation in refineries directly would be costly, the EPA delegated the data collection task to refineries. Refineries also had to report their lead credits transactions, deposits and withdrawals. A computer system was then set up, to analyse the data received and detect inconsistencies between the data on lead use and the data on lead trading and banking and between all of this data and the data on lead sales. This self-reporting system led to significant misreporting of lead use and, more frequently, gasoline production. Additionally, many refineries engaged in falsifying credits, which were then traded several times before being discovered by the EPA. Since the task of tracking these false credits to their original source was very difficult, fraudsters could leave the market before being found out (Newell and Rogers, 2004).

Fraudulent activity was severely affected by the unexpected entry in the lead credits trading system of “alcohol blenders”, which were mainly big service stations that mixed alcohol with leaded gasoline, lowering the relative concentration of lead in gasoline. These blenders were treated as small refineries by the EPA, which led to a substantial increase in the number of participants in the lead credits market, as well as the quantity of erroneous reports sent to the EPA and falsified credits sold in the market (*ibid.*).

The literature on lead in gasoline averaging and banking points out economic reasons for its adoption. Averaging is explained by Hahn (1989b) by the delay in refineries investment in the equipment necessary to produce unleaded gasoline, while Newell and Rogers (2004) find its reason for existence in the high compliance costs faced by small refineries, compared with the lower costs and lower than demanded lead content achieved by large refineries. Banking, on the other hand, appears as a necessity to Nichols (1997) due to the time that it took to install the equipment to reduce gasoline lead content, which could lead to failure to comply with the deadline for meeting the standard, while Newell and Rogers (2004) explain it on the basis on non-linear cost increases as standards became more stringent. These explanations, however, are incomplete.

Like the ETP, the market for lead in gasoline credits was created as an ad hoc extension of existing regulations that introduced some flexibility for regulated firms. In the same line, lead averaging allowed the EPA to exert its regulatory authority without clashing with the neoliberal and conservative Reagan administration. Emissions trading again was a form of creating a narrative reconciling environmental protection with economic growth, even if in reality it provided polluters a cheap and dirty way out of environmental regulation.

## 3.2 The Acid Rain Program

In the 1980s, the nascent literature on emissions trading started to take off, with several articles published in economic journals discussing design choices like the initial allowance allocation method and mechanisms for promoting market competition or price stabilization (e.g. Hahn and Noll 1982; Hahn 1984; Lyon 1985). This was complemented by the first edition of Tietenberg's (1985) influential manual on emissions trading, which generalized the case against direct regulation through econometric estimates of its inefficiencies. A related discussion was centred on pollutants with local impacts, regarding which the cap can be set according either to a desired pollution level or to a desired emissions level (e.g. Atkinson and Tietenberg, 1982, 1984, 1987; Krupnick *et al.*, 1983; McGartland, 1988). The case for emissions trading was made mostly based on arguments about superior cost-efficiency and administrative simplicity, compared with direct regulation (e.g. Tietenberg, 1980; Noll, 1982).[5] Some law scholars went further, though, and defended that emissions trading was inherently more democratic than direct regulation, since it was based on explicit environmental goals rather than obscure technological considerations (Ackerman and Stewart, 1988; Sunstein, 1991), with Stewart (1987: 154) making the extreme case that environmental regulations amounted to "nothing less than a massive effort at Soviet-style central planning of the economy to achieve environmental goals."

From the (US-based) economics literature, a scheme for emissions trading started to emerge, in which allowances are auctioned (or, if necessary to guarantee political support, given for free), banking is allowed, provisions are made to facilitate the entry of new firms and the cap is set at a maximum emissions level. The case for such a scheme was supported by the Government Accountability Office (GAO 1982), as well as the EPA (1981). In the political arena, a case for the use of emissions trading for controlling pollution from stationary sources was made by Project 88, a report commissioned by senators of both US political parties and coordinated by Richard Stavins, from Harvard University (Stavins, 1988). Contributors to the Project 88 report, which was delivered to both presidential candidates before the 1988 elections included industry representatives, environmentalists, academics and government representatives.

Of particular relevance to the evolution of emissions trading was the Project 88 proposal to create a "cap and trade" system for pollution causing acid rain, which was supported by many economists (Gollop and Roberts 1985; Raufer *et al.*, 1986; Raufer and Feldman 1987; Diemer and Eheart 1988; Dudek 1989; Tietenberg 1989; Tripp and Dudek 1989). The system was to be directed at sulphur dioxide ($SO_2$) emissions from power plants, one of the main contributors to the acidification of rain water that, as shown by scientific studies, resulted in damage to lakes and forests, as well as crops and monuments.[6]

5 The case for the superiority of emissions trading compared to direct regulation in terms of incentives to environmental innovation was not as clear cut, with Malueg (1989) arguing that this is only true for firms that are net sellers of allowances before and after the innovation.

6 $SO_2$ emissions also contribute to the concentration of fine particulate matter ($PM_{2.5}$), which were later shown to have harmful health effects, namely contributing to respiratory problems (McCormick, 1997: 32-33).

Even though acid rain was by then an important political topic, due to growing pressure by environmentalists, Congress was stuck in a regulatory gridlock, with over seventy bills directed at mitigating acid rain being rejected before 1989 (McCauley *et al.*, 1988). Until then, the CAA merely mandated NSPS for new power plants, implicitly assuming that emissions would go down as old plants were replaced by new ones. The differential treatment of old and new power plants, however, created an incentive for prolonging the life of old power plants, significantly hampering the effort to reduce emissions (Ackerman *et al*., 1998). In addition, states often chose to achieve NAAQS by requiring power plants to install taller smokestacks, to increase pollution dispersion, which resulted both in significant interstate pollution transport and a worsening of the acid rain problem (Ellerman *et al*., 2005: 14).

The impasse was overturned by the approval of the 1990 CAA Ammendments, which implemented the Acid Rain Program (ARP), an emissions trading system for $SO_2$ emissions from large coal plants. In addition to installing Flue Gas Desulphurization equipment (scrubbers), coal plants now also had the option to substitute high-sulphur coal with low-sulphur coal and/or buying $SO_2$ allowances to comply with the CAA. Since scrubbers can remove more than 90 percent of $SO_2$ emissions, coal plants that chose to install them would become net allowance sellers, while plants that chose not to could become net buyers (Winebrake *et al*., 1995).

The ARP was divided into two phases. From 1995 to 1999, Phase I applied to the 263 largest $SO_2$ sources from utilities, corresponding to 110 highest emitting power plants (almost all were coal plants from the northeast and midwest). The cap was set according to an emissions rate of 2.5 pounds of $SO_2$ per million Btu of heat input, multiplied by the historical fuel use in the 1985-1987 period, leading to an emissions reduction of about 3.5 million tons per year.[7] From 2000 to 2010, all fossil-fuelled power plants producing more than 25 MW and new energy generating units were subjected to a declining cap that would limit yearly SO2 emissions to 8.95 million tons.

7 Throughout this chapter, the term "ton" refers to the US unit of measurement, which is equivalent to 2,000 pounds. This is not to be confused with "tonne" or "metric ton", which equals 1,000 kilograms nor with the non-US ton, which equals 2,240 pounds.

The cap was intended to reduce yearly SO2 emissions by 10 million tons, or about 50 percent, from 1980 levels. This number was determined by multiplying the emissions rate specified in NSPS by total emissions from coal plants, so the 1990 CAAA merely repeated the environmental objective that was implicit in the 1970 CAA (Driesen, 2010). Findings from the National Acid Precipitation Assessment Program (NAPAP), which cost half a billion dollars and employed over three thousand scientists, as well as from other scientific studies, were ignored by Congress when deciding on the ARP cap (Heinzerling, 1995). Rather than discussing the merits of different caps, congressional debate was focused on appeasing states and industries to secure their support, through various provisions that distributed additional allowances (Joskow and Schmalensee, 1998).

Sources were allocated allowances for free. To reward early action and spur trade, sources could trade Phase I allowances from the beginning of the 90's and trade Phase II allowances in the Phase I period. Banking was also allowed without restrictions (Napolitano *et al.*, 2007a).

The initial allocation of allowances in Phase I was corrected by several special provisions that allocated additional allowances. To favour the eastern high-sulphur coal business, power plants that installed scrubbers were awarded bonus allowances. Additionally, bonus allowances were awarded to sources that invested in energy conservation or renewable energy, and some large power plants in states that successfully lobbied for cost sharing provisions. To assure that the cap was still met, a "ratchet" provision was also incorporated, reducing the allocation to all sources by the same amount as the bonus allowances given by special provisions. The same pattern of special provisions that corrected the initial allocation due either to technical issues or to successful lobbying by utilities and states was observed in Phase II, with over thirty special provisions that made allocation rules difficult to track and understand (Ellerman *et al.*, 2005: 36-48).

Another innovation was the inclusion of two "opt in" provisions, that made it possible for non-Phase I electricity generating units to enter the market in Phase I. The substitution provision allowed firms to substitute emissions reductions from Phase I units for reductions in non-covered units. The compensation provision made non-covered sources participate in the program when utilities shifted electricity production from covered to non-covered sources. About 30 percent of eligible sources chose to participate in the ARP, mostly through the substitution provision, through which units that were reducing emissions for reasons unrelated to the program could become net allowance sellers (Schmalensee *et al*., 1998; Montero, 1999).

To spur trading and give a clear price signal to affected firms, as well as accommodate the entrance of new sources of emissions, an annual allowance auction was introduced. Every year, the EPA would set aside a small part of the allowances (3.1 percent in Phase I and 2.8 percent in Phase II) for auctioning. Utilities could also sell allowances through the auctions, specifying quantity and minimum price. Revenues were then redistributed on a *pro rata* basis to utilities, making the auctions revenue-neutral (Svendsen and Christensen, 1999).

A final innovation was the Direct Sales Reserve provision, created to introduce a price ceiling for allowances. From 1993, sources could buy up to 25,000 allowances for \$1500, adjusted for inflation. Since prices were lower than anticipated, this provision was never used and in 1997 it was terminated (Joskow *et al*., 1998).

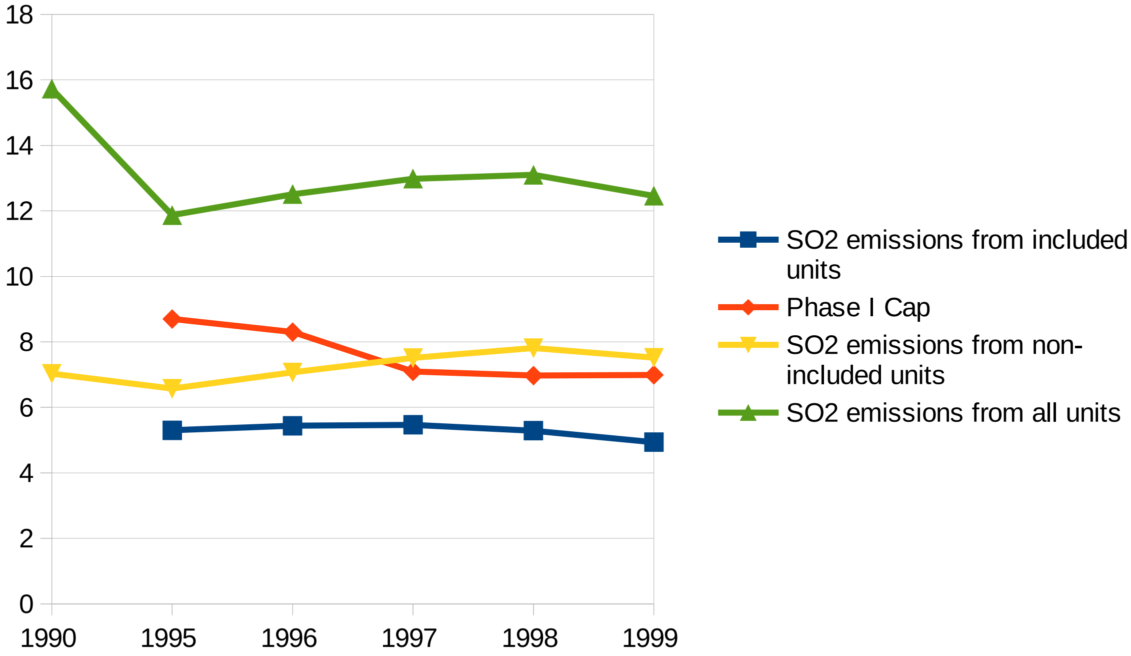
Monitoring in the ARP was assured by the mandatory instalment of Continuous Emission Monitoring Systems (CEMS), which submit data on hourly emissions in quarterly reports to the EPA. Power plants that failed to deliver allowances to cover their emissions would face a fine of \$2000 per ton of $SO_2$, corrected for inflation, as well as a deduction of excess emissions in the following year's allocation.

During Phase II, several federal and state rules intended to tighten the cap on $SO_2$ and $NO_x$ emissions were proposed and approved. In 2002, the W. Bush administration proposed the Clear Skies Act, but it failed to garner political support for it. The failed proposal gave way to the Clean Air Interstate Rule (CAIR), promulgated in 2005, which intended to reduce $SO_2$ emissions to 73 percent below the 2003 level. This new rule targeted eastern states that were in non-attainment regarding both ozone ($O_3$) and $PM_{2.5}$ NAAQS partly due to transport of precursors. Following CAIR, the ARP would be replaced by a new $SO_2$ emissions trading program in 2010, with a second phase starting in 2015. To achieve the new environmental objective, sources in states with $PM_{2.5}$ concentrations above the NAAQS would have to deliver two $SO_2$ allowances for each ton of emissions from 2010 to 2014 and 2.86 allowances for each ton of emissions after 2014. Since banking was allowed, there was a strong incentive to reduce emissions before 2010 and bank allowances for future use (Palmer and Evans, 2013; Schmalensee and Stavins, 2013).

In 2008, CAIR was successfully challenged in court by North Carolina, a downwind state, for not adequately considering the effect of emissions transport across states. The DC Circuit court initially vacated the rule but after the EPA filed petitions arguing that this would severely affect its capacity to control pollution and protect human health, the court remanded the rule. This decision maintained CAIR as originally promulgated but also placed a burden on the EPA to come up with a new regulatory scheme that would not allow upwind states to significantly increase emissions (McCubbin, 2009).

The answer to the new dilemma came in 2011 with the Cross-State Air Pollution Rule (CSAPR), which would replace CAIR in 2012. The new rule severely limited inter-state trading, so emissions trading was restricted mostly to sources located in the same state. In 2012, however, the rule was invalidated by the DC Circuit court, following petitions by a groups of states. This led the EPA to appeal to the Supreme Court, which reversed the DC Circuit decision in 2014. As a result, CSAPR was implemented in 2015, aiming to achieve by 2017 a 73 percent reduction in $SO_2$ emissions from 2005 levels.

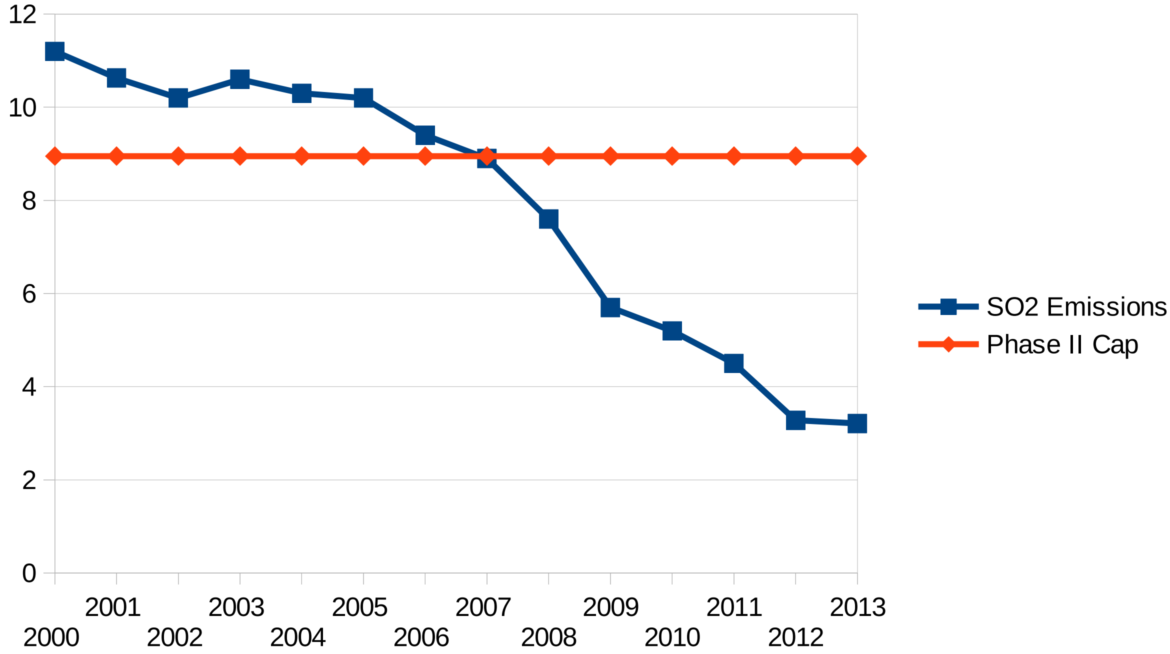
***Figure 2: $SO_2$ emissions and ARP caps, 1990-1999 ($10^6$ tons)***




*Source: EPA (2015)*

Phase I of the ARP was marked by substantial emissions reductions between 1990 and 1995, of about 25 percent for all sources. Emissions from units included in Phase I were stable between 1995 and 1999, having reduced a mere 7 percent, while non-included units actually increased their emissions. The cap, calculated by the complex set of formulas given by allocation provisions, was always significantly higher than emissions, with polluters banking an average of about 2 million allowances per year.

***Figure 3: $SO_2$ emissions and ARP caps, 2000-2013 ($10^6$ tons)***




*Source: EPA (2015)*

Emissions reductions were much steeper in Phase II. By 2007, the 10 million ton $SO_2$ emissions reduction from 1980 had already been achieved, three years before the initial deadline. From 2008, after CAIR was implemented, even if provisionally, emissions reductions accelerated again, and by 2003 SO2 emissions were 81 percent below 1980 levels. While the ARP achieved an average yearly 1 percent reduction in Phase I and 3 percent between 2000 and 2007, the introduction of CAIR led to an average yearly 13 percent reduction between 2008 and 2013.

Early reductions in the 1990s can be partially explained by significant changes in coal and electricity and markets. Productivity improvements in extraction and transport, as well as a significant reduction of rail rates following the 1989 deregulation, led to a substantial decline in the price and an increase in availability of western and eastern low-sulphur coal. During the same period, electricity demand shifted and became more concentrated in areas closer to low-sulphur coal reserves. This led the use of low-sulphur coal to become more economical and utilities invested in process innovations to blend high-sulphur coal with low-sulphur coal to use in boilers designed to burn high-sulphur coal only (Burtraw, 1996). Almost all emissions reductions prior to 1993 were from Midwest coal plants, as it became cheaper for these plants to use low-sulphur coal from Western reserves in the Powder River Basin (Ellerman and Montero, 1998).

Ellerman *et al*. (2000: 77-106) estimate that declining low-sulphur coal prices and rail rates was the main factor for emissions reductions until 1994. During Phase I, coal switching and mixing was the main means of compliance with the ARP, representing 52 percent of units and 59 percent of emissions reductions, while only 10 percent of units used scrubbers, representing 28 percent of emissions reductions (Lange and Bellas, 2005).

The market for emissions abatement equipment also changed in the early 1990s, with scrubber prices falling and their effectiveness increasing, even though patent counts related to $SO_2$ control technologies had decreased (Lange and Bellas, 2005; Popp, 2003). While the 1990 CAAA provided a disincentive for innovation in scrubber technologies, the combination of emissions standards and public R&D funding had by then already facilitated their rapid maturation (Taylor *et al*., 2005).

Additionally, as it takes two to three years to install scrubbers and allowance prices during Phase I were overestimated, many utilities chose to commit themselves to installing scrubbers before 1995, even though it was not the most cost-efficient option considering the low allowance prices observed. Since scrubber operational costs are very low, it would not make sense not to use them once installed. Some coal plants also committed themselves to buying more low-sulphur coal than what would be economical, through long-term supply contracts (Ellerman *et al*. 2000: 302-305). Curiously, then, a part of early emissions reductions can be explained by errors in estimating the cost of complying with the ARP.

Due to these early reductions, emissions were always below the cap in Phase I, allowing utilities to bank excess allowances. During the first years Phase II, coal plants were able to emit more than the cap by using banked allowances. The introduction of CAIR, however, dramatically changed the regulatory status. Since utilities were confronted with having to face a 50 percent discount factor on allowances used after 2010, and since allowances banked before this date were not subjected to the discount factor, there was a strong incentive to reduce emissions before CAIR came to effect and bank excess allowances. By 2012, 44 percent of ARP units had installed scrubbers to comply with new regulations, with only 4 percent using other abatement methods (EPA, 2015).

Regarding the distribution of emissions, empirical analyses based on Phase I data do not show signs of pollution “hot spots” being created by the ARP, nor disproportionate impacts on regions with a higher number of low income residents or people of colour. Data on the geographical distribution of trading shows that the major emissions reductions came from the biggest emitters (Swift, 2000, 2004), while crossing emissions data with census data shows that there were no significant differences in the racial and income demographics between coal plants that increased emissions and those that reduced emissions or between allowances buyers and sellers (Corburn, 2001; EPA, 2005b).

As a result of the emissions reductions imposed by CAIR, allowance prices declined rapidly. While allowance prices were always below \$200 until late 2003, discussions over stricter regulations on $SO_2$ emissions led to a demand surge that raised prices up to about \$1550 in late 2005 (EPA, 2006a). After the EPA announced CAIR prices dropped to \$400-\$600 during 2006 and 2007 (EPA, 2007). In 2008, when litigation led to CAIR being vacated and then remanded, priced dropped to about \$100 and then steadily declined (Schmalensee and Stavins, 2013). During 2011, prices were below \$5, showing that lowering the cap on $SO_2$ emissions and introducing restrictions to trading effectively killed off the acid rain market, as it led to a *de facto* mandatory installation of scrubbers on the largest $SO_2$ sources (EPA, 2012).

The higher than expected emissions reductions achieved after the 1990 CAAA led emissions trading proponents to present the first “cap and trade” scheme adopted in the world as a success, or even a “living legend of market effectiveness” (Burtraw and Palmer, 2004). None of the analyses published on the ARP, however, presents an international comparison with policies followed in other industrialized economies. Engaging in that comparison, the 29 percent reduction in SO2 emissions in the 1990-2000 period does not seem all that impressive when compared to the 61 percent reduction achieved in the EU

(EEA, 2014).

The significant progress in reducing $SO_2$ emissions in the EU is mostly due to the implementation of direct regulation that made coal plants and other major sources to use abatement technologies. This regulation, in turn, followed the 1984 Helsinki Protocol, the 1994 Oslo Protocol and the 1999 Gothenburg Protocol to the 1979 Convention on Long-Range Transboundary Air Pollution, which specified critical loads for each country and significant emissions reductions. While the 1994 protocol already opened the way for Joint Implementation, through which a country could offset its emissions by investing in emissions-reducing projects in other countries, the specification of national critical loads restricted flexibility enough to made this option too costly to be a reality (Bailey *et al.*, 1996; Rodríguez, 1999; Menz and Seip, 2004).

Within the EU, the most impressive emissions reductions were achieved by Germany. Following the approval in 1983 of the Ordinance on Large Combustion Plants, Germany started to introduce strict standards on both emissions and desulphurization rates. Compared to the 1990 CAA standards for Phase I of the ARP, the $SO_2$ emissions standards for large sources specified in the German 1988 Large Scale Combustion Act were about 85 percent lower (Schwarze, 2005: 57). Combined with voluntary agreements with private utilities, this led to emissions going down much faster than what was required, amounting to an 84 percent reduction in the 1980-1990 period in West Germany (Wätzold, 2004). In the 1990-2000 period, following reunification, Germany reduced its $SO_2$ emissions by a further 88 percent, almost three times as much as the US (EEA, 2014). While this intervention was very costly, since it demanded a massive investment in scrubbers and fuel switching on existing power plants, the final environmental goal was stringent enough to make cost savings from a hypothetical emissions trading system negligible (Bültmann and Wätzold, 2000).

The EU case-study, together with the demise of the ARP following the approval of CAIR, shows how the potential efficiency gains associated with emissions trading are negatively correlated with its environmental effectiveness. It seems, then, that the superior cost-efficiency of the ARP compared to direct regulation or taxation is more due to its unambitious emissions cap than to the flexibility in compliance.

## 3.3 Regional emissions trading systems

### The Regional Clean Air Incentives Market

In the late 1980s, the Los Angeles air basin had the worst air quality in the USA, with frequent high-ozone episodes. Tropospheric ozone ($O_3$) is formed by the interaction of $NO_x$ and Volatile Organic Compounds (VOCs) in the presence of sunlight and results in lung, throat and eyes irritation and respiratory illnesses, as well as crop damage. While $NO_x$ emissions result mostly from burning fossil fuels in power plants, industrial boilers and cars, VOC emissions can be attributed to multiple sources in chemicals manufacturing and use, as well as natural sources.

