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Using a Policy Mix to Combat Climate Change – An Economic Evaluation of Policies in the German Electricity Sector

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Using a Policy Mix to Combat Climate Change

An Economic Evaluation of Policies in the German Electricity Sector

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To Arthur

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Executive Summary

Strategies to mitigate climate change are based on a policy mix in many industrialized countries. In order to reduce greenhouse gas emissions, a complex portfolio of policies, such as emissions trading, taxes, support schemes for renewable energies, energy-efficiency standards and voluntary agreements, is employed. Within this mix, policies may complement each other. However, there is also the risk of adverse interactions between policies. Consequently, the question arises whether existing mitigation strategies represent an efficient policy mix – or rather a suboptimal policy mess. Economic answers to this question are rare and insufficient. This book makes three important contributions to improving the economic understanding of a climate policy mix:

Firstly, this book clarifies under which general conditions a policy mix for pollution control may be more efficient than a single-policy strategy. It develops an innovative framework to organize the diffuse body of existing policy mix studies. The framework introduces transaction costs into the analysis of pollution problems and of public governance structures, i.e. regulation, to solve these problems. It understands the existence of a pollution problem not only as a market failure – as it is common in neo-classical economics – but more generally as the failure of private governance structures, such as the market, the firm or bilateral bargains. Based on this framework, two overarching rationales for using a policy mix for pollution control are derived: (1) A pollution externality may be reinforced by other failures of private governance structures, such as technological spillovers or asymmetric information. In this case, one policy is needed for each failure. (2) Single policies to cope with a pollution externality may bring about prohibitively high transaction costs. A policy mix may then reduce transaction costs while providing for a similar level of pollution control. With the identification of these two general rationales, this book sets a clear counterpoint to the array of economic studies assuming that a single pollution problem can always be addressed more efficiently by a single policy than by a policy mix.

Secondly, the book overcomes the high degree of abstraction in economic policy mix research. Simplified assumptions about policy design in existing models are substituted by more complex design characteristics as they can be found for actually existing policy mixes. Thereby, the book approaches economic theory of a policy mix to real-world conditions and increases its relevance for actual policy-making. It is shown that the consideration of complex real policy design may deliver results and policy recommendations that are contrary to those derived under a simplistic model. Two selected policy combinations are considered in particular: an emissions trading scheme combined with a feed-in tariff for renewable electricity generation, and with a tax on emissions or output.

Economic theory suggests that the combination of emissions trading and a feed-in tariff is justified when a pollution externality is reinforced by spillovers related to learning-by-doing. Nevertheless, it is demonstrated in this book that the expected dynamic welfare gains of such a policy mix may be impaired by inefficient interactions when two characteristics of real feed-in tariff design are taken into account: the payment of the tariff irrespectively of the prevailing electricity price and the funding of the tariff by an add-on to the electricity price.

Existing economic analyses also point out that the combination of emissions trading and a tax is likely to be inefficient in mitigating climate change when the incentives of both policies overlap, i.e. when firms have to hold an allowance and pay a tax in addition for each unit of emission. The resulting conclusion is usually to abolish the tax. However, it is emphasized in this book that this conclusion may be flawed in the real world where taxes address a variety of policy objectives and criteria, such as raising fiscal revenues or promoting equity. Given these restrictions, this book addresses three important issues, which have been neglected so far: (1) it identifies factors that drive the actual extent of inefficient abatement under a policy mix of emissions trading and a tax, (2) it shows that the policy mix is in fact likely to reduce this inefficiency compared to a single suboptimal tax, and (3) it proposes modifications of emissions trading design which may bring down welfare losses under the policy mix even further. These new insights are necessary for the careful evaluation and design of a policy mix in the presence of multiple policy objectives and criteria.

The *third* important contribution of this book is the evaluation of the climate policy mix in the German electricity sector – a case study that is very representative for the industrialized world. This evaluation can claim to be the most extensive analysis which is available for an existing policy mix. It applies the theoretical findings made in this book but also goes beyond them by considering further details of policy design and institutional environment. Particular attention is paid to the question which policies should complement emissions trading. It is shown that the additional implementation of energy efficiency labelling and low-interest loans promoting technological innovation and diffusion clearly raises efficiency. Support schemes for renewable energies and combined heat and power plants as well as energy-efficiency standards are likely to also produce welfare gains when combined with emissions trading. However, certain design features of these policies may impair the overall efficiency of the policy mix. Finally, it is argued that only electricity taxation and voluntary agreements cannot be justified as components of a policy mix. In summary, the German case study thus confirms that there may be many instances in which a policy mix is needed – but also that the details of policy design matter for the overall performance of the policy mix.

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Abbreviations

BAT	Best available technology
C	Celsius
CAFÉ	Corporate average fuel economy
CDM	Clean Development Mechanism
CE	Conformité Européene
CFCs	Chlorofluorocarbons
CH_4	Methane
CHP	Combined heat and power
CHPD	EU Directive on the promotion of cogeneration
CO_2	Carbon dioxide
COP	Conference of the Parties
DEHSt	Deutsche Emissionshandelsstelle (German Emissions Trading Authority)
DoE	U.S. Department of Energy
EBPG	Energiebetriebene-Produkte-Gesetz (German Act on Energy-using Products)
EDD	EU Eco-Design Directive
EEG	Erneuerbare-Energie-Gesetz (German Renewable Energies Act)
EEX	European Energy Exchange
ELD	EU Energy Labelling Directive
EMAS	EU Eco-Management and Audit Scheme
EnVHV	Energieverbrauchshöchstwerteverordnung (German Maximum Energy Consumption Ordinance)
EnVKG	Energieverbrauchskennzeichnungsgesetz (German Energy Consumption Labelling Act)
EnVKV	Energieverbrauchskennzeichnungsverordnung (German Energy Consumption Labelling Ordinance)
ETD	EU Energy Taxation Directive

ETS	EU Scheme for Greenhouse Gas Emissions Trading
ETSD	EU Directive establishing a scheme for greenhouse gas emission allowance trading
EU	European Union
GDP	Gross domestic product
GHG	Greenhouse gas
GWh	Gigawatt hour
HFCs	Hydrofluorocarbons
IEA	International Energy Agency
IECP	Integrated Energy and Climate Programme
IET	International Emissions Trading
IPCC	Intergovernmental Panel on Climate Change
ISO	International Organization for Standardization
JI	Joint Implementation
KfW	Kreditanstalt für Wiederaufbau (German Reconstruction Loan Corporation)
kW	Kilowatt
kWh	Kilowatt hour
KWKG	Kraft-Wärme-Kopplungsgesetz (German Combined Heat and Power Act)
LR	Learning ratio
MW	Megawatt
MWh	Megawatt hour
N_2O	Nitrous oxide
NAP	National Allocation Plan
NO_X	Nitrogen oxide
OECD	Organisation for Economic Co-operation and Development
PFCs	Perfluorocarbons

ppm	Parts per million
PR	Progress ratio
R&D	Research and development
RECLAIM	Regional Clean Air Incentives Market
RESD	EU Directive to harmonize the promotion of electricity produced from renewable energy sources across Europe
SF_6	Sulphur hexafluoride
SO_2	Sulphur dioxide
StromStG	Stromsteuergesetz (German Electricity Tax Act)
TEHG	Treibhausgas-Emissionshandelsgesetz (German Greenhouse Gas Emissions Trading Act)
TDR	Tradable development rights
UBA	Umweltbundesamt (German Environmental Protection Agency)
UIP	Umweltinnovationsprogramm (German Environmental Innovation Programme)
UK	United Kingdom
UN	United Nations
UNFCCC	United Nations Framework Convention on Climate Change
US	United States
US$	United States Dollars
USA	United States of America
VOC	Volatile organic compounds
WTO	World Trade Organization
ZuG	Zuteilungsgesetz (German Act on the National Allocation Plan)

1 Introduction

1.1 Policy Choices for Pollution Control: Policy Mix or Policy Mess?

Since the 1970s, pollution problems have more and more drawn the attention of the public. Pollution control has become a major concern of government regulation. In the beginning, control strategies employed by governments used to be based mainly on command-and-control policies. Polluters were required, for example, to meet emission discharge limits or to use certain specified kinds of technology (Harrington et al. 2004, p. 1). During the last two decades, however, these policies have been supplemented by a variety of new approaches. These include market-based policies, such as taxes, subsidies or tradable allowance systems, as well as voluntary agreements or information measures.[1] As a result of this development, a so-called policy mix is nowadays often employed to address a single pollution problem. Policy mixes can be observed in many OECD countries and for many pollution problems including air and water pollution, waste disposal and, most outstandingly, climate change (for overviews, see Gunningham and Grabosky 1998; OECD 2007).[2]

A striking example of a policy mix is the German strategy to reduce greenhouse gas (GHG) emissions from electricity generation. The most important component of this policy mix is the European Union's (EU) Emissions Trading Scheme (ETS). Yet, it is complemented by a variety of other EU and domestic measures such as an electricity tax, feed-in tariffs for renewable energy sources, voluntary agreements to reduce GHG emissions, energy-efficiency standards for domestic appliances, electricity labeling and many more. In fact, similar strategies are employed in many other EU Member States as well (for an overview, see Sorrell et al. 2003).

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[1] In this book, the term "policy" is used synonymously for the terms "policy instrument" or "instrument", which are also often employed for measures such as taxes, tradable allowance systems or command-and-control approaches. Correspondingly, the term "policy mix" is preferred to "instrument mix".

[2] Throughout this book, a broad definition of pollution is employed. In this respect, pollution refers to the release of pollutants into the environment. It causes instability, disorder, harm or discomfort to the ecosystem, i.e. physical systems or living organisms, including humans. Pollutants can be chemical substances or energies, such as noise, heat, or light energy. Many of these substances or energies are naturally occurring. In this case, they are considered pollutants when they exceed natural levels. This definition includes classical pollutants such as sulphur dioxide, nitrous oxides, heavy metals or hazardous wastes. It also includes greenhouse gases. These have been found responsible for climate change, which may eventually impair ecosystem functions in many regions of the world.

However, the more policies are implemented for solving one pollution problem, such as climate change, the more probable are interactions between these policies. These interactions may affect the performance of pollution control policies negatively as well as positively. On the one hand, policy mixes exhibit risks. Policies may interfere with one another and produce adverse incentives if applied in parallel (Sorrell et al. 2003, p. 1). On the other hand, policy mixes may also open up opportunities. Multiple policies may reinforce each other and compensate for drawbacks of single-policy strategies (Gawel 1991, p. 56). Consequently, the question arises whether a policy mix has been designed such that it makes use of synergies and avoids contradictions between the different policies – or whether one has to fear "[…] that the policy mix will degenerate into a policy mess" (Sorrell et al. 2003, p. VI).

Among representatives of political parties and firms, there has been a lively debate in recent years on how to answer this question. Regarding Germany's climate and energy policy mix, statements have been ambiguous. The German Association of Electricity Producers blamed the German Government for promoting "climate protection with too many, inconsistent policies" (FR 2005). Wulf Bernotat, chief executive officer of Germany's largest energy producer E.ON, similarly emphasized: "The key questions are: Is the [climate policy] tool kit appropriate? Do the different policies fit together? I personally think that a single emissions trading scheme is sufficient to attain the environmental goals" (Handelsblatt 2005). Correspondingly, Germany's then-Minister of Economic Affairs, Michael Glos, announced in 2006 to "revise the energy and environmental policy mix impartially with respect to efficiency and consistency" (FAZ 2006). Recently, even members of Germany's Green Party – the impetus behind many of Germany's climate and energy policies – doubted that it was necessary to promote renewable energies by a feed-in tariff with an emissions trading scheme being in place (Waldermann 2009). In contrast, other politicians have argued in favour of a policy mix. Reinhard Loske, former Member of Parliament for the Green Party, highlighted: "There is no perfect policy to address all of our problems. What we need is a pluralism of policies" (Loske 2005, p. 45).

The debate in the political arena calls for scientific contributions regarding the evaluation of a policy mix. From an economic point of view, efficiency should be a decisive criterion to guide the debate. In this respect, the key question would be whether the policy mix provides for achieving an optimal level of pollution control at least cost. However, economic research has paid little attention to the analysis of policy mixes so far (Sorrell et al. 2003, p. 30). Traditional studies in Environmental Economics have focused on the use of single policies to correct for negative pollution externalities. They either analyzed the performance of single policies or compared two or more policies for pollution control. A common finding was, for example, that uniform command-and-control approaches provide for an inefficient resource allocation, and are therefore inferior to market-based policies (see, e.g., Stavins (2000)). In this respect, the evaluation of a policy mix does not only represent a political issue. It also poses an important challenge to economic research. Facing the political trend towards the use of policy mixes – and in a general economic interest – economists should not only ask: "How beneficial is one policy?", but rather: "How beneficial is the entire policy mix?" (Loske 2005, p. 45). Systematic economic research on this issue is still rare.

1.2 Climate Change as a Major Problem of Pollution Control

Implementing a consistent set of policies is particularly important to cope with one of the major and most urgent problems mankind is facing at the moment: climate change. According to the Intergovernmental Panel on Climate Change (IPCC), climate change is characterized by "a change in the state of the climate that can be identified (e.g. using statistical tests) by changes in the mean and/or the variability of its properties, and that persists for an extended period, typically decades or longer. It refers to any change in climate over time, whether due to natural variability or as a result of human activity" (IPCC 2007a, p. 30).

In its Fourth Assessment Report published in 2007, the IPCC once again confirmed that climate change is happening (IPCC 2007c, pp. 5-9). There was strong evidence of global warming of the climate system. Many changes in the properties of climate were found to be likely (probability of occurrence larger than 60 percent) or even very likely (probability of occurrence larger than 90 percent). Global surface temperatures had increased by an average of 0.13 °C per decade over the last fifty years. The warmth of this period was found to be unusual in at least the previous 1,300 years. Eleven of the twelve years from 1995 to 2006 ranked among the twelve warmest years in the instrumental record of global surface temperature. Polar ice caps and glaciers had melted. Ocean temperature and sea level had risen. Precipitation patterns had changed leading to increased rain and snow fall in areas such as Northern Europe, eastern parts of North and South America and parts of southern Asia. At the same time drying had occurred in other areas including the Mediterranean, the Sahel and southern Africa. The intensity and quantity of extreme events such as heavy precipitations, droughts, heat waves and tropical cyclones had increased.

The IPCC report highlighted that human activity had a significant impact on climate change. In fact, "most of the observed increase in global average temperatures since the mid-20th century is very likely due to the observed increase in anthropogenic greenhouse gas (GHG) emissions" (IPCC 2007c, p. 10). Emissions of GHGs including carbon dioxide (CO_2), methane (CH_4), nitrous oxides (N_2O), hydrofluorocarbons (HFCs), perfluorocarbons (PFCs) and sulphurhexafluoride (SF_6) reinforce the natural, so-called "greenhouse effect". GHGs stocked in the Earth's atmosphere provide that the atmosphere lets through high-energy solar radiation but traps part of the longer wave heat radiation reflected from the surface. This process results in the warming of the Earth's surface (Grubb et al. 1999, pp. 3-4).

Future projections on climate change are subject to considerable uncertainty. They crucially depend on the underlying scenario regarding the future development of anthropogenic GHG emissions and their concentration in the atmosphere. Nevertheless, even if GHG concentrations had been kept constant at year 2000 levels, a further warming of about 0.1 °C per decade would be likely to occur (IPCC 2007c, p. 12). This is due to the long time scales associated with many climate processes and feedbacks (IPCC 2007c, p. 16). More importantly, continued GHG emissions at or above current rates are very likely to induce climate changes during the 21st century that are larger than those observed during the 20th century. The best estimates for global warming for the end of the 21st century compared to the 1980 to 1999 period vary from 1.8 °C to 4.0 °C (IPCC 2007c, p. 13). Correspondingly, a rise of the sea level, changes in precipitation patterns and an increased incidence of extreme events are likely or even very likely (IPCC 2007c, p. 8).

It is likely that (regional) climate change has had and will have a discernable impact on physical and biological systems. Consequently, the human environment will experience significant changes. As climate change itself, the extent of its possible impacts is subject to considerable uncertainty. It also strongly depends on the underlying emissions scenarios and the resulting temperature increase. Moreover, impacts are also driven by non-climate stresses, such as air and water pollution, as well as socio-economic development. In addition, effects may vary between regions (IPCC 2007d, pp. 8-10). Nevertheless, there are some overarching trends. Water availability is likely to increase in high latitudes and in some wet tropical areas. In turn, it decreases in mid-latitudes and semi-arid low attitudes exposing hundreds of millions of people to increased water stress. Many plant and animal species are likely to face an increased risk of extinction. Crop productivity is expected to increase slightly in mid and high latitudes. In low latitudes it is likely to decline for even small temperature decreases raising the risk of hunger. Coastal areas are very likely to incur increased damages from floods and storms. There will be a higher burden resulting from malnutrition, diarrhoeal, cardio-respiratory and infectious diseases. Moreover, morbidity and mortality from heat waves, floods and droughts are likely to increase (IPCC 2007d, p. 16). Overall, aggregated and discounted annual costs of climate change are very likely to exceed the corresponding benefits. In general, it can be assumed that the developing world will be affected more strongly than the industrialized world (IPCC 2007d, p. 17).

The IPCC report reveals that sound strategies are needed to combat climate change and its effects. Past emissions are estimated to bring about some unavoidable warming. Consequently, adaptation to climate change will be necessary to cope with the related impacts. The need for adaptation measures will further increase if GHG emissions remain at current levels or increase (IPCC 2007d, p. 19). Unmitigated climate change is likely to exceed the capacity of natural and human systems to adapt. Therefore, measures to mitigate GHGs have to be an indispensable component of future strategies to cope with climate change (IPCC 2007d, p. 20). Stern (2008, p. 6) highlights that risk analysis calls for stabilizing aggregate GHG concentration below 550 parts per million (ppm) CO_2 equivalents. The current concentration of GHG in the atmosphere amounts to 430 ppm (Stern 2008, p. 4). To attain a stabilization level of 550 ppm, CO_2 emissions would have to be reduced globally by at least 30 percent, and probably by 50 percent, by the year 2050 compared to 2000 emissions levels (Stern 2008, p. 3). It is generally assumed that industrialized countries will have to undertake larger emission reductions than developing countries. In fact, they may be required to bring down their emissions by up to even 80 percent (Stern 2008, p. 28). Nevertheless, the corresponding stabilization level would still provide for a global mean temperature increase of 2.8 to 3.2 percent above pre-industrial levels (IPCC 2007b, p. 15). Mitigation costs to attain this concentration level would be substantial. The IPCC (2007b, p. 18) as well as Stern (2008, p. 7) estimate that they may amount to a reduction in the global gross domestic product (GDP) of around one percent. However, these GDP losses may be small compared to the gains from reducing emissions. Stern (2008, p. 21) estimates that, relative to a business-as-usual scenario, stabilizing GHG concentrations at 550 ppm may avoid damages from climate change in the amount of nine to ten percent of global GDP. These figures illustrate that mitigation measures – particularly in industrialized countries – are highly important and likely to produce overall welfare gains. However, with current mitigation policies, global GHG emissions will still continue to grow over the next few decades (IPCC 2007b, p. 4). Thus, existing global, national and regional mitigation strategies have to be revised. They have to be modified such that they provide for significant emissions reductions. The IPCC (2007b, p. 19) highlights that there is variety of mitigation policies available. However, "their applicability depends on the national circumstances and an understanding of their interactions". In order to minimize the corresponding economic burden for societies, mitigation strategies have to be designed such that policies complement each other efficiently and that negative interactions are avoided.

1.3 Importance of the German Case Study

In this book, the economic analysis of a policy mix to combat climate change is applied with reference to policies in the German electricity sector. This case study is chosen for two important reasons. Firstly and most importantly, the climate policy mix in the German electricity sector is highly representative for many OECD countries. The policy mix encompasses the entire range of policy types. It includes market-based policies, command-and-control approaches, voluntary agreements and information measures. Most of the policies set up in Germany, such as the EU Emissions Trading Scheme, the electricity tax and the schemes promoting the use of renewable energy sources and energy-efficient equipment, have been initiated at the EU level. That is, policies are implemented in an equal or similar manner in all other EU Member States as well. What is more, EU and German policies have served as an example for climate strategies in many countries outside the EU. The EU Emissions Trading Scheme was the first tradable allowance system for CO_2 emissions in place worldwide. In the meantime, similar schemes have been implemented in Switzerland, New Zealand and the north-eastern US States. They are planned for Japan, Australia and California (European Commission 2008g, p. 25). Likewise, the German scheme to promote electricity generation from renewable energy sources has been "exported" to more than 30 countries worldwide (BMU 2007d, p. 10). In addition, many countries outside the EU have also adopted further mitigation measures, such as voluntary agreements to bring down emissions or policies to increase energy efficiency (for US examples, see Kruger 2005; Bennear and Stavins 2007, p. 113). These examples illustrate that the policy mix employed in Germany can be found in many other parts of the industrialized world as well. A thorough examination of the German climate policy mix therefore yields analytical results that are not only restricted to the German case. Many of the findings can be generalized and transferred to other country contexts. They provide new and useful guidance for designing climate strategies throughout the world.

Secondly, an economic analysis of the climate policy mix in the German electricity sector is of high relevance due the importance and urgency of efficient mitigation measures in this sector. Germany is by far the largest emitter in the EU and the sixth-largest emitter worldwide (Ziesing 2007c, pp. 66-67). In 2007, Germany's CO_2 emissions amounted to 841.2 million tons. The energy sector is the largest emitter in Germany. It accounted for roughly 46 percent of Germany's total CO_2 emissions in 2007 (Ziesing 2009, p. 68). Out of these emissions, some 85 percent can be assumed to stem from the production of electricity (Ziesing 2007a, p. 85). Thus, electricity generation is responsible for more than one third of Germany's CO_2 emissions. Emissions in the energy sector have been reduced significantly in the last two decades. 2007 CO_2 emissions were by more than seven percent below 1990 levels (Ziesing 2009, p. 68). However, this reduction can be attributed above all to the modernization and decommissioning of old lignite power plants in East Germany after the German reunification in 1990 (Schleich et al. 2001, p. 369). In fact, CO_2 emissions of Germany's energy sector have started to rise again in recent years. Between 2000 and 2007, an increase by 11 percent was recorded (Ziesing 2009, p. 68). This development is despite the fact that there are many mitigation technologies for the energy sector that are already now commercially available. In this respect, the IPCC (2007b, p. 10) cites a variety of supply- and demand-side measures such as improved plant efficiency, fuel switching from coal to gas, the use of nuclear power and renewable energy sources as well as consuming electricity more efficiently in the household and industry sectors. A McKinsey report estimates that mitigation on the supply side may reduce CO_2 emissions in Germany by 20 megatons by 2020 compared to baseline scenarios at costs that are below 20 Euro per ton of CO_2. This potential increases to 75 megatons if measures with marginal abatement costs of up to 50 Euro per ton are considered (McKinsey 2007, p. 32). Demand side abatement potentials are even larger and cheaper. In the industry sectors, energy-efficiency improvements may bring about a CO_2 emissions reduction of 30 megatons by 2020 at zero or negative costs for decision makers (McKinsey 2007, p. 34). Increasing energy efficiency in the building sector may reduce CO_2 emissions by 65 megatons by 2020 at zero cost (McKinsey 2007, p. 37).

The increase of emissions in Germany's electricity sector in the presence of high mitigation potentials casts a shadow on the policies implemented currently. It becomes obvious that despite its diversity the policy mix does not provide for necessary emission reductions in the electricity sector. Questions have to be raised regarding the suitability of the policy mix to combat climate change. Are policies actually consistent? Are additional policies necessary? Could a better result be attained with less policies maybe? Does the policy mix bring emitters on track for further, even more rigorous emission reductions in the future? Answering these questions is of relevance for Germany – and many of the other industrialized countries using a similar policy mix. The coordination of mitigation policies is a decisive precondition for attaining ambitious emission reduction targets in Germany and worldwide.

1.4 Research Objectives

The principal objective of this book is to further develop and advance the economic insight on interactions between climate policies when these are implemented in parallel. Despite the focus in Environmental Economics on single-policy problems, this book can build on a variety of economic studies which have made pioneer contributions to the understanding of a policy mix for pollution control. These studies can be roughly divided into two strands: On the one hand, there are analyses which highlight that the combination of certain policies may in fact increase the efficiency of pollution control compared to single-policy strategies. On the other hand, several studies also reveal the inefficiency brought about by interactions between policies.[3] Both strands are usually subject to two important restrictions: First of all, studies usually refer to a very specific context. They analyze the combination of selected policies for a specific pollution problem, such as climate change, air pollution or waste. In addition, analyses are carried out under quite different assumptions about the real world, e.g. the behaviour of individuals and the functioning of markets. Thus, studies are conducted against a variety of theoretical and empirical backgrounds. They do not reveal general arguments in favour or against the use of a policy mix, which could be used for the evaluation of an existing policy mix. Secondly, studies are often realized on a very abstract, theoretical level. They are based on assumptions which do not represent the real world. An important restriction in this respect is that they usually analyze the combination of policies with a simplified design. For example, they consider a policy mix of a perfect emissions trading scheme and a simple emissions tax which are imposed on the entire industry. In reality, however, policy designs often deviate from these ideals. This may be due to various reasons. Policy design is usually the result of political negotiations between stakeholders (see, e.g., Kirchgässner and Schneider 2003). Moreover, policies are not implemented in a regulatory vacuum but have to be embedded into an existing institutional framework (see, e.g., Gawel 2005a). Consequently, policy design is much more complex in reality than in theoretical analyses. Due to these restrictions, existing analyses are of limited use when it comes to evaluating an actually existing policy mix and developing corresponding policy recommendations. Therefore, both restrictions provide the starting point for developing the economic analysis of a policy mix.

The book aims to advance economic theory related to a policy mix in two ways: The first objective of the book is to identify overarching economic rationales for using a policy mix for pollution control. The analysis is to reveal general conditions under which a policy mix provides for a better outcome in terms of efficiency than a single policy. Thereby, the book aims at overcoming the confusion in economic theory surrounding the use of a policy mix. It is meant to set a clear counterpoint to the array of economic studies assuming (implicitly) that a single pollution problem can always be addressed more efficiently by a single policy than by a policy mix. The rationales can be used as a guideline for designing and evaluating real-world pollution

[3] The corresponding literature will be reviewed and discussed in subsequent chapters.

control strategies. If one or more rationales can be applied for a certain policy context, there is a strong indication that a policy mix can be justified on efficiency grounds – and should be preferred to single-policy approaches.

However, applying theoretical rationales to an empirical problem has to be done carefully. This caution is particularly important due to the degree of abstraction of many existing studies on policy mixes. It is therefore the second outstanding objective of this book to approach economic theory on policy mixes to reality. Simplified assumptions about policy design are substituted by more complex design characteristics as they can be found for actually existing policy mixes. This modification of existing policy mix studies is inspired by the striking example of Germany's climate policies in the electricity sector. The development of economic theory is therefore guided by the empirical policy problem. With this approach, the book intends to increase the relevance of economic theory related to policy mix research for actual policy-making.

With respect to this second objective, two sub-goals are posed for research in this book: On the one hand, selected combinations of policies are considered which can be theoretically justified on the basis of an economic rationale. In this case, the sub-goal is to assess whether the theoretical efficiency properties of a policy mix still hold when important deviations between the theoretically recommended policy design and the actual characteristics of the policy mix are taken into account. Under certain conditions, theoretical studies may recommend a subsidy as part of a policy mix, for example. In reality, however, such a subsidy may be characterized by a complex design. It may be granted to some economic actors only, at differentiated rates. Moreover, it may not be paid by the government but by other market participants. This book aims at evaluating whether such deviations between theoretical and empirical policy design impair the theoretically predicted efficiency of the policy mix or not.

On the other hand, light is shed on selected examples of a policy mix which cannot be justified on efficiency grounds since no rationale applies. In this case, one could argue that the policy mix represents a redundancy at best and an inefficient policy mess at worst. The straightforward conclusion would be to substitute the policy mix by a single policy strategy. A more profound understanding of the policy mix would not be needed. This conclusion would assume that policy-making is one-dimensional, i.e. that the policy mix is only meant to reduce GHG emissions efficiently. In reality, however, a much more in-depth analysis of the policy mix is necessary. Policies in the policy mix may have been implemented to meet other goals and criteria as well. Fulfilling these requirements may trade off the inefficiency in reducing GHG emissions. In this case, assessing whether or not the climate policy mix is efficient is not sufficient. Rather it is necessary to determine the actual extent of the inefficiency under the policy mix. Again, this evaluation cannot be carried out on an abstract, theoretical level, as it is employed in many existing studies. Therefore, the second sub-goal is to assess whether certain characteristics of policy design ameliorate or deteriorate the inefficiency of the actually existing policy mix. The underlying task is to find a policy mix design that minimizes the inefficiency stemming from policy interactions. In this respect, the book aims at revealing drivers of the inefficiency in the policy mix as well as means to reduce it.

Apart from the two theoretical objectives, the book also pursues an important empirical research objective: It aims at using the advances in economic theory made in this book to derive a concrete evaluation of a policy mix for a specific policy context. In this context, the German climate policy mix in the electricity sector serves as a case study. This third research objective implies that theoretical insight has to be translated into recommendations for real politics. For this purpose, the complex characteristics of the German policy mix as well as its institutional environment have to be considered in great detail. Thus, the book is not limited to advancing economic theory. It also intends to build a bridge between economic theory and actual policy making. Due to its high degree of representativeness, using the German example has the appeal that the applicability of many of the results is not restricted to the German policy context but can be transferred to other country cases as well.

Thus, this book aims to make three contributions to the economic understanding of a policy mix. Two of these contributions represent important advances of economic theory. Firstly, an innovative framework of economic rationales shall clarify systematically what is already known about the economics of a policy mix for pollution control. Secondly, the integration of more complex policy design into economic models shall approach the theoretical analysis of a climate policy mix to real world conditions. Thereby, the book means to increase the relevance of economic theory for real policy problems. Thirdly, the book shall provide an extensive, applied evaluation of the climate policy mix in the German electricity sector and derive corresponding policy recommendations. Thereby, the analyses in this book are intended to provide interesting insight not only for economists but also for policy-makers in Germany and – due to the representative character of the case study – worldwide.