To address this problem, the South Coast Air Quality Management District (SCAQMD) elaborated a new Air Quality Management Plan (AQMP) in 1989 that would complement the existing LAER standards that were already applied to new sources under NSPS with BACT standards for old sources. Industry reacted by creating the Regulatory Flexibility Group, a lobby that opposed direct regulation and proposed an emissions trading program. Finding legal support in the 1990 CAAA, that introduced emissions trading for $SO_2$, this lobby managed to persuade the SCAQMD to shelve its AQMP and replace it with a proposal for two "cap and trade" markets for ozone precursor $NO_x$ and acid rain precursor sulphur oxides ($SO_x$) (Zerlauth and Schubert, 1999).[8]

The Regional Clean Air Incentives Market (RECLAIM) was approved in the end of 1993 and implemented from 1994, replacing previous SCAQMD rules. The new emissions trading system applied to facilities emitting more than four tons of either $NO_x$ or $SO_x$ per day, as well as other facilities that voluntarily opted in. Emissions allowances were called RECLAIM Trading Credits (RTCs) and were issued for a given compliance year and pollutant. Each year, each facility had to deliver enough RTCs to cover its $NO_x$ and $SO_x$ emissions, with each RTC being equivalent to a pound of emissions. Since inter-

8 A market for VOCs was originally proposed by the SCAQMD but never materialized due to concerns with monitoring and environmental justice, as well as industry and environmental groups opposition (Thompson, 2000; Egelston and Cohen, 2004).

pollutant trading was not allowed, this configuration in effect created two separate emissions allowance markets.

Trading was further restricted by dividing the LA Basin into two geographical areas, inland and coast. To account for air currents that drift emissions from the coast to inland, sources located on the coast could not buy allowances from sources located inland.

Banking allowances for future use or sale was also not allowed, due to fears of future emissions increases. To allow for some temporal flexibility and avoid a rush to the market by the end of each year, facilities were divided into two groups, one facing a reporting period from January through December, and the other facing a reporting period from July through June, and trading allowances between these groups was allowed. Each firm received in 1994 all the RTCs for future years, up to 2010, to accommodate for the predictability required in long-term investments (Schubert and Zerlauth, 1999).

The initial allowances were allocated to sources on the basis of the highest annual emissions between 1989 and 1992, estimated by multiplying throughput with an emissions factor. This calculation was then corrected upwards to account for pre-1994 emissions reductions and to counteract estimated recessionary effects on firms. In 2000, the number of allocated allowances was recalculated according to a more stringent emission coefficient and the minimum input since 1994. The number of RTCs was to diminish at a constant rate, to achieve the same objective of the direct regulation policies originally proposed, a 75 percent reduction for $NO_x$ emissions and a 60 percent reduction for $SO_x$ emissions by 2003, relative to 1994. After 2003, allowance allocations would remain constant (EPA, 2006b). As the cap was progressively lowered and economic growth led to increasing allowance demand, RECLAIM was projected to allow the LA Air Basin to reach NAAQS attainment by the 2003 deadline at a cost 40 percent lower than direct regulation would (Hall and Walton, 1996; SCAQMD, 1996).

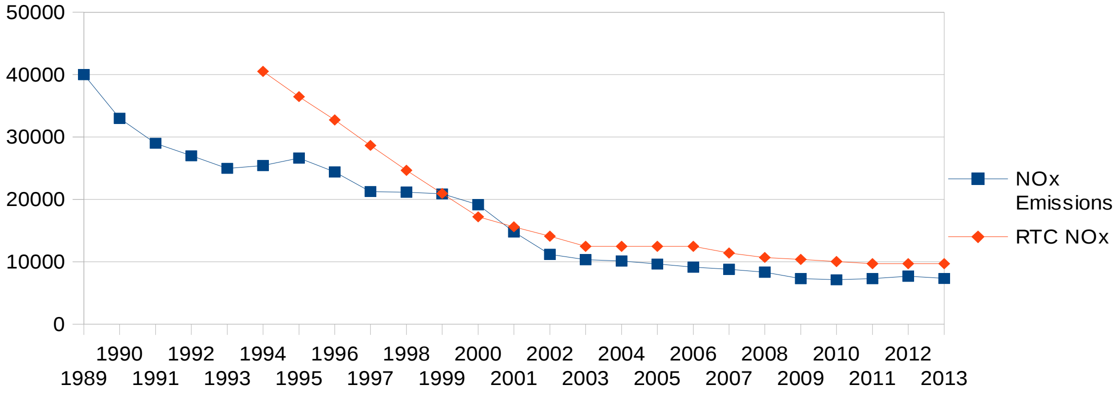
To assess compliance with RECLAIM, as well as its environmental performance and economic impacts, the SCAQMD also approved Rule 2015, which states that the program would be the target of annual audits, as well as a three-year audit in 1997. The rule further specified a mechanism for reviewing compliance and enforcement norms, to be enacted if the allowance average annual price exceeded $15,000 (SCAQMD, 1993).

Two offsetting provisions were implemented to allow RECLAIM sources to buy credits from mobile or stationary sources that could be converted to RTCs. Mobile Source Emission Reduction Credits (MSERCs), created under Rule 2008, were issued by reducing emissions from vehicles, namely by scrapping old polluting vehicles or even lawn and garden equipment and replacing them with lower emissions alternatives or by using low-emissions buses on public transport systems. Area Source Credits (ASCs), created under Rule 2506, were generated by sources that voluntarily reduced their $NO_x$ or $SO_x$ (Drury *et al*., 1999).

To monitor emissions, major sources, defined as those with the potential to emit 10 or more tons of $NO_x$ and 100 or more tons of $SO_x$ annually, had to install CEMS and report emissions daily, while large sources of $NO_x$ reported quarterly emissions estimated by multiplying measured fuel use and emission coefficients. If the SCAQMD found the emissions data lacking credibility, it could use a “missing data” clause and assume a worst case scenario volume of emissions. Facilities that failed to deliver enough allowances to cover their emissions were penalized by having their next year allocation reduced in the same amount as excess emissions. Trespassers could also face a civil penalty or even a withdrawal of the facility’s operating permit (Schubert and Zerlauth, 1999).

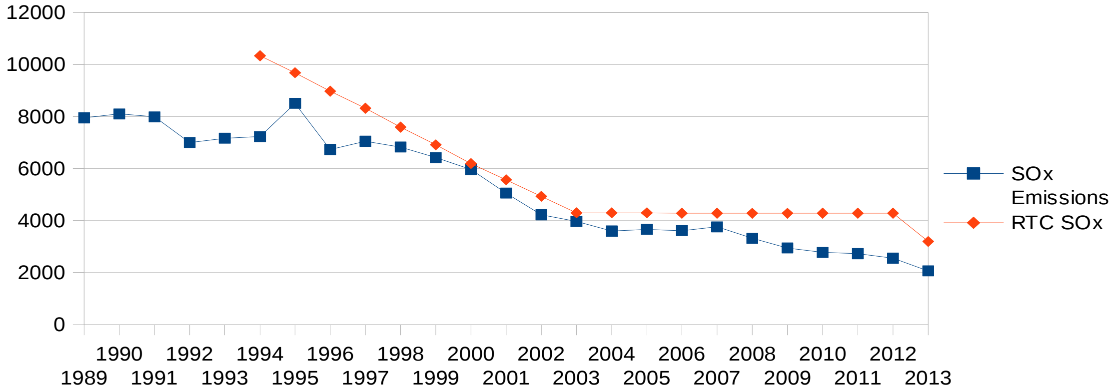
Over the years, the cap was reviewed to account for new information on emissions, new sources entering the area and credits generated by various provisions. The objective was to reflect a technologically feasible abatement volume but the revisions of $NO_x$ allowance allocation resulted in a substantial increase of the cap. In the end, initial allocations during the first two years were 40 to 60 percent above emissions (EPA, 2002) and until 2003 the aggregation of individual allowances exceeded the level compatible with NAAQS (Schubert and Zerlauth, 1999).

***Figure 4: $NO_x$ emissions and RTC $NO_x$ allocations, 1989-2012 (tons)***




*Source: SCAQMD (2015)*

***Figure 5: $SO_x$ emissions and RTC $SO_x$ allocations, 1989-2012 (tons)***




*Source: SCAQMD (2015)*

These revisions and the generous over-allocation of allowances led to a disappointing environmental performance. The cross-over point, when emissions were above allocations, was initially predicted by the SCAQMD (1996) to be achieved in 1997-1998 for $NO_x$ and 1998-1999 for $SO_x$. Since emissions reductions were smaller than predicted, this point was achieved only in 2000, when $NO_x$ emissions exceeded allocations by about 11 percent and $SO_x$ emissions were just 4 percent below allocations. This reflected the California energy crisis of 2000-2001, when market manipulation by by the

Texas energy consortium Enron, facilitated by the previous deregulation of energy markets, and higher summer temperatures led both to a lower supply and higher demand for energy, resulting in a major price increase (FERC, 2003). Satisfying energy demand led to increased production in old gas-fired plants, most of which didn't have $NO_x$ emissions controls, resulting in increased emissions (SCAQMD, 2003).

To address this crisis, the SCAQMD formed an advisory committee which presented recommendations to stabilize allowance prices. Consequently, participation of electricity generating units in RECLAIM was suspended temporarily in May 2001 and the newly drafted Rule 2009 subjected electricity generating units to direct regulation. Each facility was required to meet an $NO_x$ emissions rate compatible with the Best Available Retrofit Control Technology (BARCT), as early as possible but not later than 2004, for turbines used as peaking units, or 2003, for all other units. To prevent market manipulation, utilities subject to BARCT regulations were prevented from selling $NO_x$ allowances until 2007. Utilities that exceeded the limit on emissions could buy allowances through the Mitigation Fee Program, at a price of $15,000. The revenues generated by this measure were then applied to remove allowances from the market. Additionally, non-complying utilities were also penalized with an equivalent deduction from future allocations (SCAQMD, 2001).

The new regulations reduced emissions far more rapidly than what was achieved with emissions trading. In the 1994-2000 period, the yearly average rate of $NO_x$ reductions was a mere 4 percent, far below the 7.5 to 11.9 percent projected by the SCAQMD and the 6 to 10 percent the EPA estimated for what direct regulation could have achieved (Johnson and Pekelney, 1996; EPA, 2002). In the 2001-2003 period, when the biggest $NO_x$ sources were *de facto* removed from RECLAIM, emissions went down by a yearly average of 11.2 percent. Still, the environmental target for 2003 $NO_x$ emissions was not met, nor was the target for $SO_x$ emissions, with emissions reductions amounting to 59 percent for $NO_x$ and 45 percent for $SO_x$. After trading resumed, between 2004 and 2013, the yearly rate of $NO_x$ reductions dropped again to 3.2 percent.

With the aim of achieving further emissions reductions, the SCAQMD reviewed its rules and in 2005 implemented changes to RECLAIM. The revised allocation rule would lead to a reduction of over 20 percent for $NO_x$ emissions until 2011. Even with a more stringent cap, though, $NO_x$ emissions reductions in 2013 relative to 1994 were just 71 percent, which is still below the 75 percent target that was to be achieved a decade earlier. As for $SO_x$ emissions, the 2003 target was met only in 2010, but new regulations and a revision of the cap, which will decrease by about 48 percent between 2013 and 2019, led to a 71 percent decrease in the 1994-2013 period (SCAQMD, 2015).

Empirical evidence also points to the creation of pollution "hot spots" due to the spatial flexibility allowed by emissions trading. Lejano and Hirose (2005) modelled the spatial dispersion of $NO_x$ with and without RECLAIM, concluding that there was a large potential for the concentration of pollution in certain localities where a number of large emitters, namely refineries, converged. Drury *et al.* (1999) reach a similar conclusion by drawing a parallel with the observed concentration of VOC emissions that resulted from using credits from car scrapping to offset emissions from refineries. In both cases, the authors note that an environmental justice problem arises, since the localities where "hot spots" are likely to occur have a much larger share of poor people and people of colour than the regional average. While this problem can be attenuated through restrictions on trading between different localities, based on continuous monitoring of local air quality, the added transaction and administrative costs would significantly affect the attractiveness of emissions trading compared to direct regulation.

Mobile source credits were also problematic due to the possibility of car scrappers fraudulently earning MSERCs by scrapping cars that were no longer in circulation, even though no emissions reductions resulted from it (Drury *et al.* 1999). Since the EPA never formally approved this mechanism for generating credits, its legality was questionable. Following lawsuits filed by local environmental justice organizations, eight out of nine firms that used MSERCs for compliance were penalized by agreeing to invest in further emissions reductions and environmental improvement projects (Egelston and Cohen, 2004).

Compliance with RECLAIM regulations has been lower than acceptable. An internal audit reported that in the first year 30 percent of sources had not installed CEMS, while in 1999, about 40 percent of large sources and 80 percent of small sources still didn't comply with the reporting requirements (Drury *et al.,* 1999). Data from audit reports furthermore shows that there were always facilities that failed to deliver enough allowances to cover their emissions, indicating that penalties for non-compliance were not stringent enough, with over 10 percent of facilities not complying with RECLAIM in 1994, 1996 and 2000.

Allowance prices have been low, reflecting the generous allocation, with the exception of the 2000-2001 energy crisis period, which affected the $NO_x$ market in particular. While before 2000, average allowance prices were always below \$500 per ton, in 2000 the average price for a $NO_x$ RTC reached \$45,609 per ton. In 2001, the average $NO_x$ RTC price rose to \$52,237 per ton, reaching a maximum of \$124,000 per ton in February. After the market interventions to stabilize prices, RTC prices dropped, with 2002 NOx RTCs being traded at an average \$5,110 per ton, far below the \$15,000 backstop price (SCAQMD, 2003). From 2003 to 2012, allowance prices for current year RTCs remained below \$4,000, so the backstop provision was never again used (SCAQMD, 2014).

As with the ARP, it seems evident that the cost savings attributed to RECLAIM can be explained by the sacrifice of emissions reductions, rather than the flexibility in compliance (Moore, 2004). Furthermore, the experience with the 2000-2001 energy crisis shows that “cap and trade” systems are only viable when the cap is not binding, in particular when banking is not allowed, and can easily crumble apart when the cap becomes binding.

### The $NO_x$ Budget

In the 1990s, tropospheric ozone ($O_3$), also known as “smog”, was a major pollution problem in Northeast and Mid-Atlantic states, where many coal plants were located. Considering that the ozone precursor $NO_x$ can be transported by the wind across states and that inter-state cooperation was needed, the 1990 CAAA created the Ozone Transport Commission (OTC). In 1994 the OTC states signed a Memorandum of

Understanding to reduce their $NO_x$ emissions, which would be the basis for the creation of a "cap and trade" system in 1999, the $NO_x$ Budget Program (NBP).

Following the 1990 CAAA, large $NO_x$ sources faced Reasonably Available Control Technology (RACT) standards, which specified emissions standards compatible with technologies like low-$NO_x$ burners. The NBP built on these regulations to subject energy generating units with installed capacity of at least 25 MW, as well as industrial boilers with heat input greater than 250 mmBtu per hour, to a global "budget" on $NO_x$ emissions, calculated to achieve the same level of reductions as BACT standards. Emissions limits were to be lowered progressively, to reach a reduction of 55 percent by 2002 and 75 percent by 2005, relative to 1990 levels (Farrel *et al*., 1999; Farrel, 2000; Napolitano *et al*. 2007).

From 2003, the NBP entered a new phase, now being centrally administrated by the EPA, rather than the OTC. This decision followed the 1998 EPA $NO_x$ SIP Call, which required a total of 22 Eastern states, including the OTC states, to deliver SIPs detailing measures to achieve an 85 percent reduction of $NO_x$ emissions from the 1990 levels. Due to various court proceedings, the deadline for SIP submission was extended to October 2000, while the deadline for emissions reductions was extended to June 2004 (Burtraw and Evans, 2003).

Emissions allowances in the NBP were distributed freely by the EPA to participating states based on historic emissions and then redistributed by the states to affected sources, according to state rules. States could set aside a part of the budget to allocate allowances for new sources and energy efficiency or renewable energy projects. States could also issue additional allowances under the Compliance Supplement Pool, to reward early emissions reductions or avoid negative impacts on the electricity supply, in the first two years of each phase of the NBP (Napolitano *et al*., 2007).

The $NO_x$ market operated only during the ozone season, from May to September, when tropospheric ozone levels are higher due to the increased solar radiation. At the end of this period sources had to deliver enough allowances to cover their accumulated $NO_x$ emissions.

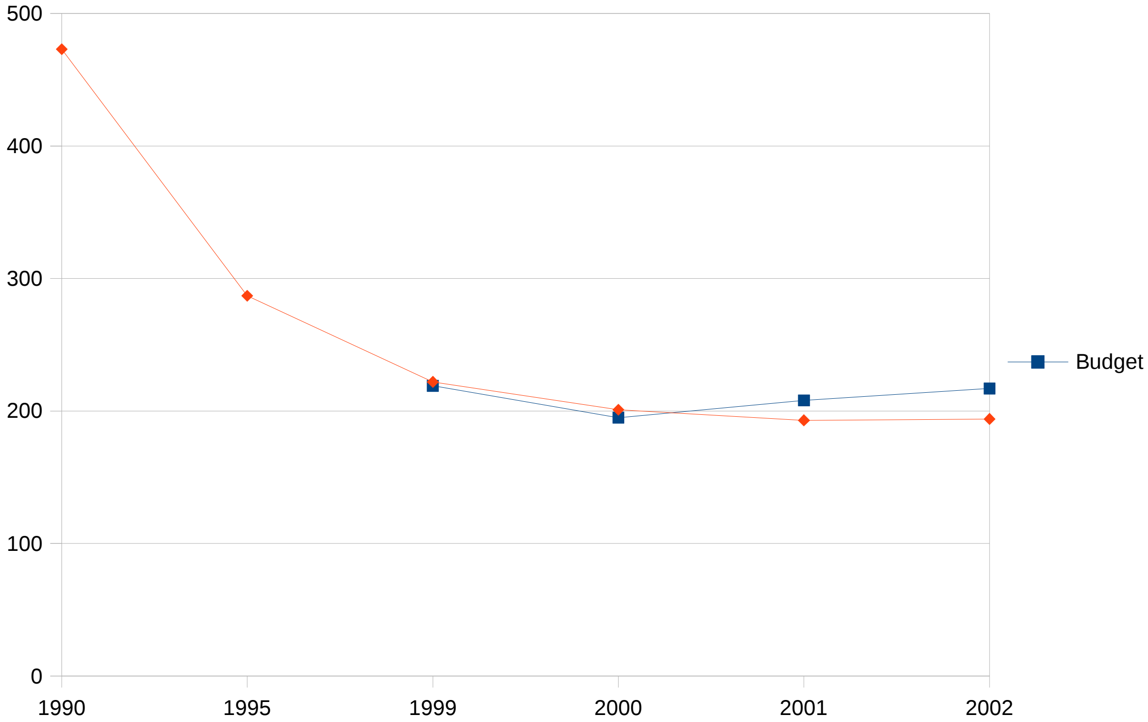
As with other emissions trading systems, emissions monitoring was guaranteed by the mandatory installation of CEMS for large sources and fuel use reporting for small sources. Non-compliance was penalized by reducing the following year's allocation by three times the number of excess $NO_x$ emissions.

Banking surplus allowances was allowed, subject to limitations given by the Progressive Flow Control mechanism (PFC). The mechanism applied when the number of banked allowances from all sources exceeded the region's budget by 10 percent or more. When this happened, a PFC ratio was calculated as 10 percent of the ratio between the current year budget and the banked allowances and a 2:1 discount factor was applied on the use of allowances above this critical level (Farrel, 2000). By having sources deliver two allowances for each ton of emissions when the ratio of banked allowances used for compliance exceeded the PFC ratio, this mechanism added a dimension of risk to banking.

Further flexibility was delivered by an opt-in provision, adopted by all states, that allowed non-affected sources to voluntarily join the NBP. Additionally, about half the states included a provision that allowed sources emitting less than 25 tons of $NO_x$ to opt out of the program and face an operating hour limit.

The NBP was to be replaced in 2005 by CAIR, which implemented both an annual and an ozone season $NO_x$ markets, aiming to reduce $NO_x$ emissions by 61 percent from 2003 levels. State $NO_x$ budgets were calculated according to what highly cost-efficient control methods could achieve and not according to each state contribution to downwind pollution, an option that motivated the court decision to vacate CAIR in 2008 (McCubbin, 2009). In 2009, following the new court decision to keep CAIR in place until a revised rule was approved, the CAIR $NO_x$ programs began. Banked allowances from the NBP programs could be used in the CAIR programs without restrictions. These programs were in turn replaced by CSAPR $NO_x$ trading in 2015, which aims to reduce $NO_x$ emissions by 54 percent from 2005 levels.

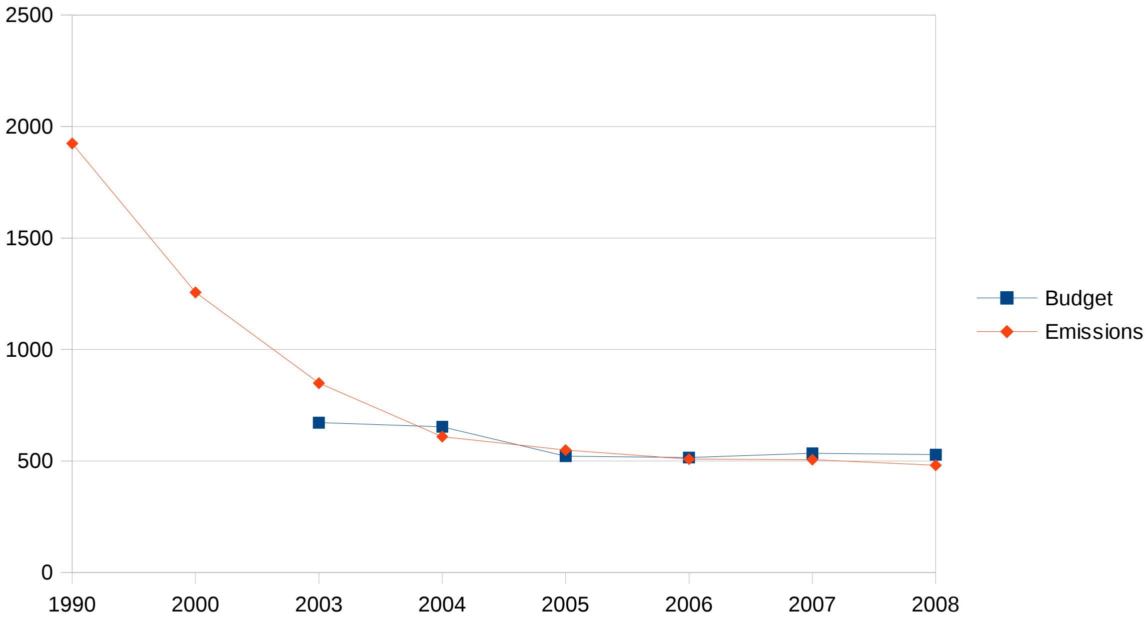
***Figure 6: Ozone season $NO_x$ emissions and budget in the OTC NBP, 1990-2002 ($10^3$ tons)***




*Source: EPA and OTC (2003)*

During the 1999-2002 period, when the NBP implemented by the OTC was in place, $NO_x$ emissions decreased by an average 3 percent per year, much less than the 8 percent yearly reductions achieved by previous regulations. Even though the environmental objective was met, with emissions being reduced by a total of 59 percent in 2002 from 1990 levels, the combination of generous allowance allocations and banking allowed polluters to avoid having to invest in control equipment or technologies that could lead to higher emissions reductions.

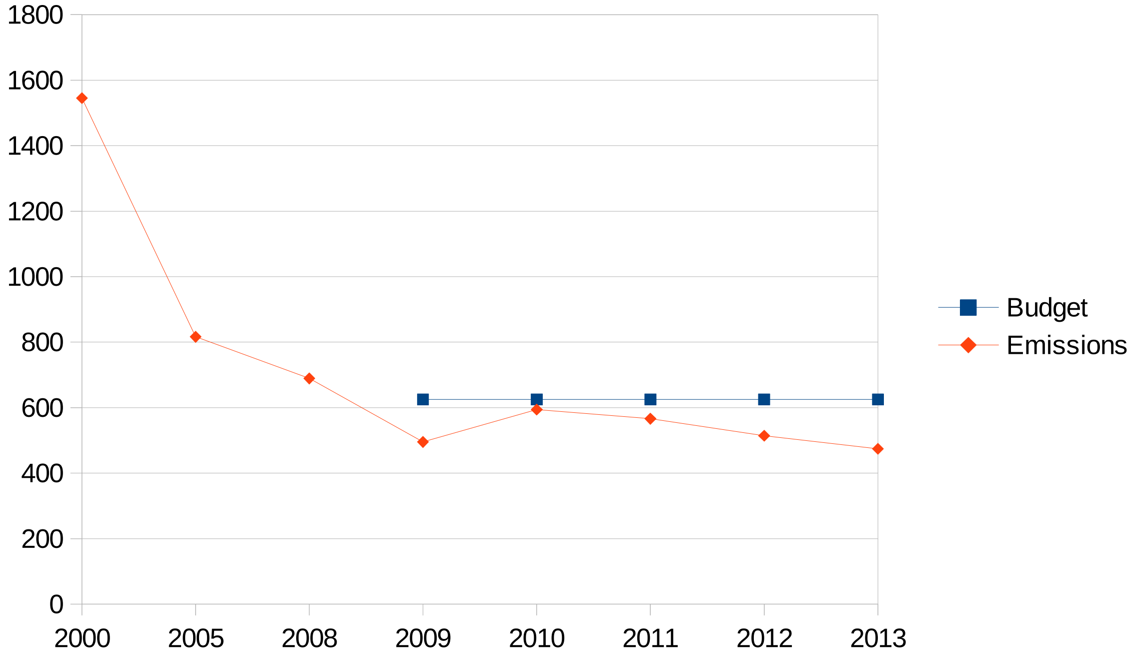
***Figure 7: Ozone season $NO_x$ emissions and budget in the SIP Call NBP, 1990-2008 ($10^3$ tons)***




*Source: EPA (2009)*

The 1998 SIP Call had a major impact on $NO_x$ emissions, which went down by 52 percent between 2000 and 2004, the final deadline for compliance. After this regulatory push, polluters in the revised NBP did not reduce their emissions substantially and the rate of yearly emissions reductions between 2005 and 2008 was again just 3 percent. Even with the push, however, emissions reductions in 2005 from 1990 levels amounted to just 71 percent, below the initial objective of 75 percent, which was reached only in 2008.

***Figure 8: Ozone season $NO_x$ emissions and budget in the CAIR seasonal program, 2008-2013 ($10^3$ tons)***




*Source: EPA (2011, 2015)*

With the replacement of the NBP by CAIR, emissions reductions are, so far, very low, amounting to just 1 percent per year. Even so, the trading system is set to achieve the 2015 environmental goal, with emissions reductions in 2013 adding up to 42 percent from 2005 levels.

The $NO_x$ market has been characterized by a large use of banking and consistently low allowance prices (Burtraw and Evans, 2003; EPA, 2011). While allowance prices increased to a record high of about $7,000 in 1999, due to fears over a possible shortage of allowances, they stabilized at an average below $750 in the following years of the NBP (EPA and OTC, 2003). The implementation of the SIP Call led to a shortage of allowances and a new price spike, with allowances being sold for about $8,000 in early 2003 and dropping to below $3,000 in the summer, but in later years allowance prices stayed below $4,000 and declined progressively, reaching $592 by the end of 2008 (EPA, 2006b, 2009). With the approval of CAIR, between 2009 and 2012, allowance prices dropped to an average below $30 (EPA, 2015).

Contrary to RECLAIM, the NBP was marked by significant emissions reductions. This is due mostly to the approval of tighter controls, with the SIP Call and CAIR, to prevent inter-state pollution transport. The seasonal $NO_x$ market, however, faces the risk of following the route of the $SO_2$ market, with new regulations increasingly leading polluters to install control equipment rather than buying allowances, thus corroding the market.

## 3.4 Conclusions

Soon after the 1970 CAA defined the NAAQS and the deadlines to achieve these standards, the EPA started to introduce market-like principles in the air quality regulations. The NAAQS were not challenged or revised according to cost-benefit analyses but rather a mix of bubbles, netting, offsets and banking provisions were introduced to allow polluters to evade some regulations by using credits generated from emissions reductions in other sources. These provisions were then unified in the ETP, which, despite its name, was not really an emissions trading system but rather a patchwork of flexibility provisions that were superimposed on pre-existing regulations and that were subjected to many restrictions. The same characterization applies to the lead averaging program, which allowed refineries to face laxer rules for lead phase-out.

Experimenting with ad hoc flexibility provisions that emulated emissions trading was a stepping stone for the development of the ARP, RECLAIM and the NBP, the first emissions trading schemes to be implemented in the world. From the early 1970s, when emissions trading literature started to take off, until the early 1990s, when several "cap and trade" schemes were approved, both the theory and practice of emissions trading was entirely US-based. The motives for the emergence of emissions trading as an alternative policy instrument cannot, therefore, be found in general statements about inefficiencies of direct regulations but rather in specific elements of US politics. In particular, the conceptualization of emissions trading as a "third option" that could break the impasse between a historical aversion of policy-makers to taxes and the polluters resistance to direct regulation follows a bias in Environmental Economics towards crude generalizations based on the US experience (Braadbart, 1998).

Emissions trading appeared firstly as a means of discursively reconciling the CAA standards with industrial growth and environmental protection with the emergent neoliberalism. While the first experiments appeased the growing anti-regulatory fervour that reached its height with the Reagan administration, the national and regional emissions trading systems implemented in the 1990s appeased the fossil fuel-based industry, in particular coal extraction and burning.

Following the US historical experience, we can trace how emissions trading schemes were set up in the real world. First, a cap on emissions was defined by considering the merits and costs of available technologies, much like with direct regulation, and then adding a margin to appease polluters that led to emissions being below the cap. Tradable emissions allowances were then distributed to polluters for free, granting them economic rents proportional to historical emissions. Compliance with the program was evaluated by installing equipment that measures emissions continuously and with a high rate of precision. If polluters could not comply with the (modest) emissions reductions mandated by the cap and allowance prices surged, government intervention in the market would assure that polluters would not have to face a too high cost of compliance. If, on the other hand, stricter environmental regulations led to increased emissions reductions, the quantity allowances in excess of the cap would be high enough to make allowance prices drop to a point where the emissions market would effectively be terminated.

Using US emissions trading schemes as case-studies, rather than analysing theoretical abstractions, we can, then, present some important lessons about this regulatory instrument. First, the practice of emissions trading largely preceded the theory, contrary to what is implied by Hahn's (1989a) metaphor on the patient following the doctor's orders. Second, optimistic arguments raised by emissions trading supporters regarding the possibility of the superior cost-efficiency and attractiveness to polluters of this instrument leading to greater emissions reductions (e.g. Stavins, 1998, Ellerman, 2005) were misguided, as in existing “cap and trade” systems the cap was set according to emissions limits present in standards from direct regulation. Third, the democratic case for emissions trading is also misguided, since the historical evidence shows that the political process of approving a “cap and trade” system, in particular regarding cap setting, was not more transparent and less technocratic than the process of approving emissions standards. Fourth, informational requirements of emissions trading systems tend to be superior to direct regulation, since these systems require gathering precise information on emissions, while compliance with emissions standards can be evaluated through audits on pollution control technologies, fuel use and other industrial data. Fifth, even though allowances can be auctioned, raising revenue and preventing rent creation and rent-seeking behaviour, governmental authorities have chosen consistently to give them away to polluters. Sixth, emissions trading can lead to the creation of pollution “hot spots”, which can only be prevented by restrictions to trading that increase transaction costs. Seventh, while emissions trading can provide important opportunities for flexibility in compliance, this flexibility diminishes and eventually disappears as the cap is lowered or technology-push regulations are approved. Eighth, given the contrast between real life experiments and idealized emissions trading systems, it is clear that implementing emissions trading did not simplify regulation but rather added new dimensions of complexity.

These lessons can, hopefully, inform the debate on instrument choice, by transcending simplistic representations based on an opposition between non-flexible, inefficient and complex “command and control” regulations and flexible, cost-efficient and simple “market-based instruments”. The debate should rather be informed about perspectives on acceptable levels of pollution and costs of pollution control, development trajectories and other societal understandings, to determine which instruments should be included or excluded from the policy-mix. This chapter, by presenting historical data and a critical review of the first emissions trading schemes, is a contribution to that debate.

# 4 Actually Existing Carbon Trading: From Kyoto to the EU ETS

“The carbon market is a politically created market and needs carefully calibrated rules.” (EC 2014c: 5)

Climate change, resulting from the accumulation of anthropogenic greenhouse gas emissions, mostly from burning fossil fuels, has resulted in increasingly widespread and considerable environmental and social impacts. According to the Intergovernmental Panel on Climate Change (IPCC), its consequences include increased and more severe extreme weather events, like heat waves, hurricanes and floods, which lead to losses namely in biodiversity, agricultural productivity and forest cover (IPCC, 2014). Given that the most important greenhouse gas is $CO_2$, which is pervasive in industrial production and cannot be eliminated through abatement technologies, climate change mitigation is arguably the hardest challenge ever presented to environmental regulation. The challenge has been met by introducing a new instrument, carbon trading, in a move that was both innovative and contentious.

Until its introduction as a means of compliance in the Kyoto Protocol, in 1997, carbon trading was an exotic idea to most policy-makers. Emissions trading theory and practice was until then a US product, and the idea that industrialized countries could replace domestic emission reductions with foreign reductions was met at first with opposition and suspicion from most government and civil society representatives. This opposition, however, quickly gave way to acceptance and reformism. Whether this was a positive or negative evolution depends on the performance of carbon markets, which this chapter critically reviews.

The structure of this chapter is as follows. The first section presents a history of carbon trading in climate negotiations, which was at first a major point of dispute between the US and most of the rest of the world and is now, as the centrepiece of international climate action, one of the most important issues on the agenda. The second section reviews the experience with the Clean Development Mechanism, which allows polluters in developed countries to finance investment projects located in developing countries that reduce emissions, receiving in exchange a number of credits equivalent to the reductions achieved, which can be then used as a means of compliance with emissions commitments. The third section reviews the history and the performance of the EU Emissions Trading System, which is since its implementation the largest carbon market in the world. The fourth section concludes by presenting a critical discussion of carbon trading based on its history and the available empirical data.

## 4.1 Carbon trading in climate negotiations

### Negotiating the Kyoto Protocol

International negotiations on climate change started with the approval of the United Nations Framework Convention on Climate Change (UNFCCC) in the 1992 Earth Summit. The UNFCCC included the non-binding commitment by countries listed in Annex I, which included Western European countries, the US, Canada, Australia, New Zealand, Japan, Turkey and economies in transition (Russia and Eastern European countries), to stabilize their greenhouse gas (GHG) emissions by 2000 at the 1990 levels. Annex I countries also committed themselves to create national inventories of GHG emissions and sinks and develop national and regional strategies for mitigation and adaptation to climate change. Furthermore, countries listed in Annex II, which only included industrialized countries not considered as economies in transition, were to promote “green” technology transfer to aid other countries. These commitments followed the common but differentiated responsibilities principle, which places the burden of early action on the countries with the largest historical emissions (UN, 1992).

Two important developments had already taken place when the UNFCCC was negotiated. First, the IPCC had released its first assessment report, in 1990, which informed decision-makers about the perils of climate change resulting from anthropogenic GHG emissions and recommended starting negotiations on an international climate treaty (Houghton *et al*., 1990). Second, the possibility of meeting national commitments to GHG emissions reductions through emissions trading had already been raised and supported by an UNCTAD (1992) report. Both the need to address climate change through reductions in anthropogenic carbon emissions and the interest in allowing industrialized countries to meet their obligations through carbon trading were, therefore, already on the negotiating table in the early 1990s.

Further negotiations under the UNFCCC proceeded with the yearly Conferences of Parties (COPs). In 1995, COP-1, in Berlin, approved a mandate to complete negotiations on a global climate treaty that would include binding commitments, with the opposition of the US and Australia. The same mandate also initiated a pilot phase of Activities Implemented Jointly (AIJ), through which carbon offsetting projects could take place, specifying that these projects should result in real, measurable and long-term benefits, that these benefits should be additional relative to what would have occurred in the absence of the project and that no credits would accrue to any party (UNFCCC, 1995).

Following the Berlin Mandate, the EU presented in March 1997 a proposal for a 15 percent emissions cut by 2010, which would be applied within in its borders through a burden sharing agreement that set differentiated targets for each of the fifteen Member States. The EU also proposed that OECD countries committed themselves to the same target (Yamin, 2000). In stark contrast, in July the US Senate approved, by unanimity, a resolution that prevented ratification of an international climate treaty that would not impose binding commitments on developing countries or was deemed to be too harmful for the US economy (Byrd, 1997). The evolution of negotiations and the compromises achieved were considerably influenced by the clash between EU and US negotiating positions.

Negotiations on the first protocol on climate change concluded in COP-3, which took place in December 1997. The Kyoto Protocol (KP) included binding commitments for industrialized countries, averaging a 5.2 percent emissions reductions in the 2008-2012 period, from 1990 levels. Emissions targets oscillated from -8 percent to the EU-15 and -7 percent to the US to +10 percent to Iceland and +8 percent to Australia. The Protocol also allowed these commitments to be met through international carbon trading and offsetting

(UN, 1998).

Carbon trading was introduced by allowing countries with binding emission commitments to trade in emission allowances, called Assigned Amount Units (AAUs). Additionally, two offsetting mechanisms were introduced: Joint Implementation (JI) and the Clean Development Mechanism (CDM). While the former would apply to economies in transition and the latter to non-Annex I countries, both would lead to carbon credits being generated from emissions-reducing projects, following the experience with AIJ. To avoid double-counting, countries that sold JI credits, called Emission Reduction Units (ERUs), had to retire an equivalent amount of AAUs, making JI essentially a different form of emissions trading among Annex I countries. Countries that sold CDM credits, called Certified Emission Reductions (CERs), did not face the same requirements, since they didn't have binding emission commitments.

Both emissions trading among Annex I countries and the flexibility mechanisms were supported by the “Umbrella Group”, which included the US, Japan, Canada, New Zealand, Russia and Ukraine, and opposed by the rest of the world, represented by the G77+China and the EU. The CDM, in particular, has its roots in a Brazilian proposal for a Clean Development Fund, which would be financed by fines payed by countries that exceeded their emissions limits and would be used to invest in adaptation and mitigation projects in non-Annex I countries. The US, opposing any type of sanctions for non-compliance, managed to convince Brazil to change its proposal into a market mechanism, in a move that was known as the “Kyoto surprise” (Werksman, 2002).

The US negotiating position in Kyoto was, therefore, based on the promotion of international carbon trading attached to (weakened) binding commitments. The contradiction between this position and the US Senate refusal to ratify any climate treaty reflects a growing division between industry coalitions following opposing strategies. Since 1989, major polluters, including fossil fuel and energy-intensive industries, were actively opposing climate action, through the corporate lobby Global Climate Coalition (GCC). This industry coalition followed a strategy that was based on discrediting climate science and presenting a possible ratification by the US of a climate treaty as a disaster for the US economy and a subsidy for foreign companies. But by 1995 there was a new coalition forming out of an alliance between industrial and financial firms and environmental NGOs to support carbon trading, through the International Climate Change Partnership (ICCP). The pro-trading lobby was significantly strengthened by BP's decision in May 1997 to leave the GCC and rebrand itself as an environmentally conscious firm, following Dupont's example. Shell soon followed suit, leaving ExxonMobil isolated as the only oil giant opposing a climate treaty (Kolk and Levy, 2001).[9]

The EU in Kyoto opposed carbon trading, proposing instead coordinated policies and measures, some of which would be mandatory. While the G77+China also opposed emissions trading, the EU failed to get widespread support for the list of coordinated policies and measures. Its stance was further weakened by the demand to negotiate a single target for the EU, which would then be disaggregated into national targets for the fifteen Member States according to an internal burden-sharing agreement. Since this was, in practice, a “bubble” scheme, the US framed it as a zero-cost emissions trade, arguing that EU's stance on carbon trading was incoherent (Cass, 2005).

At the end of the COP-3, opposition to carbon trading gave way an agreement through which trading was to be voluntary and supplemental to domestic action and its design details would be negotiated over the next years. Developing countries opposition to international offsetting was watered down by the allocation of a share of the proceeds from CDM projects to cover adaptation costs in developing countries that are particularly vulnerable to climate change impacts (Grubb *et al*., 1999: 87-107).

This evolution essentially reflected a victory for the US negotiators, which is illustrated by the similarities between the original draft protocol presented by the US and the final text of the KP (U.S. Department of State, 1997). Following the initial US proposals, not only carbon trading was included without any significant restriction but also

9 To be sure, this change of direction did not imply a support of stronger climate regulations. Both the “climate denialist” and the “free market environmentalist” factions of the fossil fuel industries followed strategies aimed at getting powerful states to block strong transnational action in the emerging regulatory institution that was then the UNFCCC (see Levy and Egan, 1998). This is evident in the speech that BP's CEO, John Browne, gave to announce the company's new “climate-friendly” strategy, in which he nevertheless opposed drastic cuts in carbon emissions or a ban on the use of fossil fuels arguing that such actions would compromise economic growth (Browne 1997).

the deadline for compliance was pushed into a 2008-2012 commitment period, GHG targets were defined in a basket that included trace gases from industrial processes – hydrofluorocarbons (HFCs), perfluorocarbons (PFCs) and sulphur hexafluoride ($SF_6$) –, on top of the most common gases – carbon dioxide ($CO_2$), methane ($CH_4$) and nitrous oxide ($N_2O$) –, included GHGs were converted into $CO_2$ equivalent ($CO_2$eq) using the relative Global Warming Potentials (GWPs) and carbon removals by forestry or agricultural activities was deducted from national emissions inventories. Even though all of these proposals were initially rejected by the EU and the G77+China, major concessions were made to assure the US participation in the protocol, deemed essential for its survival. The effort did not pay off, however, as the US chose not to ratify the protocol anyway.

**Post-Kyoto climate negotiations**

After the KP was approved, climate negotiations proceeded with defining the rules and procedures for carbon trading. The roadmap was set up in the 1998 COP-4, with the Buenos Aires Plan of Action (UNFCCC, 1999), but EU's attempt to save face by limiting the use of carbon trading and the US's attempt to expand flexibility as much as possible evolved into a standstill.

At the 2000 COP-6, in The Hague, the EU argued that no more than half of emissions reductions commitments should be realized through carbon trading, following the supplementarity principle present in the KP, and defended a “positive list” of types of eligible offset projects, which would exclude forestry and other activities related to carbon sinks. The US, on the other hand, argued against quantitative or qualitative limits to offsetting. Regarding the role of sinks in carbon emissions accounting for industrialized countries, the EU wanted to limit their inclusion, namely because of the scientific uncertainties in measuring the carbon stored in plants and soils, while the US viewed forest and agricultural activities that store carbon as a cost-effective means of compliance (Gardiner, 2002; Zapfel, 2002).

The clash was followed by the US unilateral withdrawal from the KP. The decision was firstly announced in March 2001, through a letter from US President George W. Bush to members of the Senate, declaring his opposition to the protocol and repeating the arguments given in the 1997 Senate Resolution (Bush, 2001). This was a major victory for the GCC, even though the anti-regulation lobby was losing ground to the pro-trading lobby and dissolved one year later (Meckling, 2011: 97-98). Removing the US from the negotiating table did not mean that carbon trading was shelved in climate negotiations, however, but rather that, paradoxically, it became easier for the rest of the world to reach an agreement regarding how to implement carbon trading. The EU, in particular, gradually reversed its opposition to unrestricted carbon trading, in part to appease US's allies in the "Umbrella Group" (Cass, 2005).

The CDM rules were specified in the Marrakech Accords, negotiated at the 2001 COP-7 (UNFCCC, 2002). The accords addressed some reserves regarding carbon trading by stating that developed countries had no "right, title or entitlement" over the atmosphere and that the CDM should contribute to sustainable development, rather than be a means for evading domestic reductions obligations. Since no restrictions on trading nor a quantitative limit to the use of carbon credits were specified, however, these norms were merely aspirational and symbolic.

Concerns with the CDM integrity were addressed by establishing a complex bureaucracy composed by Designated National Authorities (DNAs) to approve participation in CDM projects, Designated Operational Entities (DOEs) to validate claims made in each Project Design Document (PDD) and a CDM Executive Board (EB) to supervise the approval and monitoring of CDM projects. Regarding land-use and forestry, concerns the accuracy of measuring carbon stored in soils and forests led to a compromise solution whereby only afforestation and reforestation activities were included in the CDM and Annex I parties could only use these credits for compliance up to one percent times five of their 1990 emissions (Boyd *et al*., 2008).

As for JI, its guidelines for implementation were approved at the 2005 COP-11, in Montreal (UNFCCC, 2006b). A JI Supervisory Committee (SC) was cast in the same mould as the CDM EB but, considering that JI is very similar to emissions trading among countries with emissions commitments, projects' host countries were allowed to bypass the SC approval by following certain requirements regarding emissions quantification.

Projects could then either follow “track one” and be “party-verified” or follow “track two” and be “independently verified”.

Following Russia's ratification in 2004, the KP finally came into force in early 2005. Negotiations on the post-2012 emissions reductions then followed, with the Bali Road Map, approved at the 2007 COP-13, specifying procedures to reach a new binding agreement by 2009 (UNFCCC, 2008). At the 2009 COP-15, however, negotiations ended on a standstill and no binding agreement was produced. Instead, a voluntary agreement, the Copenhagen Accord, was submitted by the US and BASIC countries (Brazil, South Africa, India and China), through which national pledges could be made with the objective of limiting global mean temperature rise to 2ºC (UNFCCC, 2009). The accord could not become an official UNFCCC agreement due to opposition from several developing countries that wanted a stronger agreement but still it heavily influenced subsequent negotiations.

Voluntary pledges for emissions reductions to achieve by 2020 would become a part of the Cancun Agreements, negotiated at the 2010 COP-16 (UNFCCC, 2011). The second commitment period for the KP was finally introduced at the 2012 COP-18, in the Doha Amendment, which also added the gas NF3 to the basket of greenhouse gases to be controlled (UNFCCC, 2012a). The target for emissions reductions in the 2013-2020 period, relative to 1990, was set at 18 percent.

The KP first commitment period, from 2008 to 2012, was characterized by weak emissions reductions commitments by Annex I countries. Global emissions actually increased by 45 percent between 1990 and 2012, even though most industrialized countries complied with their Kyoto commitments (UNEP, 2014). The apparent paradox is explained largely by the relocation of polluting industries from Annex I countries to non-Annex I countries. Comparing production and consumption emissions, Davis and Caldeira (2010) concluded that in 2004 22.5 percent of China's emissions were exported to consumers in other countries, while the UK and France imported over 30 percent of consumption-based emissions. Since emission inventories used to evaluate Kyoto commitments are conducted according to production emissions and not consumption emissions, every time a factory moves from an EU country to China EU's emissions are reduced, even if the factory is exporting all of its production to the EU.

The second commitment period can have even worse results. Emissions pledges span from a 3 percent reduction for the US and a 20-30 percent reduction for the EU, considering 1990 as the base year (UNFCCC, 2011). These targets, while unambitious, can be weakened by the use of allowances from land-use change activities and “hot air” allowances banked from the first commitment period. Russia and Ukraine alone, having reduced their emissions by over 50 percent of what was required, accumulated 2.271 million allowances between 2008 and 2012, enough to cover nearly all 2012 emissions from Germany, the UK, France and Italy (UNFCCC, 2014c). Considering the impact of these extra allowances, which can be used subject to restrictions stipulated in the Doha COP-18, Chen *et al*. (2013) estimate that pledges can lead to a 10-11 percent emissions reduction in industrialized countries. Since economic growth in industrializing countries will inevitably lead to emissions increases, however, Rogelj *et al*. (2010) estimated that global emissions could increase by 10-20 percent in the 2010-2020 decade even if the pledges are met. The UNEP (2014) estimates a 6-12 percent increase over the same time period but even with this less pessimistic estimate global emissions in 2020 will still be about 18-23 percent higher than the level consistent with the 2$^{o}$C target.

Concerning the post-2020 period, following the Durban Platform for Enhanced Action, approved in the COP-17, a “new and universal greenhouse gas reduction protocol, legal instrument or other outcome with legal force” will have to be negotiated by 2015 (UNFCCC, 2012b). It is currently uncertain what kind of agreement will emerge but the history of climate negotiations tells us that it is unlikely that it will contain binding and ambitious commitments. This conclusion is reinforced by Canada's decision to withdraw from the KP in 2011, as well as Russia's, New Zealand's and Japan's refusal to deliver pledges for 2020 (Chen *et al*. 2013).

Despite not being able to produce measurable results regarding emissions reductions, climate negotiations have been successful in pushing forward new carbon trading and offsetting schemes. The Durban Platform introduced the New Market

Mechanism (NMM), an umbrella term that can encompass diverse mechanisms for trading in carbon credits, beyond the CDM and JI. One option on the negotiating table is complementing project-based offsetting from the CDM and JI with sector-based offsetting, through which carbon credits can be generated from whole sectors or regions, following mitigation plans predicted in developing countries' Nationally Appropriate Mitigation Actions (NAMAs) (UNFCCC, 2013). Another option is generating carbon credits from activities that plant or conserve trees, as predicted in the framework for Reducing Emissions from Deforestation and Forest Degradation (REDD+), approved at the 2013 COP-19, in Warsaw (UNFCCC, 2014b).

These new flexibility mechanisms are complemented by the expansion of regional, national and sub-national carbon trading systems. Besides the regional carbon market in the EU, national carbon markets have been established in New Zealand, Australia, Switzerland, Republic of Korea and Kazakhstan and sub-national carbon markets exist in the US, Japan, China and Canada. National or sub-regional carbon markets are also under consideration in Chile, Mexico, Thailand, Turkey, Ukraine and Brazil (Kossoy *et al*. 2015). These existing and proposed carbon markets operate under different rules, namely regarding their mandatory or voluntary nature or the existence of a cap on emissions, which makes it potentially very difficult to link different markets. As a result, instead of converging towards a global carbon market, countries engaged in carbon trading seem to converge instead towards a weakly integrated patchwork of regional, national and sub-national carbon markets.