1.5 Procedure and Structure

The book is divided into seven chapters including the introduction. *Chapter 2* introduces the German case study. It presents the climate policy mix implemented for Germany's electricity sector in its breadth and depth. The focus is on policies influencing the behaviour of either electricity generators or electricity consumers, i.e. those affecting the supply or the demand side of the electricity market. This chapter illustrates the high degree of representativeness of the German case study. It points out that most of the German climate policies in the electricity sector are framed by EU-wide regulation on climate policies. Moreover, it highlights that the German policy mix encompasses the whole variety of policy concepts that are applied in countries throughout the industrialized world, including market-based policies, command-and-control approaches, market-based policies, voluntary measures and information programs. Thereby, it is demonstrated that the German case study serves as a very useful example of a policy mix as it can be found in many countries of the EU as well as worldwide. The presentation of the German case study also reveals the complexity of the policy mix existing in reality. It shows that neither policy perfectly corresponds to the simple theoretical concept. Rather, it is highlighted that policies are characterized by a multiplicity of objectives, varying scopes as well as differentiated incentives. This introduction of the German policy mix establishes the point of reference for advancing economic theory in subsequent chapters. Moreover, the profound description of the policy mix is the basis for the derivation of concrete policy recommendations for the German case study.

Chapter 3 provides an overview of the economic literature on policy mixes. It reviews studies which bring forward arguments in favour of using a policy mix for pollution control. These studies discuss circumstances under which a policy mix may result in a better outcome in terms of efficiency than a single policy. To guide the review, an innovative analytical framework is developed, which is unique in its approach and comprehensiveness. This framework allows going beyond the specific scope and context of single policy mix studies. It organizes this diverse strand of economic theory and reveals a set of overarching, general rationales for using a policy mix. For this purpose, the framework broadens the Neo-classical perspective on policy analysis by ideas from New Institutional Economics. It incorporates transaction costs into the analysis of pollution problems and possible policy solutions. It understands the existence of a pollution problem not only as a market failure – as it is common in Neo-classical Economics – but more generally as the failure of private governance structures, such as the market, the firm or bilateral bargains.[4] Using this framework, the first research objective of this book is met. It is revealed that

[4] The framework distinguishes between public and private governance structures. Public governance structures refer to government regulation of a pollution problem by a policy. Private governance structures refer to the solution of a pollution problem by the market, firms or bilateral bargains. This conceptualization is explained in detail in Section 3.2.

there are two overarching economic rationales for using a policy mix when one pollution problem is under consideration. On the one hand, a pollution externality may be reinforced by other failures of private governance structures, such as technological spillovers or asymmetric information. In this case, one policy is needed for each failure. On the other hand, single policies to cope with a pollution externality may bring about prohibitively high transaction costs. A policy mix may then reduce transaction costs while providing for a similar level of pollution control. These two rationales can be used as a rule of thumb when evaluating an existing policy mix. However, the review also reveals limitations of the policy mix literature and avenues for further research. Among other, it is pointed out that an outstanding restriction of studies consists in the commonly held assumption of simplified policy design. Thus, available policy mix studies cannot capture the complexity of real policy design – as it has been outlined for the German case study – and its possible implications on efficiency. Consequently, these studies are of limited use only for guiding policy-making.

Subsequent *Chapters 4 and 5* are devoted to addressing and overcoming this restriction. In order to keep the analysis simple, interactions between combinations of two policies are evaluated. The focus is on two important examples of a policy mix: The combination of emissions trading with a feed-in tariff for renewable electricity generation and a tax on emissions or output. At this step of analysis, it is assumed that these policies are only meant to reduce GHG emissions efficiently.[5] It is shown that existing policy mix studies analyzing these policy combinations are far too simplistic. They do not represent real-world policy design. The analysis in this book fills this important gap in the theoretical understanding of a policy mix. For both policy combinations, it is revised whether one of the rationales for using a policy mix can be applied or not. If this is the case, it is analyzed whether the theoretical advantages of the policy mix hold when the design of actually existing policies is taken into account. If no rationale applies, it is assessed which factors of the actual policy mix design drive the inefficiency of the policy mix and whether the design has been chosen such that the welfare loss is minimized. For these purposes, the simplistic representation of climate policy mixes in existing economic models is substituted by more complex assumptions about policy design. These modifications are inspired by the real-world case of the climate policy mix in the German electricity sector. With these modifications, the second objective of this book is addressed: The economic theory of a policy mix is significantly

[5] This assumption is relaxed in Chapter 6 when it comes to evaluating the German climate policy mix.

advanced by approaching the analysis to reality. This development is a substantial contribution to raising the suitability and relevance of economic thought for real-world policy problems.

The combination of emissions trading and a feed-in tariff is analyzed in *Chapter 4*. It is highlighted that this policy mix can in fact be justified by an economic rationale. If a pollution externality is combined with technological spillovers, an emissions policy as well as an output subsidy for technology production may be required. However, the design of a feed-in tariff, as it can be found in Germany, deviates from a pure output subsidy in several respects. Therefore, two important advances of policy mix models are made in this chapter: Firstly, remuneration of renewable electricity under the feed-in tariff system is not assumed to be paid in addition to but instead of the output market price. Secondly, funding of the subsidy is not faded out but made endogenous to the model. It is demonstrated that, with these modifications, the policy mix may also provide for an efficient outcome. However, policy design is more tedious than with a simple subsidy: The emissions policy as well as the feed-in tariff have to be adapted continuously.

Chapter 5 is devoted to the policy mix of emissions trading and a tax on emissions or output. It is found that this combination is an excellent example of a policy mix which cannot be justified on the basis of any of the rationales derived before. In this case, existing studies leave it to the rather general statement that the policy mix produces inefficiency if emissions taxation is heterogeneous, i.e. if participants of the emissions trading scheme have to pay different tax rates. Consequently, they plead for an abolition of the tax. This reasoning may be appropriate under the assumption that the tax is only meant to mitigate climate change efficiently. However, this reasoning falls short if actual policy design is taken into account. As is illustrated by the German case study, taxes may not only be meant to mitigate climate change efficiently but to address other policy objectives and criteria as well. Thus, the abolition of a tax may not always be desirable or feasible. Given this insight, this chapter makes three necessary contributions to the theoretical analysis of a policy mix. First of all, the most important drivers of the welfare loss under the policy mix are identified. This insight is required to estimate the actual extent of the inefficiency in addressing climate change, and to compare it with benefits of attaining other objectives of the tax. It is demonstrated that the inefficiency depends on the heterogeneity of taxation, the slope of the marginal abatement cost curves, the total number of firms and emissions and the emissions market shares of sectors with different tax rates. It is also shown that the welfare loss is aggravated when the tax is not paid on emissions but on output (which is a complement to emissions). Secondly, the analysis departs from the first-best perspective of existing policy mix studies. It compares the policy mix not only to an optimal single emissions trading scheme but also to a suboptimal single tax. It is shown that if the tax cannot be abolished, the implementation of an additional emissions trading scheme increases the efficiency of mitigating climate change. From this perspective, the policy mix thus appears as desirable. Thirdly, it is analyzed how welfare losses can be minimized within a policy mix. It is revealed that the inefficiency can be reduced by restricting trades between sectors with different tax rates.

Chapter 6 addresses the third research question posed for this book: It provides an applied economic evaluation of the actually existing climate policy mix in Germany's electricity sector. This analysis is outstanding in its comprehensiveness and depth – not only with respect to the German case study but also compared to other applied policy mix studies in general. Particular light will be shed on complementing the ETS with different types of policies. This is because the ETS can be considered the central mitigation policy for the German electricity sector. As is shown throughout this book, the ETS determines the overall cap on emissions. Complementing the ETS with other policies either increases or decreases the cost of attaining the cap. The evaluation considers the entire variety of policies implemented in Germany. The major focus of the evaluation is on whether or not the policy combinations under consideration address climate change efficiently. The evaluation strongly benefits from the theoretical advances made in previous chapters. Nevertheless, the applied analysis of the German policy mix also goes beyond the theoretical analyses in many respects. Further details of policy mix design and their implications for efficiency are discussed. Transaction costs under a policy mix are considered explicitly where appropriate. Moreover, the evaluation also takes into account that policies are often not only addressing climate change but other policy objectives as well. In addition, additional evaluation criteria next to efficiency, such as equity, and political economy considerations are included in the analysis. This remarkable breadth in analysis allows for an extensive evaluation of Germany's climate policy mix in the electricity sector, which has not been available to this extent so far. The evaluation culminates in the derivation of concrete policy recommendations. Thanks to the theoretical advances made in this book and the broad empirical discussion, the evaluation of the policy mix as well as the related recommendations promise to be very insightful for real-world policy-making.

Chapter 7 summarizes the major findings of the book. It emphasizes once more that the appeal of the theoretical and empirical findings made in this book lies in their transferability to the context of many other country cases and environmental problems. Moreover, restrictions of the analyses carried out and possible avenues for further research are highlighted.

2 Climate Policies in the German Electricity Sector

2.1 Introduction

This chapter introduces the German case study. It presents climate policies that have been implemented to address greenhouse gas emissions from the German electricity sector. This chapter aims at demonstrating the complexity of the German policy mix – but also its representativeness for other countries contexts. *Section 2.2* illustrates that German policies have been adopted against the background of international, European as well as German climate protection strategies. International and European agreements and regulations oblige Germany to take action against climate change. National targets related to climate protection are likewise challenging. *Section 2.3* is devoted to the detailed presentation of the complex climate policies in Germany's electricity sector, by which the ambitious national, European and international goals are to be attained. For each policy, the most important features – objectives, scope and rules – will be outlined. The focus will be on government measures that set incentives for private actors supplying electricity, i.e. electricity generators and distributors, and/or consuming electricity. Purely public activities, such as funding research and development in public research institutes, remain out of the focus of this presentation. Likewise, private initiatives, such as labelling electricity generation from renewable energy sources are not considered. *Section 2.4* summarizes with an outline of the main characteristics of the German policy mix. Chapter 2 provides the basis for the subsequent theoretical analysis of a policy mix as well as the applied evaluation of the climate policy mix in the German electricity sector.

2.2 International and National Background of German Climate Policies

2.2.1 International Climate Strategy

The most important treaty that forms the basis of the international climate strategy is the United Nations Framework Convention on Climate Change (UNFCCC) (UN 1992). It was adopted in 1992 during the Earth Summit in Rio de Janeiro. Germany ratified the Convention in 1993. The UNFCCC entered into force on 21 March 1994 (Oberthür and Ott 2000, p. 63). Currently, 191 countries and the EU are Parties to the Convention (International Institute for Sustainable Development 2008, p. 2).

The principal objective of the UNFCCC, as stated in Article 2, is to achieve the "stabilization of greenhouse gas concentrations in the atmosphere at a level that would prevent dangerous anthropogenic interference with the climate system". It is moreover pointed out that this objective is to be attained "within a time frame sufficient to allow ecosystems to adapt naturally to climate change, to ensure food production is not threatened and to enable economic development to proceed in a sustainable manner".

The UNFCCC emphasizes the principle of "common but differentiated responsibility" (Article 3). It is therefore the developed countries that have to take the lead in preventing and mitigating climate change and its adverse effects. The specific needs of developing countries are to be considered. Article 4 requires all Parties to establish and publish a national inventory of anthropogenic emissions by sources and removals by sinks of greenhouse gases. In addition, each Party has to adopt national programmes to mitigate climate change by reducing emissions and enhancing sinks and to facilitate the adaptation to climate change. All developed countries including Germany and most countries from central and Eastern Europe and the former Soviet Union (the so-called Annex I countries) agree on the indicative aim of reducing their greenhouse gas emissions individually or jointly to 1990 levels by the year 2000. Developed countries are also to assist developing countries financially in complying with their commitments under the UNFCCC. The implementation and the adequacy of these commitments are reviewed annually by the Conference of the Parties to the Convention (COP).

The most obvious limitation of the UNFCCC was that it did not provide for any specific binding commitments regarding emissions reductions and mechanisms to enforce them. However, it paved the way for future agreements on binding and more specific commitments at the international level (Grubb et al. 1999, pp. 40, 43).

On COP-3, held in Kyoto in 1997, delegates agreed on the next important milestone. The so-called Kyoto Protocol strengthened the commitments of the Parties to the UNFCCC, and extended them beyond 2000 (UN 1998). The Kyoto Protocol entered into force in 2005. So far, 180 Parties have ratified the Protocol (International Institute for Sustainable Development 2008, p. 2). Germany deposited its documents of ratification in 2002. Of the Annex I Parties, the United States, being the world's largest emitter of greenhouse gases together with China, Australia and Croatia have not yet ratified the Protocol (UN 2007).

Most importantly, the Protocol sets legally binding quantitative limitation and reduction targets for greenhouse gas emissions from Annex I countries (Article 3). Developing (non-Annex I) countries are explicitly excluded from new commitments (Article 10). Annex I countries agree to reduce their overall greenhouse gas emissions by an average of 5.2 percent below 1990 levels in the (first) commitment period lasting from 2008 to 2012 (Grubb et al. 1999, p. 116). The quantified emissions limitation and reduction commitments are differentiated by country and listed in Annex B of the Protocol (see Table 2-1). Germany as all other EU Parties (excluding Poland and Hungary) committed to a joint reduction of greenhouse gas emissions by eight percent compared to its 1990 levels. Some Parties like the Russian Federation, Australia or Iceland were allowed to emit at constant or slightly increased levels. Commitments under Article 3 apply to those greenhouse gases mentioned in Annex A of the Protocol: Carbon dioxide (CO_2), Methane (CH_4), Nitrous oxide (N_2O), Sulphur hexafluoride (SF_6), Hydrofluorocarbons (HFCs) and Perfluorocarbons (PFCs). If these commitments are not met, countries can be sanctioned – in an extreme case – by the International Court of Justice, or a commission composed of members selected by all Parties (Article 19).[6]

Table 2-1: Quantified emission limitation and reduction commitments of the Parties to the Kyoto Protocol as stated in Annex B (UN 1998)

Country	Commitment (% from the base year)[7]
EU Member States (including Germany, without Malta and Cyprus, Poland and Hungary with diverging commitments), Liechtenstein, Monaco, Switzerland	-8
USA	-7
Canada, Hungary, Japan, Poland	-6
Croatia	-5
New Zealand, Russian Federation, Ukraine	0
Norway	+1
Australia	+8
Iceland	+10

The Protocol calls for Annex I Parties to undertake domestic actions to reduce emissions (Article 2). Emissions may be reduced by direct mitigation or by sinks, i.e. changes in the carbon stock during the commitment period resulting from human-induced land-use changes and forestry

[6] As, for example, Schelling (2002, p. 7) points out, it is still unclear, though, whether sanctions would actually be binding (for an overview, see Hansjürgens 2008, pp. 13-14).

[7] Some countries in transition to a market economy have base years other than 1990.

activities (Article 3). Moreover, the Protocol provides for a variety of complex mechanisms that give Parties to the Protocol some flexibility in how they actually meet their reduction targets. These measures are meant to reduce the costs of complying with the Kyoto commitments (Oberthür and Ott 2000, p. 136). Emission rights can be exchanged between Annex I countries either by an agreement to fulfil their commitments jointly (Bubbling, Article 4) or by individual trades between countries (International Emissions Trading, IET, Article 17). The Bubbling provision has been used by the European Union (see Section 2.2.2). In addition, Annex I countries can meet their commitments partly by financing climate protection projects in other Annex I countries under the Joint Implementation (JI) mechanism (particularly in central and eastern European countries in transition, Article 6) and in developing countries under the Clean Development Mechanism (CDM) (Article 12).

The Protocol pursues a process-oriented approach. Many regulations regarding the flexibility mechanisms and the monitoring and enforcement procedures have been specified during subsequent COPs. In 2005, the process of negotiating greenhouse gas reduction commitments beyond 2012 and the Kyoto Protocoll has been initiated. New commitments will be finally agreed upon at COP 15 held in Copenhagen in 2009 (International Institute for Sustainable Development 2008, pp. 1-2). The political process has been accompanied by an extensive scientific discussion on how to design agreements in a post-Kyoto world, which the interested reader is referred to (see, e.g., Böhringer 2000; Olmstead and Stavins 2006; Aldy and Stavins 2007; Hansjürgens 2008).

Despite their vagueness and uncertainties, the commitments of the Kyoto Protocol have been the guiding principle for the European climate strategy, which will be presented in the following section.

2.2.2 European Climate Strategy

The EU ratified the Kyoto Protocol in 2002, being the only supranational Party to the Protocol. In the context of the ratification, the EU declared that its Member States would fulfil their quantified emission limitation and reduction commitments jointly in accordance with the bubbling provision of the Protocol (Oberthür and Ott 2000, p. 198). The EU and its Member States provided for an overall reduction of greenhouse gas emissions by eight per cent below 1990 emission levels in the first commitment period from 2008 to 2012 as stated in the Kyoto Protocol. However, the commitments were redistributed between the Member States by the EU Burden Sharing Agreement (Council of the European Union 2002) (see Table 2-2). Commitments were differentiated to take into account differences in wealth, expected economic growth, energy mix and industry structure. Germany agreed to reduce its greenhouse gas emission by 21 per cent below its 1990 emission levels. Other major emitters like the United Kingdom or Italy also committed to emission reductions. Less developed countries like Spain, Portugal, Greece and Ireland were admitted to increase emissions to a limited extent. The EU Burden Sharing Agreement only includes those 15 countries that were part of the EU by 2002. Countries which have acceded to the EU subsequently are not subject to the agreement (Grubb et al. 1999, p. 122).

Table 2-2: Quantified emission limitation and reduction commitments of the EU Member States as determined by the EU Burden Sharing Agreement (Council of the European Union 2002)

EU Member State	Commitment (% reduction from 1990 emission levels)
Luxembourg	-28.0
Denmark	-21.0
Germany	-21.0
Austria	-13.0
United Kingdom	-12.5
Belgium	-7.5
Italy	-6.5
Netherlands	-6.0
France	0.0
Finland	0.0
Sweden	+4.0
Ireland	+13.0
Spain	+15.0
Greece	+25.0
Portugal	+27.0
EU total	-8.0

Subsequent to the ratification of the Kyoto Protocol, the European Union has emphasized its commitment to combating climate change in several occasions. In its 2005 communication "Winning the Battle against Global Climate Change", the European Commission acknowledged that climate change is happening (European Commission 2005a). It confirmed the target brought forward by the IPCC that global average temperatures should not exceed 2°C above pre-industrial levels. Moreover, the Commission highlighted that the benefits of limiting climate change outweigh the costs of doing so. Based on these statements, the Commission derived several necessary elements of a future EU climate change strategy. Firstly, the EU shall undertake efforts to make all major emitters – including developing countries – participate in combating climate change. Secondly, efforts have to cover all relevant greenhouse gases and emitting sectors, including, e.g., maritime transport and aviation as well. Thirdly, innovation has to be promoted by a mixture of "push" and "pull" policies. Fourthly, the EU should continue using market-based and flexible instruments, such as emissions trading. Fifthly, more resources have to be dedicated to adaptation to climate change in the EU.

In 2007, EU Member States agreed on quantitative targets with respect to combating climate change. Greenhouse gas emissions shall be reduced by 20 percent by 2020 compared to 1990 emission levels. This commitment was far beyond the Kyoto commitment of eight percent reduction. The reduction target may be increased to 30 percent if other developed countries agree on post-Kyoto agreements as well (Council of the European Union 2007). Moreover, energy efficiency is to be improved by 20 percent until 2020 compared to present levels. In addition, the share of renewable energy sources in total energy consumption shall be increased to 20 percent by 2020. The share of biofuels in total petrol and diesel consumption for transport purposes shall be increased to 10 percent. In January 2008, the European Commission presented a programme of measures to actually achieve these targets (European Commission 2008b). This programme includes a proposal for a burden sharing of greenhouse gas reductions beyond 2012 (European Commission 2008i). According to this proposal, Germany has to reduce its emissions by 2020 by 14 percent below its levels in 2005, the new base year chosen. Moreover, the European Commission emphasizes the need to revise the European Emissions Trading Scheme and to promote renewable energy and carbon capture and storage technologies (SRU 2008, p. 90).

The EU has translated its climate strategy into a variety of Directives and Regulations related to specific climate and energy policies. These are presented and discussed in the context of the corresponding national policies in Section 2.3.

2.2.3 German Climate Strategy

Germany's climate strategy has been strongly influenced by international and European strategies. Emissions reductions agreed upon at the international and European level have shaped Germany's national climate-related targets.

The first and second National Climate Protection Programmes adopted in 2000 and 2005 were to ensure that the 21 percent reduction target set in the EU Burden Sharing Agreement for 2012 could be attained. These programmes mandated a multiplicity of policies to be implemented or revised. Among the policies mentioned were many of importance for the electricity sector, which are presented in the remainder of this chapter. Key measures mandated by the programmes included the EU Emissions Trading Scheme, the promotion of renewable energy sources and combined-heat-and-power plants, the taxation of electricity as well as other measures fostering energy efficiency (BMU 2005; BMU 2007e, p. 24; Beck et al. 2009, pp. 24-25).

More recently, the German government has shifted the focus to greenhouse gas emission reductions beyond the year 2012 and the Kyoto Protocol. Emissions are now to be reduced by 40 percent by 2020 compared to 1990 levels (BMU 2007d, p. 9). As one means to actually achieve this target, the Federal Government adopted the so-called Integrated Energy and Climate Programme (IECP) in 2007 (BMU 2007b). It contained a set of 29 targets and measures related to energy and climate policies. Most importantly, the share of renewable energy sources in total electricity generation shall be raised to 25 to 30 percent by 2020. In this respect, the German targets are more ambitious than the European ones. Moreover, combined heat and power plants are to contribute 25 percent to total energy generation in 2020. Further measures related to emissions from electricity generation include the promotion of carbon capture and sequestration in coal power plants, smart metering, energy management systems, energy-efficient products and energy research and innovation in general. It is assumed that the IECP may contribute to a 36 percent reduction in emissions by 2020 compared to 1990 levels. Many of the IECP targets and measures still have to be implemented. However, the programme has already resulted – among other – in a revision of the Renewable Energies Act and the Combined Heat and Power Act (BMU 2007a).

The climate policies related to the electricity sector which have been implemented as a result of the National Climate Protection Programmes and the IECP – but also as a response to international and European agreements and regulations – are presented in further detail in the remainder of this chapter.

2.3 Presentation of German Climate Policies

This section is devoted to the detailed presentation of climate policies in the German electricity sector. The policy mix represents a compilation of all common types of policy design. Policies implemented pursue above all market-based approaches. This true for the EU Emissions Trading Scheme, the feed-in tariff for electricity generation, the electricity tax, the bonus for combined heat and power generation and low-interest loans promoting technology innovation and diffusion. These policies are introduced in Sections 2.3.1 to 2.3.5. Apart from market-based approaches, the policy mix also includes other common policy types. Eco-design standards (Section 2.3.6) are an excellent example of a command-and-control approach. The energy labelling for household appliances (Section 2.3.7) corresponds to an information policy. Finally, the policy mix also encompasses a voluntary agreement to reduce GHG emissions (Section 2.3.8).

2.3.1 EU Emissions Trading Scheme

Under the EU Scheme for Greenhouse Gas Emissions Trading (briefly, EU Emissions Trading Scheme, ETS) a so-called National Allocation Plan (NAP) has been set up for Germany. The NAP determines a total cap of CO_2 emissions for Germany's energy and energy-intensive industry sectors. Single CO_2 emitting installations are allocated a certain quantity of allowances, each of which incorporates the right to emit one tonne of CO_2. The total number of allowances equals the emissions cap. For each year, the operator of an installation has to surrender a quantity of allowances which corresponds to the actual emissions of his installation. If the CO_2 emissions of an installation are lower than the number of allocated allowances, the operator may sell the excess allowances. If the emissions of an installation exceed his allowance holdings, the operator may buy additional allowances. If he does not hold and surrender a sufficient amount of allowances to cover the emissions, the operator will be sanctioned (the theoretical economic basis of emissions trading and available design options are extensively discussed in several seminal books: see, e.g., Tietenberg 1985; Klaassen 1996; Hansjürgens 2005b).[8]

The ETS started its operation in Germany and all other EU Member States on 1 January 2005. The first commitment period from 2005 to 2007 was considered a trial phase (Zapfel 2005, p. 165). Subsequent commitment periods are five years long (Art. 11 ETSD, §6(4) TEHG). The

[8] Moreover, there are also important studies of emissions trading at the interface of economics and business administration (for overviews, see Antes et al. 2006; 2008).

period from 2008 to 2012 is the first in which the EU and its Member States have to meet the binding emission limitation and reduction commitments of the Kyoto Protocol.

The framework for the ETS is set by the EU Directive establishing a scheme for greenhouse gas emission allowance trading (ETSD) (European Parliament/Council of the European Union 2003). This Directive has to be implemented and specified by national laws of the Member States. The ETS has been implemented in Germany by the Greenhouse Gas Emissions Trading Act (Treibhausgas-Emissionshandelsgesetz, TEHG) (Bundesregierung 2004d). The TEHG was supplemented by Acts on the NAP for the first and second commitment period (Zuteilungsgesetz 2007/Zuteilungsgesetz 2012, ZuG 2007/ZuG 2012) (Bundesregierung 2004b; 2007). In the following, the presentation of the ETS will only refer to the regulations effective during the commitment period from 2008 through 2012.

Objectives

The primary objective of the ETS is to promote the reduction of greenhouse gas emissions in a cost-effective and economically efficient manner (Art. 1 ETSD, §1 TEHG). Thereby, the ETS shall contribute to climate protection on a global scale. Neither the ETSD nor the TEHG are specific about the quantitative extent of greenhouse gas emissions reductions to be achieved. Yet, the emission limitation or reduction has to be in line with each Member State's commitments under the UNFCCC, the Kyoto Protocol and the EU Burden Sharing Agreement (Annex III ETSD). When setting up the ETS at the European and national level, decision makers had at least two further objectives in mind. On the one hand, the ETS is expected to promote economic development as it sets incentives for investment in and innovation of new technologies (BMU 2007c, p. 7). On the other hand, the ETS has to be implemented with the least possible diminution of economic development and employment. Member States have to prevent discrimination between sectors and companies, provide for the entrance of new installations and operators into the market and allow for the consideration of early emission reductions (Annex III ETSD).

Scope

At the moment, the ETS only covers CO_2 emissions (Annex I ETSD and TEHG). The ETS pursues a downstream approach. That is, installations where CO_2 emitting activities are performed are subject to the trading scheme. The operators of these installations are required to hold emission allowances.[9] According to Annex I ETSD and TEHG, the ETS covers CO_2 emitting activities which are carried out in installations of the energy sector and some activities conducted in the energy-intensive industry sector, including activities of the iron and steel, the cement, the glass and the pulp and paper industry (see Annex A.1 of this book). Installations of the chemical industry and the aluminium industry and all emissions from households, transport and trade, commerce and service sectors are not included. Most of the installations from the covered sectors are only subject to the ETS if they exceed certain production capacity or output thresholds. Installations for generating power and heat are included if their production capacity is larger than 20 Megawatt. Energy installations which are subject to the German Renewable Energies Act (see Section 2.3.2) and receive fixed feed-in tariffs for power generated from renewable energy sources are not covered by the ETS (§2(5) TEHG). Moreover, emissions stemming from the burning of biomass are also excluded from the ETS since an emissions factor of zero is applied for biomass when CO_2 allowances are calculated for an installation (Annex IV ETSD, Annex 2 TEHG).

During the commitment period from 2008 to 2012, 1,625 German installations are subject to the ETS. Out of these, roughly two thirds – 1,072 installations – belong to the energy sector, and one third – 553 installations – belong to the energy-intensive industry sector (UBA 2008a, p. 8).

Rules

At the beginning of each period, operators of ETS installations are allocated a certain quantity of allowances. An allowance incorporates the right to emit one tonne of CO_2 equivalent during a specified period. The total quantity of allowances and the procedure of their allocation to individual installations are determined in the NAP. The NAP, first of all, defines the national emissions budget for all greenhouse gases and emitting economic sectors. For the commitment period from 2008 to 2012, the annual national emissions budget is set at 973.6 million tonnes of CO_2 equivalents (§4(1) ZuG 2012). This implies a reduction of greenhouse gas emissions in Germany by 258.9 million tonnes of CO_2 equivalents, or 21 per cent, compared to 1990 emission levels. Thus, the NAP is designed such that Germany is able to fulfil its commitments under the Kyoto Protocol (BMU 2007c, pp. 17-19).

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[9] In contrast, an upstream emission trading would refer to production inputs, such as fossil fuels, which result in CO_2 emissions once they are used. In this case, producers and importers of these inputs would have to hold emission allowances according to the carbon contents of the inputs.

From the national emissions budget, a maximum of 453.1 million allowances is allocated annually to ETS installations (§4(2) ZuG 2012). This corresponds to an emission reduction of roughly 20 percent compared to average emission levels in 2005 and 2006 (UBA 2008a, p. 24). The ETS budget includes a reserve of 23 million allowances annually for new installations set up after 2007 and installations whose allowance allocations are adjusted ex post (§5 ZuG 2012). This rule implements the corresponding Article 11(3) and Annex III(6) of the ETSD. Out of the total ETS budget, 40 million allowances are sold via carbon exchanges until 2009, and auctioned starting in 2010 (§21(1) ZuG 2012). This is in line with the ETSD which provides for up to ten percent of all allowances to be auctioned (Art. 10 ETSD).

The remaining amount of allowances is allocated to operators of installations free of charge. This allocation process is administered by the German Emissions Trading Authority (Deutsche Emissionshandelsstelle, DEHSt) within the Environmental Protection Agency (Umweltbundesamt, UBA) (§20(1) TEHG). The amount of allowances allocated to a single installation is determined according to a differentiated set of rules outlined in the NAP (see Table 2-3). Industry installations commissioned before 2003 receive allowances based on a so-called grandfathering approach (BMU 2007c, p. 30). Their quantity of allowances is calculated as the product of their average annual emissions in a historical reference period times a compliance factor of 0.9875. That is, these installations have to reduce their emissions by 1.25 percent. Energy installations commissioned before 2003, all installations commissioned after 2003 as well as new installations are allocated allowances on the basis of a benchmarking approach. The number of allowances is determined as the product of historical or expected production output times an emission factor (the benchmark). The emission factor represents emissions per unit of output. It is technology-specific and set with respect to the best available technology (BAT) in each field (see Annex A.2 of this book). For power plants using natural gas, an emission factor of 365 grams CO_2 per kWh is set. The benchmark for power generation from other fuels, such as lignite and coal, is 750 grams CO_2 per kWh. Allowances of installations decommissioned during the commitment period are withdrawn and added to the reserve unless the production of the installation is transferred to other installations of the same operator (§10 ZuG 2012).