Even though carbon trading has been expanding internationally, most trades are still concentrated in the CDM and the European carbon market (*ibid.*). To assess the relevance of carbon trading, each of these schemes will be exposed and analysed in the next two sections.

## 4.2 The Clean Development Mechanism

The first step towards creating international offset crediting mechanisms was given with the inclusion of AIJ in the 1995 Berlin Mandate. Even though these projects could not generate carbon credits, they were attractive to investors that wanted to gain experience with offsetting and enjoy “first mover” advantages when a crediting mechanism was established. The latest AIJ assessment reported that the number of projects increased from 37 in 1997 to 157 in 2006, most of which focused on the energy sector (80 percent) and a majority of which implemented in economies in transition (55 percent) (UNFCCC, 2006a).

Partially based on the experience with AIJ, the CDM was implemented in 2001, allowing Annex I countries to prepare for KP's first commitment period by buying CERs and banking them for future use. Considering the uncertainty regarding the future of the KP, though, CERs only started to be issued in early 2006. Since then, the CER market has grown rapidly and as of September 2015, a total of 1,614 million CERs were issued and 7,764 projects were registered, encompassing 108 host countries (UNEP DTU Partnership, 2015).

Project data also shows that CER supply is highly skewed in favour of some countries and technologies. Almost half of all registered projects and almost 60 percent of CERs issued until 2014 had China as the host party, while India had about a fifth of registered projects and 13 percent of CERs issued. In total, 84 percent of registered projects are located in the Asia and Pacific Region, while Africa has only 2.5 percent (UNFCCC, 2014a).

These discrepancies can be explained in part by the replication, within the CDM, of the limitations that developing countries face accessing financial markets, as well as the correlation between industrial development and emission reduction potential (Jung, 2006; Bohr and Dill, 2011). Furthermore, the considerable bureaucracy and high costs inherent to project approval discriminate against small-scale projects, namely community-based projects that can have a positive impact on poor regions (Lövbrand *et al.*, 2009). As a result, CDM projects tend to channel finance to large polluters located in industrializing countries.

This trend can also be illustrated by the distribution of CERs issued by project type. Almost half of CERs issued during until the end of the KP first commitment period came from projects that rely on end-of-pipe reductions in non-$CO_2$ gases, with a third being issued to abatement of industrial gases (HFCs, PFCs, $SF_6$ and $N_2O$) and 11 percent being issued to $CH_4$ abatement in landfills and coal mines. Although investment in renewable energy accounted for 34 percent of issued CERs, almost half of this investment was directed towards building large dams, while investments in solar energy, which can be highly beneficial for local communities, accounted for a mere 0.2 percent of total issued CERs (UNEP DTU Partnership, 2015).

An example of a particularly problematic category of CDM projects is the destruction of HFC-23, a waste gas that results from HCFC-22 production. Even though HCFCs are ozone-depleting substances and its production is being phased out under the Montreal Protocol, developing countries face laxer rules and only have to reduce its production by 10 percent by 2015 and 35 percent by 2020 (UNEP, 1987). Industries located in these countries can install HFC-23 incinerators and sell CERs according to the conversion rate specified in the KP (the GWP), so that each ton destroyed is worth 11,700 CERs. Since marginal abatement costs are very low, below $1 per ton of $CO_2$eq, HFC-23 elimination is a highly profitable business, even in China, the host country of most projects, where the tax rate reaches 65 percent (Shin, 2010).

The high profitability of HFC CDM projects raises doubts about their integrity, as it actually makes economic sense to increase HCFC-22 production beyond the demand just to earn CDM money. To address this concern, the CDM methodology for this project category limits CER generation to historical levels of HFC-23 emissions but still empirical evidence suggests that HCFC-22 production would have been lower without the CDM revenue (Schneider, 2011).

Furthermore, profits from HFC destruction projects can be seen as an indication of a waste of resources, since it is much more costly to buy CERs from these projects than to subsidize the installation of HFC-23 incinerators. Even before the KP first commitment period began, Wara (2007) estimated that paying HCFC-22 producers to install HFC-23 destruction equipment would cost only €100 million, saving €4.6 billion in CERs. Similarly, Shislov and Belassen (2014) report HFC-23 abatement costs from two CDM projects at US$0.1, far below the average carbon price of US$21 registered between 2008 and 2012. The estimates show how the CDM, taken as a subsidy, is highly cost-inefficient and results in a massive rent to HFC destruction project developers.

Another critical issue with the CDM is additionality, that is, the requirement that CERs are not attributed to emission reducing investments that would have happened even if the project developers had not received CDM funding. This requirement is crucial to guarantee the environmental integrity of the CDM. If CERs are generated from non-additional projects, then their use for compliance purposes leads to a net increase in global carbon emissions. Furthermore, developers of non-additional CDM projects receive a windfall profit, since the revenue from selling CERs is not associated with any action beyond business-as-usual.

Using a random sample of registered CDM projects, Schneider (2009) distinguishes four different approaches to assessing additionality. The first, positive lists, automatically classifies a project as additional if it fits within a certain category. This is rarely used, and in most cases the CDM EB uses a combination of the other three approaches: barrier, investment and common practice. Barrier analysis consists in demonstrating that there are barriers that prevent the project from being carried out. Investment analysis uses an opportunity cost approach to make the case that there is a better alternative for investing the funds than the project. Finally, common practice analysis requires a demonstration that the technology or practice inherent to the project type has not been extensively used in the relevant sector or region. All of these analyses are problematic, since it is not clear what counts as a barrier or common practice or how to establish financial attractiveness and project developers frequently make unsubstantiated claims and fail to provide detailed data. Furthermore, many projects (a third of the reviewed sample) are retroactively financed, which shows that the concept of additionality has been stretched to the point where the validation or registration date can be posterior to the project activity start date.

Assessing additionality, like calculating a CDM project contribution towards reducing emissions, is not merely a technical task since it involves measuring future emissions reductions relative to an imaginary baseline, which has to be somehow isolated from an undetermined and possibly infinite set of possible futures. Project developers have an incentive to inflate baseline emissions as much as possible and can be rewarded with CERs for investments that would have been made anyway because they're profitable or required by law. On the other hand, since non-additional CERs can be used for compliance, CER buyers do not have an incentive to discriminate against fraudulent CDM projects. Given the impossibility of objectively determining additionality and the lack of incentives for quality control in the CER market, the CDM becomes unregulatable, despite the considerable technical apparatus set up around it (Lohmann, 2009a).

Empirical data on project distribution by country and type, as well as on additionality analysis, exposes the deep contradiction that exists within the CDM, since, on the one hand, it was designed to minimize the costs of abatement by directing investment in emissions reducing projects to where they are cheaper and, on the other hand, it was purported to deliver sustainable development. The trade-off between these two objectives has been widely documented (Olsen, 2007; Paulsson, 2009), as well as the failure of the CDM to offer significant employment, health or environmental benefits, while having detrimental social and environmental effects (Boyd *et al*., 2009; Docena, 2010; Shin, 2010; Ghosh and Sahu, 2011; Bond *et al*., 2012).

The main source of demand for CERs has been the european carbon market, but new rules approved for the post-2012 period have led to a CDM breakdown. To understand why, it is necessary to analyse the evolution of the first large-scale carbon market in the world, which was born out of a EU volte-face on emissions trading soon after the Kyoto negotiations.

## 4.3 The European Carbon Market

### From resistance to adoption

The first attempt to regulate carbon emissions in the EU appeared at the time of the Rio Summit, in 1992, when a carbon tax was proposed. Fierce opposition by industrial lobbies and lack of consensus in the Council of Finance Ministers, however, made the proposal approval impossible and it was formally withdrawn in 2001 (Christiansen and Wettestad, 2003). The failure to implement a carbon tax, combined with the failure to promote its coordinated policies and measures approach in Kyoto and growing support for carbon trading from industry lobbies, Member States and European Commission (EC) officials, set the stage for a remarkable U-turn, with the EU quickly shifting from a major opposer to a major supporter of carbon trading (Wettestad, 2005).[10]

In the run-up or shortly after the Kyoto climate summit, a growing number of heavy polluters changed their stance from climate action opposition to carbon trading support. This trend was particularly evident in the EU as already in early 1998, the Union of Industrial and Employers' Confederation of Europe (UNICE, now called BusinessEurope) was defending that the EU should introduce emissions trading between companies to address climate change (UNICE, 1998). The industry association representing energy suppliers, Eurelectric, went one step further and ran economic models in its Greenhouse Gas and Energy Trading Simulations (GETS), starting in early 1999, to estimate the effects of emissions trading. After realizing that utilities could profit from passing through the opportunity cost of freely allocated allowances, Eurelectric became another strong advocate for emissions trading (Meckling, 2011: 112-113).[11]

The two european oil giants, BP and Shell, went even further and actually set up internal carbon trading systems, in 1998 and 2000, respectively (Christiansen and Wettestad, 2003). BP, in particular, had an instrumental role in the development of carbon trading in the EU. On the one hand, its internal emissions trading system, which ran until 2002, was presented as a success case, since BP managed to reduce GHG emissions by 10 percent below 1990 levels and save over $650 million through increased gas venting and flaring and increased energy efficiency (Victor and House, 2006). On the other hand, BP's

10 Damro and Méndez (2003) discuss the adoption of carbon trading by the EU not as a U-turn or as a concession to the US but rather as a reconciliation of the competing policy paradigms: US free market environmentalism and EU risk-prevention leadership. Carbon trading allows the flexibility that the US was promoting, while also delivering the emissions reductions commitments that the EU was defending. We do not follow this approach since it downplays the political relevance and the environmental effects of the EU not only accepting carbon trading but also dropping its proposals for limits to trading in carbon allowances and credits.

11 The opportunity cost of an allowance used for compliance is the market price that it would reach if it was sold. Accordingly, opportunity costs are calculated as the market value of freely allocated allowances.

expertise allowed it to assign members of its staff to key lobbying positions in industrial associations like UNICE, as well as being invited to informal meetings with EC's Directorate-General for Environment officials (Braun, 2009). This two-tiered strategy was complemented in 2000 with the company's rebranding as “bp: beyond petroleum”, a marketing move intended to convince the public that BP was no longer an oil giant but rather an environmentally-friendly energy company committed to solar energy (Beder, 2002).

Some EU Member States were also experimenting with emissions trading after Kyoto. In 1999, Denmark introduced a carbon trading system for electricity providers, implemented in the 2000-2003 period (Pedersen, 2001). More significantly, the UK created an Emissions Trading Group (ETG), an institution created in 1999 to engage industries and government bodies in a proposal for a national emissions trading scheme. The ETG was set up at BP's headquarters, which was symbolic of the oil company's leadership role (Meckling, 2011: 111-112). The UK Emissions Trading System (UK ETS) was launched following industry opposition to the proposed Climate Change Levy (CCL), a tax on carbon emissions that would be implemented in 2001. The UK ETS was expected to provide an early experience with emissions trading for businesses, help to establish the City of London as an international centre for emissions trading and influence the design of a future EU-wide carbon market (Von Malmborg and Strachan, 2005).

The UK ETS was a mix of different policies. Industries could choose to participate in a “cap and trade” system or in a “baseline and credit” system, facing absolute or relative targets, respectively. Firms not incorporated in any of these systems could also invest in offset projects and sell carbon credits (Rees and Evers, 2000). In any case, firms could voluntarily participate in the system by participating in the 2002 auction and bid emissions reductions pledges to be achieved over the 2002-2006 period, using historical emissions in the 1998-2000 period as a baseline. Participants were awarded carbon allowances for free according to their pledges, which were tradable and bankable. Those that complied with the pledges were also rewarded with a share of the £215 million that was made available as incentive payments. This generous subsidy raised the question of whether there were market participants that could be rewarded for emissions reductions that would have happened anyway. In fact, an assessment of the UK ETS by the House of Commons criticized this subsidy and concluded that some participants were rewarded with incentive payments for emissions reductions that they had achieved prior to 2002 (House of Commons, 2004).

Considering that the UK ETS was a complex overlap of different policies and that it wasn't really an alternative to carbon taxation, which wasn't dropped, but rather a form of compensating polluters for the costs of the CCL, its conception and implementation by an industrial lobby can be seen as a symbolic step towards the political acceptance of carbon trading (Nye and Owens, 2008). Framing carbon trading as the business-friendly alternative to taxation and regulation in the UK was seen by major polluters like BP as a stepping stone towards making the same case in the EU.

Within EU institutions, the first step towards carbon trading acceptance was given only months after the Kyoto climate summit, when the EC published its communication on a post-Kyoto strategy (EC, 1998). In this communication, international carbon trading and offsetting was already presented as a means to achieve emission commitments at a lower cost for the EU industry. The door was opened also to setting up an internal carbon trading system by 2005, to accumulate practical experience and prepare for international carbon trading, which would take off in 2008.

This first step was followed in 1999 by another EC communication, on the KP implementation, which announced a consultation with stakeholders, Member States, businesses and NGOs on a carbon trading pilot phase (EC, 1999). This consultation was conducted on the basis of a Green Paper exposing different design options for an EU Emissions Trading System (EU ETS). To bring together environmentalists, experts and industry representatives to discuss on carbon trading in the EU, the European Climate Change Programme (ECCP) was also created. Setting up the ECCP was fundamental to legitimate the creation of an EU carbon trading scheme, as initial opposition from environmentalists, represented in the federation Climate Action Network – Europe (CAN-E), and industry lobbies, namely the Federation of German Industries (BDI), gradually evolved into reformism (Braun, 2009).

Consultations based on the Green Paper exposed divisions between polluters and environmentalists (EC, 2000, 2001). On one side, industrial lobbies, represented namely by UNICE and Eurelectric, mostly supported a voluntary EU ETS, with free allocation of allowances and no restrictions on the use of international offset credits. On the other, environmentalists, represented in CAN-E, supported a mandatory ETS, with full auctioning of allowances and severe restrictions on the use of international offset credits. Even though the European Parliament's position on ETS design was largely convergent with CAN-E's, the results of the consultation were heavily skewed in favour of industrial lobbies' positions (Markussen and Svendsen, 2005; Vlachou and Konstantinidis, 2010).

Within Member States, initial reactions to the proposed EU ETS indicated a lack of consensus. While the UK, Denmark, The Netherlands, Sweden and Ireland supported carbon trading, other Member States were more skeptical. Germany, the largest emitter, sided with its industrial lobby to defend voluntary agreements and oppose a mandatory EU ETS in the pilot phase, a demand that was also supported by the UK. To tackle this initial opposition, the EC conceded to a decentralized EU ETS, allowing states to define rules for market participation (Skjærseth and Wettestad, 2009). The stage was now set for the EU ETS implementation.

**Implementing the EU ETS**

Following consultations, the EU ETS directive was approved in 2003 (EC, 2003). The directive established a mandatory carbon trading system, starting in 2005, which would incorporate $CO_2$ emissions from major stationary emitters. Carbon allowances, called European Union Allowances (EUAs), would be distributed for free to polluters according to historical emissions, following rules set out by each Member State in National Allocation Plans (NAPs). Offset credits from the CDM and JI were excluded until 2007.

To monitor EU ETS emissions, a Community Independent Transaction Log (CITL, later renamed to EUTL) was created to connect national registries (EC, 2004b). Member States assessed compliance according to self-reported emissions, verified by certified independent entities, following EU monitoring and reporting guidelines (EC, 2004a). Non-compliance was penalized with a 40€ fine for each ton of excess emissions, which would rise to 100€ after 2008 (EC, 2003).

Following the pilot phase, from 2005 to 2007, the second phase of the EU ETS lasted from 2008 to 2012, corresponding to the KP first commitment period. In Phase II, the number of participating countries increased to 30, with Norway, Iceland and Liechtenstein joining. The scope of the carbon market was also increased, by having non-$CO_2$ gases from the chemical sector included. Major reforms, however, were excluded, as the EC (2006) recommended.

An exception to this rule was the inclusion of air transport emissions in the EU ETS. Following a 2008 Directive, emissions from international flights from, to or within the European Economic Area (EEA) were included in 2012 (EC, 2008b). The 2012 cap was set at 97 percent of the average emissions in the 2004-2006 period, while the post-2013 cap was set at 95 percent of the same baseline. Airlines were allocated a new type of allowance, the EU Aviation Allowance (EUAA), which cannot be used for compliance by other polluters. Free allocation of EUAAs covers 85 percent of emissions in 2012, and 82 percent after 2013, with the remaining 3 percent being given to new entrants or fast-growing airlines. To cover the remaining emissions, airlines can buy EUAs or EUAAs. Additionally, airlines can cover up to 15 percent of their emissions with international offset credits.

Accounting for the possibility of using allowances and credits, an early impact assessment carried out for the EC estimated that emissions reductions from airlines resulting from their inclusion in the EU ETS would amount to 2.8 percent by 2020, which is equivalent to one year's growth in emissions in a “business as usual” scenario (EC, 2006). Despite the limited impact, the new directive was opposed by the International Air Transport Association (IATA), representing airlines, as well as by many foreign governments (CEO, 2008). To appease opposition, the EU suspended participation in the EU ETS for flights to and from non-EEA countries until 2016. By then, following a 2013 decision by the International Civil Aviation Organization (ICAO), an agreement is expected to be reached to develop a global market-based mechanism to regulate aviation emissions, to be applied in 2020 (EC, 2013).

Environmental targets for Phase III, from 2013 to 2020, were first presented in the "20 20 by 2020" strategy, which postulated a reduction of at least 20 percent in GHG emissions and an increase to 20 percent of the share of renewable energies in EU energy consumption by 2020 (EC, 2008a). The strategy predicted the possibility of increasing the emissions reductions to 30 percent by 2020 if there was an international agreement that committed industrialized countries to comparable targets, which became the EU official negotiating stance in climate negotiations. It was also made clear that the EU was eager to engage other OECD countries in creating an international carbon market.

The 2020 strategy also presented the guidelines for a EU ETS reform in Phase III (Skjærseth and Wettestad, 2010). The reforms included incorporating more non-$CO_2$ GHGs ($N_2O$ and PFCs) and industrial emitters that were previously excluded (namely aluminium and ammonia producers), replacing NAPs with harmonized allocation rules and a single cap on CER use across the EU. The EU cap was set by having a yearly linear reduction of 1.74 percent in allocated EUAs, to achieve by 2020 a 21 percent reduction from 2005 levels. Auctioning was to replace grandfathering as the allocation method for the power sector, while industries were to receive free allowances according to a benchmark that reflects the best available control technologies and gradually converge to full auctioning until 2020.

Regarding the use of EUA auction revenues, the new rules specify that they are distributed to Member States according to historical emissions, with the recommendation that half of the revenues are used in mitigation and adaptation measures. Additionally, 2 percent of total auction revenue is redistributed to Member States with emissions in 2005 that are 20 percent below 1990 levels and 10 percent of auction revenue accruing to relatively richer Member States is redistributed to relatively poorer Member States. New sources can participate in the auctions of the New Entrants Reserve (NER), through which 300 million EUAs will be sold to finance carbon capture and storage and renewable energy projects.

Two important changes were made by the European Council to the auctioning rule, though, to tackle industry and Member States opposition (European Council, 2008; Skodvin *et al*., 2010; Vlachou, 2014). First, power plants that provide more than 30 percent of national electricity in Eastern European countries will be given 70 percent of EUAs for free in 2013 and will only face full auctioning in 2020. Second, the benchmarking rules were changed to increase the quantity of freely allocated allowances. According to the new rules, the benchmark for each sector or sub-sector is calculated by multiplying historical production levels with self-reported average emissions of the 10 percent most efficient installations, using the pre-crisis 2005-2008 period as reference. Industries that are deemed to be energy-intensive and exposed to international competition will receive all of their benchmark for free, while the remaining industries will receive 80 percent of the benchmark for free, declining to 20 percent in 2020 and zero in 2027 (EC, 2009a, 2011a).

EU ETS reforms further affected how offset credits can be used in Phase III. Concerns with the geographical distribution of the CDM and how it led to subsidizing competitors in industrializing countries supported a decision to restrict CER use to those that were generated by projects located in Least Developed Countries (LDCs). Additional credits generated from other countries can be used only if these countries engage in international or bilateral agreements with EU. For developing countries that aren't LDCs, the EU aims to replace the CDM with a sectoral crediting approach, through which whole sectors can generate offset credits by achieving a pre-determined emissions threshold (EC, 2009b).

The limit on the use of offset credits was also revised. While the initial Phase II limit to CER and ERU use was set at 1.4 billion credits, and 1.05 billion credits were surrendered until 2012, the new rules specify a 1.6 billion limit that applies from 2008 to 2020 (Betz, 2015). With the new limit, half of Phase III emissions reductions from 2005 levels can be met by surrendering offset credits. On top of that, the new rules allow banking credits from Phase II to Phase III, up to a limit of 3 percent of 2005 emissions (EC, 2008c).

In 2011, two additional reforms were approved. First, CERs generated by projects that destroy industrial gases were banned from use in the EU ETS from May 2013. The decision targets both $N_2O$ from adipic acid production and HFC-23 from HCFC-22 production and was justified by the EC for the lack of environmental integrity of projects that destroy these gases and the unequal geographical distribution of the CDM (EC, 2011d). Second, a "backloading" provision was approved to postpone the auctioning of 900 million EUAs until the end of Phase III. This ad-hoc decision was taken to address the oversupply of EUAs, estimated at almost two billion EUAs, and will be implemented gradually, with auction volumes being reduced by 400 million allowances in 2014, 300 million in 2015 and 200 million in 2016 (EC, 2011b).

The "backloading" provision will not reduce the number of EUAs in circulation but merely reshuffle their temporal distribution. The EC actually expects the oversupply of EUAs to continue to grow and reach 2.6 billion at the end of Phase III, and is currently proposing that a Market Stability Reserve (MSR) is implemented after 2021. The MSR would allow the adjustment of auction volumes to the balance between supply and demand, by reducing the number of auctioned allowances if the excess supply is higher than 833 million EUAs or increasing the number of auctioned allowances if there is an excess demand over 400 million EUAs (EC, 2014b).

These reforms fell very short of what environmentalist NGOs advocated for: full auctioning, use of auctioning revenue in climate change related investments, stringent quantitative and qualitative limits on the use of international offset credits and a 30 percent reduction target for 2020 (CAN-Europe *et al*., 2007). Following the precedent given in the 2005 Green Paper, the 2007 consultations within the ECCP on the EU ETS review process were again skewed in favour of industrial lobbies' interests, which is unsurprising considering that the european carbon market was created to placate opposition from major polluters to environmental regulations.

On top of the changes made in the EU ETS in Phase III, EU institutions have been discussing the future of carbon trading after 2020. The 2030 framework on climate and energy indicates that the emissions target for Phase IV of the EU ETS, from 2020 to

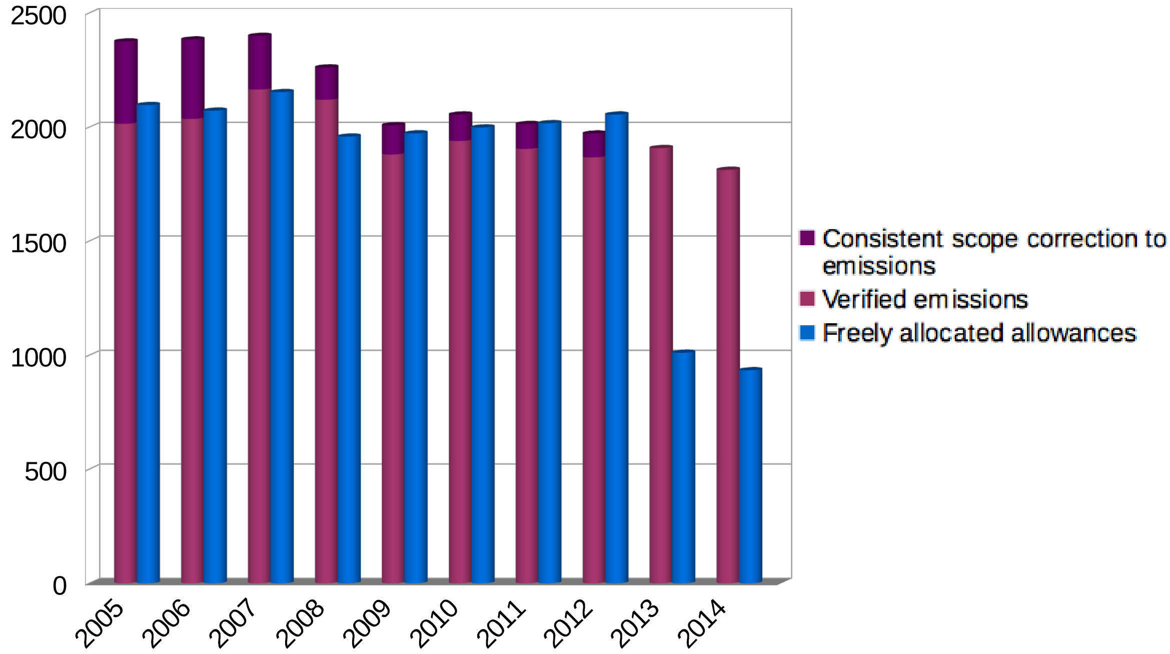
2030, will be a 40 percent emissions reduction, from 1990 levels, which will be achieved through increased investment in renewable energy and energy efficiency and without the use of international offset credits (EC, 2014a). This framework, in turn, follows a roadmap for a low carbon economy by 2050, which indicates an 80 to 95 percent emissions reduction until 2050, to be achieved through the EU ETS (EC, 2011c).

The EU ETS reforms are intended to address some of the problems that have plagued it during Phase I and Phase II. To ascertain its adequacy and effectiveness, we now turn to an empirical review of the EU ETS performance.

### The EU ETS performance

The effectiveness of the EU ETS can be evaluated firstly by the evolution of covered emissions. The picture that emerges from this evolution is not very clear-cut, however, since exogenous factors can lead to substantial variations in emissions.

***Figure 9: EU ETS verified emissions and freely allocated EUAs, 2005-2014 (Mt $CO_2$eq)***




*Source: EEA (2015)*

During Phase I, emissions from covered sources actually increased by about 7.5 percent but during Phase II, emissions decreased by about 13.8 percent, following a sharp decline in 2008-2009. Using the consistent scope correction calculated by the European Environment Agency (EEA) to retrospectively add the emissions from sectors and gases that were included only in Phase III, emissions reductions in the EU ETS from 2005 to 2014 reach a total of 23.7 percent.

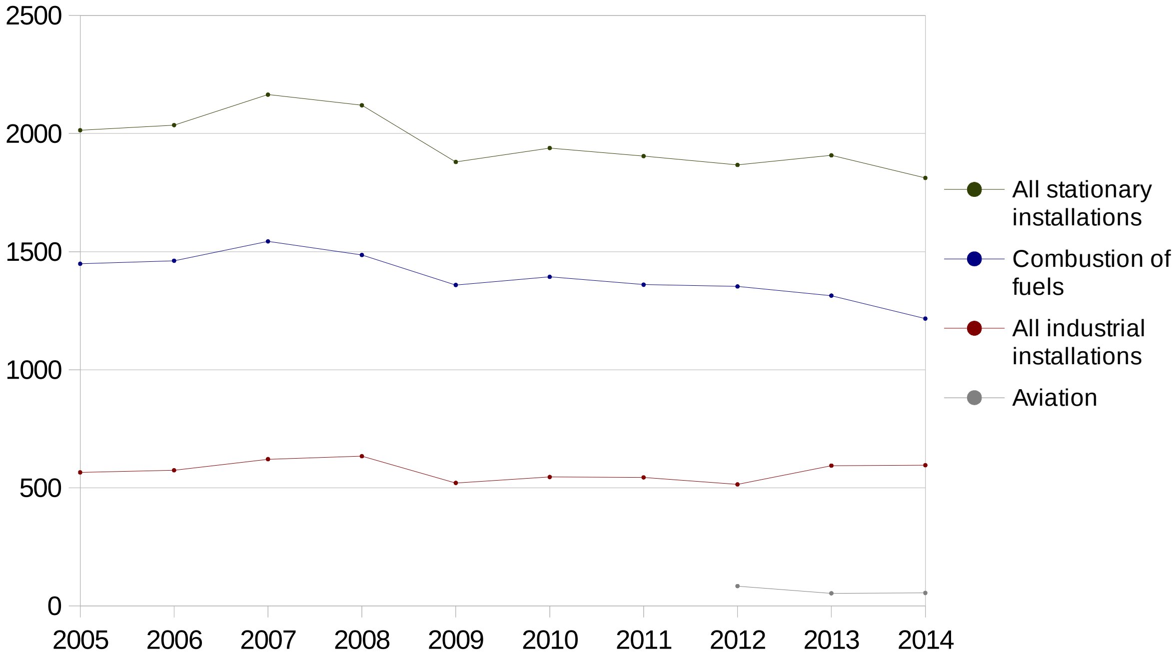
The observed downward trend in the EU ETS emissions is consistent with the 19 percent decrease in GHG emissions in the EU registered in 2013, relative to 1990 levels, which is very close to the 20 percent reduction target set for 2020 (EEA, 2014). But this is not enough to determine the environmental effectiveness of the EU ETS, since emissions reductions can be attributed to factors unrelated to the ETS. Among factors that explain emissions reductions in the early 1990s, we can point out the replacement of coal plants for gas plants in the UK and the deindustrialization of Eastern Europe (including the former Eastern Germany), as well as the relocation of polluting industries to non-EU countries. More recent factors include the approval of renewable energy and energy efficiency policies and the economic recession of 2008-2012.

Given the difficulties in separating the different factors that can lead to emissions reductions, early estimates on the EU ETS effectiveness were very disparate, with Ellerman and Buchner (2008) estimating a 270 to 420 million tonnes of $CO_2$ reduction in 2005-2006 and Anderson and di Maria (2011) estimating a reduction of 174 million tonnes during Phase I, compared to a “business as usual” baseline. Ex-post studies using empirical data instead of baselines delivered more pessimistic conclusions, indicating that the reductions can be attributed to exogenous factors. Using data from the energy sector in the 2005-2012 period, Nicolas *et al*. (2014) estimate that, even though the carbon price contributed to emissions reductions, the main factor was the increase in renewable energy production, which in turn can be explained mostly by national policies unrelated to the EU ETS like feed-in tariffs or green certificates. Gloaguen and Alberola (2013) went further and estimated that, during the 2005-2011 period, emissions reductions in EU ETS covered sectors could be explained almost entirely by a combination of increased renewable energy production, the economic downturn, improved energy efficiency and fuel switching (from coal to gas), all of which are factors dependent on EU policies and economic variables unrelated to the carbon market evolution. Similarly, Bel and Joseph (2015) estimated that, during the 2005-2012 period, only 11.5 to 13.8 percent of the reductions in EU ETS sources can be attributed to the carbon market.

Comparing the EU ETS verified emissions with freely allocated EUAs, in turn, we can confirm that overallocation was constant over the first two phases, with the exception of the years 2007 and 2008. Data from Phase III shows that the new rules resulted in a major decrease in the number of freely allocated EUAs, of about 50 percent between 2012 and 2013, even after the scope of the EU ETS was broadened to include new gases and sectors. In 2014 only about half of emissions were covered by freely allocated EUAs, implying that the other half had to be covered with banked or auctioned EUAs.

Disaggregating the EU ETS verified emissions by sector, we can infer that the total emissions from stationary installations presents a strong correlation (r = 94%) with emissions from fuel combustion. This indicates that the yearly variation of EU ETS emissions is determined mostly by the yearly variation of fossil fuel use in the energy sector, since the correlation with industrial emissions is much weaker (r = 64%).

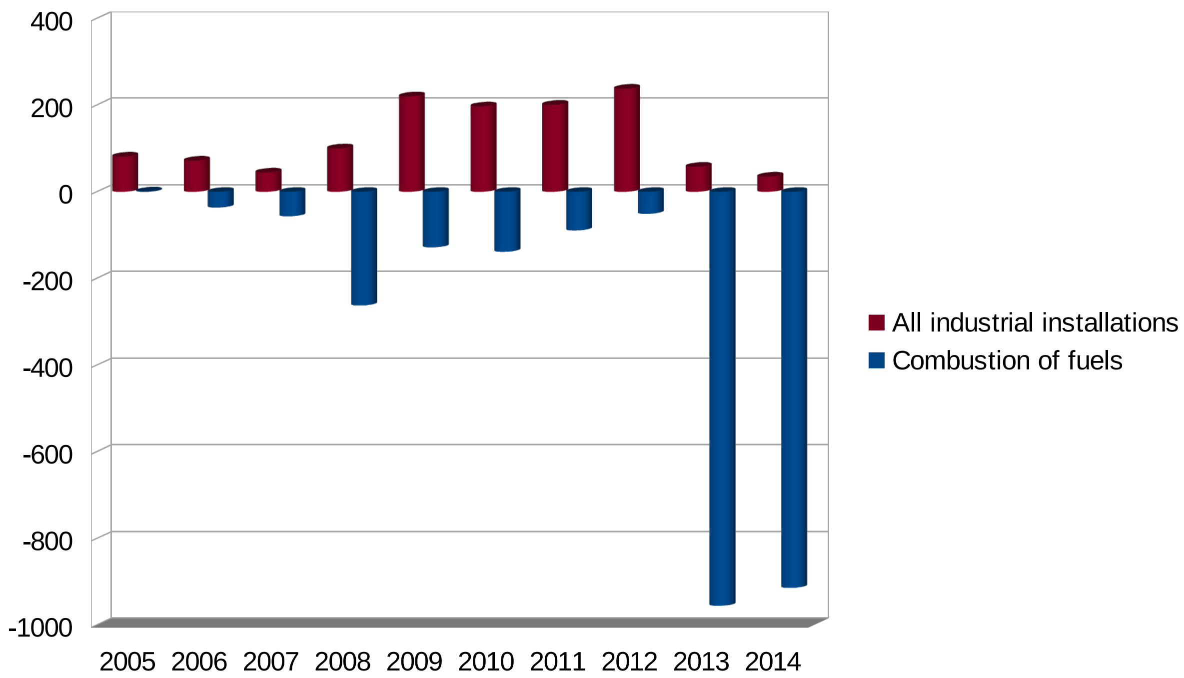
***Figure 10: EU ETS verified emissions by sector, 2005-2014 (Mt $CO_2$eq)***




*Source: EEA (2015)*

This trend reflects an important division within the EU ETS. Even though energy suppliers and manufacturing industries were integrated in the same carbon market, the two sectors have faced different allocation rules. In particular, the energy sector has faced constant under-allocation, which implies that the overallocation present in the EU ETS results entirely from a too generous allocation to manufacturing industries. This was not incidental but rather part of the EU ETS design, reflecting the general conviction that, while energy companies could easily pass through the cost of bought allowances to consumers, industries were prevented from doing the same due to international competition. In other words, the cost of the EU ETS has been borne almost entirely by consumers through higher energy prices.

***Figure 11: Difference between freely allocated allowances and verified emissions by sector, 2005-2014 (Mt $CO_2$eq)***

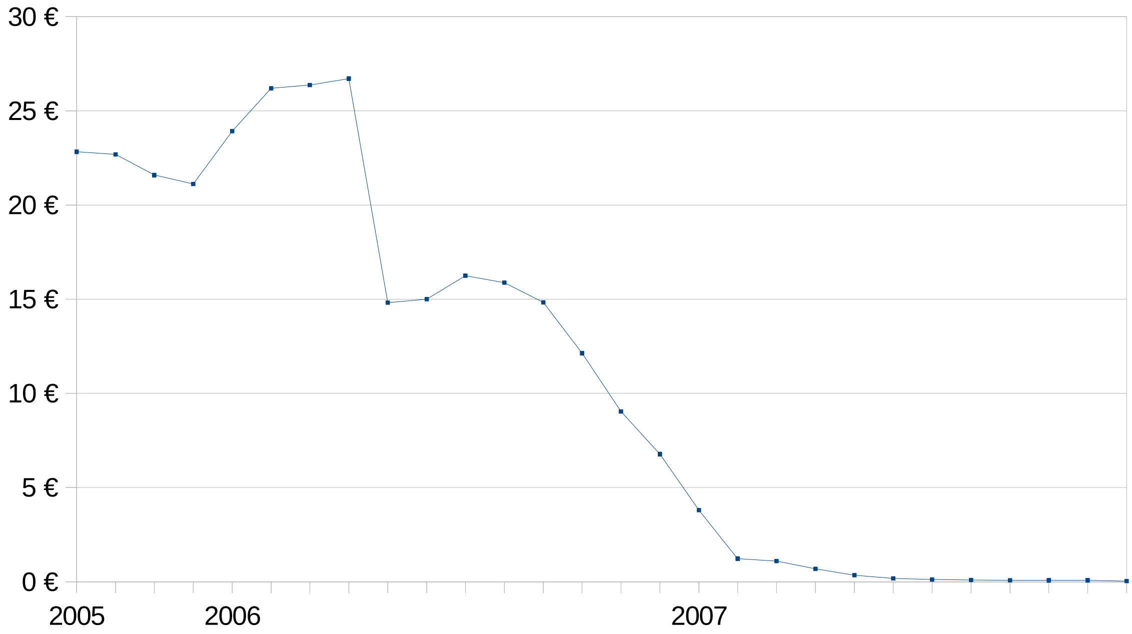



*Source: EEA (2015)*

Comparing free allocations with emissions also shows that the rift between the energy sector and manufacturing industries has broadened considerably in Phase III. While in Phase I and II, the energy sector was under-allocated by about 7 percent and the manufacturing industries were over-allocated by about 26 percent, in the first two years of Phase III the energy sector underallocation reached 74 percent and the manufacturing industries overallocation dropped to 8 percent. This reflects the effect of new allocation rules, which force most energy companies to buy all of the necessary allowances in auctions but exempt industries from the same requirement.

Since the EU ETS has been swamped with excess allowances, both EUA and CER prices have been low and declining. This trend can be illustrated with the evolution of spot markets.

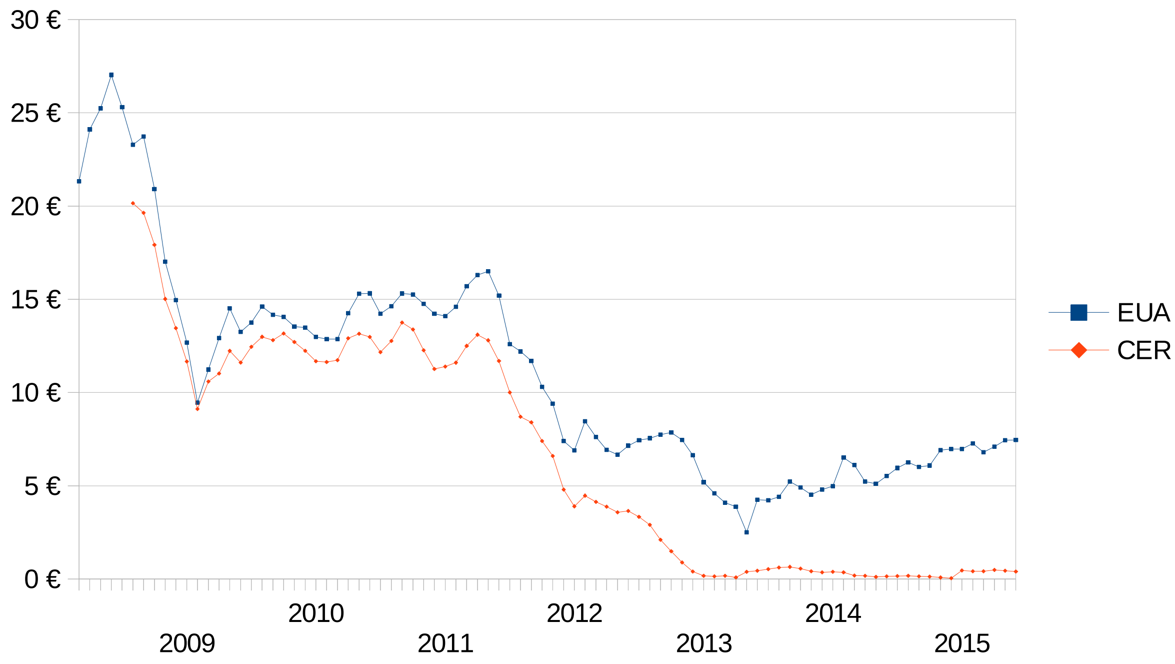
***Figure 12: Monthly average EUA spot price, September 2005 – December 2007***




*Source: Caisse des Dépôts (2006, 2007, 2008).*

EUA spot prices in Phase I were relatively high at first, reaching a peak high of 27€ in April 2006. After the data on EU ETS emissions and allocations from 2005 was released, however, the EUA price dropped to 15€ in May 2006 and exhibited a negative trend from thereafter. By early 2007, the EUA price was already near zero. This reflected both the oversupply of allowances and the impossibility of banking EUAs for use in Phase II.

***Figure 13: Monthly average EUA and CER spot price, February 2008 – June 2015***




*Source: Caisse des Dépôts (2008, 2009, 2010, 2012, 2013, 2014, 2015a, 2015b)*

Since banking from Phase II to Phase III is allowed, the EUA spot price on the two phases can be analysed together. The trend is very similar to what was observed in Phase I, with prices being above 20€ at first but then declining steadily. After reaching a peak 27€ in June 2008, the EUA price started to decrease and by September 2008, when the global financial crisis hit the EU markets, the price plunged, dropping below €10 in February 2009. After a period of recovery, during which the EUA monthly spot price averaged at about €15, there was again a price slump in June 2011, following the recession caused by the EU debt crisis. The EUA price reached a record low of 2.5€ in May 2013 and has slowly recovered thereafter, reaching 7€ in April 2015.[12]

12 Even though we concentrated on spot prices, EUA prices on futures markets have followed a similar pattern. Zhu *et al.* (2015), for instance, use econometric models to find structural breaking points for

During most of Phase II, CER prices have followed the EUA price trajectory, being traded at a discount that results from the limits on CER use and the uncertainty regarding CER delivery by project developers. Accordingly, until mid-2012, EUA and CER prices were highly correlated (r=97%), and the spread averaged 2.3€. In July 2012, however, the EUA-CER price spread reached a record high of €4.11 and then grew to more than €6 by the end of Phase II. By the end of Phase II the CER price had dropped to zero and in Phase III it has stayed consistently below 1€.

This evolution reflects the EU ETS reforms that limit CER use in Phase III. Since no bilateral CER purchase agreements were signed between the EU and third countries, only CERs from projects located in LDCs are now accepted. Furthermore, EU ETS installations have already surrendered 1.45 billion CERs and ERUs to cover their emissions, which is 90 percent of the total 1.6 billion limit that applies from 2008 to 2020 (Kossoy *et al*., 2015).

The price dynamics of both EUAs and CERs is illustrative of the incapacity of the EU ETS to deliver a stable and non-negligible carbon price that could facilitate a transition towards a low-carbon development path. This is due both to the difficulty of enacting reforms and to the insufficiency of the reforms enacted.

The EU ETS Phase I was conceived as a pilot phase, during which industries, EU authorities and Member States would gain knowledge and experience on emissions trading. During Phase I, up to 5 percent of allowances could be auctioned by Member States during Phase I, but this provision was hardly used and nearly all allowances were allocated to polluters for free. Allocations followed emissions historical data contained in NAPs, which exhibited substantial variations in the methodologies used, despite the EC's supervision (Betz *et al*., 2004). Furthermore, even after the EC revised original NAPs proposals to reduce Phase I allocations by 4.5 percent (Skjærseth and Wettestad, 2009), the final allocation was still too generous, with EUAs exceeding emissions by 3 percent.

No major changes were made in Phase II, from 2008 to 2012, even though serious problems with EUA allocation were already evident. The limit for voluntary auctioning was increased to 10 percent, but still auctioning was negligible. Allocations

December 2012 EUA futures contracts in November 2008, June 2011 and November 2011.

again followed rules set in NAPs and again, even though EC's revision reduced allocations by 10.5 percent (Skjærseth and Wettestad, 2009), allocated EUAs still exceeded emissions by about 3 percent.

By creating an asset with economic value and then giving it away for free, the EU delivered a massive windfall to polluters. In particular, the power sector has been allowed to earn windfall profits from increasing energy prices according to the opportunity cost of grandfathered EUAs. This was already evident during Phase I of the EU ETS, with Sijm *et al*. (2006) estimating a windfall profit for the power sector in The Netherlands of €300-600 million per year. By the end of Phase I, Keppler and Cruciani (2010) estimated windfall profits for the power sector from cost pass-through at €13 billion. This trend continued during Phase II, with Fell *et al*. (2013) estimating a cost pass-through rate for electricity suppliers of at least 100 percent in Germany, France, The Netherlands, the Nord Pool Market and Spain. Windfall profits for the power sector in Phase II were estimated by Point Carbon (2008) at €23-72 billion.

The practice of passing through the opportunity costs of freely allocated EUAs also extends to energy-intensive industries, despite the fears of carbon leakage, i.e., the relocation of industrial production to countries without binding emissions constraints. Econometric estimates from de Bruyn *et al*. (2010) suggest that industries from the refineries, iron and steel and plastic sectors managed to pass on the full opportunity cost of EUAs in most of their products between 2005 and 2008. This suggests that even industries exposed to carbon leakage risk were able to reap a considerable windfall profit from the EU ETS.

Similar conclusions can be reached regarding the inclusion of airlines in the EU ETS. Nelissen and Faber (2012) estimated windfall profits accruing to aviation companies in 2012 at €679-1,358 million, of which €436-872 million are due to passing through the opportunity costs of free allowances and €243-486 million result from tariff increases in intercontinental flights prior to the suspension of their participation in the EU ETS.

These empirical estimates expose a major weakness of the allocation method chosen in Phase I and II of the EU ETS, which turned it into a “polluter gets paid” system.

Grandfathering allowances, instead of auctioning them, turns allowance allocation into rent distribution. Accordingly, industries have competed among themselves for a larger share of the rent created, equivalent to the value of freely allocated allowances (Markussen and Svendsen, 2005). Since rent-seeking is a wasteful activity that adds nothing to social product (Krueger, 1974), the cost of this waste should be considered when evaluating the cost-efficiency of the EU ETS.

Considering that windfall profits are proportional to historical emissions, grandfathering also results in a redistribution of income from labour-intensive industries to energy-intensive industries, which can lower employment and increase energy use. Moreover, by subsidizing industries according to their historical pollution levels, the EU ETS has provided an incentive extend the lifetime of old dirty installations or even build new ones. During Phase I, econometric estimates already pointed out that the EU ETS made it more profitable to build new coal plants (Neuhoff *et al*., 2006). Accordingly, empirical evidence on Germany suggests that grandfathering during Phase I was a major factor for a massive increase in coal plant investments (Pahle *et al*., 2011). In other words, even though carbon trading is supposed to incentivize a transition to a low-carbon development path, in reality the EU ETS made european energy consumers face tariff increases to support a subsidy to the dirtiest industries.

The new allocation rules applied in Phase III are intended to address the problems associated with grandfathering by replacing it with auctioning as the allocation method. Since the change is gradual, however, only by 2027 full auctioning is expected to take place. Until then, EUAs will continue to be distributed for free, but the basis for allocation is a benchmark representing historical emissions of the least polluting industries, instead of historical emissions.

Allocating allowances according to benchmarks provides in theory an incentive to reduce emissions up to the level of the less polluting installations in each sector or sub-sector. In reality, though, calculating benchmarks is as much a scientific as a political process and, therefore, it is amenable to being influenced by industry lobbying. Environmentalists represented by CAN-E, for instance, noted that, following EC consultations with industry lobbies, the cement benchmark excluded from calculations the production of cement substitutes and the steel benchmark was increased by 25 percent. Since the baseline for production levels was set at the 2005-2008 median, i.e. at pre-crisis levels, environmentalists estimated that these industries will not have to reduce emissions or buy EUAs during Phase III (CAN-Europe, 2010).

Further problems lie in the free allocation of allowances to industries considered to be at a significant risk of carbon leakage. According to EU ETS rules, these sectors are defined as those with carbon intensity greater than 5 percent and trade intensity greater than 10 percent or either carbon or trade intensity greater than 30 percent. The carbon intensity is calculated as the ratio between the costs of allowance auctioning, using €30 as a reference price, and the gross value added of a sector, while the trade intensity is calculated as the ratio between the total value of international trade with third countries (exports plus imports) and the total market size of the EU (annual turnover plus imports from third countries) (EC, 2009a). But many industries that fall within this category can easily switch to inputs with lower carbon intensity or pass through the cost of allowances bought in international markets, namely those that benefit from country specific factors. Taking this into account, Martin et al. (2014) suggested a new carbon leakage measure, after interviewing managers in regulated firms, that extends auctioning to industries with high trade intensity and low carbon intensity and redefines trade intensity to include only international trade with developing countries. The authors estimate that even with a low allowance price of €5 this decision could generate an additional revenue of €0.5 to €3.5 billion per year.

Industries deemed to be at risk of carbon leakage are also being awarded subsidies in the form of state aid, to cover the costs of the expected increase in electricity prices. A total of thirteen sectors and sub-sectors, including aluminium, steel, paper and chemicals, are targeted by state aid, which can amount to 85 percent of eligible costs from 2013, 80 percent from 2016 and 75 percent from 2019. Additionally, electricity providers can apply to state aid to cover up to 15 percent of investments in new coal or gas plants that are “CCS-ready”, i.e., that can be retrofitted to install carbon capture and storage equipment (EC, 2012).

To sum up, the EU ETS has been characterized by a chronic excess supply and allocation rules that result in massive giveaways to polluters. Even though the reforms applied from 2012 were intended to address these problems, they fall short of what was required. Without further changes in the horizon, at the end of Phase III, in 2020, the EU ETS will still not provide a binding cap for emissions and carbon prices will likely remain low.

## 4.4 Conclusions

Following the Kyoto Protocol ratification, a complex network of carbon markets emerged, of which the CDM and the EU ETS are the most relevant. International climate negotiations have since then put several new market mechanisms on the table but failed to deliver binding post-2012 emission reductions commitments. While this evolution closely follows the interests of industry lobbies supporting carbon trading, the environmental performance of market mechanisms is, at best, dubious and it is unclear how it could improve in the future.