Apart from the basic allocation rules, several special rules may apply. Operators of installations may receive additional allowances for by-gases, such as blast furnace gases, coke oven gases and converter gases (§11 ZuG 2012). These are a by-product of the production of basic materials and can only be reduced by reducing output. Industry installations may benefit from an extra allocation for emission-reducing modernizations between 1994 and 2002, which was granted during the first commitment period (§12 ZuG 2007, §6(8) ZuG 2012). This rule excludes installations from the application of the compliance factor for twelve years subsequent to the commissioning of the modernization. Likewise, the compliance factor does not apply to small installations which were commissioned before 2003 and emit less than 25,000 tonnes of CO_2 per year (§6(9) and §7(4) ZuG 2012). Combined heat and power plants are privileged by a dual benchmark for power and heat (§7(3), (§8(1) and §9(4) ZuG 2012).[10] They receive allowances according to a technically comparable installation for exclusive power generation and, in addition, according to a technically comparable installation for exclusive heat generation.[11] Moreover, installations commissioned until 2002 may receive additional allowances if the allocation posed an extraordinary hardship to the operator otherwise (§6(6), §7(5) and §12 ZuG 2012).

Table 2-3: Allowance allocation procedures under the ETS according to the ZuG 2012

Year of commissioning	ETS sector	Allocation type	Annual quantity of allowances allocated	ZuG 2012
Before 2003	Industry	Grandfathering	Average annual emissions in reference period[a] X Compliance factor 0.9875	§6
	Energy	Benchmarking	Average annual production output in reference period[a] X Emission factor[b]	§7
2003–2007	Industry and energy	Benchmarking	Capacity of installation X Standard Utilization Factor[c] X Emission factor[b]	§8
After 2007	Industry and energy	Benchmarking	Capacity of installation X Standard Utilization Factor[c] X Emission factor[b]	§9

[a] The reference period is 2000 to 2005 for installations commissioned before 2000. For installations commissioned after 2000, the reference period in the year subsequent to commissioning and lasts until 2005.

[b] Technology-specific emission factors are provided in Annex A.2 of this book.

[c] Activity-specific and installation-specific standard utilization factors are provided in Annex A.3 of this book.

[10] For a definition of combined heat and power plants see Section 2.3.4.

[11] If only one benchmark, power or heat, was employed, combined heat and power plants would be put at a disadvantage compared to conventional power or heat plants as they are characterized by higher specific emissions per unit of power or heat (BMU 2007c, pp. 38-39).

The total budget of emission allowances for the ETS sectors – including the reserve and the allowances to be sold – is in fact 451.9 million, slightly below the legal limit (UBA 2008a, p. 4). General and specific allocation rules result in an allocation of 135.1 million allowances – or tonnes of CO_2 equivalents – to industry installations (UBA 2008a, p. 32). The calculated quantity of allowances allocated to energy installations generating electricity amounted to 243.6 million allowances. This amount had to be reduced by 38 million allowances to be sold or auctioned (§20 ZuG 2012).[12] To achieve this reduction, allocations to installations generating electricity were cut uniformly by 15.6 percent. Taking this reduction into account, the number of allowances allocated to energy and industry installations free of charge still amounted to 402.1 million (UBA 2008a, p. 4). This would still have been in excess of the legally defined limit of 388.9 million allowances which could be allocated to existing installations free of charge (§4(3) ZuG 2012).[13] Consequently, the allocation to energy installations had to be adjusted proportionally to an efficiency standard (Annex 5 ZuG 2012). This standard related hypothetical BAT emissions of each installation to its actual emissions in a reference year. Energy (electricity as well as non-electricity) installations eventually receive an annual total allocation of 253.8 million allowances (UBA 2008a, p. 32). This translates into an average reduction of allowance allocation of 35 percent compared to average emissions in 2005 and 2006 (UBA 2008a, p. 5). Thus, the emissions reduction mandated by the ETS compared to pre-ETS emission levels is mainly born by energy installations.

Allocated allowances are issued to operators of ETS installations by 28 February each year (Art. 11 ETSD, §11 TEHG). They are credited to the operator's account in an electronic emissions registry (Art. 9 ETSD, §14 TEHG). Similar to a banking system, this account keeps track of the issue, the holding, the transfer and the cancellation of allowances (European Commission 2005b). In Germany, the registry is administered by the DEHSt (§20 TEHG). Once issued Allowances can be freely transferred between operators of installations subject to the ETS and other natural and legal persons in the EU, such as individuals, institutions or non-governmental organisations, i.e. anybody can acquire allowances (Art. 12 ETSD, §6(3) TEHG). This allows operators to sell allowances if they are planning on emitting less than their initial allowance allocation and to purchase allowances if they want to emit more than that. Credits from CDM and JI projects can

[12] Two million out of the total of 40 million allowances to be sold or auctioned are attributed to the allowance reserve.

[13] This number is computed by subtracting the 40 million allowances to be sold and the 23 million allowances of the reserve from the total budget of 451.86 million allowances.

also be traded under the ETS and are considered as equivalent to EU greenhouse gas emission allowances (European Commission 2005b, p. 17). Traded allowances are subtracted from the account of the selling and added to the account of the acquiring party (§16 TEHG). Neither the ETSD nor the TEHG determine how and where allowances are to be transferred and traded. Thus, trades are organized via private trading platforms (UBA 2004, p. 5). Trades may be carried out directly, via brokers (over-the-counter) or via exchanges (European Commission 2005b, p. 14).

The trading volume of allowances has increased continuously since the introduction of the ETS (UBA 2009, p. 98). Allowance prices emerging at exchanges provide an indication of the prevailing emissions price. Figure 2-1 depicts the development of the ETS allowance price for the commitment period from 2008 to 2012 at the European Energy Exchange (EEX) in Leipzig. The illustration is based on forward prices observed until March 2008 and spot prices thereafter. Until the end of 2008, allowance prices had ranged between 15 and 30 Euro per ton of CO_2. As a consequence of the worldwide financial crisis, prices dropped subsequently. In February 2009, the spot price of one ETS allowance amounted to roughly ten Euro.

Figure 2-1: Development of ETS allowance price for the commitment period 2008 – 2012
Source: EEX

Operators are obliged to monitor their emissions and to submit an emissions report to State Immission Control Agencies by 1 March of year subsequent to the reporting year (Art. 14 and 15 ETSD, §5 TEHG). The State Agencies review the reports on random basis and submit them to the DEHSt by 31 March. The report has to be approved by independent verifiers. Monitoring of CO_2 emissions may be based on the direct measurement of emissions or on the calculation of emissions (Annex IV ETSD, Annex 2 TEHG).

By 30 April each year, operators of ETS installations have to surrender a quantity of allowances equal to the total emissions in the preceding year as stated in the verified emission report (Article 12(3) ETSD, §6(1) TEHG). Allowances surrendered are withdrawn from the operator's registry account and subsequently cancelled. Credits from JI and CDM projects can be used by operators to cover up to 20 percent of their installation's emissions (§18 ZuG 2012).

If an operator holds more allowances than he has to surrender for the preceding year, these allowances can be used to cover emissions in subsequent years of the current commitment period as well as of subsequent periods (Art. 13 ETSD, §6(4) TEHG). This is referred to as banking. Operators which do not surrender a sufficient amount of allowances to cover their CO_2 emissions in the preceding year, have to pay a fine of 100 Euros per tonne of CO_2 emitted for which no allowance has been surrendered (Art. 16 ETSD, §18 TEHG). In addition, the operator has to surrender the lacking allowances by 31 January of the following year.

2.3.2 Feed-in Tariff for Electricity Generation from Renewable Energy Sources

German grid system operators are obliged to connect renewable energy plants to their grids, purchase and transmit electricity from these plants and remunerate this electricity at legally defined feed-in tariffs. Via a nation-wide equalization scheme among transmission system operators, the feed-in tariffs eventually result in a uniform add-on to the electricity prices for final consumers.

Feed-in tariffs are mandated by the German Renewable Energies Act (Erneuerbare-Energie-Gesetz, EEG) adopted in 2000 (Bundesregierung 2000a). In 2001, the EU adopted a Directive to harmonize the promotion of electricity produced from renewable energy sources across Europe (RESD) (European Parliamant/Council of the European Communities 2001). As response to this Directive, the German EEG was amended in 2004 (Bundesregierung 2004c). A further amendment was agreed upon in 2008 (Bundesregierung 2008c).

Objectives

As §1(1) EEG sets out, the primary objective of the EEG is to facilitate the sustainable development of energy supply in Germany. In particular, the promotion of electricity from renewable energy sources is expected to contribute to the protection of climate, nature and environment. Moreover the EEG pursues several goals that are related to the primary objective.

It aims at reducing the long-term costs of energy supply to the national economy (including external costs) and at promoting the further development and diffusion of technologies for generation of electricity from renewable sources. In addition, the promotion of this kind of electricity shall also help to prevent conflicts over fossil fuels.

In line with the Integrated Climate and Energy Programme of the German Federal Government (see Section 2.2.3), the EEG pursues the specific goal to increase the percentage of renewable energy sources in total electricity supply to at least 30 percent in 2020 (§1(2) EEG). This target is above the indicative 22.1 percent goal set by the RESD for the same time horizon (Art. 3 RESD).

Scope

According to §3(1) EEG, eligible plants are facilities generating electricity from renewable energy sources and mine gas. Renewable energy sources include hydropower, wind energy, solar radiation, geothermal energy and energy from biomass, landfill gas, sewage treatment plant gas and the biodegradable fraction of municipal and industrial waste (§3(2) EEG). Feed-in tariffs only have to be paid for electricity stemming from plants that *exclusively* use renewable energy sources or mine gas (§16(1) EEG). Moreover, certain restrictions are set out in the EEG with respect to specific energy sources. New and modernized hydropower plants have to meet certain environmental standards (§23(5) EEG). Wind energy plants only receive remuneration if they achieve at least 60 percent of the reference yield at the intended site and are not located in protected offshore areas (§29(3) and §31(3) EEG). Operators of solar radiation plants not mounted on buildings only receive payments for generated electricity if the plant is erected before 1 January 2015 within the scope of application of a local development plan according to the Federal Building Code (§32(2) EEG). Electricity from biomass is only eligible for remuneration if it is generated in a plant with a capacity of up to 20 MW (§27(1) EEG). Biomass installations with a capacity between five and 20 MW have to use combined heat and powet technologies (§27(3) EEG). Electricity from plants using landfill gas or sewage treatment gas is subject to remuneration if the plant's capacity does not exceed 5 MW (§24 and §25 EEG).

Rules

The EEG imposes a variety of obligations on grid system operators. Grid system operators are persons running interconnected facilities used for the transmission and distribution of electricity (§3 EEG). They are required to connect renewable energy plants to their grids immediately and preferentially compared to conventional power plants (§4 EEG). Moreover, grid system operators have to guarantee priority purchase and transmission of electricity from renewable energy plants (§8 EEG). These obligations are in line with Article 7 RESD. Costs of connecting the renewable energy plant to the grid have to be born by the operator of the plant (§13 EEG). Grid system operators are required to bear the costs of grid system upgrades, which may be necessary to accommodate electricity from renewable energy plants (§14 EEG).

Grid system operators have to remunerate the electricity from eligible renewable energy plants at fixed minimum prices, so-called feed-in tariffs (§16 EEG). Feed-in tariffs under the EEG are meant to provide for an economic operation of renewable energy plants. They have been set to cover investment, operation, measurement and capital costs of a certain type of plant, and to consider common interest rates for the capital employed (BMU 2004c, p. 28). Tariffs are differentiated with respect to energy sources, the capacity of the plant, the location of the plant and the technology employed (see Table 2-4). If different tariffs apply for different classes of capacities, the tariff applicable for a certain plant calculates according to the share of the plant's total capacity in relation to the capacity thresholds (§18(1) EEG).[14] Feed-in tariffs are paid for 20 years subsequent to the commissioning or modernization of the plant as well as the year of commissioning or modernization (§21(2) EEG).[15] Tariffs are stated in the EEG for 2009 and are subsequently subject to an annual degression, which varies with the energy source and the technology employed (§20 EEG). This implies that plants set up in one year are paid a higher tariff for 20 years than plants commissioned in the subsequent year. Special bonuses which are paid for the use of efficient technologies and renewable raw materials are added to the feed-in tariffs and are also subject to degression.

[14] For example, for small hydroelectric power plants, feed-in tariffs amount to 9.67 cents per kWh if their capacity does not exceed 500 kW and 6.65 cents per kWh if their capacity is below five MW. The applicable feed-in tariff for a 1.5 MW plant calculates as 0.33 X 9.67 + 0.67 X 6.65 = 7.66 cents per kWh.

[15] Hydropower plants with a capacity exceeding 5 MW receive remuneration for only 15 years.

Table 2-4: Overview of Feed-in Tariffs under the German Renewable Energies Act (EEG)

Energy source	Technology constraint/bonuses	Capacity thresholds	Tariff in Cent/kWh in 2009	Annual degression (starting 2010)	EEG
Hydropower	New plants with a capacity ≤ 5 MW	≤ 500 kW	12.67	0 %	§23(1)
		≤ 2 MW	8.65		
		≤ 5 MW	7.65		
	Modernized plants with a capacity ≤ 5 MW	≤ 500 kW	11.67		§23(2)
		≤ 5 MW	8.65		
	Modernized plants with a capacity > 5 MW with an increase in capacity due to modernization (tariff applies to extension only)	≤ 500 kW	7.29	1 %	§23(3), §23(4)
		≤ 10 MW	6.32		
		≤ 20 MW	5.80		
		≤ 50 MW	4.34		
		> 50 MW	3.50		
Wind energy	Onshore plants		9.20 (reduction to 5.02 after 5 years or later)[a]	1 %	§29
	Bonus for grid system services of onshore plants		0.50		
	Bonus for onshore plants substituting existing plants with smaller capacity (repowering)		0.50		§30
	Offshore plants		15.00 (reduction to 3.50 after 12 years or later)[b]	1 % (5 % starting 2015)	§31
Solar radiation	Plants not attached to or integrated on top of buildings		31.94[c]	10 % (9 % starting 2011)	§32
	Plants attached to or integrated on top of buildings	≤ 30 kW	43.01	8 % (9 % starting 2011)	§33
		≤ 100 kW	40.91		
		≤ 1 MW	39.58	10 % (9 % starting 2011)	
		> 1 MW	33.00		
Biomass		≤ 150 kW	11.67	1 %	§27
		≤ 500 kW	9.18		
		≤ 5 MW	8.25		
		≤ 20 MW	7.79		

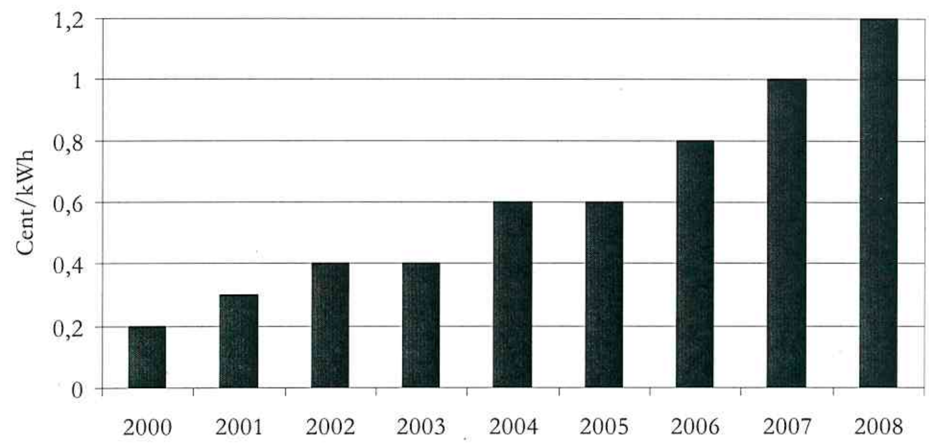
	Bonus for using certain efficient combustion technologies	≤ 5 MW	1.00-2.00[d]		
	Bonus for using renewable fuels or manure	≤ 500 kW	6.00-11.00[e]		
		≤ 5 MW	2.50-4.00[e]		
	Bonus for using combined heat and power technologies		3.00		
	Bonus for using biogas generated by anaerobic fermentation	≤ 500 kW	1.00		
Landfill gas		≤ 500 kW	9.00	1.5 %	§24
		≤ 5 MW	6.16		
	Bonus for using certain efficient combustion technologies		1.00-2.00[d]		
Sewage treatment plant gas		≤ 500 kW	7.11	1.5 %	§25
		≤ 5 MW	6.16		
	Bonus for using certain efficient combustion technologies		1.00-2.00[d]		
Mine gas		≤ 1 MW	7.16	1.5 %	§26
		≤ 5 MW	5.16		
		> 5 MW	4.16		
	Bonus for using certain efficient combustion technologies	≤ 5 MW	1.00-2.00[d]		
Geothermal energy		≤ 10 MW	16.00	1 %	§28
		> 10 MW	10.50		
	Bonus for using heat		3.00		
	Bonus for using petro-thermal technologies		4.00		

[a] The initial period of higher remuneration is extended for plants whose yield is below 150 percent of the reference yield.
[b] The initial period of higher remuneration is extended for plants whose locations exceeds a distance of 12 nautical miles to the shoreline and a water depth of 20 metres.
[c] Remuneration is reduced to 25.01 Cent/kWh if operator uses the electricity for own purposes.
[d] Bonus depends on the technology used.
[e] Bonus depends on the fuels used.

The EEG includes a mechanism which equalizes the quantities of electricity purchased, transmitted and remunerated as well as the remunerations paid under the EEG among grid system operators, transmission system operators and utility companies in Germany. This nation-wide equalization scheme is to avoid regional inequalities in the EEG burden borne by these actors (BMU 2004a, p. 11). Upstream transmission system operators are required to purchase and transmit electricity which grid system operators have purchased from renewable energy plant operators under the EEG (§34 EEG). Transmission system operators are also obliged to pay remuneration for this electricity to grid system operators according to the tariffs set out in the EEG (§35 EEG). The quantities of renewable electricity and remunerations are equalized across all German transmission system operators (§36 EEG). Transmission system operators transfer their quantities of electricity from renewable energy sources to the utility companies they serve, according to the total quantity of electricity received by each utility (§37 EEG). Thus, each utility company is obliged to purchase and remunerate the same share of electricity from renewable energy sources. The remuneration utilities have to pay refers to average remuneration paid per unit of electricity from renewable energy sources. The EEG is not specific about how utility companies are to proceed with the quantities of renewable electricity they were obliged to purchase and remunerate. This implies that the resulting additional costs can be passed on to final customers. The EEG therefore results in a uniform, tax-like add-on to the electricity price for final customers. This add-on computes as the share of renewable energy sources in total electricity generation times the difference of average remuneration paid per unit of renewable electricity produced and the average wholesale electricity price (BMU 2008b, p. 30). Due to the growth of electricity generation from renewable energy sources, the add-on has increased continuously since the implementation of the EEG in 2000 (see Figure 2-2). It amounted to 1.0 Cent per kWh in 2007 and was expected to raise to 1.2 Cent per kWh in 2008 (BMU 2008b, p. 30).

Figure 2-2: Development of the EEG add-on to the electricity price 2000 – 2008 (in 2007 prices)

Source: BMU (2008b, p. 30)



For manufacturing companies and railway operators with a relatively high electricity consumption of more than 10 GWh per year, the EEG add-on is limited to 0.05 Cent per kWh (§40, § 1 and §42 EEG). For a manufacturing enterprise to be eligible, the ratio of the enterprise's electricity costs and its gross value added must have exceeded 15 percent in the last financial year. The reduced add-on applies without restrictions to manufacturing enterprises whose annual electricity consumption is beyond 100 GWh and whose ratio of electricity costs and gross value added exceeds 20 percent. For manufacturing enterprises that do not reach this thresholds, and railway operators, the provision only applies to the quantity of electricity exceeding 10 percent of the total electricity consumed in the last financial year. The reduction of the EEG add-on for eligible manufacturing enterprises and railway operators is compensated by an increase of the add-on for final customers who are not subject to this provision.

2.3.3 Electricity Tax

The electricity tax was introduced in Germany in 1999. It formed part of the so-called ecological tax reform. This reform implemented taxes on the use of environmental resources. The electricity tax has to be paid upon the consumption of electricity. The tax revenues, in turn, are used to stabilize and reduce employers' and employees' contributions to pension insurances.

As a first step, the German Act on the Introduction of the Ecological Tax Reform (Gesetz zum Einstieg in die ökologische Steuerreform) was adopted in 1999 (Bundesregierung 1999a). It incorporated the Electricity Tax Act (Stromsteuergesetz, StromStG) introducing a tax on electricity. This legislation has been modified subsequently, mainly by the 2000 Act on the Continuation of the Ecological Tax Reform (Gesetz zur Fortführung der ökologischen Steuerreform) and the 2003 Act on the Further Development of the Ecological Tax Reform (Gesetz zur Fortentwicklung der ökologischen Steuerreform) (Bundesregierung 1999b; Bundesregierung 2002a). National laws were complemented by European legislation in 2004 when the EU Energy Taxation Directive (ETD) entered into force in 2004 (Council of the European Union 2003).

Objectives

The electricity tax is expected to provide a "double-dividend" by shifting the tax burden from labour to polluting electricity generation (BMU 2004b, p. 3). On the one hand, it aims at combating climate change. The electricity tax is meant to internalize externalities of electricity generation, particularly those related to the emission of greenhouse gases. In order to reduce greenhouse gas emissions, the tax is to provide incentives to reduce electricity consumption and to develop and use energy-efficient products and production processes.

On the other hand, it is the explicit goal of the electricity tax to promote economic growth in Germany. Using the revenues of the electricity tax to stabilize and reduce employers' and employees' contributions to pension insurances, non-wage labour costs are to be brought down. This is expected to improve the international competitiveness of the German industry and, eventually, to reduce unemployment in Germany. What is more, companies and sectors producing energy-efficient products may benefit from a rising demand and, thus, further contribute to economic growth (BMU 2004b, p. 4).

Scope

The electricity tax is an excise tax that has to be paid upon the consumption of electricity (§1 StromStG). The tax is levied if electricity is taken from the electricity supply network by an end-consumer (§5 StromStG). For the ease of administration, the tax is not raised from the consumer but from the utility. Yet, utilities are allowed to pass the tax over to consumers by increasing the electricity price correspondingly (BMF 2004, p. 2).

Electricity is exempt from taxation if it is used for certain activities of the manufacturing sector listed in Table 2-5 (§9a StromStG).

Table 2-5: Activities of the Manufacturing Sector Exempt From the Electricity Tax

	Activity
1	Electrolysis
2	Production of glass and glassware, ceramic products, ceramic tiles and bricks, cement, lime and gypsum, products made of concrete, cement and gypsum, mineral insulation materials, asphalt and mineral fertilizers; baking, smelting, warming and thermal stress relief of these products or of preliminary products used for the production of these products
3	Production and processing of metals and, in the context of the production of metal products, the production of forged, pressed, drawn and punched parts, rolled rings and powder-metallurgic products and surface refinement and thermal treatment for smelting, warming, stress release and other thermal treatment
4	Chemical reduction

Rules

Article 10 and Annex I Table C ETD require that EU Member States introduce a tax on electricity of 0.50 Euro per MWh for business uses and 1.00 Euro per MWh for non-business uses. The Electricity Tax Act sets out a general tax rate of 20.50 Euro per MWh, which is therefore well above the EU requirements (§3 StromStG).

The StromStG provides for a variety of tax breaks. Public rail and trolleybus transport is subject to a tax rate of 11.42 Euro per MWh, i.e. roughly 56 percent of the regular tax rate (Art. 5 ETD, §9(2) StromStG). A tax rate of 12.30 Euro per MWh, i.e. 60 percent of the regular tax rate, applies for electricity used by the industry sector, including the mining, brick and earth and manufacturing industry, agriculture, forestry and fishery (§9(3) StromStG). For the first 25 MWh of electricity consumed by these sectors an even further reduced tax rate of 8.20 Euro per MWh is raised (§9(5) StromStG). In addition, electricity consumers of the industry sector are eligible for a tax cap (§10(2) StromStG). If the tax burden of a company from the industry sector is higher than its tax relief from the reduction in contributions to the pension funds, the company is refunded 95 percent of the differential amount. This translates into to a marginal tax rate of 3 percent (5 percent of 60 percent). The tax cap only refers to the tax burden that exceeds a base amount of 512.50 Euro (§10(1) StromStG). The application of the tax cap is provided until the end of 2009. Whether it will be extended beyond that deadline depends on the extent to which the goals of the voluntary agreement between the Federal Government and the German industry associations are actually attained (see Section 2.3.8). It may be extended until 2012 if 96 percent of the agreed emission reductions are achieved by the end of 2009 and 100 percent fulfilment can be expected for 2012 (§10(1a) StromStG). All tax breaks are in line with Article 5 ETD providing for the differential taxation electricity for business uses and non-business uses. Moreover, Article 17 ETD allows tax breaks for energy-intensive business and firms subject to a voluntary agreement on emissions reduction or an emissions trading scheme. However, the average tax rate across all electricity consumers still has to meet the EU requirements on minimum taxation.

Revenues from the ecological tax reform as a whole, i.e. electricity tax and other energy taxes, were expected to amount to 18.7 billion Euro in 2007 (Ziesing 2007b, p. 278). Out of these, some five billion Euro are generated by the electricity tax (Frondel and Hillebrandt 2004, p. 331). Roughly 90 percent of the revenues were used for the gradual reduction and stabilization of employers' and employees' contributions to the pension insurances. Due to the revenues of the entire ecological tax reform, contribution rates were reduced from 20.3 percent in 1998 to 19.1 percent in 2001 and 2002 and 19.5 percent in 2003. Without the reform, the rate would have been 1.7 percent points higher in 2003. With the remaining part of the tax revenues, the use of renewable energy sources for heating in private homes has been promoted (BMU 2004b, p. 15).

2.3.4 Bonus for Combined Heat and Power Generation

Quite similar to the feed-in tariff for renewable energy sources, grid system operators are also required to connect combined heat and power (CHP) plants to their grids and to purchase electricity from these plants. The grid operator has to remunerate this electricity at the prevailing whole sale electricity price and pay a legally defined CHP bonus in addition. A nation-wide scheme equalizes the CHP remunerations borne by grid system operators. This results in a uniform add-on to the electricity price of final consumers. CHP bonuses are legally defined in the German Combined Heat and Power Act (Kraft-Wärme-Kopplungsgesetz, KWKG), which was adopted in 2002 and amended in 2008 (Bundesregierung 2004a; 2008b). In 2004, national legislation was complemented by the European Directive on the promotion of cogeneration (CHPD) (European Parliament/Council of the European Union 2004).

Goals

The KWKG is to contribute to increasing the share of CHP electricity generation in total electricity generation to 25 percent (§1 KWKG). For this purpose, the KWKG aims at protecting existing CHP plants and promoting the modernization and new construction of CHP plants. Moreover, it shall support the diffusion of fuel cells and district heating networks using heat from CHP plants. By these means, the KWKG shall help to save energy, protect the natural environment and attain Germany's climate protection goals. In its introductory remarks and Article 1, the CHPD also highlights that the promotion of CHP may contribute positively to the security of energy supply and to the competitive situation of the EU and its Member States. This Directive established a European framework for CHP support and provides basic definitions related to CHP technologies.

Scope

The KWKG covers electricity generated in CHP plants using hard coal, lignite, waste, waste heat, biomass and gaseous and liquid fuels (§2 KWKG). Electricity remunerated under the EEG is not subject to the KWKG. Installations are considered as CHP plants if they simultaneously transform input energy into electric energy and useful heat (§3(1) KWKG). Eligible CHP technologies include steam turbines, gas turbines, combustion engines, Stirling engines, steam engines, Organic Rankine Cycle plants as well as fuel cells (Annex I CHPD, §3(2) KWKG). The KWKG mandates remuneration for all CHP plants commissioned before April 1, 2002. Plants newly commissioned between April 1, 2002 and 2008 are only covered if their capacity does not exceed two MW. Plants commissioned between 2009 and 2016 are promoted irrespective of their capacity. However, they have to be highly efficient. According to Annex III CHPD, CHP generation can be considered as highly efficient if energy savings amount to at least ten percent compared to the separate generation of electricity and heat. Moreover, new plants set up in 2009 and thereafter must not crowd out district heating provided by existing CHP plants. Electricity generation by fuel cells is remunerated without restrictions (§5 KWKG).

District heating networks are within the scope of the KWKG if they supply heat to facilities which are not located on the site of the CHP plant and not owned or operated by the same person as the CHP plant (§3(13) KWKG). The KWKG promotes the construction of new or the significant extension of existing district heating systems starting in 2009. Systems have to account for a share of at least 60 percent of CHP heat provision in total heat provision to be eligible (§5a(1) KWKG).

Rules

The KWKG requires grid system operators to connect CHP plants to their grids and to purchase electricity from these plants preferentially compared to conventional power plants (§4(1) KWKG). Remuneration paid to operators has to amount to the sum of the common electricity price and an additional CHP bonus (§4(3) KWKG). The common price refers to the average price of base load electricity offered at the European Energy Exchange (EEX) in Leipzig in the preceding quarter. The CHP bonus as well as the duration of remuneration is differentiated with respect to the year of commissioning the plant and the technology employed (see Table 2-6). If different tariffs apply for different thresholds of capacities, the tariff applicable for a certain plant calculates according to the share of the plant's total capacity in relation to the capacity thresholds (for a numerical example, see the footnote in Section 2.3.2). The total of bonuses paid to operators of CHP plants must not exceed 750 million Euros annually less the bonuses paid to operators of district heating networks (see below). In case this threshold is exceeded, bonuses are reduced correspondingly for plants commissioned before April 1, 2002 and modernized after 2008 and new plants commissioned after 2008. This reduction only applies to plants with a capacity of more than 10 MW (§7(9) KWKG).