The experience with CDM exposes the difficulties associated with attaching climate mitigation investment in developing countries to a financial instrument that allows industrialized countries to increase their emission levels. Sustainable development benefits from CDM investments are negligible, since the logic of the market in carbon offsets is based on minimizing regulated industries costs and not on maximizing social and environmental benefits from emission reductions. Furthermore, since project additionality is impossible to prove, the CDM inevitably leads to offset credits being generated from emission reductions that would have happened anyway, thus resulting in inefficient investments and increased global emissions. None of these problems are solvable in a market where no participant has an incentive to care about the quality of the product being traded.

The experience with EU ETS, on the other hand, shows what can and does go wrong with trading in carbon emissions allowances. Considering that the premise of carbon trading is that industries can be pushed into transitioning to a low-carbon development path through the incentive that is provided by the carbon price set in the market, the failure of the EU ETS to deliver a non-negligible and stable carbon price is a failure of the system on its own terms.

This failure, to be sure, can be traced to a combination of overallocation of allowances and use of international offset credits that took the “cap” out of “cap and trade”. The allowance overallocation, as well as the grandfathering that allowed polluters to reap windfall profits, is an example of design problems that could be addressed by EU ETS reforms, as opposed to inherent and consequently unsolvable problems. But the distinction between design issues and inherent problems becomes less important as we step outside textbook economics, as the inability of EU authorities to approve reforms that could effectively address identified problems shows.

Considering the dismal performance of existing carbon markets, it seems that adopting carbon trading as an alternative to direct regulation and taxation was a costly mistake. This begs the question of whether these carbon markets should be reformed or dropped from the policy-mix, or, to put it another way, whether carbon trading could work in theory. This question will be addressed in the following chapters, which deal with the foundations and consequences of carbon trading.

# 5 Slicing the pie in the sky: The science and politics of the carbon commensuration black box

"Potentially at least, far from restricting the space of contestation, further scientific calculations may serve to open it up." (Barry, 2002: 274)

Following the Kyoto Protocol, multiple carbon markets were created to allow polluters to trade in emissions rights. In these markets, allowances from "cap and trade" programs and credits from offset projects can be traded not only by polluters but also by speculators interested in making a profit from price fluctuations and NGOs engaging in market environmentalism by retiring allowances or credits bought. The question that is rarely asked, while numbers flow from one computer to another, is what exactly is being traded. This chapter engages with this important question by shedding some light on how carbon market numbers are produced and how they relate to carbon emissions.

Carbon trading is a form of governing and managing carbon as a financial commodity. Turning carbon into a commodity, in turn, requires abstracting carbon from its social, historical and political context, which is achieved by commensurating carbon across space and time. Carbon commensuration, as the social process that creates equivalences between carbon emissions by measuring them according to conventions of quantification, is a "black box" that transforms the complex phenomenon of anthropogenic climate change into a simple, quantifiable problem of too much (abstract) carbon emissions. This transformation is not without its costs, though, as everything that is deemed to be irrelevant is obfuscated, including quantification uncertainties.

Because the numbers produced by the carbon commensuration black box are fundamental to carbon trading, we propose to open the black box and explore the accuracy and accountability of the quantification of carbon emissions, as well as expose the political assumptions that lie at its core. While the first section presents an analysis of carbon commensuration as a mechanism for producing objective knowledge, the second section shows how the numbers produced by the commensuration process are misleading, due to scientific uncertainties and social indeterminacies. The final section concludes with a reflection on the role of numbers in climate policy.

## 5.1 Carbon commensuration as a black box

Commensuration is the process of using a common metric to measure different entities (namely persons, countries or institutions), in order to compare them according to intervals or ratios.[13] By transforming qualities into quantities and delivering numbers that can be easily compared, the quantification inherent to commensuration produces (apparently) objective, rational and impersonal knowledge, which can then be used to legitimize and simplify decision-making (Porter, 1996). Objectifying knowledge through numbers, however, is costly because "it implies the existence of a heavy socio-technical infrastructure that can assure its production" but also "in terms of reducing the normative complexity of phenomena" (Centemeri, 2008: 119). Furthermore, by creating "relationships between virtually anything", commensuration "simultaneously overcomes distance (by creating ties between things where none before had existed) and imposes distance (by expressing value in such abstract, remote ways)" (Espeland and Stevens, 1998: 324).

Carbon commensuration follows from the quantification of emissions using methodologies grounded in scientific knowledge, but goes beyond this measurement to create equivalences between emissions attached to different realities (MacKenzie, 2009). This process is central to carbon trading. Setting up a carbon market involves firstly creating the carbon commodity, which is specified both by its boundaries of tradability and by its connection with carbon emissions. Creating the carbon commodity, in turn, necessarily involves commensurating carbon emissions across space and time, a process that can be described as a "black box", to borrow Latour and Woolgar's (1986) term, that abstracts carbon emissions from their context.

Levin and Espeland (2002) described how emissions commensuration, in the

13 Following Chang's (1997) suggestion, when two entitites can be ordinally ranked but not cardinally compared, we can conclude that the two entities are comparable but not commensurable.

context of emissions trading, follows three dimensions: technical, value and cognitive. We can illustrate these three dimensions using carbon markets.

Technical commensuration consists of the use of knowledge and technologies to create accounting methods that allow for the monitoring of emissions. While scientifically and technically complex, this operation is essential to evaluate compliance with the mandated emissions limits, both at the aggregate and at the individual level. It is also essential to create uniform commodities that can be traded in carbon markets.

Value commensuration consists of attaching a single monetary value to all carbon emissions. This is achieved by the regular functioning of carbon markets, which, working within the limits set by government regulation, set a price for allowances and credits.

Cognitive commensuration consists of the amalgamation of polluters within a category of "abstract polluter", through the decontextualization of emissions made possible by technical and value commensuration. In other words, carbon markets do not allow us to see the qualitative differences between polluters, as the quantitative reductionism that is at its base only allows differences to be expressed in quantities.

Following this framework, the three dimensions of carbon commensuration, which are mutually reinforcing, create and reify new objects (abstract carbon and the attached allowances and credits) and new actors (abstract polluters), while rendering invisible or irrelevant information regarding how, where and when emissions occurred. Carbon trading therefore operates on a reality that was reconstructed by the commensuration processes, which frames climate change as a problem that flows from an excess of abstract carbon emissions. These processes, in turn, are a fundamental part of both commodification practices and of the pollution as externality discourse, produced by what Lohmann called the "endless algebra of climate markets":

> Commodity solutions always reinterpret and transform the social and environmental challenges that they confront. Their goals are never exogenous but are incessantly reshaped by the very process of addressing them. Hence the ''internalization of environmental externalities'' associated with market environmentalism is better conceived not as a (successful or failed) attempt at ''environmental problem-solving'' but rather as a continuous changing of the subject. In order to be ''internalized,'' environmental harms of any complexity must be simplified, reformatted, made abstract, quantifiable, and transferrable in a process that obscures many of their characteristics while introducing fresh problems. (Lohmann, 2011: 112)

Carbon commensuration is also fundamental for to make carbon "manageable". Following the old management adage "you cannot manage what you cannot measure", many corporations, under the guise of Corporate Social Responsibility, are measuring their carbon emissions, in order to plan and enact actions to reduce them. The resulting carbon accounting, which can be seen as a tool to generate knowledge, frame policy debates and/or governing people (Lovell and MacKenzie, 2011), is able to generate sustainability reports for corporations only at the expense of leaving out what cannot be translated as physical information (Lippert, 2012). Likewise, "carbon footprints" value individual human actions using abstract carbon as the commensurator, resulting in a measure of individual responsibility that ignores social, political and historical constraints (Dalsgaard, 2013).

Carbon accounting reproduces at the corporate or individual level the same carbon commmensuration process that frames climate change as a problem that can be "fixed" by carbon trading. While calculative practices are arguably indispensable in any climate policy, a market-based policy is highly susceptible to its shortcomings, since it demands a high level of accuracy. In other words, unlike, say, technological standards, a carbon market cannot be implemented on the basis of an interval for the emissions of each regulated polluter but rather requires a specific number. Whether or not this number can be accurately produced by the commensurative "black box" is the question that the next section addresses.

## 5.2 Opening the black box: Uncertainties, assumptions and misleading numbers

The Kyoto Protocol (KP) created binding carbon emissions reduction targets for industrialized countries, according to which each party is attributed carbon allowances (UN, 1998). Following Article 5, compliance with emissions targets is evaluated considering carbon emissions from sources and carbon removals from sinks, which are measured according to guidelines prepared by the Intergovernmental Panel on Climate Change (IPCC) that reflect the best available scientific knowledge (IPCC, 1996, 2000).

The targets are calculated as the carbon dioxide equivalent ($CO_2$eq) of six greenhouse gases (GHGs): carbon dioxide ($CO_2$), methane ($CH_4$), nitrous oxide ($N_2O$), hydrofluorocarbons (HFCs), perfluorocarbons (PFCs) and sulphur hexafluoride ($SF_6$). The $CO_2$eq of each gas, in turn, is calculated by multiplying emissions with the respective Global Warming Potential (GWP). Article 8 of the protocol further stipulates that national inventories (emissions minus removals) are verified by expert review teams. Since these inventories reflect estimates and not real emissions (which are unknown), it is theoretically possible for a party to the protocol to be in legal compliance even if its emissions exceed the target.

The issue of scientific uncertainty in emissions measurement is particularly problematic considering that compliance with the KP can be achieved through carbon trading. If estimated emissions from a party are below its target, the party can sell its excess allowances to another party, even if real emissions are actually above the target. While the technical dimension of carbon commensuration presupposes that the scientific knowledge necessary to accurately measure and commensurate carbon emissions is available, significant uncertainties remain in carbon emissions measurement. Furthermore, the use of GWPs to commensurate GHGs obscures both the scientific uncertainties and the normative assumptions present in its calculation. Both of these two issues can result in profit opportunities for entities engaged in carbon trading created by the measurement methodologies, even while their emissions are not decreasing.

National carbon inventories are obtained using a mix of direct measurements and estimations based on activity data multiplied by emissions factors. Uncertainty estimates for industrialized countries' emissions inventories reviewed by Gupta *et al*. (2003) reach 20 percent for $CH_4$, 34 to 200 percent for $N_2O$, 15 to 50 percent for HFCs, 20 to 100 percent for PFCs and 25 to 50 percent for $SF_6$. Accordingly, Rypdal and Winiwarter (2001) estimate uncertainties in the inventories from five industrialized countries (UK, Netherlands, US, Norway and Austria) at 5 to 20 percent. More relevant for compliance evaluation, trend uncertainties, i.e., uncertainties in yearly changes in emissions, reach 4 to 5 percentage points for countries with available estimates (*ibid.*). Since this level of trend uncertainties is comparable with Kyoto's emission reduction targets for 2012, it follows that compliance cannot be evaluated unambiguously (Winiwarter, 2004).

The use of GWPs to convert each GHG into $CO_2$eq, the common metric used in Kyoto's carbon trading systems, further adds uncertainty to emissions measurements. The GWP was developed by the IPCC following a proposal made by the US government in 1989, which successfully argued for a “comprehensive approach” to climate policy and research that included and commensurated all emissions and sinks, in order to prioritize cost-efficiency (see Stewart and Wiener, 1992: 85-86). The first definition of the GWP in the scientific literature is found in Lashof and Ahuja (1990) who present it as an extension of the work done to commensurate ozone-depleting chemicals according to their Ozone Depletion Potential (ODP) and define the GWP as a metric that measures the relative radiative forcing of each GHG, relative to $CO_2$.

Estimates for the GWPs for several GHGs were then presented in the IPCC First Assessment Report, calculated as:

$$GWP = \frac{\int_0^n a_1 c_1 \, dt}{\int_0^n a_{CO_2} c_{CO_2} \, dt}$$

Where $a_x$ is the instantaneous radiative forcing due to a unit increase in the concentration of the gas $x$, $c_x$ is the concentration of the gas $x$ remaining at time $t$ and $n$ is the number of years considered for the calculation (Houghton *et al*., 1990: 58). Radiative forcing measures the difference between the solar radiation absorbed by the Earth atmosphere system and the radiation emitted back to space. This indicator is relevant because disturbances in the equilibrium between planetary radiation absorption and emission, caused by increased carbon concentrations, lead to climate change.

According to Article 5 of the KP, compliance is evaluated using GWP estimates from IPCC's (1996) Second Assessment Report, with n being 100 years. Since then, the GWP estimates have been periodically recalculated by the IPCC. Table 1 shows how, when recalculations were made, the GWPs for non-$CO_2$ GHGs rarely remained unaltered. Comparing the estimates used in the KP with the most recent estimates from the IPCC, we can see that the GWPs increased by 62 percent for $CH_4$ and 26 percent for HFC-23 while decreasing by 4 percent for $N_2O$ and 5 percent for $SF_6$. Since GWPs are used to calculate the $CO_2$eq of each GHG, updating the GWP estimates has a proportional effect on the profitability of carbon offsetting projects.

**Table 1 – Global Warming Potentials (100 year integration time horizon) for greenhouse gases included in the Kyoto Protocol**

| Greenhouse Gas | IPCC Assessment Report / Year | | | | |
|---|---|---|---|---|---|
| | I / 1990 | II / 1995 | III / 2001 | IV / 2007 | V / 2013 |
| $CH_4$ | 21 | 21 | 23 | 25 | 34 |
| $N_2O$ | 290 | 310 | 296 | 298 | 298 |
| PFC-14 | | 6,500 | 5,700 | 7,390 | 7,350 |
| PFC-116 | | 9,200 | 11,900 | 12,200 | |
| HFC-23 | | 11,700 | 12,000 | 14,800 | |
| HFC-134a | 1,200 | 1,300 | 1,300 | 1,430 | 1,550 |
| $SF_6$ | | 23,900 | 22,200 | 22,800 | |

*Source: Houghton, Jenkins, and Ephraums (1990: 60), Houghton et al. (1996: 121), Houghton et al. (2001: 386), Solomon et al. (2007: 211), Stocker et al. (2013: 114).*

GWP recalculations made according to new scientific evidence are predicted and even encouraged by the KP, but this has no practical effect. The formulation of Article 5 of the KP precludes the possibility of changing the GWPs used to evaluate compliance during the first commitment period (2008-2012). Since the Doha Amendment to the KP did not incorporate any change to Article 5, the 1995 estimates for GWPs will also be used in the third commitment period (2013-2020) (UNFCCC, 2012a). Precluding GWP estimates updating is fundamental to assure predictability in carbon trading investments, which apparently was deemed more important in climate negotiations than assuring the quality of carbon emission estimates.

Apart from the scientific uncertainties, using the GWP estimates to commensurate different GHGs is also problematic due to its sensitivity to the assumptions made in calculations, including if and how to incorporate indirect effects, the atmospheric lifetime of each gas, the evolution of $CO_2$ concentrations and the relevant time horizon.

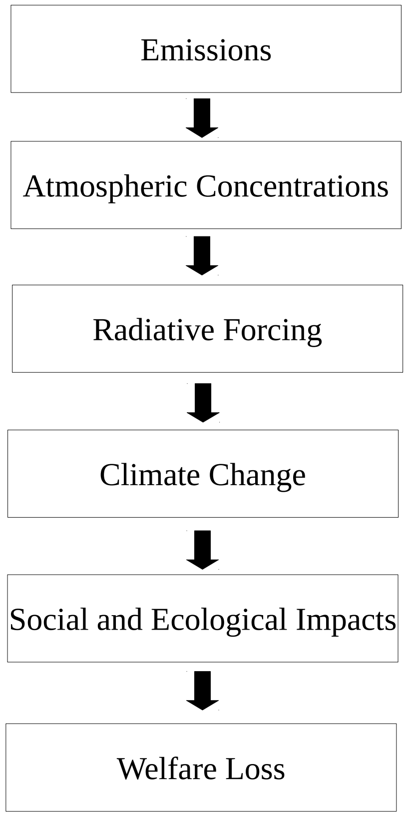
One crucial aspect of these assumptions is that the GWP estimates have to be periodically adjusted to reflect calculations of $CO_2$ concentrations according to the current treatment of the carbon cycle. These recalculations would be pertinent even if no scientific uncertainties existed regarding the estimation of GWPs, given that the climate system is not at an equilibrium state and that $CO_2$ concentrations are increasing and will continue to increase due to current and past emissions (Wuebbles *et al*., 1995). On the other hand, if the objective of climate policy is to follow a certain emission reduction path, rather than assure a certain level of emissions concentration, an index that varies with time would be more adequate than the GWP (Wigley, 1998; Tanaka *et al*., 2009).

Another crucial aspect is the non-scientific nature of the choice of integration time horizon. Choosing a shorter time horizon emphasizes the rate of climate change over its magnitude, so it would be adequate for formulating a climate policy aimed at preventing medium-term abrupt changes in the climate system. Conversely, choosing a longer time horizon emphasizes the magnitude of climate change over its rate, so it would be adequate for formulating a climate policy aimed at preventing long-term, chronic changes in the climate system (Harvey, 1993).

Regarding this choice, the IPCC's First Assessment Report presented GWP estimates for three different time horizons, of 20, 100 and 500 years, but made it clear that these were merely “candidates for discussion” that “should not be considered as having any special significance” (Houghton *et al*., 1990: 59). The KP ended up using the 100-year integration time horizon, but this was “not based on any published conclusive scientific discussion” (Fuglestvedt *et al*., 2003: 292), nor, which is more surprising, on any explicit political justification. It can be argued that choosing a long time horizon was coherent with the objective of the UNFCCC, stated in Article 2, which is the “stabilization of greenhouse gas concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system.” But this does not explain why a 100-year time horizon was chosen over, say, a 500-year time horizon.

A final critical issue with using GWPs to commensurate GHGs is the choice of variable used to compare the climate impact of each gas. Contrary to what its name might suggest, GWPs do not commensurate GHGs according to their contribution to global warming, which can be measured by global temperature change, but rather commensurate GHGs according to radiative forcing. If the objective of the KP is to stabilize global mean temperatures, sea level rise and/or other climate change indicators, then the GWP is not an adequate commensurator and climate models must be used to convert radiative forcing into these indicators. If, on the other hand, the objective is to minimize ecological and social impacts or welfare loss, socioecological and/or econometric models must also be used. Of course, the further we go down on the cause-effect chain from emissions to impacts and damages, the increased relevance of the commensurator used comes at the price of increased complexity and uncertainties in calculations (Fuglestvedt *et al*., 2003).

**Figure 14 - Cause-effect chain Emissions—Impacts—Damages**




*Source: Adapted from Fuglestvedt et al. (2003)*

Each step in the cause-effect chain involves is marked by complex and non-linear relationships. According to Smith and Wigley (2000b), the GWP overestimates forcing changes, since it assumes constant background concentrations of GHGs and ignores the non-linearities in the relationship between emissions and concentrations and between concentrations and radiative forcing. Some authors, however, have proposed to go further down the chain.

Shine *et al*. (2005, 2007), suggest going one step further in the chain to calculate the Global Temperature Change Potential (GTP) for each GHG. Hammit *et al*. (1996) further suggested going to the last step of the chain, to calculate an Economic Damage Index (EDI) that commensurates GHG emissions according to their contribution to discounted GDP loss. Also taking into account economic variables, Manne and Richels

(2001) proposed instead to calculate the relative importance of each GHG on the basis of the willingness to pay to emit one additional ton of the gas, considering its contribution for global mean temperature change. An IPCC meeting on alternative metrics, however, concluded that the GWP continues to be useful, since alternative metrics lack the scientific basis to address the shortcomings identified with the GWP (IPCC, 2009). In other words, using a more complex metric requires further research.

Changing the metric used to commensurate GHGs significantly changes the relative importance of non-$CO_2$ short lived gases (Aamaas *et al*., 2013), with proportional effects on the relative profitability of different investments in carbon offsetting. But any metric that commensurates $CO_2$ emissions reductions with non-$CO_2$ emissions reductions effectively creates an incentive to replace the former with the latter, whenever it is cheaper to do so. Yet, since non-$CO_2$ GHGs are short-lived and $CO_2$ can remain in the atmosphere for centuries, the equivalence established by the chosen metric does not hold in the long-run. If, for instance, Kyoto's carbon trading leads to a $CO_2$ emissions increase, compensated by a $CH_4$ or other short-lived GHG emissions reduction, in the long-run radiative forcing and climate change consequences are increased (IPCC, 2009; Smith and Wigley, 2000a). This problem follows from the choice of the GWP as the common metric, as it can lead to a significant overestimation of the importance of non-$CO_2$ emissions reductions for the stabilization of global temperature change (Johansson *et al*., 2008; Fuglestvedt *et al*. 2000).

To sum up, the GWP methodology followed by the KP implicitly assumes that GHGs can be traded-off according to their contribution to radiative forcing, $CO_2$ concentrations are stable, the magnitude of change is more important than the rate of change and 100 years is the relevant time horizon. While all of these assumptions reflect political choices, including the choice to sideline scientific uncertainties to promote carbon trading with non-$CO_2$ GHGs, none of these choices was made explicitly nor justified. Instead, the political nature of the GWP was obscured by “black boxing” its calculations and presenting the numbers as indisputable scientific facts.

The available scientific knowledge, then, cannot assure the quality of the numbers produced by the technical dimension of carbon commensuration. But the scientific basis for the value and cognitive dimensions of carbon commensuration also exhibits significant weaknesses, considering the location effects of GHGs and the differences between emissions from land use and fossil fuel burning.

Global carbon trading, created by the KP, relies on the assumption that all carbon emissions should be given the same value since the contribution of carbon emissions to climate change does not depend on its location or timing. The assumption, however, does not hold when carbon emissions are mixed with emissions of gases and aerosols that have an indirect climatic effect, which is the case of fossil fuel burning and most industrial processes that emit GHGs. Burning fossil fuels, for instance, releases not only $CO_2$ and $CH_4$ but also nitrous oxides ($NO_x$), volatile organic compounds (VOCs) and carbon monoxide (CO), which contribute to the formation of the GHG tropospheric ozone ($O_3$), as well as affect the atmospheric lifetime of $CH_4$. Considering these indirect effects, Berntsen *et al*. (2006) conclude that the climate effect of reducing carbon emissions depends not only on the time horizon over which the effects are considered but also on which gases and aerosols are affected, the duration of the reduction and the chemical, physical and meteorological conditions of the region where the reduction occurred. Accordingly, Shindell and Faluvegi (2010) estimate strong regional variations in climate impacts of multi-pollutant emissions from coal plants, which cannot be captured by global metrics such as the GWP.

Attributing the same value to all carbon emissions is also problematic when it involves accounting for sinks, that is, negative carbon emissions from land use activities. This is the case of KP's inclusion of Land Use, Land Use Change and Forestry (LULUCF) in industrialized countries' carbon inventories, as well as the inclusion of afforestation and reforestation activities in the CDM, which allow sink enhancement to be used to offset fossil fuel emissions. It is also the case of the proposed inclusion of Reducing Emissions from Deforestation and Forest Degradation (REDD+) as an offsetting mechanism, which would generate carbon credits from avoided deforestation. While the carbon stock in unexplored fossil fuel reserves is stored geologically, the land carbon stock, i.e., the carbon stored in forests and soils, is part of the active carbon cycle and therefore circulates in the atmosphere. In a policy-relevant time horizon (which can go from decades to centuries but is commonly taken to be 100 years, as the GWP formula shows), only the fossil fuel carbon is permanently stored and the potential contribution of sinks to climate mitigation is negligible (Mackey *et al*., 2013). Commensurating land use and fossil fuel emissions, therefore, leads to an increase in the transfer of carbon from fossil fuel reserves to the carbon cycle, through fossil fuel burning, which aggravates climate change.

Commensurating carbon across sources and sinks also leads to an increase in scientific uncertainties regarding the measurement of national inventories. While measuring emissions from sources can be done directly (with monitoring equipment) or indirectly (through calculations on the basis of fuel consumption), measuring removals from sinks necessarily depends on estimates which rely on complex calculations. To unambiguously estimate variations in carbon sequestered in the world's forests, for instance, it is necessary to not only have a common definition of forest but also use a common methodology to provide accurate data on deforestation and its impacts on the carbon cycle (Ramankutty *et al*., 2007). But the best available scientific knowledge, combined with advanced technologies (like satellite remote sensing), is still unable to deliver such accurate data, due to scientific uncertainties and the high cost of measurement technologies (Spalding *et al*., 2012). The lack of comparable and reliable historic data further adds considerable uncertainties to the process of establishing baselines against which to determine variations in forest cover (DeFries *et al*., 2002; Harris *et al*., 2012). What is worse, even if accurate data on forest cover was available, inaccuracies could still remain in the methodologies used to convert this data to estimates of carbon sequestration. Likewise, carbon sequestration estimates for other carbon stock, like soils, are also prone to significant uncertainties (Ogle *et al*., 2003; Goidts *et al*., 2009).

The high cost of gathering data on carbon sinks further leads to significant omissions and imprecisions, namely in developing countries' carbon inventories (DeFries *et al*., 2007). Transferring data from well-known carbon sinks is not an option, considering that the carbon sequestered on terrestrial carbon sinks depends on regional variables and exhibits seasonal changes. It is also climate-dependent, meaning that carbon sinks estimates have to be revised downwards as the global climate changes (Friedlingstein *et al*., 2006). Regional, seasonal and climatic variations are important enough to convert sinks into sources, temporarily or permanently, and magnify uncertainties in measurements.

In short, uncertainties and inaccuracies in land carbon measurement do not make it possible to unambiguously determine the carbon outcome of land use changes, or even its sign, with the exceptions being the transition from agriculture to forest regeneration (which is certainly positive) and the transition from mature forest to agriculture (which is certainly negative) (Ziegler *et al*., 2012). But this is not merely a scientific problem. Since it is impossible to predict future human actions, namely regarding land-use practices, social indeterminacies have to be acknowledged, which leads to abandoning the idea that advances in scientific knowledge will eventually eliminate all uncertainties and produce objective and value-neutral estimates of carbon sinks (Lövbrand, 2004). The inclusion of sink enhancement as a means of compliance in the KP, therefore, can significantly widen the gap between emissions estimates and real emissions.

## 5.3 Conclusions

Carbon trading requires carbon commensuration, that is, the social process by which carbon emissions are quantified, given a single monetary value and abstracted from their context. This process is not just an interpretation of the underlying reality, but rather a transformation that renders irrelevant or invisible all the information about how, where and when carbon was emitted, while creating and reifying new categories and actors. Opening the black box of carbon commensuration to analyse how its numbers are produced, however, reveals significant uncertainties, as well as political choices made without justification.

The implementation of a comprehensive approach to climate change, interpreted as the creation of carbon markets where all carbon emissions can be traded, was based on sidelining measurement uncertainties, in order to transform intervals into numbers. These uncertainties in carbon emissions quantification can make emissions estimates unreliable to the extent that compliance with reduction commitments cannot be evaluated unambiguously. This is aggravated by the use of a common metric, the GWP, for commensurating different GHGs, which is prone to its own uncertainties and dependent on implicit political assumptions that were never justified. Ignoring the effect of co-pollutants and including sinks without considering the consequences of non-permanence and inaccuracies in data gathering further contributed to sidelining uncertainties. As a result, a source can avoid reducing its emissions by buying allowances or credits even if these do not correspond to real emissions reductions but rather were generated as a result of measurement errors.

These errors could, to be sure, be partially corrected by advances in scientific knowledge and measurement technologies. Berntsen *et al*., (2005) argue that it could be possible to directly measure national emissions using inverted models that use data on the spacial and temporal distribution of a pollutant in the atmosphere, the atmospheric circulation and the natural emission sources and sinks. The authors note, however, that this method would have to be substantially refined to deliver estimates for each country and greenhouse gas with a low level of uncertainty, which implies that, two decades after Kyoto, carbon markets still lack the knowledge necessary to measure emissions with a high degree of confidence, even though this is supposedly a major precondition for the implementation of emissions trading.

Measurement errors, moreover, do not merely reflect scientific uncertainties but also deep uncertainties regarding the evolution of societies and political choices regarding the objectives of climate policy. There is no technical solution that would render the comprehensive approach a workable alternative for climate mitigation. A reasonable alternative that would be compatible with an effective climate policy would rather be based on accepting uncertainties regarding possible outcomes of human actions, which would necessarily lead to different approaches for each type of carbon emissions source or sink. Accepting uncertainties as a fact of life would also lead to following the precautionary principle and committing to deep emissions reductions, steering climate policy into the lower end of the interval of possible climate change impacts. Clearly, none of this is compatible with including carbon trading on the policy-mix.

# 6 All carbon emissions were not created equal: Why carbon trading fails

"The carbon market doesn't care about sustainable development. All it cares about is the carbon price…the carbon market is not going to be able to put sustainable development and everything else into one price." (Jack Cogen, 2005 *apud* Erion, 2008: 446).[14]

The Kyoto Protocol sets binding commitments for greenhouse gases (henceforth designated as carbon) emissions for industrialized countries. These targets can be met through the use of two types of flexibility mechanisms: "cap and trade", which allows polluters to trade pollution allowances among themselves, and offsetting, which allows polluters to exceed their caps by buying credits generated from projects that reduce emissions elsewhere.

Both instruments, which will be approached here using the umbrella term "carbon trading", are justified on cost-efficiency grounds. As the argument goes, by not discriminating carbon emissions, carbon trading allows polluters to choose their level of emissions and abatement technologies according to their cost structure, therefore minimizing abatement costs (cf. Dales, 1968b). This follows mainstream economics focus on finding the best instrument to put a price on pollution, which is understood as a market failure that results from unpriced external effects of production (or externalities). The climate policy debate, then, is reduced to the relative merits of a carbon tax versus a carbon trading system, in terms of their effects on economic efficiency (Parry et al., 1999; Hepburn, 2007).

Efficiency arguments disregard how efficiency is but one of several values that societies can pursue. From a value pluralist perspective, a richer climate policy debate can

14 President and co-founder of carbon asset management firm Natsource and ex-Chairman of the International Emissions Trading Association (2008-2010), personal communication at a 2005 Montreal Climate Conference side-event.

be had by understanding climate change impacts as social costs of private economic activities that are borne by societies as a whole. Following Kapp (1975: 13-25, 67-79), these social costs are multi-dimensional, in the sense that they affect different dimensions of human well-being, and, as extra-market phenomena, fail to be fully captured by quantitative evaluations. This framework allows us to acknowledge both the existence of value conflicts in climate policy and the impossibility of addressing these value conflicts by assuming their commensurability, i.e., assuming that values can be traded-off with each other (O'Neill et al., 2008: 70-81).

The Kappian approach on social costs also allows us to question the ends-means dichotomy that is inherent to the characterization of carbon trading as the best means to achieve a given end, which is taken to be a certain level of emissions reductions. As this chapter will illustrate, even though carbon trading was purportedly conceived to address then social costs of climate change, it also generates social costs of its own. This means that different means (policies) lead to different ends (policy outcomes) and, therefore, that the design of climate policies is both a process of adjusting means to ends and of adjusting ends to means.

This approach can be further enriched by articulating the problem of climate policy choice with the debate on the limits of markets. Carbon trading is a fundamental part of the neoliberalization of nature, which comprises relations of governance, privatization, enclosure and valuation (Heynen and Robbins, 2005). It grants polluters the possibility of delaying structural change by buying rights to pollute, while also creating new financial markets that expand the frontiers of capital accumulation. Through what Castree (2003) called the proxy commodification of the environment, trading in rights to pollute and "ecosystem services" is discursively justified as a means of "selling nature to save it" (McAfee, 1999). Yet, as Polanyi (2001: 75-79) warned, the commodification of nature cannot "save it" but rather contribute to its destruction. Free market ideology, which leads to costly and ultimately futile attempts to disembed the economy from society, is particularly damaging when applied to nature which, as a fictitious commodity, is not something that is produced to be sold in a market but rather a condition for production and life itself. Following Polanyi, then, environmental policies should not hand out decisions about the fate of the environment to the laws of the market.

Using these references as inspiration, this chapter aims to present a radical critique of carbon trading, focusing on the negative consequences of carbon commodification and of the processes of commensuration and abstraction that support it. First, we present the methodology for devising a taxonomy of normative critiques, based on published critical literature. Second, we argue, following the taxonomy of normative critiques, that carbon trading is ineffective, unjust, undemocratic and unethical. Third, we discuss the possibilities and limitations of carbon trading reforms that could address some critiques. The chapter concludes with a discussion on alternative ways to evaluate climate policies.

## 6.1 Reviewing the critical literature on carbon trading

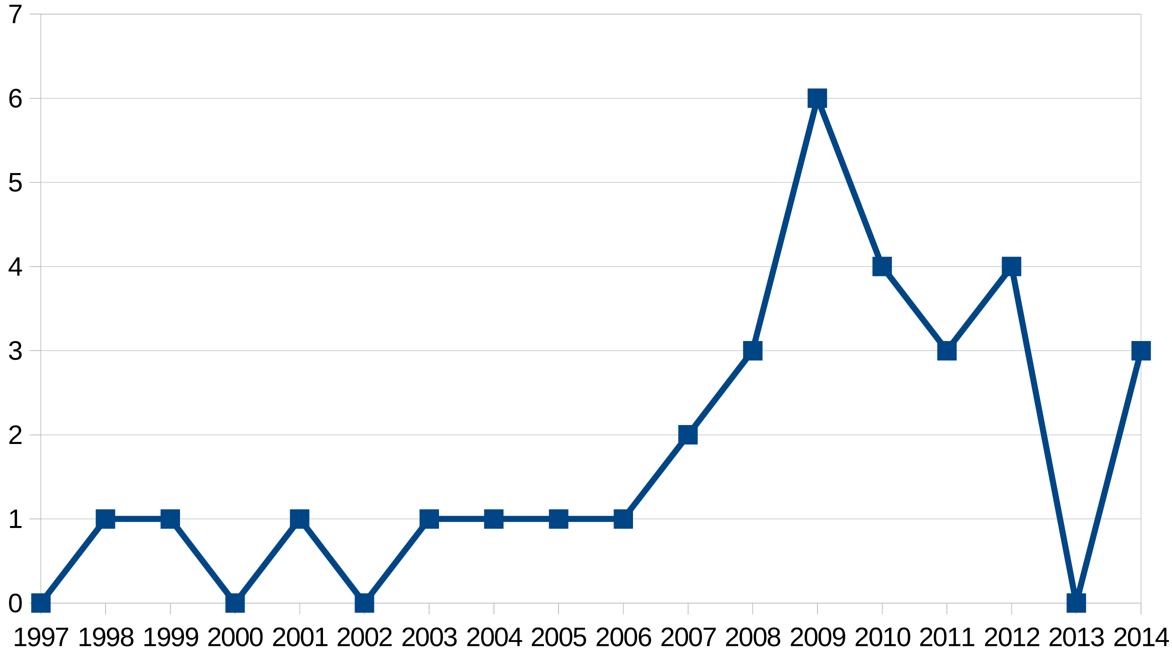
To produce a normative critique of carbon trading, we did a comprehensive and systematic review of the relevant critical literature. We retrieved publications using the generic search terms “carbon trading”, “carbon markets” and “Clean Development Mechanism” in full text academic databases provided by major publishers (Elsevier, Sage, Springer, Taylor and Francis and Wiley) and library services (EBSCO's Academic Search Complete and Business Source Complete). Additionally, the same search term was used on Google Scholar, to retrieve open access versions of publications not included in these databases. We then selected academic journal articles and books written in English, and excluded book reviews, news articles, interviews, conference proceedings, reports, comments and presentations. Given the objective of the review, publications were selected on the basis of their abstracts. This resulted in the exclusion of four types of publications: (1) reformist critiques, i.e. those that criticize a certain correctable problem in carbon markets and propose ways to overcome the problem; (2) pro-emissions trading publications that reject its applicability to carbon emissions on the basis of technical, economic or scientific obstacles; (3) empirical publications that present case-studies; (4) publications that do not focus primarily on carbon trading, including those that present a broad critique of market-based environmental policies.

The database thus gathered of critical articles and books was remarkably small: even though our original search in academic databases resulted in retrieving over 2,000 publications, our final database included only thirty one articles (Driesen, 1998, 2007a, 2007b; Byrne and Yun, 1999; Byrne *et al,* 2001; Richman, 2003; Bachram, 2004; Lohmann, 2005, 2008, 2009b, 2009c, 2010, 2011a, 2011b; Sandel, 2005; Baldwin, 2008; Bumpus and Liverman, 2008; Jones, 2009; Pearse, 2010; Spash, 2010; Vlachou and Konstantinidis, 2010; Bohr and Dill, 2011; Aldred, 2012; Böhm *et al*., 2012; Childs, 2012; Fletcher, 2012; Spash and Lo, 2012; Bryant *et al*., 2015; Ervine, 2014; Vlachou, 2014; Pearse and Böhm, 2015) and three books (Lohmann, 2006; Böhm and Dabhi, 2009; Reyes and Gilbertson, 2009).

Most of these publications (nearly 80 percent of the total) were published after 2007, that is, a decade after the Kyoto Protocol was signed. One reason why it took a decade for critical literature to take off might be that only in 2005 the first carbon trading system was implemented, in the EU. Another possible reason was the publication in 2006 of a critical book by activist and independent researcher Larry Lohmann, which was made freely available on the internet and is one of the most widely quoted references on this carbon trading.[15]

15 To compare citations for carbon trading publications we used the citations analysis software Publish or Perish (Harzing, 2007) to search in Google Scholar, as it is the most comprehensive database available. Lohmann (2006) is the most quoted reference when we use the search term “carbon trading”, both in terms of total number of citations and of the number of citations per year.

***Figure 14: Number of critical publications on carbon trading per year, 1997-2014***




After compiling and reviewing the critical literature, we grouped the arguments given against carbon trading according to four normative principles that they appeal to: effectiveness, justice, democracy and environmental ethics. By focusing on how carbon trading disregards these principles, we arrived at four normative critiques, which were substantiated, as needed, by complementary literature.

The four normative critiques are, necessarily, interconnected: an ineffective climate policy unjustly penalizes those who cannot adapt to climate change, an undemocratic climate policy is also procedurally unjust and thus penalizes those who are not consulted, an unethical climate policy undermines the motivation for mitigation action, and so on. For analytical purposes, however, a taxonomy of normative critiques, which can also be seen as a taxonomy of the social costs of carbon trading, is both useful and illustrative, even if the boundaries between different categories are sometimes blurred.

## 6.2 Four normative critiques of carbon trading

**1. Carbon trading is ineffective**

Addressing climate change requires substantial emissions reductions, which can only be achieved if most fossil fuel reserves are left unexplored (McGlade and Ekins, 2015). This, in turn, requires major changes in production and consumption patterns, affecting all economic sectors, including industry, energy, agriculture and transportation. Carbon trading is ill-designed to promote the transformations that are required to overcome fossil fuel dependency and drives resources away from political actions and technological innovations that can forestall fossil fuel extraction and burning (Lohmann, 2005).

Modern industrial economies are characterized by carbon lock-in, that is, the lock-in of high-carbon technological systems and the lock-out of alternative ones (Unruh, 2000). This is a result of increasing returns to scale on the adoption of new technologies, which makes competition between technologies a path-dependent process that can "cause the economy gradually to lock itself in to an outcome not necessarily superior to alternatives, not easily altered, and not entirely predictable in advance" (Arthur, 1989: 128; see also David 1985). In addition, the public and private institutions that govern adopted technologies can create non-market forces of lock-in (Unruh, 2000).

Transitioning to a low-carbon development path necessarily involves implementing low-carbon technological systems, which can only be achieved by positively discriminating investments in radical innovations, to the detriment of incremental innovations (Unruh, 2002). Investments in renewable energy, energy conservation and other alternatives to fossil fuel energy, then, should be valued more highly than end-of-pipe investments in technologies that reduce emissions but do not displace fossil fuels in the long run, as the former lead to higher emissions reductions in the future than the latter. Since carbon trading fails to differentiate between different emissions reductions of the same magnitude, it does the opposite, directing investments to cheap emissions reductions that are usually associated with incremental innovations

(Driesen, 2007b).[16] As a result, carbon trading reinforces carbon lock-in and makes the transition away from fossil fuel dependence even more difficult.

This incapacity of carbon trading to acknowledge path dependencies on technological innovations is on par with the failure of the economic theory that supports it to acknowledge path dependencies in control costs. Future control costs depend on the choices made in the present on how to distribute carbon reductions across sectors, time and greenhouse gases. These choices affect the price of carbon permits and credits, which will have an effect on all prices in an economy, due to the fact that, in the present, all goods use (fossil) energy in their production process as an input. The present and future effect on prices, including prices of pollution control technologies, from variations in the carbon price is uncertain and unpredictable (Spash, 2010; Spash and Lo, 2012). A climate policy based on carbon trading, however, cannot incorporate this unpredictability, as it implicitly assumes a stable and predictable relationship between the carbon price and carbon emissions.

The incapacity to differentiate emissions reductions also undermines carbon trading's effectiveness in the present, due to the differences in the local effects of pollutants. As mentioned in the previous section, carbon emissions are usually released into the environment mixed with emissions of other pollutants. Many of these co-pollutants have not only a direct effect on air quality but also have an indirect climate effect, namely by contributing to the formation of tropospheric ozone ($O_3$), which is a greenhouse gas, and by affecting the lifetime of methane emissions. The timing and location of carbon emissions, then, matters when considering their climate effect (Berntsen *et al*., 2006; Shindell and Faluvegi, 2010).

A third reason why carbon trading is ineffective is the failure to distinguish

16 It can be argued that carbon trading provides a lower incentive to innovate than performance standards since sources with high abatement costs can choose to buy carbon allowances instead of investing in emissions-reducing technological innovations (Driesen, 2007a). The incentive to innovate might, however, be larger with emissions trading for sources with low abatement costs, since these can choose to invest in emissions-reducing technological innovations to sell carbon allowances at a profit, so the relative inferiority or superiority of emissions trading in terms of incentives to innovation depends on the balance between allowance buyers and sellers in the market (Malueg, 1989). Furthermore, the argument presented here is unrelated to the discussion on incentives for innovation from different regulatory instruments and is rather a variation of the argument on the relative inferiority of price instruments regarding incentives for radical innovations (Mokyr, 1991).

emissions reductions driven by policy changes from “paper” reductions derived from exogenous events or from non-additional CDM projects. The first source of “paper” reductions is the “hot air” allowances that are generated as a result of economic crises. One source of “hot air” is the early 1990s crisis that followed the disaggregation of the USSR, which is not reflected in Russia and Eastern European countries' emissions commitments. Another source is the 2008-2012 global financial crisis, which led to an increase in excess allowances held by EU countries. Allowing industrialized countries to buy “hot air” allowances from countries that reduced their emissions due to factors unrelated to the KP results in a net emissions increase.

The second source of “paper” reductions is non-additional CDM projects. For a CDM credit to be considered eligible for compliance, it has to result from an emissions reducing investment in a country from the global South which would not have happened otherwise. Yet, as explained in chapter 5, this additionality requirement is both essential and impossible to enforce, since scientific knowledge and technical expertise cannot isolate a single baseline scenario, representing the business as usual emissions, from the undetermined and possibly infinite set of possible futures (Lohmann, 2009c, 2010). The classification of a project as additional and the estimates of emissions reductions achieved by it are therefore exercises in futurology or even in story-telling, and any objectivity claims attached to the numbers that these exercises produce are unfounded. Consequently, the CDM, by replacing real emissions reductions in the present for imaginary emissions reductions in the future that might happen for motives unrelated to the CDM or fail to materialize, actually leads to a net increase in carbon emissions (Lohmann, 2011b).

Given the non-technical nature of the additionality problem, a technical solution will necessarily fail. Furthermore, the nature of the CDM leads “bad” credits to drive away “good” credits. In a market where consumers cannot evaluate the quality of the products sold, lower quality producers will drive higher quality producers out of the market (Akerlof, 1970). In carbon offset markets, not only are consumers unable to evaluate the quality of the credits but will likely even be indifferent to it, since all that matters is that credits are usable for compliance. Credit buyers will then prioritize low-cost emissions reductions, disregarding the environmental and social circumstances in which the credits were produced (Byrne *et al*., 2001).

Finally, a fourth reason why carbon trading is not an effective instrument to address the climate crisis is that it is incompatible with an ambitious climate policy. Reducing the cap on emissions to the level that is compatible with avoiding runaway climate change, which is a 40 to 70 percent reduction by 2050 relative to 2010 (IPCC, 2014), would undermine and eventually eliminate opportunities for trading (Childs, 2012). Moreover, approving regulations that lead to further emissions reductions, to compensate for the ineffectiveness of carbon trading, inevitably leads to conflicts, as the reductions generate an allowance surplus that can be sold to other sources or banked for future use.

When carbon trading is a part of the policy-mix, the emissions reduction achieved by additional regulations can be compensated by an emissions increase of the same magnitude, rendering the regulations ineffective. Additional regulations can only be effective if they mandate polluters to reduce their emissions irrespective of the number of allowances and credits held. This would, however, lead to a chronic excess supply in carbon markets and eliminate opportunities for trading. Carbon trading, then, can only work when both the emissions cap and the additional climate regulations are unambitious.

## 2. Carbon trading is unjust

Climate change impacts are not spread equally across societies but rather hurt the vulnerable and the poor disproportionately, since affluent individuals and groups can adapt to the changing climate more easily (Barker *et al*., 2008; Comim, 2008). Since carbon trading is ineffective in addressing climate change, it necessarily follows that it aggravates climate injustices. But carbon trading is also unjust because it relies on abstracting carbon emissions from their context and does not allow for the distinction of different emissions-reducing projects or emissions-enhancing investments regarding their impacts on social inequalities (Lohmann, 2008).

Any climate policy encompasses a quantitative target for carbon emissions (which can be a number or an interval) and the means to ration the carbon below the target. Carbon trading rations carbon according to willingness to pay and income, extending the extreme inequalities of access to goods that exists within and between countries to carbon (Aldred, 2012). But carbon is not a product, rather it is an undesirable byproduct, one that is pervasive across production processes, given industrial economies' dependence on fossil fuels (Spash, 2010). Inequalities in access to carbon lead, therefore, to inequalities in access to goods that are essential for subsistence. Since carbon trading disregards the distinction between subsistence emissions and luxury emissions, to use Narain and Riddle's (2007) terminology, it inevitably aggravates inequalities. Furthermore, carbon trading can result in the concentration of carbon emissions and hazardous co-pollutants in local “hot spots”, aggravating living conditions for poor communities living in the most polluted areas (Kaswan, 2008).

The deleterious effects of carbon trading on social justice are amplified as its scope increases. The CDM, in particular, by allowing industrialized countries to appropriate cheap opportunities for emissions reductions in the global South and evade their climate commitments, amounts to a form of carbon colonialism (Bachram, 2004). International carbon offsetting supports accumulation strategies of polluting industries in the North by allowing them to export emissions reductions to the South, even though the historical responsibility for climate change lies in the North (Bryant *et al*., 2015). Even though developing countries can profit from selling CDM credits, this benefit comes at the cost of sacrificing the “common but differentiated responsibilities” principle that is present, even if only symbolically, in the KP, their sovereignty and the interests of the poorest and most vulnerable countries (Byrne *et al*., 2001; Richman, 2003; Bohr and Dill, 2011). Furthermore, the CDM benefits for the global South are allocated under conditions of unequal exchange between buyers and sellers, making the CDM a form of accumulation by dispossession (Harvey, 2003) that Bumpus and Liverman (2008) dubbed as accumulation by decarbonization.

Freely allocated allowances, which are a pervasive feature of carbon markets, also lead to shifts in the distribution of income depending on these allowances are distributed across polluters. On the one hand, if firms are able to pass through to consumers the opportunity cost of allowances they got for free, they get a windfall profit, which reflects a transfer of income from consumers to polluters. On the other, if energy-intensive industries face relatively laxer caps than labor-intensive industries, justified by the exposure to international competition and the risk of "carbon leakage", income is transferred from the latter to the former, which can aggravate unemployment and stimulate higher energy use. In short, while auctioning allowances leads to inequalities in the access to carbon, freely allocating allowances leads to having governments distribute economic rents to polluters, in a regressive and potentially corrupt slicing-the-pie game (Vlachou, 2014).

Carbon trading also conflicts with intergenerational justice, which requires that present generations abstain from actions that increase the probability of future harm and compromise future generations' needs (Ekeli, 2004). Banking provisions in carbon markets, which allow polluters to retain emissions allowances and credits that can be used in the future, can induce greater emissions reductions in the present only at the expense of increased emissions in the future. But, while the chemical composition of emissions does not change over time, its social meaning and even its environmental impact (given the cumulative nature of environmental problems) is constantly evolving (Levin and Espeland, 2002). Future generations, therefore, should have the right to review the goals of climate policies to reflect changes in values or scientific evidence, namely by increasing the ambition in emissions reduction targets, which inevitably colludes with polluters' rights to use banked allowances and credits.

### 3. Carbon trading is undemocratic

A democratic climate policy presupposes that citizens can have a voice and participate in decision-making, while decision-makers are held accountable to citizens. Carbon trading conflicts with these goals, as public participation is a transaction cost that hampers its efficiency (Kaswan, 2008).

According to the democratic argument for emissions trading, while direct regulation focuses the political debate on the cost and potential of technologies, emissions trading rather focuses the debate on the acceptable levels of pollution (Ackerman and

Stewart, 1988; Sunstein, 1991). Since most people do not have the knowledge necessary to participate in a debate on technologies but nevertheless have an idea about what is an acceptable environmental state and what is the level of emissions reductions needed to achieve that state, it follows that replacing direct regulation with emissions trading increases the accountability and democratic participation in the decision-making process. This argument, however, does not hold when we take into account that it is not possible to separate the discussion on means and ends, and therefore discussing what is an acceptable level of pollution necessarily implies discussing what are the costs of reducing emissions to reach that level of pollution, bringing into fore the discussions on control technologies that emissions trading was supposed to dismiss (Heinzerling, 1995). Historical experience with carbon trading illustrates this impossibility, since the EU ETS Directive determined that emissions caps for covered sectors should be set according to the technological potential for reductions, which is the same calculation made in Best Available Technology (BAT) standards (Driesen, 2010). Furthermore, the assumption that debates over the acceptable levels of pollution are (always) more accessible than debates over control technologies amounts to a leap of faith of dubious validity (Heinzerling, 1995).