Table 2-6: CHP bonuses

CHP plant	Capacity threshold	CHP bonus in Cent/kWh										KWKG
		2002	2003	2004	2005	2006	2007	2008	2009	2010	post 2010	
Commissioned before 1990		1.53	1.53	1.38	1.38	0.97	-	-	-	-	-	§7(1)
Commissioned between 1990 and April 1, 2002		1.53	1.53	1.38	1.38	1.23	1.23	0.82	0.56	-	-	§7(2)
Commissioned before 1990 and modernized between April 1, 2002 and 2005 [a]		1.74	1.74	1.74	1.69	1.69	1.64	1.64	1.59	1.59	-	§7(3)
Commissioned before April 1, 2002 and modernized between 2009 and 2016	≤ 50 kW								5.11 for 6 years subsequent to modernization [b c]			§7(4)
	≤ 2 MW								2.10 for 6 years subsequent to modernization [c]			
	> 2 MW								1.50 for 6 years subsequent to modernization [c]			
Commissioned between April 1, 2002 and 2008 with a capacity > 5 kW and ≤ 2 MW [a]		2.56	2.56	2.40	2.40	2.25	2.25	2.10	2.10	1.94		§7(5)
Commissioned after 2008 with a capacity > 5 kW and ≤ 2 MW	≤ 50 kW								5.11 for 6 years subsequent to commissioning [c]			
	≤ 2 MW								2.10 for 6 years subsequent to commissioning [c]			
Commissioned starting April 1, 2002 with a capacity ≤ 50 kW or fuel cell plants		5.11 for 10 years subsequent to commissioning										§7(6), §7(7)
Commissioned between 2008 and 2016 with a capacity > 20 MW	≤ 50 kW								5.11 for 6 years subsequent to modernization [c]			§7(8)
	≤ 2 MW								2.10 for 6 years subsequent to modernization [c]			
	> 2 MW								1.50 for 6 years subsequent to modernization [c]			

[a] Remuneration starts as soon as the plant/the modernization of the plant is commissioned.

[b] Modernized plants have to be highly efficient.

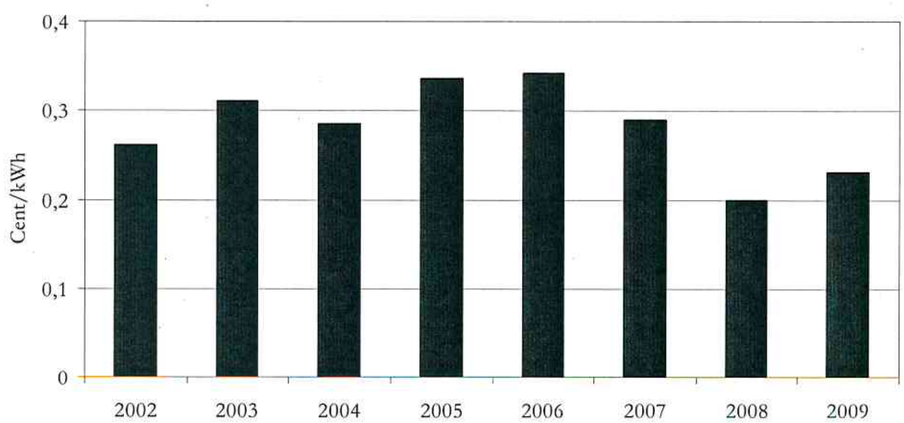
[c] Remuneration is paid for a maximum of 30,000 full hours of operation. If the CHP plant mainly provides process heat to a manufacturing enterprise, remuneration is only paid for four years subsequent to commissioning.

The bonus for district heating networks amounts to one Euro per millimetre of diameter for each meter of the network newly constructed. The total bonus paid for each project must not exceed 20 percent of total investment costs, or five million Euro (§7a KWKG).

The KWKG also incorporates a scheme to equalize the burden of CHP bonuses. Transmission system operators have to compensate grid system operators for bonuses paid for CHP electricity (§9(1) KWKG). These bonuses are subsequently equalized across all transmission system operators according to their individual quantities of electricity transmitted (§9(3) KWKG). In turn, transmission system operators have to be compensated for the CHP burden by downstream utilities (§9(4) KWKG). In order to cover these costs, utilities are allowed to charge final electricity customers an increased system usage fee (§9(7) KWKG). This increase is uniform across Germany and can be referred to as the CHP add-on. The CHP add-on has been fluctuating since the implementation of the KWKG in 2002 (see Figure 2-3). It is computed to amount to 0,231 Cent per kWh in 2009 (BDEW 2008b). However, this level is only valid for final customers which consume less than 100 MWh per year. If a final customer consumes more than that, the CHP add-on is limited to 0.05 Cent per kWh for any electricity consumption exceeding that threshold. If this final customer is a manufacturing company or a railway operator, the CHP add-on is further reduced to 0.025 Cent per kWh. In either case, electricity costs have to amount to at least four percent of the company's annual turnover.

Figure 2-3: Development of the KWK add-on to the electricity price for final customers with an annual consumption of less than 100 MWh 2002 – 2009

Source: BDEW (2008b)



2.3.5 Low-Interest Loans Promoting Technology Innovation and Diffusion

2.3.5.1 Environmental Innovation Programme

The Environmental Innovation Programme (Umweltinnovationsprogramm, UIP) provides low-interest loans for technological demonstration projects that have the capacity of avoiding or reducing environmental harms (BMU 2009). The Programme is based on a German Directive adopted by the Federal Ministry of the Environment in 1997 (BMU 1997).

Objectives

The UIP aims at avoiding or at least reducing environmental harms resulting from the production as well as the use and consumption of products. It promotes the development of environmentally friendly technologies. It focuses on projects that demonstrate for the first time how production processes can be adapted in an environmentally friendly manner and how environmentally friendly products and substitutes can be produced (BMU 1997, p. 1). Such projects are expected to have an important positive effect on technology diffusion (BMU 2009).

Scope

The programme only promotes technological demonstration projects at a large scale. It covers installations for environmental protection, supply, disposal or treatment, environmentally friendly production processes as well as processes for the production of environmentally products and substitutes. These projects must significantly reduce emissions of pollutants, waste, noise or soil contamination compared to existing technologies (BMU 1997, pp. 1-2). Eligible projects include, among other, measures to save energy, to improve energy efficiency and to use renewable energy sources (KfW 2009c, p. 2). Installations have to correspond to a progressive state of technology, represent an innovative combination of processes or apply processes integrated into the production process. Projects are only eligible if they would not be realized otherwise. Moreover, they must not be initiated until they are approved by the responsible authority (BMU 1997, pp. 1-2). Expenditures for which support may be granted under the UIP include costs of investments, commissioning and necessary expert opinions but not costs of purchasing a site and maintenance and operation (BMU 1997, p. 3). Moreover, research and development related to the demonstration projects is explicitly excluded from promotion (BMU 2009). Support under the UIP can be combined with the Environment and Energy Efficiency Programme as well as the KfW Programme Renewable Energies (see Sections 2.3.5.2 and 2.3.5.3) (KfW 2009c, p. 2).

Support under the UIP can be applied for by manufacturing enterprises as well as any other private or public actors. The programme puts particular emphasis on the promotion of small and medium-sized enterprises (BMU 1997, p. 2).

Rules

The UIP usually provides low-interest loans for eligible projects. These loans may cover up to 70 percent of the total investment volume of the project (BMU 1997, pp. 2-3). In exceptional cases, the programme may also grant subsidies to investment costs (BMU 2009). These subsidies must not exceed 30 percent of the total investment volume. The eventual extent of support provided by the programme depends on a variety of variables including the environmental impacts, the technological progress and technological and economic risks of the project (BMU 1997, pp. 2-3).

The duration of loans is limited to a maximum of 30 years, including five years free of redemption. The interest rate is fixed for the first ten years of duration. The prevailing market-based interest rate applies thereafter. The UIP provides support to borrowers by reducing the market-based interest rate by five percent points during five years of the loan's duration. Borrowers have to provide the usual securities. Loans are allowed by the state-owned Kreditanstalt für Wiederaufbau (KfW). Loans are settled via the borrower's bank (BMU 1997, pp. 2-3). The technical revision of projects is realized by the Environmental Protection Agency (Umweltbundesamt, UBA) (BMU 1997, p. 4).

2.3.5.2 Environment and Energy Efficiency Programme

Under the Environment and Energy Efficiency Programme (Umwelt- und Energieeffizienzprogramm), the state owned Kreditanstalt für Wiederaufbau (KfW) provides low-interest loans for investments protecting the environment. Particularly low interest rates are granted to small and medium-sized enterprises implementing energy efficiency measures.

Objectives

The Environment and Energy Efficiency Programme aims at fostering environmental protection in general and energy efficiency in particular. For this purpose, it promotes the adoption of environmentally friendly technologies. It covers a broad variety of measures including the support of energy-efficient technologies to generate and use electricity as well as heat. It puts a particular focus on the promotion of improvements in the sector of small and medium-sized enterprises (KfW 2009a, p. 1).

Scope

The program funds investments that contribute significantly to improving the situation of the environment. Loans can be applied for by privately owned, German and foreign companies from the manufacturing and service sectors, free-lancers, companies providing contracting services for third parties as well as public private partnerships (KfW 2009a, p. 2). Environmental protection measures covered include investments in efficient energy generation as well as efficient energy use. Moreover, a variety of other measures may be financed varying from air pollution control to wastewater treatment. The program does neither allow loans for purchasing real estate nor for setting up installations using renewable energy sources (these are funded under the KfW Programme Renewable Energies, see Section 2.3.5.3) (KfW 2009a, pp. 3-4).

Energy efficiency investments by small and medium-sized enterprises may be eligible for loans with particularly low interest rates. Investments may be related to building equipment and energy management, machinery, process measuring and control technology as well as information and communication technology. If these investments replace old installations, they have to provide for a reduction of energy consumption by at least 20 percent compared to average consumption of the pre-existing installation during the three preceding years. Investments in new installations have to reduce energy consumption by 15 percent compared to average energy consumption with common technology in the same sector (KfW 2009a, p. 4).

Rules

Loans are allowed by the state-owned KfW. They may be provided for up to 100 percent of a project's investment costs. They are usually limited to a total of two million Euros per project. This restriction may be relaxed for projects of particular environmental and political importance. Small and medium-sized enterprises may receive loans of a maximum of ten million Euros. Loans cannot be combined with other KfW programmes. The duration of loans may be up to 20 years with a maximum of three years free of redemption. Interest rates are lower for small and medium-sized enterprises than for other borrowers. Rates depend on the development of the capital market. Moreover, as for conventional loans, interest rates depend on the rating and the quality of securities provided by the borrower. However, it is below average rates at the market. In March 2009, rates varied from 2.25 to 6.95 percent. The interest rate is fixed for the first ten years of the duration of a loan and may be adapted afterwards (KfW 2009a, pp. 5-6). Small and medium-sized enterprises applying for loans for energy efficiency investments may also be granted financial support for energy efficiency consulting (KfW 2009a, p. 1). Applications for loans are not submitted directly to the KfW but settled via each borrower's bank. This bank will also review the borrower's securities as well as the appropriate use of the loan (KfW 2009a, pp. 7-8).

2.3.5.3 KfW Programme Renewable Energies

Under the KfW Programme Renewable Energies, the KfW grants long-term loans for investments in installations using renewable energy sources. These loans are characterized by interest rates below the market average.

Objectives

The KfW Programme Renewable Energies aims at contributing to a sustainable energy supply. This is understood as an important pillar of protecting the environment and mitigating climate change. For this purpose, it pursues the goal of funding measures using renewable energy sources in the long term (KfW 2009b, p. 1).

Scope

The KfW Programme Renewable Energies covers investments to set up, extent and purchase installations subject to the Renewable Energies Act (see Section 2.3.2). Moreover, it provides financial support for the set-up, extension or purchase of combined heat and power plants (see Section 2.3.4 for a definition). Loans are allowed for German as well as foreign enterprises that undertake investments in Germany or abroad (for German companies only). Enterprises may be owned privately or by municipalities and religious and charity organizations. Moreover, free-lancers, private persons and non-profit organizations may apply for loans if they do not use the energy for own purposes but feed it into the public network (KfW 2009b, p. 2).[16]

Rules

Loans are allowed for up to 100 percent of investment costs net of the value added tax. The volume of the loan is usually limited to ten million Euros per project (KfW 2009b, p. 3). Loans are allowed for a duration of up to 20 years with a maximum of three years free of redemption. Interest rates depend on prevailing rates on the capital market. However, interest rates are below the average of capital market rates. In March 2009, they ranged from 2.20 to 7.25 percent. All

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[16] The programme also encompasses a „premium" part with a more restricted scope. This part particularly promotes the use of renewable energy sources for heat generation. Since the focus is on the electricity sector, this section only introduces "standard" regulations concerning electricity generation from renewable energy sources.

other requirements and procedures correspond to those under the Environment and Energy Efficiency Programme (see Section 2.3.5.2) (KfW 2009b, pp. 9-13).

2.3.6 Eco-Design Standards

Eco-design standards mandate the implementation of minimum requirements regarding the environmental performance – including the energy efficiency – of household appliances, consumer electronics and other products.

These standards are based on the EU Eco-Design Directive (EDD) (European Parliament/Council of the European Union 2005). This Directive establishes a framework for the setting of eco-design requirements for energy-using products. The German Act on Energy-using Products (Energiebetriebene-Produkte-Gesetz, EBPG) implementing this Directive into national law was adopted in 2008 (Bundesregierung 2008a). Both legislations are not specific about the minimum requirements. They have been and will be complemented by individual implementing measures for each type of energy-using product.

Objective

As laid out in the introductory considerations and Article 1, the EDD aims at contributing to sustainable development by improving the environmental impact of energy-using products. Greenhouse gas emissions are highlighted as an important environmental impact of these products. To reduce environmental impacts, the entire life cycle of energy-using products shall be considered. The EDD puts particular importance on increasing the energy efficiency of these products. It is considered that the promotion of energy efficiency and energy saving is an important means to reduce environmental impacts in general and greenhouse gas emissions in particular. Moreover, energy saving is perceived as the most cost-effective way to increase security of energy supply and reduce import dependency.

Scope

The EDD and the EBPG cover energy-using products in general. Energy-using products depend on energy input including electricity, fossil fuels and renewable energy sources to work as intended. Likewise, products for the generation, transfer and measurement of such energy are considered as energy-using products. The regulations also cover components or sub-assemblies of energy-using products (Art. 2 EDD, §2 EBPD). Transportation means and products with predominant military use are explicitly excluded from requirements (Art. 1(2) EDD, §1(1) EBPD). Neither the EDD nor the EBPD are specific about the types of products considered. However, requirements shall only be imposed on products which represent a volume of annual sales of more than 200,000 units within the EU, and which have significant environmental impacts as well as significant potential in improving them (Art. 15(2) EDD). Implementing measures including specific requirements have been introduced for water boilers fired by liquid or gaseous fuels, household fridges and freezers, ballasts for fluorescent lighting and stand-by electric power consumption of household and office equipment (Council of the European Union 1992a; European Parliament/Council of the European Union 1996; 2000; European Commission 2008a). Implementing measures have been proposed by the Commission for simple set-top boxes, external power supplies, fluorescent lighting and domestic lighting products including incandescent bulbs (European Commission 2008c; 2008d; 2008e; 2008f). They are planned, among other, for televisions, washing machines, dishwashers, computers and air conditioners (European Commission 2008h).

Rules

The EDD provides for the setting of standards for energy-using products in order for these to be placed on the market and/or put into service (Art. 1(2) EDD). That is, requirements cover products produced within the EU as well as those imported (Art. 4 EDD). Requirements may refer to the entire life cycle of energy-using products, including the manufacturing, distribution, use and disposal of these products (Annex I EDD). Neither the EDD nor the EBPG define specific standards for different types of energy-using products. This specification is to be undertaken by implementing measures at the European or national level (Art. 15 EDD, § 3 EBPG).

So far, electricity-related requirements have been specified for a limited number of products. A limit on maximum electricity consumption has been implemented for refrigerators, freezers and their combinations, ballasts for fluorescent lighting and standby and off modes of electric household and office equipment (European Parliament/Council of the European Union 1996; European Parliament/Council of the European Union 2000; European Commission 2008a).[17] For example, standby and off mode power consumption is limited to one to two watts until 2012 and 0.5 to one watts thereafter, depending on the technology used.

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[17] Requirements for refrigerators, freezers and ballasts have been implemented into German law by the Maximum Energy Consumption Ordinance (Energieverbrauchshöchstwerteverordnung, EnVHV) (Bundesregierung 2002b). Requirements for standby and off modes are defined in a Commission Regulation, which applies directly to national law.

The Commission has provided draft Regulations to limit electricity consumption of non-directional household lamps, fluorescent and high discharge lamps, external power supplies simple set-top boxes (European Commission 2008c; 2008d; 2008e; 2008f). Requirements under these regulations are tightened in several phases. The Regulation concerning household lamps implies that conventional incandescent bulbs will be phased out by 2012.

Manufacturers or importers have to issue a declaration of conformity by which they ensure that their energy-using products are in line with the provisions of the corresponding implementing measures. Moreover, a CE conformity marking has to be attached to these products (Art. 5 EDD, §4 EBPG). Violations of these provisions may be sanctioned by fine of up to 50,000 Euros (§13 EBPG).

2.3.7 Energy Labelling for Household Appliances

Suppliers and dealers of household appliances, such as refrigerators, washing machines or dish-washers, are required to provide information about the energy consumption and energy efficiency of their products to customers by the means of a label.

The 1992 EU Energy Labelling Directive (ELD) required Member States to implement harmonized and legally binding regulations on labelling and standard product information related to energy consumption and energy efficiency of household appliances (Council of the European Union 1992b). Regulations of the Directive were specified subsequently by several implementing Directives for each type of household appliances. The ELD was implemented into German law in 1997 by the Energy Consumption Labelling Act (Energieverbrauchskennzeichnungsgesetz, EnVKG) and the Energy Consumption Labelling Ordinance (Energieverbrauchs-kennzeichnungsverordnung, EnVKV) (Bundesregierung 1997a; Bundesregierung 1997b).

Objectives

The principal goal of the EU labelling approach is to improve the energy efficiency of household appliances and, thereby, to contribute to saving energy and to a reduction of CO_2 emissions stemming from energy generation (introductory considerations and Art. 1 ELD). The provision of information to customers by a label is expected to guide their consumption choices towards household appliances that require least energy. Thus, suppliers of household appliances may be stimulated to develop and supply energy-efficient products. In addition, the EU Labelling Directive also aims at reducing the consumption of other essential resources, such as water, by providing relevant information to customers.

Scope

The obligation to provide information about the consumption of energy and other resources applies to selected household appliances as soon as they are offered for sale, hire, hire-purchase or displayed to final customers (Art. 2(1) ELD, §3(1) EnVKV). This obligation does not apply to second-hand appliances (Art. 1(5) ELD, §3(2) EnVKV). Appliances covered by the EU Directive and national legislation include household refrigerators, food freezers, household washing machines, household tumble driers and dishwashers. Household electric lamps and fluorescent lamps have to be labelled unless they have a luminous flux greater than 6500 lumens or an input power of less than 4 kW, they are reflector lamps, they are not primarily marketed for the production of light in the visible range or they are marketed as part of a product, the primary use of which is not illuminative. The labelling requirement also applies to air-conditioning appliances except for air-to-water and water-to-water appliances and units with a cooling output exceeding 12 kW. Household electric ovens – including those being part of larger appliances – are covered unless they are portable and have a mass of less than 18 kilograms (Art. 1(1) ELD, Annex 1(8) EnVKV). Water heaters and hot-water storage appliances are deemed to fall under the EU Labelling Directive as well. Yet, no implementing Directive has been adopted so far.

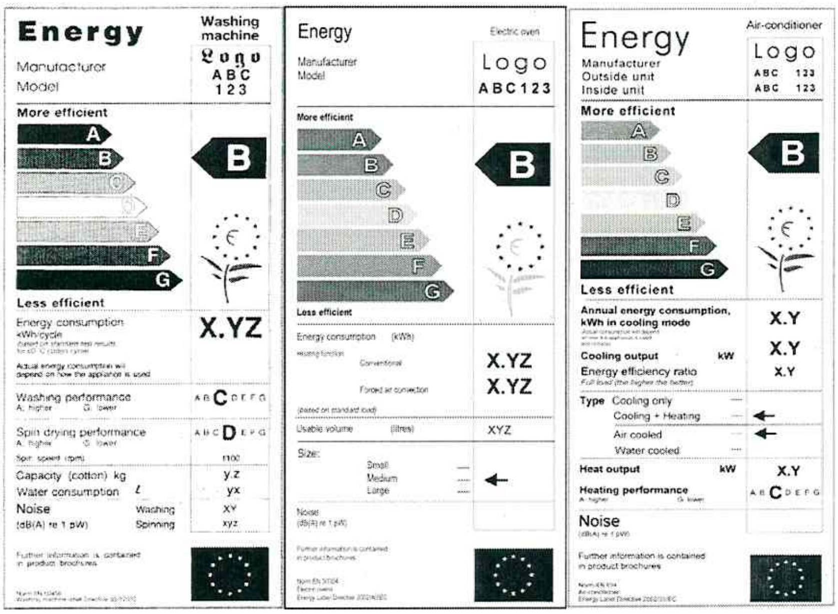
Rules

According to Article 2(1) ELD and §3(1) EnVKV, information about the consumption of electricity, other forms of energy and other essential resources and supplementary information has to be brought to the final customers' attention for selected household appliances. Suppliers of household appliances are required to provide dealers with a label for their product free of charge (Art. 3(1) ELD, §4(1) EnVKV). Dealers are obliged to attach the label to the appliance in a clearly visible position – on top or on the front of the appliance – whenever it is displayed for sale (Art. 4 ELD, §4(2) EnVKV). The label has to be in the relevant language version and its comprehensibility and visibility must not be hampered by other information or imprints.

The major component of the label is the indication and illustration of the energy efficiency class of an appliance (see Figure 2-4). Classes range from A (A++ for refrigerators and freezers) for most efficient appliances to G for least efficient appliances. Classes are determined on the basis of energy efficiency indicators which vary with the type of appliance (see Annex A.4 of this book). For example, for washing machines, kilowatt-hours spent for washing one kilogram of laundry in a standard 60°C cotton cycle is used as an indicator. Washing machines are ranked A if the indicator is below 0.19 kWh per kilogram of laundry and G if it exceeds 0.39 kWh per kilogram of laundry. In addition, energy consumption has to be indicated for each appliance in kilowatt-hours per year, cycle or standard load except for lamps. Moreover, labels for all types of appliances besides lamps have to contain information about the supplier of an appliance and the corresponding model identifier of the supplier. Further information requirements vary with the type of appliance (see Annex A.5 of this book). They may refer, among other, to the capacity and water consumption of an appliance, specifications about the type of appliance or expected noise exposure.

In addition to labels, suppliers of appliances are obliged to provide fiches for their products (Art. 3(2) ELD, §4(1) and §4(4) EnVKV). Fiches have to contain the information provided by the label and further information which supplements and specifies this information (see Annex A.5 of this book). Suppliers have to include the fiche in the product brochures. Dealers have to keep the fiches ready for interested customers (§4(2) EnVKV).

Figure 2-4: Energy Labels for Washing Machines, Electric Stoves and Air Conditioners



Suppliers of appliances are required to furnish technical documentations for their products (Art. 2(3,4) ELD, §6 EnVKV). This documentation has to be made available for inspection upon request in order to verify the accuracy of the information provided in the labels and fiches. If the obligations according to relevant EU and national regulations are not met, suppliers and dealers of appliances may be sanctioned by a fine of to 50,000 euros (§2 EnVKV).

2.3.8 Voluntary Agreement

According to the 2000 voluntary agreement with the Federal Government, German industry associations have committed to reducing their specific greenhouse gas emissions by 35 percent by 2012 compared to 1990 emissions levels (Bundesregierung 2000b). This agreement was amended in 2001 when associations of energy producers committed to reducing CO_2 emissions by 45 million tons annually by 2010 compared to 1998 levels (Bundesregierung 2001). In turn, the Federal Government commits to abstaining from implementing too stringent climate policies for the industry sector.

Objective

The voluntary agreement aims at climate protection. This is seen as an integral part of Germany's environmental as well as industry and energy policy – and of sustainable development in general. It is emphasized that climate protection has to meet ecologic, economic and social criteria. Moreover, it has to be in line with other goals such as resource conservation, security of energy supply and recovery of energy supply costs (Bundesregierung 2000b, p. 1).

Scope

The 2000 agreement covers emissions of firms belonging to the 19 associations of the German industry that have signed the agreement (Bundesregierung 2000b, p. 1). These associations represent firms from a variety of industrial sectors, including electricity, mining, chemistry, petroleum, minerals, ferrous and non-ferrous metals, electrotechnology, textile, pulp and paper as well as sugar (RWI 2008, pp. 10-11). The agreement refers to all greenhouse gases listed in the Kyoto Protocol (Bundesregierung 2000b, p. 4). The 2001 amendment to the agreement only binds associations of the energy industry including the Association of the Electricity Industry, the Association of the Integrated Industry, the Association of Municipal Utilities, the Association for Gas and Water and the Association of Industrial Energy Producers and Consumers. Moreover, the amendment only refers to CO_2 emissions (Bundesregierung 2001, p. 3).

Rules

Under the 2000 voluntary agreement, industry associations commit to reducing their specific greenhouse gas emissions by 35 percent by 2012 compared to their 1990 emission levels (Bundesregierung 2000b, p. 4). Specific emissions refer to the emissions per unit of output reduced, not to overall emissions. Since the agreement is made between the Federal Government and 19 industry associations as a whole, each association is supposed to adopt declarations on its individual reduction aims and on how these aims are to be attained (Bundesregierung 2000b, p. 7). Electricity producers commit to reducing their annual CO_2 emissions by 25 million tons by 2015 compared to 1990 emissions levels. This implies a reduction to 264 million tons of CO_2 emitted annually. However, this commitment is based on the condition that electricity generators are free to decide about the mix of energy sources. In particular, the operation of nuclear power plants must not be restricted. Moreover, operators must be allowed to extend the useful life as well as the capacity of their nuclear power plants (RWI 2008, p. 248). Most other associations have agreed to reduce the specific emissions in the corresponding sectors by 10 up to 69 percent (RWI 2008, p. 10).

The 2001 amendment requires the energy industry to reduce CO_2 emissions by 45 million tons annually until 2010 compared to 1998 emission levels. A 20 million tons reduction is to be attained by maintaining and modernizing existing CHP plants as well as constructing new CHP plants. This commitment applies particularly to electricity generators and industrial operators of power plants. The remaining 25 million tonnes have to be reduced by other measures in the energy sector. Electricity generators commit to modernizing their power plants, accelerating the employment of renewable energy sources and launching efficiency campaigns. The natural gas industry is to improve heating and hot water technologies (Bundesregierung 2001, p. 1).

If these commitments are successfully implemented by the industry associations, the Federal Government abstains from implementing command-and-control policies and a mandatory energy audit to attain climate targets. However, this commitment does not include policy initiatives mandated by the European Union. Moreover, the tax breaks for the manufacturing industry under the electricity tax (and the Ecological Taxation Reform in general) have been granted to reward the commitments of industry (see Section 2.3.3). The Federal Government promises to safeguard that further developments of the Ecological Taxation Reform do not hamper the international competitiveness of the German industry. Moreover, the overall energy tax burden shall not compromise the firm's profitability. The government will also engage in preventing distortions of competition possibly resulting from the Kyoto Protocol, the EU Burden Sharing

Agreement or flexible mechanisms such as International Emissions Trading, Joint Implementation or Clean Development Mechanism (Bundesregierung 2000b, pp. 5-6; 2001, p. 2) (see Section 2.2.1).

The voluntary agreement between the industry associations and the Federal Government as well as the individual declarations of the industry associations are continuously monitored by an independent economic research institute. Costs of monitoring are borne by the Federal Government and the industry associations in equal shares (Bundesregierung 2000b, p. 6; 2001, p. 2).

2.4 Summary of the Main Characteristics of the German Policy Mix

Chapter 2 has introduced the case study of the German climate policy mix in the electricity sector. This case study is meant to inspire the theoretical discussion of a policy mix in subsequent chapters. Moreover, it is employed in Chapter 6 to derive an extensive economic evaluation of an actually existing policy mix. The presentation of German climate policies has revealed two important characteristics of the German policy mix: its high representativeness for climate policies adopted throughout the industrialized world and its outstanding complexity in background, design and implementation. To summarize Chapter 2, it is worth to emphasize both characteristics once more.

The *high representativeness* of the German policy mix is first of all owed to that fact that most of the policies are guided by EU legislation. In fact, only the low-interest loans and the voluntary agreement are within the sole responsibility of the German Federal Government. All other policies are determined by EU Directives (which establish a regulatory framework and have to be implemented into national law) or EU Regulations (which come into force in Member States directly). This fact implies that a similar policy mix can be found in all other EU Member States as well. Moreover, the German policy mix represents the whole range of possible policy types. It includes market-based approaches (EU Emissions Trading Scheme, electricity tax, feed-in tariff for renewable energy sources, bonus for combined heat and power, low-interest loans), command-and-control measures (eco-design standards), information programmes (energy labelling) and voluntary agreements. Therefore, the German policy mix encompasses many characteristic policy combinations that can be observed not only in other EU Member States but worldwide. The high representativeness makes the German example a very useful case study to guide the theoretical as well as the empirical analysis of a policy mix. It implies that many of the expected findings are not restricted to the German case study but are of relevance for other country contexts as well.

The second important characteristic of the German policy mix is its *high degree of complexity*. The presentation in this chapter revealed that the complexity can be attributed inter alia to sequential and multi-level policy-making. Policies have been implemented at different points in time throughout the last decade of the twentieth century and the first decade of the twenty-first century. They were designed and implemented step by step by governments with different political objectives and strategies. The political culture with respect to designing environmental policies has been subject to remarkable shifts. Policy decisions have been taken in the light of an increasing awareness of climate change and its impacts. Moreover – as has been outlined above – climate policies have been adopted at different political levels. Some measures are only driven by national legislation. Other policies were mandated by European Directives and had to be implemented into German law thereafter (EU Emissions Trading Scheme, energy labelling). In contrast, some measures were first implemented by the Federal Government and had to be integrated into a European legal framework which was adopted later on (electricity tax, feed-in tariff for renewable energy sources, bonus for combined heat and power). Finally, eco-design standards are an example of a policy implemented by the EU without any discretion left to the Member State.

The complexity of the policy mix becomes obvious in many respects. As has already been emphasized above, the policy mix includes the entire panoply of policy types that are commonly discussed in economics. In addition, policies affect the entire product chain of electricity. Some aim at greening electricity supply by substituting emission-intensive fuels and/or improving the energy-efficiency of electricity generation (EU Emissions Trading Scheme, feed-in tariff for renewable energy sources, bonus for combined heat and power, voluntary agreements, low-interest loans). Others focus on reducing electricity demand by reducing electricity consumption and substituting electricity-intensive consumer goods (electricity tax, energy labelling, eco-design standards, low-interest loans). Finally, the individual design of each single policy is very diverse and complex. Notably, many policies do not only pursue the goal of mitigating climate change. Rather they may address multiple objectives. This has been highlighted, for example, for the feed-in tariff for electricity generation from renewable energy sources. Apart from reducing GHG emissions, it aims at resource conservation, energy security and technological as well as economic development. In addition, the scope of regulation may vary with policies. Some measures, such as the feed-in tariff for renewable energy generation, the electricity tax or the eco-design standards are restricted to the electricity sector. Other policies, most noteworthy the EU Emissions Trading Scheme, also cover other sectors next to the electricity sector. Moreover, in several cases, rules and incentives set out by policies are not homogeneous for economic actors affected. For example, different electricity tax rates are imposed on household and industry customers. Consequently, policies hardly correspond to simple, theoretical concepts of policy design.

The diversity and complexity of the policy mix seems to be desirable on the one hand. Climate change being a complex challenge, one may conclude that policy makers should undertake every action possible to reduce greenhouse gas emissions. On the other hand, this diversity raises concerns about the integrity of the policy mix. Can policies actually complement each other? Or do contradictions between policies and adverse effects of certain policy combinations impair the overall performance of the policy mix? These questions cannot be answered on a general level. An appropriate economic analysis has to take into account that policies in the mix exhibit quite different and complex characteristics and have been adopted against a variety of backgrounds. This requirement poses an important challenge to the economic theory of a policy mix. Addressing this challenge promises to deliver new and valuable insight about the economics of a policy mix – which is useful for evaluating climate policies in Germany and worldwide.

The evaluation of the policy mix will be conducted in the following chapters. Chapters 3 to 5 are devoted to advancing the economic understanding of a policy mix. In many instances, theoretical developments are inspired by the German case study as it has been presented in this chapter. Chapter 6 then applies this new theoretical insight to provide an extensive applied evaluation of the German climate policy mix.

3 Rationales for Using a Policy Mix for Pollution Control

3.1 Introduction

The main criterion guiding the analysis of policies and their mixes throughout this as well as subsequent chapters will be efficiency. Neo-classical theory suggests that markets may fail to allocate resources efficiently to different uses under certain circumstances. Environmental problems are usually dedicated to the presence of a pollution externality, one of the most prominent types of market failure. It is commonly highlighted in environmental economics that a pollution externality can be corrected efficiently by single policies. This recommendation is derived under the assumption that markets are perfect in other respects, and that policies can be perfectly implemented at zero cost. Under these conditions, there is no need for implementing a policy mix to cope with an environmental problem. A policy mix – as it has been described for Germany's climate strategy in the electricity sector in the previous chapter – must be redundant – or even inefficient.

However, there is an emerging field of economic research that discusses rationales for using a policy mix for pollution control. Studies highlight that in a second-best world – where deviations from the ideal of the perfect market are numerous and policy implementation is subject to a variety of constraints – a policy mix may be superior to single policies in terms of efficiency. Policies applied simultaneously may reinforce each other and compensate for disadvantages of single-policy strategies.

Available economic studies on policy mixes are conducted against a large diversity of backgrounds. On the one hand, they refer to different kinds of pollution problems, such as air pollution, water pollution, climate change or waste management. On the other hand, conclusions in favour of a policy mix are drawn under a variety of different conditions, such as knowledge spillovers or uncertainty. This large diversity raises the demand for a systematic and comprehensive overview of the economic literature on using a policy mix. Moreover, the question arises whether there is a set of general rationales for using a policy mix which overarches the economic literature despite its diversity.

This chapter provides a review of existing economic studies that bring forward arguments in favour of a policy mix for pollution control. The studies taken into account focus on situations in which the combination of policies improves the efficiency of resource allocation. The review does therefore not address economic research which finds an existing policy mix to be inefficient.[18] The review organizes the diverse body of policy mix literature. It identifies general economic rationales for using a policy mix that underlie all of the reviewed studies. Thereby, this chapter reveals substantial limitations of the dominant strand of theoretical economic analyses considering one policy only. It is clarified that one cannot assume that a single policy is a superior strategy under most of the realistic conditions. Studies arguing in favor of using a policy mix do not refer to exceptional cases. Rather it is shown that there is an important systematic set of economic reasons for using a policy mix.

Rationales can be employed to evaluate an existing policy mix as it has been presented in Chapter 2 for Germany. If some of the rationales hold for (some of) the policy combinations existing, the policy mix can be assumed to contribute to an efficient solution of the climate change problem. Otherwise, policy combinations are presumably redundant at best – or inefficient at worst. However, it may be questioned to what extent results from the studies reviewed can actually be transferred to the case of the German policy mix. The review therefore also identifies and highlights possible restrictions of existing studies. These restrictions reveal promising strands for advancing the economic theory of a policy mix. Some very promising avenues are addressed in the subsequent chapters.

To guide the review, an analytical framework is developed. It helps to identify overarching rationales for using a policy mix as well as avenues for future investigation. The framework is based on the seminal work by Coase (1960). It incorporates transaction costs into the analysis of pollution problems and of possible solutions to overcome them. Moreover, the framework distinguishes between private governance structures (market, firm and bilateral bargains) and public governance structures (government regulation) to address a pollution externality. It is shown that the presence of a pollution externality is not only attributed to a market failure – as widely presumed in neo-classical economics – but to the failure of private governance structures in general. Thus, the framework introduces concepts of New Institutional Economics into the analysis of a policy mix. Based on this insight, it is argued that two deviations from the traditional theoretical model may provide a rationale for using a policy mix: On the one hand, a pollution externality may not exist in isolation but be reinforced by other types of failures of private governance structures, such as technological spillovers and asymmetric information. In this case, multiple policies may be needed to correct all failures simultaneously. On the other hand, a single

[18] Where appropriate, some studies analyzing the possible inefficiency of a policy mix are discussed in subsequent chapters.

first-best policy may not be implemented perfectly because the corresponding transaction costs would be too high. A policy mix may then be a useful means to reduce transaction costs. Along these two rationales, the literature review is organized.

The chapter is structured as follows: Section 3.2 presents the analytical framework. It introduces the guiding criterion of efficiency. It points out the restrictions of neoclassical rationales for using regulation based on market failure. Coase's rationale for using regulation is proposed as a more appropriate concept for justifying government intervention. Based on this elaboration, the two rationales for using a policy mix for pollution control are derived. Section 3.3 presents the strands of policy mix literature referring to the first rationale. Section 3.4 introduces studies related to the second rationale. Section 3.5 provides a summary and highlights possible avenues of future research.

3.2 Analytical Framework

3.2.1 Efficiency Criterion

Above all, economic theory is dedicated to analyzing the allocation of scarce resources, including natural resources such as atmosphere, air, water or soil. The allocation of resources determines what goods are produced, which combinations of inputs are employed for producing these goods, and how output is distributed between individuals (Perman et al. 1999, p. 105). The most important criterion for evaluating the allocation of resources is efficiency. In terms of welfare economics, an allocation is efficient if it maximizes social welfare. Social welfare is a function of private utilities of all individuals belonging to the society. Social welfare is maximized if resources are allocated such that the utility of an individual cannot be improved without making another individual worse off. This is also referred to as Pareto efficiency (Pindyck and Rubinfeld 2001, p. 567). Thus, efficiency implies that scarce resources are allocated to different economic activities such that the highest degree of social welfare possible is produced per unit of resource.

Is somewhat weaker criterion is that of cost-effectiveness. This criterion requires that a certain level of welfare be attained with the least possible use of resources. Thus, cost-effectiveness analysis reflects on whether a certain goal is achieved at least costs – but not on whether this goal has been chosen optimally. In this respect, cost-effectiveness does not necessarily mean efficiency. This will only be the case if the goal has been determined to be Pareto-optimal. In turn, an efficient allocation of resources will always be cost-effective.

Efficiency will be the guiding criterion throughout the subsequent chapters of this book. In reality, however, it may be a challenging task to determine the efficiency of resource allocation –

particularly due to the difficulties in determining the related cost and benefits. Cost-effectiveness may then be a more pragmatic criterion since it only requires insight about costs. Where appropriate, the criterion of cost-effectiveness will therefore also be discussed in the remainder of this book.

Policies implemented by governments to guide the behaviour of individuals will have an impact on the allocation of resources. The efficiency criterion requires that policies or policy mixes contribute to attaining a maximum of social welfare. Whether policies and policy mixes may actually provide such a contribution and how they should be designed optimally depends on the assumptions about the characteristics of the economy and the policy-making process. In the following, different assumptions and their implications for optimal policy design will be highlighted.

3.2.2 Neoclassical Rationale for Using Regulation

According to the first theorem of welfare economics, mutually beneficial trades in perfectly competitive markets result in an equilibrium allocation of resources that is Pareto efficient. That is, decentral market transactions result in an optimal outcome (Pindyck and Rubinfeld 2001, p. 574). The model of perfectly competitive markets rests on a variety of assumptions. In order to work properly, market economies require well-defined and allocated private property rights (see, e.g., Bromley 1997, p. 6). Perfect competition implies that suppliers and buyers on the market take the price as given. Suppliers can enter and exit the market at no special costs. Buyers can easily switch from one supplier to another. Goods traded on the market are homogeneous. That is, they are perfectly substitutable with each other (Pindyck and Rubinfeld 2001, pp. 252-253). Individuals behave rationally and have no personal, spatial or temporal preferences. They are perfectly informed about prices, goods and the behaviour of other market participants, and react infinitely fast (Kreps 1990, pp. 264-265). In addition, the decision problems market participants have to face are independent from each other (Schlieper 1969, p. 10). If these assumptions held for all resources and goods, there would be no need for government regulation (Pindyck and Rubinfeld 2001, p. 574). The role of the government would be limited to protecting private property rights. Natural resources would be allocated and used in an efficient manner. In reality, however, markets deviate from the model of perfect competition. These deviations may provide for market failures. The presence of a market failure is commonly presumed in neoclassical economics as a rationale for government regulation. In the following, types of market failures as well as corresponding policy recommendations and their restrictions are introduced.

3.2.2.1 Market Failures

Market failures are conditions under which market transactions do not result in an efficient allocation of resources, i.e. do not produce Pareto optimal welfare (see, e.g., Bator 1958). Three types of market failure, which are of particular importance for environmental problems, will be discussed in the following: pollution externalities, technology spillovers and asymmetric information.[19] [20]

Pollution Externalities

Environmental problems are commonly assumed to be associated with pollution externalities. Baumol and Oates (1988, p. 17) identify two conditions for the presence of an externality. First, individual decision problems are not independent from each other. Rather, "[...] some individual's (say A's) utility or production relationships include real (that is, non-monetary) variables, whose values are chosen by others (persons, corporations, governments) without

[19] With respect to environmental problems, public goods are usually mentioned as an important type of market failure as well (Sterner 2003, p. 27). Public goods are characterized by non-excludability (if a public good is provided for some individuals, others cannot be excluded from using it) and non-rivalry (the consumption of a good by one individual does not diminish the amount of the good available to others). As will be explained in Section 3.2.3, non-exclusiveness is a prerequisite for the existence of externalities. In this respect, (partially) public goods can be considered the underlying source of externalities rather than a separate type of market failure.

[20] Market power in the output market is another important type of market failure (see, e.g., Pindyck and Rubinfeld 2001, p. 592). Market power will not be discussed in further detail in the context of this book since it is of limited importance for discussing environmental policy mixes. It is highlighted that the efficiency of environmental taxes may actually be distorted if they are imposed on monopolistic or oligopolistic polluters (Buchanan 1969; Lee 1975; Asch and Seneca 1976; Barnett 1980; Misiolek 1980; Levin 1985; Simpson 1995; Carraro and Soubeyran 1996; Katsoulacos and Xepapadeas 1996a). Likewise, market power in the output market may also impair the efficiency of tradable permit schemes, although the distortion is of less significance (Lee 1975; Malueg 1990; Requate 1993; Mansur 2004). In the presence of a pollution externality and market power in the output market, an optimal policy mix would encompass a Pigovian tax on pollution and an output subsidy to the monopolist or oligopolist (Baumol and Oates 1988, pp. 80-81; Ebert 1992; Gersbach and Requate 2004). However, this policy mix is not addressed in this chapter since it is of little practical relevance. In fact, a regulator will usually have neither the authority nor the inclination to implement a subsidy to firms executing market power (Cropper and Oates 1992, p. 684).

Market power may not only be an issue in output markets but also in markets for tradable permits. Several authors highlight that market power in the permit market may hamper the efficiency of permit trading (see, e.g., Hahn 1984; Misiolek and Elder 1989). However, suggestions for improvement do not include policy mixes but rather focus on modifications of the permit trading design (see, e.g., Smith 1981; Hahn 1984; Ledyard and Szakaly-Moore 1994; Chavez and Stanlund 2003).

particular attention to the effects on A's welfare". These variables may improve (positive externality) or diminish (negative externality) the well-being of A. Second, the resulting benefits or costs are not compensated by the market mechanism and, thus, not taken into account by the originator of an externality. As a consequence, social and private costs of an activity do not correspond and the activity is performed to an extent that is below or above the social optimum if it is regulated by the market mechanism only. Pollution problems are typically attributed to negative externalities. Climate change caused by greenhouse gas emissions is one example. Emitters of CO_2 maximize their utility without considering the costs other individuals have to incur – e.g. due to changes in average temperatures or extreme weather events. Consequently, the emitter will produce and emit more than what would be socially optimal. However, environmental problems may also be related to positive externalities. For example, a farmer that afforests his farmland produces a positive externality. The trees sequester carbon dioxide and may thus contribute to the reduction of global warming and its effects on humans. Yet, the farmer considers his own benefits only and will therefore plant fewer trees than what would be desirable at the social level.

Technology Spillovers

Technology spillovers can be considered a second type of market failure. They arise in the process of technological change. Technological change is an essential force in our economy which makes it possible over time to use given input vectors to produce output vectors that were not previously feasible (Jaffe et al. 2002, p. 42). Schumpeter (1942) distinguished three stages of technological change: invention (when a new product or process is first developed), innovation (when the new product or process is introduced to the market) and diffusion (when the new product or process gradually becomes widely adopted). Spillover effects may arise with respect to innovation as well as diffusion.

Innovation is driven inter alia by firms' investments in research and development (R&D). These investments generate new, innovative knowledge about technologies. On the market, other actors may benefit from the innovator's knowledge about a new technology without having invested in R&D themselves and without paying compensation to the innovator. This is because knowledge is a public good. Despite patents, which are meant to protect intellectual property rights, an innovator will usually be unable to perfectly protect the knowledge generated by him. Thus, a spillover in fact represents a positive externality. Consequently, the innovator may not be able to appropriate the complete social returns of his innovation. His incentives to innovate are reduced to those returns that he can actually realize at the private level. This results in significant underinvestment in R&D, relative to the social optimum (Jaffe et al. 2005, p. 167). The market fails. This effect is only marginally offset by strategic overinvestment in R&D by firms that try to innovate ahead of their rivals (Grubb and Ulph 2002, p. 93).

Spillovers also occur in the context of technology diffusion. The gradual adoption of a new technology is characterized by dynamic increasing returns, i.e. the cost of adopting the technology is the lower for an additional user the more actors already use the technology. Dynamic increasing returns can be attributed to (1) learning-by-using, (2) learning-by-doing and (3) network externalities. (1) Learning-by-using refers to a situation where the adopter of a new technology produces a positive externality for others by generating information about the existence, the characteristics and the success of this technology. This information may be available to everybody. (2) Learning-by-doing producers of a new technology gain experience as their cumulative output increases. This experience results in a reduction of production costs with increasing levels of cumulative production. The experience may spill over to other firms and, thus, represent a positive externality as well. (3) With network externalities a new technology becomes more valuable to a user as other users adopt a compatible product – as with telephones, for example. That is, new adopters benefit from positive externalities generated by existing users of a technology. In either case, the incentive to invest in diffusion is unlikely to correspond to the social benefits of diffusion. Thus, market allocations result in inefficiently low levels of diffusion of new technologies (Jaffe et al. 2005, p. 167). Instead, a lock-in of dominant existing (but potentially inefficient) technologies and a lock-out of viable alternatives may occur (see, e.g., Arthur 1989).

Asymmetric Information

Asymmetric information is abundant in economies which are based on the division of labor. These economies are characterized by principal-agent-relationships. Instead of producing a good or service himself, the principal assigns an agent to do so. Principal-agent relationships are typically characterized by information asymmetries in favour of the agent. Asymmetric information on markets can be attributed to the public good character of information (Jaffe et al. 2005, p. 168). In order to demonstrate the value of his information, the provider has to reveal at least part of it. Moreover, information may be known to others once it is transferred or used. Other actors may take advantage of the information without compensating the provider for its efforts.[21] Therefore, the possessor of information may not be able to appropriate the total social value information provision brings about (Keck 1987, p. 139). His incentives to reveal information are reduced to his private returns. In fact, these returns may well be zero or negative. For example, if a producer provides information about the bad quality of his products, he will not be rewarded by the market but rather lose clients. The possessor of information may therefore not have an incentive at all to disclose it. Consequently, the provision of information on the market is suboptimal and asymmetric information arises between market participants. The market fails.

Asymmetric information gives rise to so-called agency problems (Jaffe et al. 2002, p. 54). Economic actors may make adverse selections. If a buyer (i.e. the principal) cannot distinguish between the characteristics of goods he will tend to purchase cheaper goods with potentially worse quality. Consequently, low-quality goods crowd out high-quality goods. Akerlof (1970) demonstrated that adverse selection may even provide for markets to collapse. If a principal-agent-relationship does not only refer to a singular action (such as the purchase of a good) but to a sequence of several actions subsequent to contracting (such as an insurance of an agent or the employment of an agent by a firm), additional problems may arise. The principal may often only be able to detect a certain outcome. However, relating an outcome to actions of a specific agent may be difficult if agents are numerous. Moreover, the outcome may not only depend on the effort of the agent but on stochastic, exogenous factors as well (Häder 1997, p. 66). Agents may use their information advantage and deliberately cheat others to maximize their private utility – a behavior called moral hazard (see, e.g., Becker 1968).

3.2.2.2 Policy Recommendations and Restrictions

In neoclassical economics, the existence of a market failure is usually understood as a rationale for government regulation. Pigou (1932, p. 173) argues that in the presence of an externality, decentral market transactions will not "make the national dividend a maximum; and, consequently, certain specific acts of interference with normal economic processes may be expected [...] to increase the dividend." Likewise, Samuelson (1954, p. 388) points out that no decentralized pricing mechanism can be used to determine the optimal level of goods characterized by collective consumption, such as many natural resources and information.

[21] In fact, asymmetric information can therefore be interpreted as a result of externalities of information provision. These can be considered positive if an actor is presumed to have the right to conceal information and negative if he has the duty to disclose information.

In order to reduce negative externalities, several types of policies have been suggested. The oldest approach is a command-and-control policy that prohibits or restricts harmful economic activities. Market-based approaches were developed by Pigou (1932) and Dales (1968). Pigou suggested that a tax equal to the marginal external costs of an economic activity, i.e. the difference between social and private costs, should be imposed on producers of this activity. Due to this incentive, these activities would be limited to an optimal level. Dales proposed a laterally reversed solution. According to his approach, the government determines a socially optimal level of an activity. Subsequently, allowances that allow for the performance of a certain level of this activity are allocated to individuals and can be traded among them. Consequently, a price emerges on the allowance market that reflects the difference between the social and the private costs of the activity and adjusts the amount of individual activities to the socially optimal level. This latter approach has been implemented, for example, in the EU by the ETS (see Section 2.3.1).

To correct for the inefficiencies stemming from positive externalities, Pigou (1932) also suggested paying a subsidy to the producer of the externality. The subsidy should be equal to the marginal external benefits of the respective economic activity. This subsidy would promote the extension of the activity to the socially optimal level. This concept has been applied to cases of knowledge spillovers, i.e. positive externalities of knowledge generation. Studies argue in favour of subsidies to private R&D expenditures in the case of R&D spillovers (see, e.g., Spence 1984; Dixit 1988; Stoneman and Vickers 1988; Segerstrom 1991; Hinloopen 1997). Likewise, output subsidies are recommended in the presence of positive externalities related to the diffusion of technologies (see, e.g., Bardhan 1971). Policies which correspond to these approaches have been implemented in Germany by the feed-in tariff for electricity generation from renewable energy source (see Section 2.3.2), the CHP bonus (see Section 2.3.4) and the low-interest loans for technology innovation and diffusion (see Section 2.3.5).

Measures recommended to overcome information asymmetries in markets are quite diverse. Policies proposed include licensing, certifying and labelling (see, e.g., Akerlof 1970, p. 500; Shapiro 1986). These measures provide information to market participants. They may be either mandatory or voluntary. Moreover, costs of these measures may be born by the informing actor only or be subsidized by the government. An example is the energy label for household appliances implemented throughout the EU (see Section 2.3.7). Alternatively, minimum quality standards have also been found to be welfare-increasing in the presence of asymmetric information (see, e.g., Leland 1979). The EU eco-design standards follow this approach (see Section 2.3.6).

The overarching statement made by the theoretical studies is that one single policy can correct a market failure efficiently and thus contributes to attaining the optimum of social welfare. This implies that the use of multiple policies has to be considered a redundancy at best and an inefficient "policy mess" at worst (Gawel 1991, p. 7; Johnstone 2003, p. 5). This reasoning also reflected in Tinbergen's (1952) statement that a certain number of independent policy targets could be solved by an equal number of policies. Tinbergen's rule is often cited as a rationale for a policy mix – but not in the sense of a policy mix as it is defined in this book. The rule basically implies that in the presence of only one target, such as correcting a pollution externality, a single policy is sufficient. The majority of environmental-economic studies has therefore analyzed the performance of single policies or compared two or more policies for pollution control.

However, this approach is subject to several limitations. First of all, it usually considers only one type of market failure: a pollution externality, a knowledge spillover or asymmetric information. Apart from the market failure under consideration, markets are assumed to correspond to the model of perfect competition. Secondly, policies suggested to overcome the market failure are expected to be implemented perfectly. The regulator is assumed to incur no additional costs, which may hamper implementation. This assumption leads to an even more profound criticism of the concept of market failure as a rationale for government regulation. As Coase (1988, p. 26) and Zerbe and McCurdy (1999, p. 561) highlight, market failures are ubiquitous in real economies. This would provide a universal rationale for governments to intervene in the market. Thus, this concept does not seem to be appropriate to guide real politics (Coase 1988, p. 26). What is more, there is evidence of market failures which were corrected efficiently by private action. This can be illustrated by two examples: (1) Operators of lighthouses are said to be unable to appropriate the benefits they provide to ship owners by saving lives and cargoes (Samuelson 1964, p. 45). Thus, lighthouses produce a positive externality which stems from the public good character of the service they provide. Neoclassical theory would suggest that lighthouses are underprovided and have to be supplied or regulated by the government. However, Coase (1974) points out that most 17th century lighthouses in England where not built or run by the government but by private actors. These levied tolls and collected them from ships at the ports. Government intervention was restricted to providing a patent to lighthouse operators which allowed them to levy the toll. (2) Likewise, positive reciprocal externalities between beekeepers and owners of apple orchards could be seen as rationale for government regulation. With respect to this example, Meade (1952) argues in favor of a system of taxes and subsidies, which have to be imposed to achieve efficiency. In contrast, Cheung (1973) provides evidence of contractual agreements between farmers and beekeepers to overcome these externalities without any government regulation.

Therefore the concept of market failure fails as a normative rationale for regulation (Randall 1983, p. 147; Zerbe and McCurdy 1999, p. 572). An alternative approach is required. Zerbe and McCurdy (1999, p. 570) emphasize that a more appropriate normative theory of government intervention rests in transaction costs. This relates to economic insight on regulation gained by Ronald Coase, which will be presented in the following section.

3.2.3 Coase's Rationale for Using Regulation

Coase (1960) was the first to show that the concept of a market failure as a rationale for using regulation was somewhat flawed. In his work, he focuses on externalities. He finds that externalities come into being because of transaction costs. Transaction costs have to be considered as well when discussing possible solutions, such as government regulation, to correct for externalities. Based on this finding, Coase concludes that an unambiguous plea for government regulation to correct for externalities cannot be made.

Transaction costs can be broadly defined as the costs of establishing, maintaining and transferring property rights in goods and resources (Cheung 1970, p. 67; Allen 1991, p. 3).[22] [23] In this sense, a transaction is defined as a transfer of property rights (which requires the

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[22] The economic attempts to define transaction costs are numerous and diffuse. Allen (1991, pp. 5-6) and McCann et al. (2005, p. 530) provide overviews. Coase (1960, p. 15) himself specifies that the costs of market transactions are the costs incurred in order "[...] 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 the 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". Dahlman (1979, p. 148) distinguishes search and information costs, bargaining and decision costs, and policing and enforcement costs as categories of transaction costs. Likewise, Bromley (1991, p. 63) mentions information costs, contracting costs, and enforcement costs. Birner and Wittmer (2004) make a distinction between decision-making costs that arise before a contract is made on the market, and implementation costs attributed to monitoring and enforcing a contract. These definitions of transaction costs mainly refer to the costs of transferring property rights. For the purpose of this chapter, it seems more appropriate though, to define transaction costs more broadly – including the costs of establishing, maintaining and transferring – as suggested by Cheung (1970) and Allen (1991).

[23] To have a property right implies that a person has secure control over a future benefit stream from a good (Bromley 1997, p. 3). Property rights include the rights to use a good and to exclude others from using it, to derive income or utility from a good, to change its form and substance, and to transfer all the rights in the good, or some rights, as desired (Furubotn and Richter 1997, p. 72).

establishment and maintenance of rights). Coase (1960, pp. 16-17) outlines that transactions can be carried out under different governance structures. On the one hand, transactions can be guided by private governance structures including the market, bargains and firms. On the market, the transfer of property rights is organized decentrally by the pricing mechanism. With bargains in the Coasian sense, property rights are exchanged with individual side payments. In firms, the decentral distribution of property rights is replaced by hierarchical decisions for certain economic activities. On the other hand, transactions can be carried out under public governance structures, such as government regulation.[24] If the government intervenes in the market by regulations, it acts as a "super-firm" that coordinates the transfer and distribution of property rights for the entire society. Coase emphasizes that with none of these governance structures the transfer of property rights is costless. Transaction costs may well differ between governance structures. However, with none of these governance structures transaction costs are *per se* higher or lower than with another.

Using the insight on transaction costs, Coase (1960, pp. 15-16) explains the presence of pollution externalities in the absence of government regulation. Property rights in natural resources, such as air or water, may not exist or may not be distributed optimally among private actors under neither private governance structure because the costs of establishing and transferring the rights would exceed the expected benefits. Therefore, many natural resources are non-excludable goods. This may allow some agents to use (and pollute) a natural resource and produce uncompensated damages to other resource users to an extent that is beyond the socially optimal level, i.e. externalities are generated. Thus, pollution externalities persist because under any private governance structure transaction costs of internalizing an additional unit of pollution damage (without government intervention) are higher than the net value of internalization, i.e. the balance of welfare gains from reducing an additional unit of pollution damage and the costs to abate it. The presence of an externality does therefore not only result from a market failure – as widely presumed in economics – but from the failure of private governance structures, including markets, bargains and firms, in general. Externalities come into being because of

[24] Williamson (1999) further developed Coase's ideas on governance structures. He distinguishes between markets, firms, hybrids of markets and firms (bargains), regulation and public agencies. Regulation and public agencies are both types of public governance structures. With regulation, the government makes private actors carry out certain transactions and produce goods. With public agencies, the government itself produces a good, and all necessary transactions are carried out within the agency. An example for the latter is national defence.

transaction costs. Dahlman (1979, p. 142) puts it like this: "In the theory of externalities, transaction costs are the root of all evil."