Given that decisions on where and how to reduce emissions, how to allocate carbon allowances and whether and in what terms is offsetting allowed have a profound impact on social justice and climate change mitigation, it is to be expected that citizens will care about these decisions and not just about the emissions cap. Carbon trading, however, hands decisions on emissions reductions over to polluters, traders and other carbon market participants, while other decisions are made in an opaque process that makes policy-makers accountable to corporations instead of populations. Accountability to consumers is also lost in the abstraction of carbon emissions, as carbon trading provides little information regarding the emissions abatement efforts made by producers or the location of the reductions (Baldwin, 2008). Likewise, democratic accountability is lost when emissions commitments made by governments can be overturned by purchasing allowances and credits, making it possible for KP parties to comply with their reduction commitments even if their emissions increased and even if the allowances and credits purchased to cover excess emissions do not reflect real emissions reductions (Driesen, 1998).

Accountability, to be sure, is not just about implementing accounting schemes that accurately measure emissions and trades, using the best available scientific and technical knowledge. While such schemes could be of crucial importance to monitor the environmental performance of market participants, their monopolization by a narrow professional class of quantifiers and validators excludes citizen participation in decisions regarding the future climate and actions directed at securing this desired environmental state (Lohmann, 2005). A quick perusal at documents produced over the years within the climate negotiations on carbon markets by technocratic elites is enough for any non-expert to feel increasingly flooded with indecipherable technical jargon, making it increasingly hard for environmental NGOs and local communities to participate in the discussions (Pearse and Böhm, 2015). Paradoxically, the increased accountability that improvements on measurement and verification methods can bring to carbon markets is often achieved at the cost of the decreased accountability of these methods.

## 4. Carbon trading is unethical

A climate policy necessarily aims to change the individual and corporate conduct, by fostering personal and societal values that have a positive effect on environmental morale. Carbon trading can have the opposite effect, by weakening environmental morale and thus undermining motivations for behaviour change. Furthermore, carbon trading is based on pricing and commodifying nature and its implementation runs counter the ethics of shared responsibility and care for the environment that should be at the heart of climate policies.

Carbon trading allows polluters to evade their obligations by buying allowances and credits, even if these do not correspond to emissions reductions beyond business as usual. While regulation is based on the principle that pollution is a wrong and thus subjects polluters to fines, carbon markets are based on trading pollution rights, turning fines into fees and removing the moral stigma that is properly associated with pollution (Sandel, 2005). As a consequence, carbon trading can crowd out the environmental morale that supports behaviour change. This effect is stronger than with environmental taxes, which do not entirely remove the moral stigma associated with pollution, and much stronger than with direct regulation, which can even reinforce environmental morale (Frey and Stutzer, 2008; see also Goodin, 1994; Frey, 1997). This is because direct regulations and taxes have a clear expressive function which reinforces citizens' and firms' motivation to refrain from causing harm to the environment, while buying rights to pollute legitimises pollution, inducing a cynical attitude comparable to that of buying indulgences to absolve past or future sins (Smith, 2007).[17]

A second moral objection to carbon trading is that it undermines the sense of shared responsibilities and sacrifices that cooperation in addressing climate change requires (Sandel, 2005). Much like war time rationing systems, a carbon rationing system will only be seen as fair and worthy of compliance if no individual or firm is able to buy its way out. Carbon rationing, then, requires that everyone participates directly in reducing emissions. Furthermore, the moral obligation not to cause harm cannot justifiably be discarded by paying someone else to avoid causing harm, which is what carbon markets allow participants to do (Hyams and Fawcett, 2013).[18]

Carbon trading is also unethical due to the commodification of the carbon cycle and the concomitant creation of property rights over the atmosphere.[19] This assertion holds whether we consider the atmosphere to be common property or no one's property. If the atmosphere is humanity's common property, then carbon trading is to be rejected because it leads to an uneven access to the privatized atmosphere. If, alternatively, the atmosphere is not the property of humanity but rather we are just its stewards, then no one can own the atmosphere nor a right to pollute it (Aldred, 2012).

17 Even though firms are not individuals, the argument still stands for firms if they behave as if they were motivated by environmental concerns. This is the case when firms follow the environmental ethic of consumers to increase their market share or avoid boycotts, when firms want to appease governments in order to prevent environmental regulations or when firm owners are environmentalists (Frey, 1992).

18 This argument was made eloquently, if also sarcastically, by Cheat Neutral (http://cheatneutral.com), a spoof website that advertises cheat offsets, to be bought by people who cheat on their partners and sold by people who commit to being faithful.

19 A common counter-argument from carbon trading supporters is the denial of the status of property rights to carbon allowances and credits, as these only grant temporary use rights on the atmosphere (Caney, 2010). This argument, however, ignores how a lease is a *de facto* legal property right, a fact illustrated by the legal difficulties inherent to removing from circulation rights to pollute (Spash, 2010; Aldred, 2012).

A fourth ethical objection to carbon trading is that it relies on the incorrect assumption that the environment should be valued in market terms. Carbon markets set a price for carbon, which can be taken to express the social cost of carbon, i.e., the value that compensates for the harm caused by emissions, and thus used as a reference for climate policies (Aldred, 2012). This conflicts with people's commitments to incommensurable environmental values, which can emerge from an ethic of care for the environment that is irreconcilable with the notion that the environment should only be preserved if monetary benefits exceed costs (O'Neill *et al*., 2008: 70-88).

These objections expose two major flaws in the consequentialist ethic that carbon trading promoters follow (e.g. Caney and Hepburn 2011; Page 2013). First, focusing a climate policy on its consequences, measured by emissions reductions, neglects legitimate concerns with justice and democracy. Second, the connections between motivations for acting and the consequences of the act, illustrated with the consequences of removing the moral stigma of pollution, creating a way for firms to buy their way out of reducing emissions and pricing nature, are sufficient to show how carbon trading crowds out the environmental ethic that is needed to induce behaviour change. This is yet another reason why carbon trading is ineffective, which invalidates the consequentialist defence of carbon trading in its own terms.

## 6.3 The limitations and contradictions of carbon trading reformism

There are, broadly speaking, two types of answers to the normative critiques presented *supra*. On one side, the radical answer, given in the critical literature that was reviewed, consists in arguing that carbon trading should not be seen as an alternative for climate change mitigation. On the other side, the reformist answer consists in arguing that carbon trading can be improved to increase its effectiveness, reduce injustices, promote public participation and protect environmental ethics (Boyd *et al*., 2009; Caney, 2010; Caney and Hepburn, 2011; Page, 2011, 2012, 2013; Newell *et al*., 2013). Social movements and environmental NGOs that follow the former approach include Friends of the Earth and the Climate Justice Now! coalition, while WWF, Greenpeace and most organizations affiliated with the Climate Action Network follow the latter approach.

Carbon trading reformism is supported by "pragmatic" arguments. As the argument goes, there is no alternative to carbon trading and that the debate that environmentalists should have is on how to improve it (MacKenzie, 2009; Dirix *et al*., 2015). In a more optimistic version of this political "pragmatism", carbon markets are seen as an on-going *in vivo* experiment that gradually incorporates solutions for the new problems it creates (Callon, 2009). Since this type of arguments inevitably emerge in climate policy debates, it is necessary, at this point, to explain in detail why carbon trading reformism is doomed to fail.

A first response to reformist arguments is to discuss what carbon markets are for. As instruments of capital accumulation, carbon markets facilitate the commodification and financialization of nature, while allowing polluters to evade emissions reductions and creating perverse profit opportunities for polluters and financial corporations from the climate crisis (Lohmann, 2006; Böhm and Dabhi, 2009; Jones, 2009; Pearse, 2010; Vlachou and Konstantinidis, 2010; Böhm *et al*., 2012; Fletcher, 2012). In fact, carbon trading stands out as one of the few regulatory instruments that was not only supported but even promoted by some major polluters, including fossil fuel companies which could go out of business if an effective climate policy was approved (Meckling, 2011). Some authors, though, even while acknowledging that carbon markets are instruments for capital accumulation, maintain that they can also be instruments for decarbonization, ushering in a new era of "climate capitalism", if adequate reforms are made (Newell and Patterson, 2010; Hahnel, 2012a, 2012b). A critical view of carbon trading does not, therefore, provide a decisive argument for their abandonment, since it might be a necessary evil in capitalist economies. The decisive argument is, instead, the insufficiency and infeasibility of carbon market reforms.

That some of the problems with carbon trading can be mitigated through design changes in existing carbon markets is unquestionable. The negative impact of carbon markets on income distribution could be attenuated by auctioning allowances using the revenue to compensate the poor for the loss of income. Increased monitoring and positive discrimination of emissions reductions associated with renewable energy and energy efficiency would increase the effectiveness of carbon markets. A set of discount factors and/or spatial restrictions to trading could be devised to account for the differences in carbon emissions contribution to climate change. We could even have different markets for different greenhouse gases or restrict trading to $CO_2$. The banking of permits and credits can be banned or penalized with a fee. The possibilities are endless.

All of these reforms involve restrictions to trading, diminishing the attractiveness of carbon trading. Furthermore, it is is not clear if these reforms, even if they work as intended, will not just end up creating new problems. New regulations will bring about new loopholes which will be exploited by polluters seeking to evade costly emissions-reducing investments, and the need for more reforms will emerge. As carbon markets are patched up, they become more complex and harder to regulate and transaction costs inevitably increase. Increasing the regulatory apparatus, in turn, comes at the cost of expanding the power of the regulatory bureaucracy, which can increase corruption and fraud even if the intention was to decrease it (Lohmann, 2009b).

Proposed carbon market reforms also fail to address problems that cannot be designed away. Establishing additionality for carbon offsets is not a technical issue but rather a scientific impossibility, given that no one can predict the future. An ideal carbon trading system would still conflict with other climate policies, since emissions reductions achieved by these policies would be compensated by emissions increases elsewhere. Carbon markets can never foster public participation and the democratic accountability of climate policy because markets are not democratic institutions through which people can express their values and vote on desired outcomes. Likewise, trading in pollution rights inevitably implies the privatization and pricing of the atmosphere, the crowding-out of environmental morale and the undermining of the shared responsibilities principle.

A final line of defense for a carbon trading proponent might be to argue that we could design a completely different system of carbon trading in which tradable allowances would be given to individuals, perhaps on an equal per capita basis, which would be used to cover their consumption emissions (Ayres, 1997; Fleming, 1997; Fawcett and Parag, 2010). The overwhelming complexity of such a system, due to the number and diversity of participants, as well as the difficulties of having a carbon currency alongside national currencies in all markets, however, is enough to understand how the administrative costs of implementing personal carbon trading would be far superior to other regulatory instruments. Personal carbon trading would also generate new injustices, as it amounts to a rationing system in which the relatively rich can buy their way out. Furthermore, it is left unexplained by personal carbon trading proponents how it would help societal transition to a low-carbon development path, when individual citizens do not have the power to affect patterns in infrastructure investments through their shopping habits. In other words, personal carbon trading would just unduly penalize citizens with higher consumer prices for choices made at the government and/or firm level on issues such as transport and energy production.

Regardless of the direction that the never-ending process of reforming carbon trading takes, it inevitably raises the fundamental question of whether it would still make sense to use the term "market" to describe an increasingly complex and regulated system for trading emissions allowances and credits. Carbon market reforms entail restrictions on trading, which conflict with their cost-efficiency by raising transaction costs and reducing polluters' flexibility in compliance. Much like lowering the emissions cap to a level that requires deep emissions reductions from all polluters, such reforms gradually erode opportunities to trade, eventually rendering carbon markets irrelevant. Carbon trading reformism, then, if taken seriously, is a failed project, not only because it undermines the political justifications for adopting carbon trading in the first place but also because it ends up being contradictory with the objective of saving carbon trading from itself. It is hard to see how this project can be politically more realistic than dropping carbon trading from the policy-mix.

## 6.4 Conclusions

The discussion presented here, based on the critical literature on carbon trading, allows us to see that carbon trading is ineffective, unjust, undemocratic and unethical. Turning the discussion on its head, we can postulate the criteria that climate policies should ideally follow. An effective climate policy would facilitate escaping carbon lock-in and fossil fuel dependence, acknowledge the interactions between different pollutants, exclude “paper” reductions, minimize conflicts with other climate policies and be based on an ambitious emissions reductions commitment. A just climate policy would ration access to carbon according to social needs and not income, address international and intranational injustices in adaptation to climate change, exclude any means of subsidizing polluters and allow future generations to review it according to their needs. A democratic climate policy would promote citizen participation and accountability, instead of market participation and accountability. An ethical climate policy would recognize the wrongness in pollution, promote a sense of shared responsibilities and exclude the privatization of the environment or its valuation in market prices.

While perfect climate policies would be impossible to devise, several examples of policies that respect at least most of these criteria can be easily brought to the fore, including those that support community-based renewable energy, energy conservation, public transport, agroecology or waste reduction, reuse and recycling initiatives. Such policies can foster a plurality of values and provide significant social benefits, whereas carbon trading generates significant social costs.

Trying instead to patch up a broken carbon trading system is a dead end for discussions on climate policy. Reforming existing carbon trading systems could alleviate some of the problems that they face, but only at the cost of increasing barriers to trade and decreasing its political support by polluters. Since some problems would remain unaddressed, while other new problems would emerge, the never-ending process of carbon trading reformism would inevitably lead to an erosion of opportunities to trade in carbon markets. How advocating for reforms that slowly kill carbon trading is more politically feasible than dropping carbon trading from the policy-mix is a question to which reformists have yet to provide an answer.

# 7 Conclusions

In the 1960s, while the modern environmental movement was blooming, mainstream economic theory incorporated concerns with excessive pollution by framing pollution as an externality, a market failure that resulted from polluters not having to pay for the costs of pollution. Addressing this market failure through Pigouvian taxes that internalized the cost of pollution would maximize efficiency and bring pollution to its optimal level. The Coasean critique, however, had already raised the possibility that, in the real world, transaction costs and government failure would be significant enough to make the case for intervention unwarranted, since in many situations the solution that maximizes efficiency is to do nothing. Furthermore, while formally elegant, the taxation scheme was hardly applicable outside the textbook, considering that pollution costs tend to be non-linear, non-separable and uncertain, if not outright incalculable and incommensurable.

Emissions trading theory, which was emerging at the same time, offered a way out of this conundrum. The general idea was to get rid of the externality concept and the project of trying to calculate the optimum level of pollution, allowing instead a politically constructed market of emissions rights to define the price of pollution. But Crocker (1966) was still trying to devise a way to optimize the two air value dimensions (life support and waste dump), while Dales (1968b) restricted his proposal to a market that would allow reaching an exogenously determined pollution level at the least cost. The latter proposal, which became known as “cap and trade”, was appealing, in theory, for its cost-efficiency, simplicity and flexibility, when compared with taxation or direct regulation. Still, it took two decades for it to be implemented.

The US started to experiment with emissions trading in the 1970s and 1980s, with several provisions being juxtaposed with existing environmental regulations to allow some flexibility in compliance. The Emissions Trading Program consisted of netting, offsetting, bubbling and banking provisions that allowed polluters to trade emissions reductions in some sources for reductions in other sources, while lead trading allowed refineries to trade in lead credits, instead of meeting individual limits to lead in gasoline.

These experiments were not yet real emissions trading schemes, but rather “baseline and credit” schemes through which polluters could exchange credits without an emissions cap being defined. Their ad-hoc nature reflected the conflict between environmental preservation and economic growth that the schemes were meant to resolve, but failed to do so, since increased market oversight to guarantee environmental effectiveness increased transaction costs, while increased regulatory flexibility to promote efficiency resulted in widespread fraud.

The early attempts to introduce emissions trading gave way, in the 1990s, to the first “cap and trade” systems in the world: the Acid Rain Program (ARP), at the national level, and the Regional Clean Air Incentives Market (RECLAIM) and $NO_x$ Budget programs at the regional level. The experience with these systems clearly disproves the purported simplicity and low informational requirements of emissions trading. In reality, emissions trading systems are characterized by a level of complexity that exceeds even direct regulation, which makes monitoring harder and more expensive and discourages public participation. Similarly, claims made by emissions trading proponents regarding the possibility of it leading to increased emissions reductions do not resist the test of reality. In contrast, emissions trading schemes in the US were characterized by unambitious caps and rent distribution through free allocation of allowances. Since emissions reductions were far below what could have been achieved with regulation, the US government had to implement further regulations to counteract the detrimental effects of air pollution on human health, which ended up eroding the opportunities to trade in emissions markets to the point where these markets only exist in theory.

Given that the environmental performance of regulation and taxation in Europe was far superior to the performance of emissions trading in the US, it seems surprising that carbon trading was introduced in the Kyoto Protocol. Under Kyoto, not only are industrialized countries allowed to trade in carbon allowances but also an international offsetting scheme, the Clean Development Mechanism (CDM), was introduced that effectively allows these countries to increase their emissions beyond the global cap. Despite being opposed by a clear majority of countries, carbon trading was accepted in climate negotiations as a way to guarantee the US participation, and justified as a first step towards an increasingly stringent global climate regulation. But the US decided not to ratify the protocol anyway and global emissions actually increased in the first commitment period. Even worse, climate negotiations have not evolved towards binding and increasingly tight emissions commitments, but rather towards the expansion of carbon trading through the creation of new regional and national carbon markets and new offsetting mechanisms.

Experience with Kyoto's carbon trading has proved disappointing. The CDM has mostly financed investment projects that reduce emissions from trace gases with high Global Warming Potential (GWP), which are very profitable but do not contribute to sustainable development. It has also been plagued with the impossibility of proving additionality and the inevitable attribution of offset credits to investments that would have happened even without the CDM funding. The experience with the CDM is, therefore, a good illustration of the trade-off between cost-efficiency and sustainable development.

As for the biggest carbon market in the world, the EU Emissions Trading System (EU ETS), it resulted in negligible incentives for emissions reductions and a massive subsidy to large polluters. Since one major reason for the EU ETS failure to deliver a stable and significant carbon price is the systematic overallocation of allowances and since the free allocation represents an unacceptable windfall profit, some major reforms were approved for the post-2012 period that will purportedly address these issues. Benchmarking was to replace grandfathering as the method for free allocations, in order to attribute allowances based on environmental performance benchmarks instead of historical emissions. Auctioning was to gradually replace free allocations. Some of the most problematic offset credits, associated with industrial gas CDM projects, became unusable for compliance and only credits from Least Developed Countries can now be used. A “backloading” provision delayed auctioning a part of allowances until 2020, in order to temporarily remove some of the excess allowances from the market. But these reforms were insufficient to effectively address the problems that the EU ETS faces, in particular after the EU institutions backed down on their original proposals and approved a series of extensions and exemptions to the full auctioning rule. In short, not only did the European carbon market face the same problems of the US “cap and trade” systems but also showed how politically unrealistic it is to expect that a “polluter gets paid” system eventually evolves through internal reforms into the “polluter pays” system that it was supposed to be.

The history of Kyoto's carbon trading also shows how it faced new problems, namely regarding the quantification and commensuration of emissions, relative to US emissions trading schemes. While the latter regulate emissions from a single, easily measurable pollutant, emitted by a relatively small number of sources, the former regulates emissions from several pollutants, some of which cannot be measured with a low degree of uncertainty and emitted by a large and diverse group of sources. This raises the question of whether the “black box” of carbon commensuration that measures, values and abstracts carbon emissions and thus supports carbon trading is based on accurate data or rather ignores inaccuracies and uncertainties. Analysing the scientific literature leads to the latter conclusion.

Even though carbon trading's environmental effectiveness is supposedly assured by the emissions cap, the uncertainties regarding national carbon inventories are significant enough to make it impossible to unambiguously monitor compliance. These uncertainties were magnified by the use of the GWP as a commensurator of greenhouse gases, taking into account that its methodology relies on implicit political assumptions, namely on the relevant time horizon and metric. The inclusion of land-use emissions and removals in carbon inventories further aggravated difficulties in monitoring emissions, since the data these calculations rely on is plagued by uncertainties and inaccuracies. On the other hand, attributing the same value to all carbon emissions ignores important scientific facts regarding the role of co-pollutants in climate change and the difference in permanence between fossil and land carbon. Furthermore, deep uncertainties regarding possible future states of societies cannot be mitigated through advances in scientific knowledge and technology.

If carbon emissions from some activities and sectors cannot be measured with a high degree of accuracy, then a carbon trading system that includes all carbon emissions, like the Kyoto system with its "comprehensive approach" will inevitably be based on erroneous data, which negatively impacts its effectiveness and has significant distributional effects. But the processes of commensuration and abstraction that are present in carbon trading raise more significant issues, that cannot be addressed with more accurate data, since they relate to how these processes reconstruct reality and render invisible or irrelevant information on the emissions "how, where and when". Using a critical literature review as a starting point, these issues were grouped in four normative critiques of carbon trading, exposing its ineffective, unjust, undemocratic and unethical nature.

Carbon trading is *ineffective* because it ignores the role of carbon "lock-in" and the need to positively discriminate some relatively costly investments to overcome fossil fuel dependence, disregards the contribution of co-pollutants to climate change and local pollution problems, does not distinguish between real and "paper" emissions reductions and is not compatible with an ambitious emissions cap and/or ambitious climate policies. Carbon trading is *unjust* because it rations access to carbon (across individuals, firms and/or countries) according to relative income, subsidizes polluters through free allocations and conflict with future generations needs through banking. Carbon trading is *undemocratic* because it not only fails to avoid obstacles to public participation that are present in environmental regulation but also hands decisions on climate policy to polluters and carbon market actors, with accountability to consumers and citizens being inevitably lost. Finally, carbon trading is *unethical* because it turns wrongs into rights, undermines the sense of shared responsibilities in climate mitigation, privatizes the atmosphere and values the environment in market terms.

Not only are these critiques, for the most part, not dependent on how carbon markets are designed but also the possibility of addressing some of the problems with internal reforms is precluded by the very functioning of carbon markets. Even if some reforms address some of the problems without creating new ones, their political feasibility is always questionable, considering that they always imply restrictions to trade and the concomitant increase in transaction costs and decrease in cost-efficiency. Furthermore, reformism, taken seriously, would inevitably erode and kill carbon markets, as opportunities to trade are constrained by increasing restrictions.

To sum up, then, carbon trading has failed in all of its purported objectives, namely deliver larger emissions reductions, compared to regulation, provide a stable carbon price that would push the economy towards decarbonization, simplify climate policy, increase democratic participation, make polluters pay for their pollution and, in international politics, assure the Kyoto Protocol's ratification by the US. On the flip side, carbon trading was successful in “marginalizing certain types of futures and actors” (Lohmann 2005: 230), namely the type of technological and institutional innovations, individual, communitarian and corporate behaviour changes, government regulations and social actions that ease the transition towards a low-carbon development path.

Not only are actually existing carbon markets very different from the textbook ones but carbon trading, even in theory, fails to provide an effective, just, democratic and ethical climate policy. While a carbon trading proponent can, at this point, ask about alternatives, we argue instead that carbon trading is itself an alternative to both direct regulation and taxation and that the question that must be asked is rather whether or not it should be a part of the policy-mix. The present thesis, naturally, concludes that the answer to this question is a clear “no”, given that carbon trading is ineffective, imposes unacceptable social costs and conflicts with other climate policies.

The question that climate policy theory has to answer, then, is what policies are worthy of being pursued and what policies are not. To put it in other words, the most important question is what policies should be on the menu, considering the social values that they foster or hinder. Answering this question is beyond the scope of this thesis, but obvious candidates include feed-in tariffs to support renewable energy, an environmental tax reform that does away with fossil fuel subsidies and other dispositions incompatible with climate mitigation, public investment in energy efficiency and conservation, improved regulation to cut down on packaging waste and increase the lifetime of consumer goods, increased waste recycling, agricultural policy reform to promote agroecological modes of production, direct regulation of polluting industries through emissions standards and transport policies that favour public transport and non-polluting modes of transportation. At the international level, financing climate mitigation and adaptation in less developed countries should be separated from market mechanisms and rather be based on mechanisms of aid and cooperation, namely by foreign debt cancelation and technology transfer.[20]

These policies can and should be implemented in a general policy framework that respects value pluralism and acknowledges conflicts between incommensurable values, which is not compatible with market-based policies that take the minimization of a narrow definition of costs as the overriding societal goal to which all others must pay tribute. The justification for these policies rests on a false ends-means dichotomy, which must be rejected in favour of policy-making that continuously adjusts means to ends and ends to means. Similarly, the false distinction distinction between “command and control” policies and “free markets” should be questioned, considering that creating a market through government regulation is no less of an intervention in existing markets than direct regulation is. New markets in renewable energy credits or energy efficiency credits, as well as new markets in environmental services, such as biodiversity offsets or water quality allowances, have to be created and tightly regulated by governments, in order to reduce the opportunities for fraud, manipulation and corruption that they create, as well as the ones that already existed in previous regulatory schemes. The experience with emissions trading is enough to show how, far from being a simple and efficient alternative to regulation, these markets are complex and can generate significant social costs.

In short, emissions trading fails to be a satisfactory alternative when dealing with pollution, particularly in the case of carbon emissions. Dropping the existing emissions trading schemes would not lead to inaction but rather open the way for better alternatives. Considering the urgency of climate change and the high cost of carbon trading's “cost-efficiency”, the never-ending effort to fix a broken system is no longer an option. We can do better by re-centring environmental policy on the same social values that emissions

20 Some of these policies have been proposed by civil society organizations that are critical of carbon trading (Clifton 2010; Coelho 2012; Reyes 2014).

trading sidelines, which precludes applying “cap and trade” or offsetting schemes to environmental policy that create new financial markets and allow polluters to evade regulations.

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