According to Coase (1960, p. 18), the failure of private governance structures to correct for pollution externalities is a necessary, but not a sufficient condition for regulation. Governments should only adopt a regulation to internalize a pollution externality if the expected net value of internalization exceeds the transaction costs of setting up and implementing the regulation. The Coasian rationale for using regulation for pollution control can therefore be generalized as follows:

> *A government should implement a regulation if a) the difference D of the net value of internalizing an additional unit of a pollution externality and the transaction costs of bringing it about is positive for regulation and if b) D is larger for regulation than for any private governance structure.*

It may be supposed that the net value of internalization is the same under public and private governance structures. Condition b) then requires that the transaction costs of regulation be lower under regulation than under any private governance structure. If the net value of internalizing an additional unit of an externality is below the corresponding transaction costs under government regulation, further internalization by government regulation is not desirable from an economic point of view. In this case, existing pollution externalities can be considered as (Pareto-) irrelevant (Dahlman 1979, p. 147).[25]

Coase (1960, pp. 17-18) highlights that government intervention may lead to a reduction of transaction costs compared to private governance structures when the problem to solve involves a large number of people – which is the case with most pollution problems such as smoke emissions. Above all, this can be attributed to the government's power of coercion. The government may decide about the establishment, transfer and distribution of property rights, and it monopolizes the forces within a certain territory to enforce its decisions, e.g. by the police. Yet, any government imposing a regulation may also face significant transaction costs, e.g. for determining the appropriate distribution of property rights and monitoring and enforcing the compliance of the regulated parties. Unlike firms, governments do not face competition. Thus, they are not forced to revise their operation and undertake attempts to minimize the transaction

[25] Dahlman (1979, pp. 155-156) even argues that if the government does not do any better than the market (or any other private governance structure), then there is no externality. In this sense, one can only speak of "market failure" (or failure of private governance structures in general) if the market fails to do as well as other governance structures.

costs involved on a regular basis. However, changes in the demand and supply conditions, technological innovations and improved methods of inspection may lower transaction costs over time. Goods whose provision needs to be regulated by the government today may be cheaper to provide privately in the future. In the past, markets have in fact proven to respond much quicker to such changes than governments and their legal systems (Cheung 1970, p. 68). Moreover, one has to bear in mind that transaction costs with government intervention are not only incurred by the regulator but by the regulated agents as well, e.g., for signalling their behaviour to the regulator. Therefore, it cannot be determined on a general level whether or not government regulation provides an efficient means to correct for an externality. Decisions can only be made on a case-by-case basis analyzing transaction costs empirically (Dahlman 1979, p. 156; Zerbe and McCurdy 1999, p. 570). Any economic analysis and comparison of alternative governance structures has to start from the status quo of existing institutional arrangements in order to be able to derive appropriate recommendations (Hansjürgens 2000b, p. 102). Zerbe and McCurdy (1999, p. 571) point out: "No theory captures the variety of institutional arrangements that people have developed to resolve collective problems." Economists neglecting institutions and transaction costs have to be blamed for practising "blackboard economics": "They paint a picture of an ideal economic system, and then, comparing it with what they observe (or they think they observe), they prescribe what is necessary to reach this ideal state without much consideration for how this could be done. The analysis is carried out with great ingenuity but it floats in the air" (Coase 1988, p. 28).

3.2.4 Rationales for Using a Policy Mix

When discussing the use of regulation, Coase (1960) is not specific about what kind of policies he refers to, and whether there should be one or multiple policies to correct for an externality. Nevertheless, based on his rationale for using regulation, at least two rationales for using a policy mix for pollution control can be derived: First, if a pollution externality is reinforced by other types of failures of private governance structures, one corrective policy may be needed for each failure (Section 3.2.4.1). Second, a policy mix may help to correct for a pollution externality where regulation with a single first-best policy would involve (prohibitively) high transaction costs (Section 3.2.4.2).

3.2.4.1 Multiple Failures of Private Governance Structures

Pollution externalities may be coupled with and reinforced by failures of private governance structures in other fields. In this case, correcting for the pollution externality only does not necessarily provide for an optimal allocation of resources. Rather, it may be necessary to correct all failures simultaneously by implementing at least one policy for each failure. Two types of failure of private governance structures are of particular importance for environmental pollution problems: (1) Technological spillovers and (2) asymmetric information (see Section 3.2.2.1). Spillovers reinforce pollution externalities in a dynamic perspective while asymmetric information does so in a static perspective.

(1) In Coase's (1960) terms, technology spillovers arise because the transaction costs of defining and transferring property rights in technological knowledge generated by innovation or diffusion are higher than the net value of reducing spillovers. Transferring Coase's insight on pollution externalities to the case of technology spillovers reveals that the latter cannot only be attributed to the failure of the market – as would be done by neoclassical economics. Rather, the presence of technology spillovers implies that other types of private governance fail as well. Actors could form research joint ventures, which pool and coordinate the R&D efforts of several firms (which can be considered as a type of bargain) (Katsoulacos et al. 2001). The innovating firm could also merge with other competitors into one firm (see, e.g., D'Aspremont and Jacquemin 1988). Both approaches would allow the innovator to appropriate more of the social benefits of his innovation. The presence of knowledge spillovers thus indicates that under either type of private governance the net value of reducing an additional unit of spillover is smaller than the transaction costs to bring this reduction about.

Technological spillovers may reinforce pollution externalities positively, i.e. the worse the spillover, the worse the pollution externality. For example, technological spillovers may hamper the development of new and less costly technologies to mitigate pollution externalities. In the future, pollution reduction will be costlier than necessary and the level of the pollution externality will be worse compared to a situation with no technological spillovers.

(2) Applying Coase's approach to asymmetric information reveals that this problem arises because the transaction costs of defining and transferring property rights in information are too high. As pollution externalities and knowledge spillovers, asymmetric information does not only represent a market failure. Its presence indicates that other private governance structures fail as well. Economic actors could privately initiate labeling and certification systems – which can be considered as one form of bargains. Moreover, information asymmetries could be overcome in a firm. This is an important reason for vertical integration between producers and suppliers (see, e.g., Crocker 1983). However, information asymmetries may also prevail within a firm and cause internal agency problems (Gabel and Sinclair-Desgagné 1998, pp. 90-91; Jaffe et al. 2002, p. 54).

The presence of asymmetric information implies that under any private governance structure the transactions costs of reducing an additional unit of asymmetric information is higher than the net value of the reduction.

As technological spillovers, asymmetric information may reinforce pollution externalities positively. For example, buyers who have a preference for non-polluting goods may not be able to identify such goods because of asymmetric information between the buyer and the supplier of the good. Buyers are therefore likely to acquire cheaper, but more polluting goods. Consequently, the pollution externality will be worse and correcting it will be costlier than in a situation with perfectly informed market participants.

The presence of multiple failures of private governance structures may provide a rationale for combining several policies. But as has been pointed out before, the failure of private governance structures does not necessarily justify government regulation in each case. Rather, a more complex, first rationale for using a policy mix for pollution control can be derived on this basis:

> *A government should implement a policy mix for pollution control if a) a pollution problem can be attributed to multiple failures of private governance structures which reinforce each other, b) for each of the failures, the difference D of the net value of reducing an additional unit of the failure and the transaction costs of bringing it about is positive for regulation, and c) for each of the failures, D is larger for regulation than for any private governance structure.*

If these conditions are met, the optimal policy mix encompasses at least one policy for each type of failure of private governance structures.

3.2.4.2 High Transaction Costs of Regulation with Single First-Best Policies

Coase (1960) points out that with different governance structures, such as markets, firms or regulation, the net value and the transaction costs of internalizing an additional unit of an externality may differ. Likewise, it can be assumed that transaction costs may vary with different types of regulation, such as command-and-control or market-based policies. These can be understood as different types of public governance structures. For example, different policies may bring about different information requirements for designing a policy and monitoring and enforcing its implementation. With some single first-best policies, the transaction costs of correcting a pollution externality may be very high. Transaction costs may even attain prohibitive levels where they exceed the expected net value of internalization. Alternative single-policy options may provide for lower transaction costs. However, they may not allow for an optimal allocation of resources and thus decrease the net value of internalization. If only single-policy strategies are considered, this finding might result in the conclusion to abstain from regulation. Under certain conditions, however, combinations of the prohibitively costly first-best policy with further policies or of several alternative policies may yield a similar net value of internalization as the first-best policy at substantially reduced transaction costs. Thus, a policy mix may be desirable from an efficiency point of view even though only the pollution externality, i.e. only one type of failure of private governance structures is present.

A second rationale for using a policy mix for pollution control is therefore as follows:

> *A government should implement a policy mix for pollution control if a) the difference D of the net value of internalizing an additional unit of the externality and the transaction costs of bringing it about is larger for regulation with a policy mix than for regulation with a single policy, b) D is positive for regulation with a policy mix, and c) D is larger for regulation with a policy mix than for any private governance structure.*

Condition a) includes the case where implementing a single first-best policy would involve prohibitively high transaction costs and implementing a policy mix decreases transaction costs such that condition b) is met. It also includes the case where alternative single policies to the first-best policy could be implemented, i.e., D is positive for the single policy, but introducing a policy mix is superior since it increases the net value of internalization and/or decreases the transaction costs of internalization.

Along both general rationales for using a policy mix that have been developed above the review of economic literature is organized in the following sections. Section 3.3 refers to studies of multiple failures of private governance structures. Section 3.4 includes analyses of policy mixes in the presence of high transaction costs of regulation with first-best single policies.

3.3 Using a Policy Mix to Cope with Multiple Failures of Private Governance Structures

This section reviews the economic literature on using a policy mix to cope with multiple reinforcing failures of private governance structures. The focus will be on cases where a pollution externality coincides with technological spillovers or asymmetric information. It is brought forward that with reinforcing failures of private governance structures, single pollution control policies do not provide for an efficient outcome. Instead, a policy mix may be required to correct for all failures simultaneously and to provide for a superior outcome.

Available economic studies usually do not explicitly take the transaction costs of implementing such a policy mix into account. Rather, most studies implicitly assume that the balance of the net value of internalization and corresponding transactions costs is positive for regulation and larger than for any private governance structure. Moreover, most studies consider regulation as a means to correct for market failures while only few explicitly refer to the failures of the firm also.

3.3.1 Pollution Externalities and Technological Spillovers

3.3.1.1 Inefficiency of Single Policies

Pollution control policies designed to overcome a pollution externality provide incentives to invest in technological change only to a certain extent. Technological change may allow firms to comply with pollution policies at less cost. For example, firms may reduce their emissions tax burden if they discover and apply new and cheaper technologies for emission reduction. There has been a broad scientific debate on how high these incentives are under different types of pollution control policies. It has been highlighted that market-based policies, such as taxes and tradable permits, are superior to command-and-control policies (Zerbe 1970; Magat 1978; Downing and White 1986).[26] Incentives to foster innovation and diffusion may also vary with market-based policies (Milliman and Prince 1989; Biglaiser and Horowitz 1995; Jung et al. 1996; Fischer et al. 2003). Yet, an unambiguous ranking of market-based policies appears to be impossible (Grubb and Ulph 2002; Fischer et al. 2003).

More importantly, however, the incentives provided by either type of pollution policy are unlikely to be sufficient to overcome technological spillovers. Grubb et al. (1995, p. 428) highlight that for climate change, the effects of emission mitigation policies may be dominated by technological spillovers. They estimate that the benefits of stimulating and adjusting R&D and diffusion directly may be up to seven times larger than the direct Pigovian benefits from initial emission reductions. Parry (1995) shows that firms subject to a Pigovian emissions tax may invest too little in R&D in the presence of R&D spillovers.[27] The optimal tax rate has to be higher than the

[26] In Chapter 3, the term "permit" is used as a synonym for "allowance". This is because most of the theoretical economic literature on policy mixes uses this term. However, the legally correct term – as it is defined under the ETS, for example (see Section 2.3.1) – is "allowance". Therefore, this term will be preferred in the other chapters.

[27] Parry (1998) emphasizes, though, that lacking R&D investments due to spillover effects may be counteracted by the incentive for firms to innovate ahead of their rivals. This so-called "common pool effect" may theoretically result in excessive investment in R&D in order to capture monopolistic gains available to the first supplier of a new technology. Parry estimates that with the spillover effect partly offset by the common pool effect, inefficiencies in the R&D market are not necessarily large. In fact, if the innovator can appropriate 50 percent or more of the social benefits of his innovation, i.e. if spillovers are relatively small, the R&D efficiency gain under an emissions tax is at

Pigovian tax rate. However, this solution requires that all emission-reducing investments carry the same potential for innovation. Otherwise, increasing the tax beyond the Pigovian level will result in inefficient distortions for polluters with little innovation potential (Grubb and Ulph 2002, p. 94). This implies that a single pollution control policy is unlikely to provide for an efficient solution in a dynamic perspective in the presence of pollution externalities and technology spillovers. Rather, a more focused stimulation of innovation and diffusion is needed in addition.

3.3.1.2 Superiority of a Policy Mix

In the presence of pollution externalities and technological spillovers, a combination of pollution policies and technology policies may be superior to the exclusive use of the former. Several authors therefore suggest on a general level to combine tradable permits or taxes correcting for pollution externalities with subsidies addressing the spillovers (Johnstone 2003, p. 22; Sorrell et al. 2003, p. 69; Jaffe et al. 2005). Katsoulacos and Xepapadeas (1996b) analyze the case of a pollution externality combined with R&D spillovers. Using a simple model of a duopoly with non-cooperative behaviour, they show that the combination of a subsidy to R&D expenditures and a Pigovian emissions tax may achieve first-best levels of output, emissions and R&D spending. The optimal emissions tax is less than the marginal pollution damage. The optimal level of the subsidy depends on the deviation between the emission tax and the marginal pollution damage, the deviation between the private and social marginal product of R&D and the firms' strategic incentives to invest in R&D. Likewise, Goulder and Schneider (1999) demonstrate in a general equilibrium model that an R&D subsidy reduces the costs at which an emissions tax attains an emissions goal. In their model, the optimal subsidy should not exceed 60 percent of R&D expenditures. Fischer (2008) argues that the combination of an emissions policy with a strong support for R&D is economically justified if knowledge spillovers are significant. Moreover, she demonstrates that welfare gains from adding a R&D support policy are particularly high when the emission policy has not been set at a Pigovian level.

Similar recommendations are derived by studies which analyze the combination of pollution externalities with spillovers related to the diffusion of technologies, particularly to learning-by-doing. Fischer and Newell (2008) evaluate a policy mix by using a partial-equilibrium model of the electricity sector with fossil and renewable generation. The latter technology is subject to learning-by-doing effects and spillovers. The optimal policy mix consists of an emissions tax

least 90 percent of that in a first best outcome. Other empirical evidence suggests, however, that the spillover effect may be way more important than the common pool effect (Griliches 1992).

imposed on fossil generators and an output subsidy to renewable generators. Bläsi and Requate (2007) employ a more differentiated model, which accounts for the vertical structure of the electricity sector and free entry to the market. They differentiate between fossil and renewable generators and producers of technologies for using renewable energy sources. Only the technology producers experience learning-by-doing and related spillovers. Under these assumptions the optimal policy mix combines an emissions tax and an output subsidy as well as a market entry premium for technology producers. This rationale may also provide a justification for combining an emissions trading scheme and a feed-in tariff system for renewable energies as they have been presented for the German case study in Chapter 2. However, both policies deviate from the simplistic theoretical approaches presented above in several respects. These deviations may have important implications for actual efficiency of the policy mix. They are analyzed and discussed in detail in Chapters 4 and 6 (Section 6.2).

Analogously, Braathen (2003, p. 110) highlights that complementing a voluntary agreement to reduce pollution by a technology subsidy allows for a faster and more focused development of new technologies than if only the voluntary approach was introduced. Yet, he qualifies that paying the subsidy only to participants in the voluntary scheme will lead to a slower technological change than if the subsidy had been introduced in isolation since the conditional combination biases the selection of firms that apply for subsidies.

Katsoulacos et al. (2001) emphasize, however, that the coexistence of a pollution externality and technological spillovers does not necessarily call for a policy mix if the assumption of non-cooperative behavior between firms is relaxed. In fact, research joint ventures may help firms to overcome R&D spillovers. Katsoulacos et al. demonstrate that under certain conditions it may be optimal to have a pollution control policy only and to allow for research joint ventures in addition. The study by Katsoulacos et al. stands out in that it does not only address a market failure but also the failure of other private governance structures – particularly the firm and bargains between firms. They reveal that the presence of a technology spillover does not necessarily imply direct government intervention. Instead, it may also be fruitful to promote and strengthen private governance structures. This study underpins the finding that a careful revision is required to determine under which governance structure – government regulation by a subsidy, research joint ventures or other types of private governance structure – the balance of the net value of reducing spillovers and the corresponding transaction costs is actually highest in a specific context.

3.3.2 Pollution Externalities and Asymmetric Information

3.3.2.1 Inefficiency of Single Policies

If pollution externalities coincide with asymmetric information, policies that only address the pollution externality may be inefficient at reducing pollution. This is particularly emphasized for the presence of (external) information asymmetries between market participants. Bennear and Stavins (2007, p. 121) highlight the case of markets for energy-efficient technologies remaining underdeveloped despite the existence of pollution control policies. Pollution control policies increase energy costs. In theory, agents would have an incentive to switch to energy-efficient technologies, e.g. for domestic appliances or production processes. Yet, agents often do not respond efficiently to pollution control policies because they are not well informed about available options of energy-saving technologies. Jaffe et al. (2002, p. 48) point out that this is particularly evident in the housing sector. In this sector, the tenant pays for fuel consumed for heating and for electricity. The landlord decides about appliances, such as the insulation of walls and windows or the installation of stoves, which affect the fuel and electricity demand of the tenant. With perfectly informed tenants and landlords, a single emissions tax on fuel or electricity may be efficient. In order to reduce their tax burden, tenants will prefer the most energy-efficient housings. Landlords, in turn, have an incentive to invest in energy-efficient technology. As Jaffe et al. (2005, p. 168) highlight, however, tenants are hardly informed about the energy efficiency of buildings. Thus, they will be unable to identify the most energy-efficient options. This implies that landlords may be unable to recover the cost of energy-efficiency investments, and may not undertake them at all. Therefore, single emission policies cannot provide for energy-efficiency in the housing sector in the presence of asymmetric information.

Gabel and Sinclair-Desgagné (1998, pp. 91, 99) point out that the efficiency of pollution control policies may also be hampered by (internal) information problems within the firm. Firms' reactions to pollution control policies are guided by complex, multi-step and imperfect management systems. Information transmission from the top management to the employee level is characterized by a variety of principal-agent-relationships. The firm's final reaction to a policy – and thus the actual efficiency of a policy – is therefore a function of the incentives and capacities of all agents within the firm and may well deviate from the outcome with rational, profit-maximizing agents.

3.3.2.2 Superiority of a Policy Mix

The failure of pollution control policies to overcome external and internal problems of asymmetric information provides a rationale for combining them with information policies. This is particularly brought forward in the economic literature with respect to external information problems between market participants. Petrakis et al. (2005) argue that information provision may dominate environmental taxation in terms of welfare. They assume that a good produces a pollution externality during production (which does not affect the utility of consumers) and health damages during consumption (which affect utility of consumers). Consumers are not perfectly informed about the health damages resulting from consumption. Information provision then shifts demand to healthier products and at the same time reduces the environmental externality. Consequently, the required level of taxation is reduced and overall welfare increases.

A broad body of literature pleads for supplementing pollution control policies by information measures to foster the diffusion of energy-efficient technologies (Rolfe et al. 1999, p. 5; Sorrell and Sijm 2003, p. 431; Jaffe et al. 2005, p. 172; Bennear and Stavins 2007, p. 121). It is argued that additional information measures may help to overcome asymmetric information on the technology market. They provide agents with knowledge about different energy-efficient options. Agents are thus enabled to respond efficiently to the incentives set out by pollution control policies.

Measures to correct for external information problems may be twofold: On the one hand, the regulator may directly provide market participants with information. An example is the US Green Lights program, which provides information on energy-efficient lightning technology to businesses. On the other hand, the regulator may require market participants to provide information to other market participants by certification or labeling. Producers of domestic appliances may be obliged to display a label on their products indicating the annual energy use and how this energy use compares to other products of the same category. In the US, this is done by the EnergyGuide (Bennear and Stavins 2007, p. 113). A similar and prominent EU example of such policy is the energy label for domestic appliances (see Section 2.3.7). In the housing sector, the landlord may be required to inform the tenant about the energy efficiency of his building, e.g. the level of insulation or the energy consumption of the appliances installed. In the European Union, this approach has been implemented by the energy certificate which has to be issued for each building by the owner (Sunikka 2003). Apart from mandatory information measures, voluntary labeling and certification programs, such as the Energy Star for particularly energy-efficient domestic appliances in the United States, may be implemented to foster the diffusion of energy-efficient technologies (for an overview of existing US programs see Banerjee and Solomon 2003). Empirical studies on the Green Lights and Energy Star programs suggest that both have been successful at lowering energy consumption (DeCanio 1998; DeCanio and Watkins 1998; Howarth et al. 2000). Sorrell and Sijm (2003, p. 432) argue that it eventually depends on the specific features of the market, the technology and the policies under examination whether or not information policies are a useful component of a policy mix. Sorrell and Sijm expect that information provision will particularly improve the efficiency of pollution policies for households and small and medium-sized firms which are characterized by low energy price elasticity and large energy-saving potential.

Jaffe et al. (2005, p. 172) emphasize that pollution control policies may also be complemented by command-and-control policies when technology diffusion hampered by asymmetric information. Command-and-control policies may mandate certain technological standards and force less expensive and less environmentally friendly technologies from the market. Such standards restrict the choices of individuals. Thereby, they avoid costly information and assessment processes which are undertaken by individuals to trade off capital and energy operating costs of different technological options. Asymmetric information is then an less important issue. As an example, Jaffe et al. cite energy-efficiency regulations in the US, such as the Corporate Average Fuel Economy (CAFE) standards. These have been implemented in addition to gasoline taxes in order to improve the fuel efficiency of light-duty vehicles, such as passenger cars and small trucks. Agras and Chapman (1999) provide a rough idea of how such a policy mix may perform compared to the isolated utilization of each policy. They compute that when both policies are used jointly, only 41 percent of the levels determined for the single use of either CAFE standards or gasoline taxes would be necessary to achieve the US reduction targets for greenhouse gases under the Kyoto Protocol. In this respect, the eco-design standards presented in Section 2.3.6 could be understood as a useful complement to the European ETS. However, Jaffe et al. (2005, p. 172) also point out that such command-and-control policies bear the risk of going beyond an economically justified minimum, at which point they can impose limits to product choice, and undesirable costs if adopters are heterogeneous. These issues are addressed for the German case study in Section 6.6.

Gabel and Sinclair-Desgagné (1998, p. 113) emphasize that supplementing pollution control policies by further policies may also be useful to overcome problems of internal asymmetric information within the firm. In this respect, they highlight the importance of standards for voluntary environmental management systems such as the International Organization for Standardization's ISO 14001 and the European Eco-Management and Audit Scheme (EMAS).[28] Among other, participating firms are required to set up organizational structures and to define responsibilities and procedures to attain pre-defined environmental goals. Thereby, environmental management systems may improve the flow of environmentally-relevant information within the firm and overcome possible information asymmetries. Other programs have a similar intention. The U.S. Green Lights program requires firms to survey their facilities in exchange for providing them with information on energy-efficient lighting (Bennear and Stavins 2007, pp. 113-114). The U.S. Department of Energy (DoE) provides free energy audits to small and medium-sized firms that recommend desirable energy-saving projects (Anderson and Newell 2004, p. 31). The possible effects of these programs have been subject to economic analysis. Anderson and Newell (2004) regard the inexpensive DoE energy audits a success since roughly 50 percent of the suggested projects were actually adopted.[29] Likewise, Wätzold and Bültmann (2001) find that the majority of firms participating in EMAS have reduced the use of polluting materials and have implemented ecologically motivated improvements of products and production processes. They estimate that the net value of reducing asymmetric information exceeds the corresponding transaction costs (borne by the government) for EMAS. Thus, information measures have proven in many cases to be a desirable complement to pollution control policies from an efficiency perspective.

3.4 Using a Policy Mix to Cope with High Transaction Costs of First-Best Single Policies

Economic studies highlight that the implementation of single first-best policies for pollution control may bring about high transaction costs under three assumptions: (1) if marginal damages of pollution are distributed heterogeneously, (2) if marginal abatement costs are heterogeneous, or (3) if polluters have incentives not to comply with a policy. It is emphasized that with each of these assumptions, transactions costs of a single first-best policy may well be prohibitively high, i.e. exceed the net value of internalizing the pollution externality. It is further demonstrated that

[28] EMAS can also be understood as a means to overcome external information asymmetries between market participants since it requires firms to publish environmental statements. Wätzold and Bültmann (2001) estimate, however, that this effect is limited since EMAS is directed to a specific site or company and not a specific product.

[29] Anderson and Newell (2004) qualify, however, that the fact that 50 percent of the projects were not adopted suggests that asymmetric information is not the only reason for non-adoption.

the implementation of alternative single policies may reduce transaction costs but usually comes at a decreased net value of internalization as well. The balance of the net value of internalization and transaction costs may be positive for these alternative policies but pollution abatement is likely to be inefficient. The important point brought forward is that a policy mix may then reduce transaction costs and provide for a high net value of internalization. Thus, the balance of the net value of internalization and transaction costs may be positive for the policy mix and larger than for any single policy. Available studies implicitly seem to assume that the policy mix also outperforms private governance structures although not much attention is drawn to this question.

3.4.1 Heterogeneity of Marginal Pollution Damages

3.4.1.1 Inefficiency of Single Policies

It may be assumed that the spatial distribution of marginal pollution damages is homogeneous, i.e. the marginal damage produced by one pollution unit does not depend on the location of its emission and reception. Homogeneity is provided if pollutants are uniformly mixed in receiving airsheds. This applies, for example, to CO_2 and chlorofluorocarbons (CFCs). In this case, a pollution externality may be corrected efficiently by a single emission-based policy (Bennear and Stavins 2007, p. 124). For designing such a first-best policy, damages caused by one unit of pollution do not have to be determined and considered by the regulator. Therefore, it may be assumed that transaction costs related to the emission-based policy are relatively low and that it can be set up and implemented easily. The balance of the net value of internalization and related transaction costs is likely to be positive.

However, policy design and implementation become more complex with heterogeneous marginal pollution damages. Heterogeneity arises due to non-uniformly mixed pollutants, such as sulfur dioxide (SO_2), nitrogen oxide (NO_X), and most water pollutants. In these cases, the concentrations – and thus the marginal damages – caused by a pollution unit may vary considerably between locations – depending on ecological, technical, and socioeconomic conditions at the point of emission as well as at the receptor point (Sterner 2003, p. 83).[30]

[30] The damages of pollution may vary over time, as well. For example, marginal damages of car emissions in urban areas will peak at daytime when the daily rhythm of the labour market causes congestion and rapid build-up of pollutants to coincide with the highest concentration of people affected. Taking this into account would make policy design and implementation even more complex.

Montgomery (1972) demonstrates that an efficient allocation of pollution units under these circumstances could theoretically be provided for by a single-policy strategy based on tradable "ambient permits", which considers not only emissions but also the damage produced by each unit of pollution. Such permit systems require policy-makers to specify a vector of transfer coefficients for each emitter, linking emissions from each source with concentrations at each of the predefined receptor points. An emitter has to acquire separate pollution permits for each receptor that is affected by its emissions. Krupnick et al. (1983, pp. 235-236) find that under this first-best permit system, administering agencies incur relatively low transaction costs. Their efforts are limited to specifying the transfer coefficients and allocating permits to polluters. Officials do not need information regarding abatement costs. In contrast, such permit systems result to be very costly for polluters since they have to participate in different permit markets for each receptor point. With many receptor locations, an emitting firm has to acquire and manage a very complex portfolio of necessary permits. Tietenberg (1995, p. 97) points out that high transaction costs exceeding the expected net values of internalization have prevented first-best ambient permit systems from being implemented. Similarly, Kolstad (1987, p. 340) argues that other spatially differentiated regulations like taxes may also fail due to high transaction costs.

Tietenberg (1995, p. 101) points out that alternative policies based on emissions rather than damages resulting from emissions (e.g., emissions trading or emissions taxes) are not desirable in the presence of heterogeneous marginal pollution damages either. Compared to a complex ambient permit system, they may bring about relatively low transaction costs but provide for inefficiency in the allocation of abatement. Thus, the marginal net value of internalization is reduced. For example, when few large sources are concentrated around a receptor requiring large improvements in ambient air quality, they have to be controlled to a higher degree than distant sources, which hardly affect the receptor. Emission-based policies, however, provide for marginal abatement costs to be equalized across all sources over-controlling distant sources and under-controlling adjacent sources. Thus, these approaches may aggravate existing or give rise to new so-called "hot spots", where concentrations of pollutants and resulting damages exceed socially optimal levels. Smith (1999, p. 213) highlights that hot spots can affect the efficiency of emission-based instruments particularly negatively if damage functions for pollutants exhibit the characteristics of thresholds and other non-linearities. Under these circumstances, marginal damages increase substantially once a certain level of pollution is exceeded.

Attempts to improve the efficiency of single emission control policies in the presence of heterogeneous marginal pollution damages may bring about high transaction costs. One modification that has been developed for emission permit trading makes use of trading zones to prevent hot spots. While trading within each zone is not limited, trading across zones is restricted.[31] Krupnick et al. (1983, p. 237) and Tietenberg (1995, p. 104) report that, regarding transaction costs, this approach reverses the findings made for ambient trading schemes. Polluters incur relatively low transaction costs since they only have to participate in the permit market of the zone in which their facilities are located. In contrast, the administering agency has to determine and continuously readjust efficient amounts of permits for each zone. To do so, it must gain knowledge of source-specific abatement costs in addition to the air-modeling data needed for ambient permit system. Moreover, Atkinson and Tietenberg (1982, p. 104) highlight that zonal systems may provide for inefficiencies in other fields as well since they prevent cost-saving trades between emitting sources in different zones. Thus, zonal trading increases transaction costs but does not necessarily increase the net value of internalization compared to an emission-based trading scheme. Consequently, single-policy strategies may not efficiently cope with pollution problems if pollutants are non-uniformly mixed in the environment.

3.4.1.2 Superiority of a Policy Mix

In the presence of spatially heterogeneous marginal pollution damage, a policy mix combining an emissions tax or an emissions trading scheme with command-and-control policies may be superior to single policies. Regarding a tradable permit system, Krupnick et al. (1983) propose the approach of "pollution offsets" allowing trading among sources as long as it does not violate ambient air quality standards at any receptor point. New emitters have to acquire permits from existing sources in order to completely "offset" the effects of the new emissions on pollutant concentrations at receptor points. This procedure provides for ambient quality goals to be attained at least cost and is relatively easy to administer. The approach is advanced by McGartland and Oates (1985). They examine "modified pollution offsets" allowing trading among sources if neither the initial level of environmental quality nor certain predetermined ambient standards established by the environmental authority (whichever is more stringent) are exceeded. The approach promises to prevent the costly deterioration of air quality in areas where pollution concentration is already far below ambient air standards. McGartland and Oates show that their approach is cost-effective and achieves a higher level of environmental quality

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[31] This approach was pursued in the context of the RECLAIM program, a tradable permit system for NO_X and SO_2 in the Los Angeles basin area. The basin was divided into a coastal and an inland zone. Within each zone, trading is not restricted. However, trades across zones are only allowed if permits are sold from facilities in the heavily polluted coastal zone to facilities in the inland zone (Tietenberg 1995, p. 107; Fromm and Hansjürgens 1996).

compared to the approach of "pollution offsets". An alternative approach is proposed by Atkinson and Tietenberg (1982). They suggest "nondegradation offsets" that allow trading among sources as long as local ambient air quality standards for the worst receptor are met and the overall emissions do not increase.

Either of the presented approaches combining market-based and command-and-control approaches provides for a second-best solution in the presence of non-uniformly mixed pollutants by sharing tasks between both policies. Baumol and Oates (1988, p. 187) highlight that emission permit systems allow for the reduction of marginal abatement costs across emitting sources. Ambient standards provide for guaranteed minimum benefits due to abatement and prevent hot spots. Compared to the ambient permit system, this approach saves transaction cost since firms are not required to participate in a multitude of separate markets. Compared to a zonal emission permit system, the approach involves only modest information demands on part of the environmental authority.[32]

A combination of emission trading and command-and-control policies was introduced in the US by implementing the Acid Rain Program for tradable sulphur dioxide emission permits on top of existing ambient air quality standards.[33] [34] Technically, trading is not restricted under this program. However, firms are only allowed to use purchased permits if the resulting emissions do not violate ambient air standards at the location of their facilities or State regulation on air quality. Thus, emitting sources actually must comply with both market-based and command-and-control policies (Hansjürgens 2000a, pp. 269-270; Swift 2000; Burtraw and Palmer 2004). Likewise, participants of the trading system implemented for nitrogen oxides in the north-eastern states of the US simultaneously have to comply with local ambient air quality standards (Burtraw and Evans 2004).

Requiring polluters to comply with ambient standards at the location of their emitting facilities – not at all existing receptor points as suggested by the theoretical studies – however, has been found to have an important drawback: It does not take into account the effects of pollutants

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32 See also Hansjürgens (2000a)

33 For an introduction to the Acid Rain Program, see, e.g., Hansjürgens (1998), Ellermann et al. (2000) or Burtraw and Palmer (2004)

34 Johnstone (2002, p. 9) mentions European examples as well where permit trading schemes are combined with command-and-control policies in order to prevent hot spots, e.g., in the context of the Swiss trading program for volatile organic compounds (VOC), and NO_X and SO_2 trading programs in the UK.

transported over long distances. This became obvious when emitting companies in the US started to build tall stacks that emitted pollutants high into the atmosphere in order to comply with local ambient standards. This procedure resulted in pollutants being transported over long distances and causing new damages (and hot spots) at remote locations, especially in the northeast of the US and the southeast of Canada (Burtraw and Palmer 2004, p. 42). The problem of long-distance transport of pollutants requires additional regulation. In the case of the Acid Rain Program, Congress in 1977 decided to simply prohibit the use of tall stacks to achieve ambient standards (Swift 2001, p. 317).

However, it is emphasized that the actual threat of hot spots and, thus, the actual advantages of a combination of market-based and command-and-control policies over pure market-based policies in terms of efficiency depend on at least two aspects. Firstly, Swift (2001, pp. 349-351; 2004) highlights that the relationship between the spatial distribution of marginal damages and the spatial distribution of marginal abatement costs appears to be important. Swift's empirical examination of the US Acid Rain Program for sulfur dioxide emissions suggests that marginal abatement costs were lowest for the largest polluters. These polluters had the highest incentives to reduce their emissions under the emissions trading system. Thus, emissions trading abated pollution at the most heavily emitting sources first and rather "cooled" existing hot spots instead of exacerbating them. Secondly, Portney (2000, p. 118) and Swift (2000) report for the example of the Acid Rain Program that the implementation of market-based emissions policies provided incentives to reduce emissions substantially at all existing sources. Thus, even given spatial reallocations of emissions, pollution at hot spots everywhere was reduced. Third, when pollution sources are more ubiquitous and no cluster of sources dominates the receptor point, inefficiencies of emission trading (and market based instruments in general) and cost savings gained from implementing additional localized standards may be limited (Tietenberg 1995, p. 101). Thus, it has to be evaluated carefully whether the balance of the net value of internalization and related transaction costs is actually higher under the policy mix than under emissions trading only.

3.4.2 Heterogeneity of Marginal Pollution Abatement Costs

3.4.2.1 Inefficiency of Single Policies

Price-based policies (such as taxes) and quantity-based policies (such as tradable permits) provide for efficient pollution control if the price or quantity for pollution set under each policy, respectively, matches the intersection of the marginal abatement cost curve and the marginal damage cost curve. Determining the marginal abatement cost curve to determine the tax rate or the total quantity of permits is easy for the regulator if the costs of abating one unit of pollution are the same no matter by whom, where and when this is done. However, marginal pollution abatement costs are usually characterized by large heterogeneity. Polluters exhibit differences in the technologies they employ, the inputs they use, and the outputs they produce, all of which may have an effect on the costs of pollution abatement. In order to determine the aggregated marginal abatement cost curve, the regulator has to find out about the characteristics of each polluter. Designing a first-best price-based or quantity-based policy may thus bring about significant transaction costs (Sterner 2003, p. 165). Kwerel (1977) shows that, in theory, a regulator could learn about the true abatement costs by successively implementing and revising policies based on polluters' responses to the policies, i.e. by a trial-and-error appraoch. However, Bennear and Stavins (2007, p. 122) point out that the marginal abatement cost curve may shift over time as well, for example, because of technological innovation or changes in input and output prices. Thus, an iterative approach does not necessarily reduce the information requirement for the regulator. Moreover, a lot of investment decisions related to pollution abatement are long-term and cannot be reversed easily by policy revisions.

If transaction costs impede a correct assessment of heterogeneous marginal abatement costs, regulators may choose to accept a certain degree of uncertainty regarding abatement costs and rely on more or less rough estimates. Under these circumstances, the implementation of price-based policies as well as of quantity-based policies is likely to result in inefficient over- or under-regulation. If, on the one hand, marginal abatement costs turn out to be lower than expected, the quantity restriction, which appeared to be optimal ex ante, will be not stringent enough compared to the social optimum resulting in too little abatement, whereas the tax chosen ex ante will be too high causing too much abatement. On the other hand, if marginal abatement costs prove to be higher than expected the reverse will be true: Abatement levels will be too high under the quantity policy and too low under the tax (Johnstone 2003, p. 15). Thus, reducing transaction costs comes at the expense of a reduced net value of internalization.

Weitzman (1974) demonstrates that price-based and quantity-based policies are not equally inefficient in the presence of uncertain abatement costs, i.e. the costs of over-regulation by one policy do not exactly correspond to the costs of under-regulation by the other. The inefficiency of the respective policy and, thus, the choice of policies depend on the characteristics of the marginal abatement cost function and the marginal damage function of emissions.[35] If (the absolute value of) the slope of the (expected) marginal abatement cost function is greater than the slope the marginal damage function, i.e. the marginal abatement costs change faster with the level of emission than the damages caused by the emissions, it is more important to control for the cost of abatement by fixing a price. The social cost of a misjudged price for pollution is lower than that of a misjudged quantity for pollution. Therefore, the regulator should favour a price-based policy. In turn, if the marginal damage curve is steeper than the marginal abatement cost curve, i.e. damages vary more strongly with the emission level than abatement costs, the regulator should control for the quantity of pollution – and choose a quantity-based policy.

Pizer (1999) applies Weitzman's finding to climate change. He assumes that marginal damages of further carbon dioxide emissions are rather flat. Based on this assumption, he finds that price-based policies should be preferred concerning climate change. Using a stochastic computable general equilibrium model, Pizer (2002) shows that net benefits of a tax policy implemented to cope with climate change are five times the gain of a permit policy. Hoel and Karp (2001) and Newell and Pizer (2003) confirm this finding for a dynamic context where the marginal damage of emissions is a function of the stock of pollutants accumulated over time – as in the case of CO_2. However, the assumption of flat marginal damage functions may be challenged. Hansjürgens (2005a, p. 224) points out that there is some evidence of threshold effects related to GHG emissions. That is, under certain conditions, an emissions increase may result in major changes in the climate system which are highly economically relevant. In this case, permit trading systems may outperform tax solutions. These considerations notwithstanding, a policy mix may be superior to both price-based and quantity-based policies, i.e. provide for a higher net value of internalization, irrespective of the assumptions about marginal cost and benefit curves.

3.4.2.2 Superiority of a Policy Mix

Roberts and Spence (1976) demonstrate that the combination of quantity-based and price-based policies may provide for a better outcome in terms of social costs than either policy individually. They propose a permit trading system which is supplemented by taxes and subsidies.[36] Their

[35] Weitzman (1974) actually examines the relation of the marginal abatement cost curve and the marginal benefit curve of abatement. The marginal damage curve of emissions is the inverse of the marginal benefit curve of abatement. Using the marginal damage curve does not change Weitzman's findings, since they are based on the absolute values of the slopes of the functions.

[36] See also Weitzman (1978).

three-part policy combines the advantages of price-based and quantity-based policies and compensates for their deficiencies. On the one hand, the tax protects against unexpectedly high permit prices if true marginal abatement costs are higher than anticipated and a pure permit trading system would result in too much abatement. The mechanism works as follows: The regulator allows polluters to generate emissions in excess of their permit holdings but charges a tax per unit of such emission (or requires polluters to buy additional permits from the regulator at a fixed price). If the permit price actually rises above the tax rate, polluters with even higher marginal abatement costs will choose to pay the tax rather than buy permits on the market. Hence, the tax acts as a "safety valve" that caps the permit price and prevents inefficiencies due to extraordinary price increases.[37] On the other hand, compensation in the form of subsidies may stimulate further abatement if marginal abatement costs are lower than expected and a pure permit trading system would result in too little abatement.[38] In order to do so, the regulator pays firms for every unit of pollution that falls below their actual permit holdings. The subsidy sets a floor for the permit price since any polluter will rather collect this payment than sell his permits on the market at a lower price. Within the price range set by the tax and the subsidy, the permit system provides for the attainment of the pollution reduction target determined ex ante. Thus, such three-part policy sets binding emission targets as long as costs are reasonable and allows the target to rise (tightens the target) if costs are unexpectedly high (low). Similarly, Smith (1999, p. 212) suggests to implement taxes and subsidies as a supplement to command-and-control policies, which guarantee quantity, but at uncertain abatement cost.

The use of a safety valve has been examined particularly for the context of climate change policies. Fearing high costs of carbon abatement that may even exceed its benefits, McKibbin and Wilcoxen (1997a; 1997b; 2002; 2007) suggest an international agreement on climate change establishing a system that combines national emission permits and fees.[39] According to their

[37] Similarly, a regulator can also provide for cost certainty by creating a permit reserve which he can bring on the market in the case of unexpectedly high permit prices. This approach has the disadvantage, however, that the price can only be capped for as long as the permit reserve holds (Johnstone 2003, p. 17). Jacoby and Ellerman (2004) show that price certainty under a permit trading scheme can be provided for by banking and borrowing provisions as well – however not to the same extent as by a safety valve.

[38] Baumol and Oates (1988, pp. 222-223) demonstrate, however, that subsidizing firms for pollution abatement involves dynamic inefficiencies.

[39] Apart from high abatement costs, McKibbin and Wilcoxen (1997a; 1997b) especially fear that a pure permit trading system on an international level would result in unacceptably high wealth transfers from developed to

approach, each country would be allowed to allocate emission permits equal to its 1990 carbon emissions, and would agree to sell additional permits at a fee specified in the agreement. They propose an internationally uniform initial fee of US$ 10 to 20 per ton of carbon. Kopp et al. (1997; 1999) focus on designing a domestic permit trading scheme for greenhouse gas emissions in the United States in the presence of abatement cost uncertainty. They suggest the allocation of permits for the emission of 1,460 million tons of carbon (this corresponds to the actual carbon emissions in the US in 1996) and a price cap of US$ 25 per ton of carbon.[40] With respect to a global climate change policy, Pizer (2002) finds that the hybrid approach performs slightly better than the price-based approach but considerably better than the quantity-based approach. An optimal hybrid policy would set a global carbon emission target of 5 gigatons and a safety valve of US$ 7 per ton of carbon emission. Jacoby and Ellerman (2004) emphasize that national safety valves may impair the operation and efficiency of an international permit trading scheme. If the safety valve in one country is set below the clearing price on the international permit market, polluters will rather buy their permits at the domestic safety valve price than on the international permit market. The lowest national price cap would therefore determine the international permit price. This problem may arise if several domestic permit trading systems with different price caps are linked.

At first sight, the safety-valve argument may also seem to provide a case for having an emissions trading scheme and a tax as components of the German climate policy mix (see Chapter 2). However, the design of the German electricity tax (Section 2.3.3) hardly corresponds to the emissions tax as it has been proposed by economists above. It will be analyzed in detail in Chapter 5 what the implications of this deviation for efficiency are.

3.4.3 Non-Compliance by Polluters

3.4.3.1 Inefficiency of Single Policies

As Cropper and Oates (1992, p. 695) point out, it is often assumed in economic literature that firms and individuals comply perfectly with pollution policies they are subject to. For example, firms are supposed to keep their emissions within the limitation prescribed under a standard or to

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developing countries, that it would put enormous stress on the world trade system, that it would be inoperable due to the allocation of excess permits, particularly to countries of the former Soviet Union, and that it would be difficult to monitor and enforce.

[40] For a detailed comparison of the approaches presented by McKibbin and Wilcoxen and Kopp et al., see Morgenstern (2007).

accurately report their emissions and pay the required fees. The policy does not bring about transaction costs since the regulator does not have to invest resources in monitoring and enforcement. Under otherwise perfect conditions, a single first-best policy can then be implemented to correct for a pollution externality.

However, Becker (1968) highlights that individuals may have an incentive not to comply with regulation. This incentive arises because pollution policies usually bring about additional costs for the individual (either for abatement measures or tax payments and permit purchases), which can be avoided by non-compliance (Gersbach 2002, p. 45).[41] Therefore, a firm which is liable to an emissions tax, for example, will tend to understate its actual emissions in order to reduce its tax burden. Consequently, the regulator has to incur transaction costs for the proper monitoring and enforcement of a pollution policy. Transaction costs rise with the level of information asymmetry between the regulator and the individuals (Sterner 2003, p. 106). Moreover, Bohm and Russell (1985, p. 428) emphasize that monitoring and enforcement costs are particularly high with numerous and mobile sources of pollution polluters. This applies, for example, to waste disposal and car emissions.

Reducing monitoring and enforcement efforts to bring down transaction costs comes at rising costs of non-compliance. Fullerton and Kinnaman (1995, p. 79) state, for example, that the illegal disposal or burning of waste in order not to pay a tax on garbage may even deteriorate the pollution problems related to waste and produce new externalities. Palmer and Walls (1997, p. 194) make the point that the costs of monitoring and enforcement, or the costs due to non-compliance, may (partly) outweigh or even exceed the net value of reducing pollution, i.e. implementing the policy may not be advisable from an efficiency point of view. Consequently, the presence of positive monitoring and enforcement costs has a significant effect on the overall efficiency of policies for pollution control. Downing and Watson (1974) demonstrate that with positive enforcement costs, the optimal level of pollution abatement will be less than when these costs are ignored. This finding is confirmed for the specific case of a pollution tax by Harford (1978). Malik (1990) introduces enforcement issues into the analysis of permit trading schemes.

[41] The actual extent of non-compliance is debatable. On the one hand, Harrington (1988) points out that substantial compliance may exist in spite of only modest monitoring and enforcement efforts. On the other hand, there is also a strong body of literature pointing out an important lack of enforcement for environmental policies (for the German context, see Mayntz et al. 1978; Mayntz 1983; Lübbe-Wolff 1993; Bültmann and Wätzold 2002).

He reveals that non-compliance affects the market-clearing price in the permit market – and thereby compromises the efficiency of permit trading schemes.[42]

If monitoring and enforcing a first-best direct regulation on a polluting good is very costly, policy-makers may bring down transaction costs by regulating related goods that are easier observable.[43] Eskeland and Jimenez (1992, p. 154) point out that the intuitive suggestion is to tax complements to a polluting good and subsidize cleaner substitutes. For example, if it is too costly for a regulator to monitor and enforce a direct tax on car emissions, he may choose to tax fuels (complementary input to car emissions) or to subsidize catalyzers or public transportation (substitutes to car emissions). If the regulator faces difficulties in implementing a tax on waste disposal, he may likewise tax packaging or consumption in general or subsidize returning and recycling waste. Subsidies bear the additional advantage that they provide incentives to disclose information (e.g. a polluter may only receive a subsidy if he provides adequate evidence of recycling) and increase the utility of compliance by reducing the cost of pollution abatement (e.g. by subsidizing abatement technologies or appropriate waste disposal).

In spite of reducing the cost of monitoring and enforcement and non-compliance, these policies do not necessarily provide for cost-minimizing pollution abatement – and thus decrease the net value of internalization – if implemented in isolation.[44] Inefficiencies arise because taxes on complements to pollution and subsidies to substitutes of pollution do not perfectly mimic the substitution and output reduction effects that provide for efficient resource allocation under a direct tax (see, e.g., Fullerton and Wolverton 1999). Eskeland and Jimenez (1992, p. 155) point out that the taxation of dirty inputs to the production of polluting goods, such as fuels, provides

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[42] Harford (1978) and Malik (1990) make the interesting finding that under certain stringent conditions, polluters underreport emissions but set their actual level of emissions such that marginal abatement costs equal the tax rate or permit price. Nevertheless, this behaviour does not necessarily imply efficient abatement. As the studies cited above highlight, optimal tax rate or permit trading system has to be designed such that, in equilibrium, marginal damages equal the sum of abatement costs and monitoring and enforcement costs.

[43] Several authors point out that in order to bring down monitoring and enforcement costs, the regulator may also increase fines (see, e.g., Becker 1968) or implement differentiated monitoring and enforcement systems (see, e.g., Harrington 1988).

[44] Moreover, Sandmo (1976) and Wijkander (1985) emphasize that taxing complements and subsidizing substitutes may result in inefficient distortions in a general equilibrium framework if complements and substitutes are related to other, non-polluting goods as well, or if the demand for complements and the demand for substitutes are related to one another.

incentives to switch to cleaner inputs and to reduce the production of a good. However, it does not stimulate polluters to employ potentially cheap technologies to reduce emissions with dirty inputs, e.g. catalytic converters or scrubbers. Taxing dirty technologies, in turn, does neither provide incentives to switch fuels nor guarantee the proper maintenance of cleaner alternatives. Alternatively, the regulator may tax the output or consumption of related goods. Palmer et al. (1997) point out that the taxation of disposable goods – instead of their actual disposal – reduces waste disposal since it discourages their consumption. However, it fails to stimulate waste reduction by recycling and the use of recycled inputs rather than virgin inputs. In fact, Palmer et al. find that the overall amount of recycling will decline with decreasing consumption.

Subsidies to clean inputs, such as low-emission fuels, pollution abatement technologies, and recycled resources, or on clean activities, such as recycling, stimulate substitution processes but lack sufficient output reduction effects. For example, Palmer et al. (1997) state that a subsidy to recycling sets an incentive to substitute virgin inputs. Yet, they highlight that the subsidy may encourage consumption and thus waste generation because it reduces the effective price of a potentially polluting good for users. This finding has been made for the case of direct subsidies for pollution abatement as well. Several authors emphasize that subsidies will be inefficient in this context because they will foster the entry of new polluters, which in turn increases overall production, consumption and pollution (Porter 1974; Baumol and Oates 1988; Kohn 1991).

3.4.3.2 Superiority of a Policy Mix

If monitoring and enforcing first-best direct pollution control policies is very costly and alternative single policies bring about inefficiencies in pollution abatement, the implementation of a policy mix may be a superior strategy. In particular, it may be useful to combine taxes on complements to pollution and subsidies to substitutes of pollution. As Bohm (1981) makes clear, the classic application of this concept is a deposit-refund system.[45] At some point in the chain of market transactions, a potential polluter is subjected to a tax (the deposit) in the amount of the

[45] Bohm (1981) distinguishes between publicly (government) initiated and privately initiated deposit-refund systems. Since the focus of this paper is on using a policy mix, only publicly initiated deposit-refund systems are discussed. However, the existence of privately initiated deposit-refund systems demonstrates that pollution externalities do not necessarily require government intervention when the market fails. They can also be corrected by private governance structures that are somewhere in between the market and government intervention if only the private benefits of doing so are high enough – or if transaction costs are sufficiently low.

potential damage he may produce, which he has to pay under the presumption that the pollution and the related damage actually occur. At a later point, he receives a subsidy (the refund) if certain conditions are met, e.g. if he can prove that a certain product has been returned to a specified place or that a specified type of damage has not occurred. Bohm and Russell (1985, pp. 431-432) point out that deposit-refund systems can be implemented for the consumption of products which may create pollution damages, e.g. beverage containers, mercury and cadmium batteries, refrigerators, vehicles, or waste in general, and production processes potentially providing for hazardous emissions of chemicals or the disposal of toxic wastes.

Deposit-refund systems combine the advantages of single taxes on complements to pollution and subsidies to substitutes of pollution, and compensate for their disadvantages. Yohe and MacAvoy (1987, p. 177) and Fullerton and Wolverton (1999, p. 40) highlight that deposit-refund systems provide for a substantial reduction in transaction costs compared to direct pollution control policies. Difficult and costly direct monitoring of pollution is avoided, since related and easier observable goods are taxed or subsidized. The subsidy component furthermore reduces the cost of monitoring and enforcement by shifting the burden of proof for pollution abatement to the polluter. Indeed, the reduction of public transaction costs comes at the expense of higher private transaction costs, since polluters have to invest resources in signalling their behaviour. However, Russell (1988) points out that private transaction costs are likely to be lower than public transaction costs. This is because firms and individuals can be supposed to have more knowledge of the production and disposal processes than the regulator. What is more, deposit-refund systems do not exhibit the inefficiencies of taxes on complements to pollution and subsidies to substitutes of pollution with respect to pollution abatement costs. In fact, Fullerton and Wolverton (1999) show that a deposit-refund system may perfectly mimic a first-best Pigovian tax on pollution. The tax component provides for an optimal reduction of output, while the subsidy component efficiently stimulates substitution of dirty goods and activities by cleaner ones. Thus, the policy mix provides for a high net value of internalization at relatively low transaction costs.

There is a large body of literature discussing the use of deposit-refund systems to reduce waste generation and its illegal disposal. Dinan (1993) demonstrates that the combination of a tax on goods that may eventually be disposed and a subsidy to end-users of recycled materials is equivalent to a waste-end tax and performs better than a tax on virgin inputs. Fullerton and Kinnaman (1995) suggest a combination of a tax on output and a rebate on proper disposal through either recycling or garbage collection. They point out that this policy mix may yield a first-best outcome and outperforms single output taxes or recycling subsidies. Their finding is supported by Palmer et al. (1997) who compute that single-policy alternatives may be at least twice as costly as the proposed deposit-refund system. In addition, Palmer and Walls (1997) show that the combination of an output tax and a recycling subsidy exhibits an efficiency that is far above that of recycled content standards, which require that a certain percentage of total material use be comprised of secondary materials. The same finding is made by Sigman (1995) for the special case of lead disposal. Walls and Palmer (2001) extent the analysis of this type of deposit-refund systems to a setting with multiple lifecycle externalities. They show that under these circumstances, the optimal tax on output will be greater than the optimal subsidy for recycling. An alternative deposit-refund system is brought forward by Fullerton and Wu (1998). They highlight that the combination of a tax on packaging with a subsidy to recyclability of goods may provide for an optimal resource allocation as well. On a more general level, Fullerton and Wolverton (2005) analyze how deposit-refund systems can yield an optimal outcome in a second-best world with pre-existing tax distortions.

A case for the use of deposit-refund systems is also made with respect to air pollution. Regarding car emissions, Fullerton and West (2000) show that the combination of a tax on gasoline and a subsidy to newer cars can attain 71 percent of the welfare gain of a Pigovian emissions tax, whereas a tax on gasoline alone will only yield 60 percent. With respect to emissions from stationary sources, Porter (1974) discusses the combination of a periodical "entry tax" – by which the polluter purchases the rights to produce and emit and which is based on prospected baseline emissions of production in the absence of emission abatement – with a subsidy per unit of emission that is actually abated during the period. This approach is efficient and makes cheating more risky since emission reports may be double-checked with reported abatement. While Porter rejects his idea as impractical, it is revived by Kohn (1991) who states that the emission data required is already researched and published by the U.S. Environmental Protection Agency. A similar proposal is made by Yohe and MacAvoy (1987). They suggest combining a tax on the sulfur content of coal used in production with a subsidy for proven removal of sulfur from the industrial effluent and show that this two-part policy yields a first-best outcome. A different approach is pursued by Swierzbinski (1994). He develops a system in which the firm has to report its emissions and pays an initial tax based on this report. If the firm is monitored, it receives a subsidy if actual emissions turn out to coincide with reported emissions. Whether or not this system is efficient depends among other on the limitations to enforcement.

A somewhat concluding point on the applicability of deposit-refund systems is made by Fullerton and Wolverton (1999). They emphasize that the idea of this two-part policy can be transferred to a very general level. The refund may be paid to an economic actor other than the one who paid the deposit, the optimal refund may be a different amount than the deposit, and refund and deposit may even refer to different commodities.

Alternatively to deposit-refund systems, some authors suggest combinations of taxes and standards. For stationary emission sources, Walls and Palmer (2001) demonstrate that in order to provide for a first-best outcome, an output tax has to be combined with a standard that limits emissions per unit of output. Likewise, an input tax should be supplemented by an emission standard per unit of input. Eskeland (1994) highlights that when car emissions cannot be observed directly, the simultaneous implementation of a tax on gasoline and an emission abatement requirement for cars performs better than the isolated use of either policy. The policy mix mimics the incentives of a first-best direct emissions fee since the gasoline tax provides for fewer miles driven while the abatement standard reduces the emissions per mile driven. Similarly, Innes (1996) suggests gasoline taxation and automobile standards. As an alternative to the latter component, he discusses the taxation of automobiles. He recommends that this should be imposed in relation to vehicle features that affect emissions, such as fuel economy or vehicle age. The optimal automobile tax equals the social costs of the vehicle's predicted emissions, less the portion of these costs that are internalized by the gasoline tax.

Apart from using combinations of policies that are not directly related to the pollutant, some studies also propose to supplement direct policies for pollution control by further policies in order to overcome problems of monitoring and enforcement. In this respect the combination of pollution control policies with subsidies, punitive taxes or legal liability rules may be considered. Gunningham and Sinclair (1999, pp. 58-59) argue that a subsidy for certain inputs needed for pollution abatement may improve the compliance of firms with direct pollution policies. Their rationale is that subsidies, e.g. for abatement technology or low-emission fuels, reduce abatement costs, and, thus, increase the utility of compliance compared to that of non-compliance. This policy mix has to be drawn into question on efficiency grounds, though. Johnstone (2002, p. 20) emphasizes that the subsidy component will result in using abatement technologies and other subsidized clean inputs in excess of optimal levels and distort firm entry and exit. Moreover, Gawel (1991, p. 78; 1993, pp. 465-466) highlights that regulators usually increase the amount of the subsidy with the level of under-compliance. If firms are aware of this correlation and the regulator cannot monitor their emissions, they will have an incentive to under-comply strategically in order to obtain higher subsidies and thereby deteriorate the monitoring and enforcement problem. As an alternative, Johnstone (2003, p. 17) suggests combining an emissions trading scheme with a punitive tax which has to be paid per unit of emission which is not covered by emission allowances. Like a fine, the tax reduces incentives for non-compliance by diminishing its expected utility, and thus reduces monitoring requirements for the regulator. Gawel (1991, pp. 87-88; 1993, pp. 467-468) and Knorring and Welzel (1998) highlight that compliance with standards may also be improved by punitive taxes. Several authors highlight that the German effluent fee is actually effective in this respect since it exempts polluters partly or totally from paying the fee if they are in compliance with effluent standards (Maas 1987; Meyer-Renschhausen 1990). Similarly, rules to hold firms liable for potential damages from their pollution may improve compliance with pollution control policies. Bohm and Russell (1985, p. 435) emphasize that liability rules may be particularly useful if monitoring polluting activities is expensive but the source of discharges and spills can be easily identified ex post. Schwarze (1995) demonstrates that liability fosters the enforcement of pollution standards even when the compensation to be paid by the polluter can be expected to be below the actual damage from pollution. Nevertheless, Eskeland and Jimenez (1992, p. 159) state that low likelihood of detection, high costs to victims for representation and litigation, underdeveloped judicial processes, or the potential insolvency of a liable party may limit the usefulness of liability rules in addition to pollution policies.

3.5 Summary and Avenues for Further Research

There is an increasingly large body of economic literature discussing the use of a so-called policy mix to cope with a single pollution problem. Studies argue that under certain conditions, a combination of multiple policies may outperform single-policy strategies in terms of efficiency. Studies are conducted against a wide variety of backgrounds and based on very diverse assumptions about the real world. As a consequence, there is quite some confusion on whether and when a policy mix actually increases the efficiency of pollution control. The theoretical understanding of overarching economic rationales for combining policies is still very poor. In order to overcome this deficiency, this chapter provides a comprehensive review of relevant economic analyses that bring forward efficiency arguments in favor of implementing a policy mix. To guide and structure the review, an innovative analytical framework is developed and applied. Based on seminal work by Coase (1960), the framework integrates transaction costs into the analysis of environmental problems and policies to overcome them. Moreover, it is highlighted that pollution externalities come into being not only because the market fails but due to the failure of private governance structures – including the market, bargains and firms – in general. This framework allows resolving the confusion and complexity surrounding the economic analysis of a policy mix for pollution control. It helps to derive two basic rationales which underlie the policy mix studies reviewed: Firstly, it may be necessary to combine policies to correct for multiple reinforcing failures of private governance structures, such as pollution externalities, technological spillovers and asymmetric information. Secondly, a policy mix may be used if the implementation of single first-best policies brings about high transaction costs – e.g., when marginal pollution damages and marginal abatement costs are heterogeneous or polluters are unlikely to comply with a policy. With the identification of these two basic rationales for using a policy mix, this chapter provides an important qualification of the dominant strand in economic theory preferring single-policy solutions for single pollution problems. What is more, the rationales identified may provide useful guidance for understanding, evaluating and possibly justifying an actually existing policy mix – such as that implemented in Germany to reduce greenhouse gas emissions from electricity generation (see Chapter 2).

The review has also revealed that both rationales for using a policy mix for pollution control can be applied to the specific problem of combating climate change. Multiple failures of private governance structures as well as high transaction costs of single-first best policies may be important challenges in this context. Only the specific case of high transaction costs resulting from heterogeneous marginal pollution damages (see Section 3.4.1) seems to be of minor importance for designing climate policies since CO_2 and other GHGs are uniformly mixing pollutants.

The framework is also useful to identify limitations of available economic research on using a policy mix. On the one hand, these limitations may restrict the use of the rationales for the actually existing policy mix. On the other hand, these limitations may open up avenues for future research approximating theoretical discussions to real problems.

First of all, the economic literature drawing on the rationale of using a policy mix to correct for multiple reinforcing failures of private governance structures typically neglects transaction costs of policy implementation – or rather assumes that these are sufficiently low. Yet, the second part of this review (Section 3.4) has demonstrated that this cannot be taken for granted. Future research in this field therefore has to consider the transaction costs coming along with the policy mix as they may preclude the use of certain policies in the mix or of regulation in general. The incorporation of transaction costs into the analysis of a policy mix for pollution control brings about a particular challenge for economics: It calls for more empirical work. As Zerbe and McCurdy (1999, p. 571) point out, it can only be determined on a case-by-case basis what the actual transaction costs are and whether a policy mix should be implemented or not. In order to provide for reliable recommendations, future analyses have to take into account a variety of aspects determining the level of transaction costs, such as the characteristics of the environmental resources, of the resource users, of the transactions taking place and of the institutional environment all of them are embedded in.

Secondly, most studies on policy mixes do not reflect on the use of private governance structures rather than regulation. This seems to be straightforward since pollution externalities, technological spillovers and asymmetric information can be considered as a failure of private governance structures. Yet, this failure may stem from institutional, including regulatory, constraints. It may be worthwhile to explore what these constraints are and whether modifying or abolishing some of them may allow for private solutions that are superior to regulation in terms of efficiency. It has been pointed out in Section 3.3.1.2, for example, that if pollution externalities are reinforced by externalities of technological change, research joint ventures can do as well as technology subsidies in supplementing emission policies under certain conditions (see Katsoulacos et al. 2001). The only requirement with respect to regulation would be to allow for research joint ventures. Likewise, it may be promising to test for the potential of, e.g., private certification and labeling agreements to overcome asymmetric information or privately initiated deposit-refund systems to avoid the monitoring problems of single pollution policies. There may be instances where supporting such private schemes is more efficient than implementing government-initiated programs.

Thirdly, existing studies usually analyze the combination of policies with idealized and simplified designs. However, policies that are actually implemented are often far from matching the theoretically ideal design. An example has been indicated in Section 3.3.1.2: The design of the German feed-in tariff for renewable electricity is much more complex than the simple output subsidy recommended to overcome learning-by-doing spillovers. Deviations between the theoretical and the actual design may well impair the theoretical efficiency of the policy mix, i.e. reduce the net value of internalization or increase transaction costs. Economists intending to give advice to policy-makers will therefore have to consider the actual design of the policy mix.

Fourthly, studies presented in the review analyze how an optimal policy mix should theoretically look like under a quite narrow restriction: Pollution is to be controlled efficiently. Yet, it may well turn out that a policy mix exists in reality although none of presented rationales applies – or that no policy mix has been implemented despite economic insight to do so. In order to provide for applicable economic policy recommendations, it is necessary to broaden the focus of policy analysis. Policies may not only be meant to address pollution control but other goals, such as security of energy supply or economic development, as well. Similarly, efficiency may not be the only criterion guiding policy makers. Equity, for example, may be similarly important. Moreover, it is important to shed light on the institutions and the political economy of policy making. For example, regulatory decisions on policies for pollution control may be taken at multiple, possibly competing levels, e.g., by the EU and national governments. The responsibilities of different levels may overlap and result in simultaneous policy-making. In addition, policy-makers may be influenced by multiple stakeholders of the policies with possibly contrary interests, and they may pursue their own, private interests (for emissions trading, see, e.g., Gagelmann and Hansjürgens 2002, pp. 198-199). Thus, policy-makers are subject to a variety of constraints and incentives that guide their decisions. Only if these are taken into account, it is possible to understand how an existing policy mix has come about – and to derive reasonable recommendations for the design and implementation of an optimal policy mix. It may be one result of this positive analysis, that the existing policy mix exhibits inefficiencies but represents the optimal choice if all constraints are considered. It may be another result, that in order for a superior policy mix to be implemented, some of the constraints have to be eliminated or changed.

These limitations provide the starting point for advancing theoretical economic insight on policy mixes in the subsequent chapters. It will be the explicit goal to approximate the economic analysis to real conditions. This analysis will be realized on a theoretical level – but be inspired by policy design as it can be found in Germany's climate strategy. It is obviously impossible to address all of the limitations identified extensively in this book. The focus will be on the third and the fourth limitation listed above.

On the one hand, light will be shed on the case of a policy combination which can be theoretically justified on the basis of efficiency: The combination of an emissions policy with a support scheme for renewable energy sources. The question will be addressed whether this policy mix actually provides for an efficient outcome when deviations from the theoretically proposed design of the policy mix are considered. The analysis will take into account design features which are derived from the actual design of the EU Emissions Trading Scheme and Germany's feed-in tariffs mandated under the German Renewable Energies Act (see Sections 2.3.1 and 2.3.2). The analysis will be carried out in Chapter 4.

On the other hand, the case of a policy combination which exists in reality but cannot be justified on the basis of the economic rationales derived in this book will be researched. The combination of an emissions trading scheme with different types of taxes will serve as an example. The analysis will seek an answer to the question whether this policy combination only represents a redundancy or rather results in an inefficient outcome. In this respect, possible determinants of the inefficiency will be identified. Moreover, it will be investigated how a possible inefficiency could be reduced by modifying the design of the policy mix. This analysis is important when a policy mix is inefficient in controlling pollution but necessary to address multiple policy objectives and criteria. In this context, it is necessary to determine the actual extent of welfare losses related to pollution control as well as strategies to minimize them. These losses then have to be compared with gains from meeting other policy objectives and criteria. The discussion will again be inspired by the example of the EU Emissions Trading Scheme and the German electricity tax (see Sections 2.3.1 and 2.3.3). It will be provided in Chapter 5.

Other restrictions mentioned above, such as transaction costs of policy-making, will be addressed in the context of the extensive applied analysis and evaluation of the German policy mix in Chapter 6 if necessary.

4 Emissions Trading and Feed-in Tariffs for Renewable Electricity Generation

4.1 Introduction

This chapter evaluates analytically the policy mix of an emissions trading scheme and a feed-in tariff for renewable electricity generation. An emissions trading scheme imposes a cap on emissions of a certain pollutant (see Section 2.3.1 for the German example). A feed-in tariff promotes the adoption of innovative renewable energy technologies (see Section 2.3.2 for the German example). It has been demonstrated in Section 3.3.1 that the combination of an emissions policy and a technology support scheme can be justified in the presence of two failures of private governance structures. The emissions policy is necessary to overcome a negative pollution externality. The technology policy corrects positive knowledge externalities related to technological change. In partial equilibrium models of the electricity sector, it has been shown that an output subsidy to the innovative industry – as it is implemented under a feed-in tariff – is a useful supplement to an emissions policy in the presence of learning spillovers (Bläsi and Requate 2007; Fischer and Newell 2008). Learning implies that firms can reduce their production costs as their production experience increases (Arrow 1962, p. 155). At least some of the experience, or knowledge, may spill over to other firms. Consequently, investment in learning may fall short of the socially optimal level (Arrow 1962, p. 168). There is strong evidence that learning-by-doing as well as related spillovers are significant for renewable energy technologies (Neij 1997; IEA 2000; Hansen et al. 2003). Consequently, there seems to be a strong case for combining emissions trading and a feed-in tariff for renewable electricity generation in order to reduce GHG emissions efficiently.

However, existing economic studies are subject to a major restriction. They do not refer to a specific policy context. Instead, they assume a general case with very simplified policy designs. They recommend a fixed uniform emissions price as well as a simple uniform output subsidy for technology production. It may be doubted whether such a "pure" policy design can actually be implemented in reality. Existing policies are likely to exhibit a much more complex and differentiated design. This implies that the relevance of existing economic studies for real policy-making is very limited so far. An important research question is left to be answered: Do the efficiency properties of the policy mix hold when the complex design of an actually existing policy mix is taken into account?

The importance of this question is underpinned by comparing the policy mix assumed in theoretical studies with that observed in the German case study. Two deviations between theory and reality are particularly outstanding: First of all, theoretical studies suggest paying an output subsidy in addition to the prevailing market price of electricity. In Germany, however, operators of renewable energy installations receive a fixed remuneration in the form of a feed-in tariff per unit of electricity produced. In contrast to theoretical studies, operators' production incentives are therefore irrespective of the prevailing market price of electricity. Secondly, German feed-in tariffs are not funded exogenously – as assumed in the existing theoretical models – but endogenously by an add-on to the electricity price.

Both deviations provide the starting point for advancing economic theory on combing an emissions policy and a technology policy. The major innovation of this chapter is that a partial equilibrium model is developed which accounts for these deviations. Compared to existing models, the simplistic assumptions about policy mix design are substituted by more complex but more realistic ones. Thus, this chapter provides an advance of policy mix theory that is inspired and guided by a real policy problem. The result is a better economic understanding of a real-world policy mix – and, consequently, a higher relevance of economic theory for policy-making.

The chapter is structures as follows. Section 4.2 revises whether and to what extent model assumptions of existing studies are applicable to the case of an emissions trading scheme combined with a feed-in tariff for renewable electricity. It summarizes the evidence for learning-by-doing and related spillovers for renewable energy technologies. Moreover, it reviews existing policy mix studies and highlights their restrictions. Section 4.3 introduces and applies the model. First of all, it presents the optimal regulatory approach. Subsequently, it analyzes the efficiency properties of a combination of an emissions policy with a revenue-neutral feed-in tariff as it can be found in Germany. Section 4.4 summarizes and highlights restrictions of the analysis.

4.2 Applicability of Existing Theoretical Insight

Theoretical studies argue in favour of combining emission policies and technology policies when a pollution externality is coupled with learning spillovers. This section will thoroughly revise whether the assumptions made in the theoretical studies actually hold for the case of an emissions trading scheme supplemented by a feed-in tariff for renewable electricity generation. Three questions are decisive for the progress of this analysis: (1) Are learning effects actually present in the case of renewable energy technologies? (2) If so, does part of the knowledge generated by some firms through learning spill over to other firms? (3) Does the design of the emissions trading scheme and the feed-in tariff as it is observed in Germany, for example, correspond to that of the policy mix suggested in theory? The first two questions are addressed in Section 4.2.1. The third question is considered in Section 4.2.2. Answering these questions will reveal avenues for further research, along which the subsequent theoretical analysis in this chapter is guided.

4.2.1 Applicability of Rationales for Using a Policy Mix

4.2.1.1 Learning with Renewable Energy Technologies

To assess evidence of learning related to renewable energy technologies, it is first of all necessary to understand the concept of learning. In fact, there is considerable confusion surrounding the definition of learning. Terms like learning, learning curve, learning-by-doing, progress curve or experience curve are used as synonyms in some studies (e.g., Argote and Epple 1990, p. 920; Ibenholt 2002, p. 1182), while others prefer to distinguish between them (e.g., Isoard and Soria 2001, p. 621). A somewhat overarching theoretical foundation of learning was provided by Arrow (1962). Firstly, he argued that learning is a result of experience. Economic actors learn through the attempt to solve a problem, e.g. to produce one unit of a good (or service). Therefore learning only occurs during activity. Secondly, learning which results from the repetition of essentially the same problem is subject to sharply diminishing returns. That is, experience economic actors gain from producing one unit of a good will always exceed experience gained from producing any subsequent unit (Arrow 1962, p. 155). Thus, learning results in unit costs of producing a good to decrease at a decreasing rate as manufacturers gain production experience (Jaffe et al. 2005, p. 167).

Measuring learning or experience directly is difficult, if not impossible. Most commonly, cumulative output is used as proxy variable for knowledge acquired through production (Argote and Epple 1990, p. 920). Cumulative output refers to the total number of units of one good produced from the beginning of time (Arrow 1962, p. 157). Regarding energy technologies, cumulative output may refer to the number of plants produced or installed, the capacity of plants produced or installed, or the amount of electricity generated by these plants (Junginger et al. 2005, p. 139). Alternative proxy measures for learning may include cumulative investment of firms or calendar time but are barely used in economic studies (Lieberman 1984, p. 214).[46]

To understand why learning actually comes about, it needs to be defined in further detail. Argote and Epple (1990, p. 921) distinguish between individual learning and organizational learning.

[46] For example, Arrow (1962) uses cumulative gross investment instead of cumulative output.

Individual learning implies that individuals become better at their particular jobs as cumulative production increases. This may lead to increased labour efficiency, work specialization and method improvements. Other authors refer to similar effects as learning curve effects (Isoard and Soria 2001, p. 621) or learning-by-doing (Junginger et al. 2005, p. 135). Organizational learning is a broader concept. It includes individual learning but also an improved coordination of the entire production process and technological developments that occur during the production process. Plant operators may learn how to improve the management of operations, modify job assignments or reduce material waste. Moreover, experience may result in product and process innovations. Consequently, operators may be able to use new materials, modify or replace their machines or implement new production processes. Isoard and Soria (2001, p. 621) and Junginger (2005, p. 135) consider these effects as experience-curve effects and learning-by-searching, respectively. Thus, this concept of learning reflects technological change induced by experimentation, implementation and R&D throughout the production process (Isoard and Soria 2001, p. 622).[47] This broad concept of learning – or the learning curve – will be used in the remainder of this chapter.

The learning curve is usually assumed to be a simple power function as given in equation (4.1) (see, e.g., Argote and Epple 1990, p. 920; Neij 1997, p. 1099):

$$c_{CUM} = c_0 \cdot CUM^b \tag{4.1}$$

where c_{CUM} is the unit cost of a good as a function of cumulative output, c_0 is the cost of the first unit produced, and CUM is cumulative output over time. Parameter b measures the rate at which unit costs change as cumulative output increases. It is typically negative and thus provides for decreasing returns of increasing cumulative output. It has to be emphasized, though, that this function is an empirical operationalization without any thorough theoretical foundation (Ibenholt 2002, p. 1182). Two indicators of learning can be derived from this functional concept: (1) Learning can be measured by the elasticity of production costs with respect to cumulative output. This is represented by parameter b in the above equation. An elasticity (or value of b) of -0.20

[47] Learning is commonly referred to as a process that occurs during the diffusion of a technology (Jaffe et al. 2005, p. 166). Diffusion leads to learning and cost reduction, which, in turn, again fosters diffusion (Söderholm and Klaassen 2007, p. 166). However, as has been outlined in this paragraph, learning may also promote innovation. In this respect, relating learning to diffusion only may be somewhat flawed. Nevertheless, learning provides a formidable example that technological change should be treated as endogenous in economic models of innovation and diffusion.

implies a cost reduction of two percent when cumulative production increases by ten percent (Hansen et al. 2003, p. 328). (2) A progress ratio (PR) can be determined as $PR = 2^b$. It determines the percentage of initial unit costs to which unit costs are reduced every time cumulative output doubles. A progress ratio of 0.8 – or 80 percent – implies that each doubling leads to a 20 percent decrease in unit costs.

It is worth mentioning that the concept of learning is subject to several limitations. Firstly, it assumes that production costs can fall infinitively as cumulative output increases. However, costs cannot fall below input costs (Spence 1981). Secondly, production is assumed to cumulate exogenously over time. Yet, production is itself a function of demand, which depends on factor prices and costs and, thus, on total output produced previously (Klaassen et al. 2005, p. 231). Thirdly, learning curves assume a standardized product that remains largely unchanged over the time period. It is obvious, however, that products are often modified to increase their quality (Christiansson 1995, p. 5). Fourthly, using ex post estimations of learning curves to forecast future cost reductions may be a tricky task due to the properties of the underlying power function. Christiansson (1995, p. 14) estimates that varying a progress ratio slightly between 80 and 90 percent may lead to variations in cost reductions of wind technology from 10 to 42 percent in 2025 compared to 1990 levels. However, these limitations cited do not draw into question that learning occurs – at least in the early stages of production. They rather highlight possible difficulties in estimating past and future learning curves.

Difficulties in measuring learning effects may also be attributed to the fact that decreasing unit costs are not only a function of cumulative output. Most importantly, economies of scale may also contribute to declining unit costs. In contrast to learning, this cost reduction does not result from experience gained from production but from fixed production costs being shared by more units of a product as cumulative output increases (Isoard and Soria 2001, p. 622).[48] Thus, economies of scale can be realized without any technological change. Moreover, private and public investments in R&D, reductions in input prices, market competition affecting the internal efficiency of the firms and government policies promoting technological development may also

[48] Junginger et al. (2005, p. 135) identify two reasons for economies of scale. An increased standardization of a product may allow a firm to scale up its production facilities and to initiate mass production. Moreover, an individual product may be redesigned and scaled up. For example, producing a larger gas turbine usually results in lower specific costs per kilowatt.

drive down unit costs (Ibenholt 2002, p. 1182). Consequently, these factors have to be controlled for when learning effects are to be assessed empirically.[49]

The first empirical evidence of production costs to decrease as cumulative output increases was provided by Wright (1936). He found that the number of labour-hours required to produce an airframe was a decreasing function of the total number of airframes of the same type previously produced. He observed a progress ratio of 79 percent.[50] Subsequently, learning curves have been confirmed for a multiplicity of products in the manufacturing and service sectors (for overviews see, e.g., Lloyd 1979, p. 222; Argote and Epple 1990, p. 920). Progress ratios in these sectors tend to lie between 70 and 80 percent. McDonald and Schrattenholzer (2001) provide a good overview of learning curves for energy technologies. They identify a median value of 83 to 84 percent for the corresponding progress ratio. These studies usually regress total production costs over cumulative output, and do not control for other drivers of cost reductions, such as economies of scale.

Empirical studies in the field of renewable energy technologies have focused on wind energy. There is a variety of studies that estimate cost reductions as a function of cumulative output without controlling for economies of scale and other factors. Using international data, Neij (1997, pp. 1101-1102) estimates a progress ratio of 91 to 96 percent. Country-specific studies assess progress ratios of 92 percent for Germany (IEA 2000, p. 54), 92 to 98 percent for Denmark (Neij 1999, p. 380; Ibenholt 2002, p. 1185), 75 to 81 percent for the United Kingdom (Ibenholt 2002, p. 1185; Junginger et al. 2005, p. 145), 82 to 85 percent for Spain (Junginger et al. 2005, p. 145), and 68 to 84 percent for the United States (Christiansson 1995, p. 11; IEA 2000, p. 42). Isoard and Soria (2001) and Hansen et al. (2003) control for scale effects when estimating learning effects. Isoard and Soria identify decreasing returns to scale, which they argue may occur at the outset of the deployment of new technologies. They use international data and find that the progress ratio decreases from 85 to 83 percent when scale effects are controlled for (Isoard and Soria 2001, p. 630). In contrast, Hansen et al. provide evidence of increasing returns to scale for the Danish context. In their study, controlling for scale effects increases the progress ratio from

49 There are several studies which use very broad concepts of learning or experience curves including also (some of) these factors (e.g., BCG 1968; Lloyd 1979; Neij 1997; Neij 1999).

50 Several studies do not estimate progress ratios (PR) but rather elasticity or learning ratios (LR), where $LR = 1 - PR$. For ease the comparison of different empirical studies, only the corresponding progress ratios will be provided in the remainder of this chapter.

89 to 91 percent (Hansen et al. 2003, p. 332). Moreover, several authors conduct multi-country studies in which they also consider the impact of private and public expenditures for R&D on cost reduction (Kouvaritakis et al. 2000; Klaassen et al. 2005; Söderholm and Klaassen 2007). They derive values of progress ratios between 84 and 97 percent.

Progress ratios found for photovoltaic technologies are usually below those for wind technologies. Ratios estimated for the world, the United States and Japan vary from 78 to 82 percent (Christiansson 1995, p. 11; Neij 1997, p. 1102; IEA 2000, p. 11). Photovoltaics in the European Union even exhibit a progress ratio of 65 percent (IEA 2000, p. 21). Decreasing returns to scale seem to be more important for photovoltaic technologies than for wind technologies. Isoard and Soria (2001, p. 629) show that controlling for scale effects reduces the progress ratio from 91 to 72 percent. Considering R&D expenditures, Kouvaritakis et al. (2000, p. 111) provide a similar progress ratio for learning of 75 percent.

Learning has also been observed for other renewable energy technologies. An 85 to 91 percent progress ratio has been identified for electricity generated from biomass in the EU (IEA 2000, p. 80; Junginger et al. 2006). Biogas production displays a progress ratio of 82 to 85 percent (Kouvaritakis et al. 2000, p. 111; Junginger et al. 2006). Learning effects are minor but existent for the costs of small as well as large hydro power plants, exhibiting progress ratios of 98 percent (Kouvaritakis et al. 2000, p. 110).

A variety of reasons can be employed to explain differences in progress ratios. Obviously, relatively more mature technologies such as hydro or wind exhibit smaller learning effects than relatively immature technologies such as photovoltaics. The time period under consideration is also of importance as progress ratios tend to be lower in earlier stages of technology diffusion (Christiansson 1995, p. 3; Ibenholt 2002, p. 1185; Junginger et al. 2005, p. 145). In addition, studies employ different measures of performance (investment costs versus production costs) and experience (cumulative capacity versus cumulative production) (McDonald and Schrattenholzer 2001, p. 260). What is more, studies may be based on different concepts of learning and fail to correct for economies of scale, R&D investments, differences in the political context and other factors driving production costs. Finally, differences in progress ratios may be explained by the fact that experience may depreciate as well, e.g. during strikes or production cuts, and spill over to other firms and countries (Argote and Epple 1990, p. 921; McDonald and Schrattenholzer 2001, p. 260) (see Section 4.2.1.2 for a more extensive discussion of spillovers).

Notwithstanding variations in the estimation of the progress rate and empirical as well as theoretical limitations of the learning concept, the presented studies provide an overwhelming amount of evidence of the existence of learning. It can therefore be assumed in the remainder of this chapter that learning effects related to renewable energy technologies are significant. Uncertainty merely seems to surround the actual extent of learning. As the following analysis is of qualitative character, this uncertainty can be neglected.

4.2.1.2 Learning Spillovers with Renewable Energy Technologies

As has been outlined already in Section 3.3.1, learning effects may be internal or external to an organization. Economic actors may be unable to perfectly protect their experience gained through production. Arrow (1962, p. 168) was among the first to emphasize that at least some of this knowledge may spill over to other actors. These may benefit from learning without compensating the learning firm and without having invested in learning themselves. This is the classical case of a positive externality (Jaffe et al. 2002, p. 48). These externalities may occur between firms within one country, but also across countries. In turn, incentives for the learning firm to invest in learning are limited to expected private returns on investment. The aggregate amount of investment in the absence of regulation will therefore fall short of the socially optimal level. If cumulative output is assumed to be a proxy for learning, spillovers imply that firms produce too little compared to the socially optimal level of output (Irwin and Klenow 1994, p. 1201). Suboptimal levels of learning result in renewable energy technologies being more expensive than in the absence of spillovers. Neuhoff (2005, p. 98) highlights that renewable energy technologies may therefore be subject to a competitive disadvantage compared to mature fossil-fuel technologies, which have already experienced significant cost reductions through learning. In this respect, spillovers may strengthen existing path dependencies in the energy sector. Path dependencies imply that decisions in the future depend on previous patterns of investment (for an extensive discussion see Arthur 1989). As a consequence, learning spillovers may contribute to a lock-in of a fossil-fuel-based energy system and a lock-out of viable alternatives provided by renewable energy technologies (Jaffe et al. 2002, p. 49; Neuhoff 2005, p. 97).

There are many modalities by which experience may be transferred from one firm to another. First of all, there seems to be a significant degree of mobility of engineers and other skilled personnel across firms (Irwin and Klenow 1994, p. 1205). In addition, knowledge may spill over through (informal) communication, participation in meetings and conferences, and trainings. What is more, firms may engage in "reverse engineering", using available products of competitors or suppliers to derive insight on the production process (Argote and Epple 1990, p. 923). Neuhoff (2005, p. 97) points out two reasons why it may be difficult to prevent such spillovers for renewable energy technologies: First of all, it is generally difficult to perfectly define engineering patents. Secondly, renewable energy technologies consist of many components supplied by a multitude of companies. Consequently, expertise from many firms is required to improve the system. It may be difficult to arrive at an agreement to share the costs of learning investments due to the related transaction costs of negotiating.

That knowledge generated within a firm is hard to protect has been shown extensively for R&D (Jaffe 1986; Jaffe et al. 1993; Jaffe and Trajtenberg 1996; Margolis and Kammen 1999). Branstetter (2001) shows that firms actually benefit more from total domestic R&D expenditures than from their own R&D expenditures. Using firm panel data from different US industry sectors, he finds that the elasticity of an individual firm's level of patenting is 0.63 with respect to own R&D investments but 0.97 with respect to total domestic R&D investments (Branstetter 2001, p. 74). Moreover, several studies also provide evidence of substantial international R&D spillovers (Coe and Helpman 1995; Coe et al. 1995; Bernstein and Mohnen 1998; Keller 1998). They find a significant positive relationship between aggregate R&D capital accumulated abroad and own country growth in factor productivity. These spillovers can in fact happen quite quickly. Mansfield (1985) reveals that 70 percent of new product innovations "leak out" within only one year.

However, a somewhat different methodology has to be employed to assess spillovers related to learning. Empirical analyses have to control simultaneously for the effects of an individual firm's cumulative output of a product and the corresponding cumulative production in the entire industry. There is only a limited amount of empirical studies on learning spillovers. For the semiconductor industry, Irwin and Klenow (1994) find – and are confirmed by Gruber (1998) – that spillovers between domestic and international firms are statistically significant.[51] Lieberman (1984, p. 226) assumes a link between the rate of cost reduction and multiplant operation in the chemical processing industry. He tests the hypothesis that if knowledge is purely private and shared only across multiple plants belonging to the same firm, cost reductions should be faster if production in a fixed number of plants is controlled by a smaller number of firms. Results show that this hypothesis has to be rejected. Lieberman concludes that these results suggest that information diffuses at a high rate not only within firms but also across firms. Learning spillovers

[51] However, it is also shown that internal, private learning is more important than learning spillovers: An individual firm learns about three times as much from an additional unit of its own cumulative output as from an additional unit of another firm's cumulative output (Irwin and Klenow 1994, p. 1217).

have also been observed for other manufacturing industries (Barrios and Strobl 2004), nuclear power plants (Zimmerman 1982; Lester and McCabe 1993) and agricultural production (Foster and Rosenzweig 1995). However, there are also some studies which do not find empirical evidence of cumulative industry output fostering firm-level learning, e.g. for energy technologies (Joskow and Rose 1985) or cars (Levin 2000, p. 643).

There are hardly any empirical analyses available regarding learning spillovers for renewable energy technologies. Some studies provide an indication at least. The IEA (2000, p. 56) observes that learning effects for wind turbines are stronger – i.e. that progress ratios are lower – in Germany than in Denmark (a 92 percent ratio compared to a 98 percent ratio). The IEA argues that spillovers may be a possible explanation of this difference. German wind turbine manufacturers may have "imported" experience from Denmark. Hansen et al. (2003, p. 328) highlight that the Danish wind industry is dominated by four firms, which account for 90 percent of Denmark's production of wind turbines. These firms operate in an industrial cluster. They draw on the same pool of highly skilled labour and profit from the same public-sector facilities. Hansen et al. find it therefore reasonable to assume that learning spillovers between Danish firms are significant.

The review of literature on learning spillovers reveals an important avenue for future research. Empirical analyses are required which shed light on the actual extent of learning spillovers related to renewable energy technologies. There are indications that spillovers may be significant. However, empirical evidence is available which confirms knowledge spillovers in general and learning spillovers in particular for many other manufacturing sectors. It is therefore reasonable to assume that the pollution externality caused by CO_2 emissions is in fact coupled with learning spillovers in the field of renewable energy technologies. This provides a rationale for combining an emissions trading scheme and a feed-in tariff for renewable electricity generation. It has to be revised, though, whether the design of the theoretically proposed policy mix actually corresponds to the design of actually existing policy mixes as they are in place, for example, in Germany. This revision is undertaken in the subsequent section.

4.2.2 Applicability of the Policy Mix Design

Two studies are available that address explicitly the question of optimal policy mix design in the presence of a pollution externality and learning spillovers. Fischer and Newell (2008) use a partial equilibrium model of the electricity sector with two time periods. They assume that electricity can be produced by fossil-fueled electricity generators and operators of renewable energy installations. Fossil-fueled electricity generators produce emissions and cause a negative pollution externality. Renewable operators experience learning effects. The cost of electricity produced by one operator in the present period decreases as a function of its own and total cumulative electricity generation in the previous period. Experience gained by one operator spills over to other operators to a certain extent. Fischer and Newell show that a first-best outcome can be attained in their model by a policy mix consisting of an emissions policy and a support scheme for renewables. A fixed emissions price should be imposed on fossil-fueled generators. Operators of renewable energy installations should receive an output subsidy per unit of electricity generated in addition to the output price.

Bläsi and Requate (2007) adopt a more differentiated two-period model of the electricity sector. They account for the vertical structure of the sector and distinguish between fossil-fueled electricity generators, operators of renewable energy plants and producers of renewable energy technologies. As Fischer and Newell (2008), they assume that fossil-fueled generators produce a negative pollution externality. However, in contrast to Fischer and Newell (2008), operators of renewable energy installations are not subject to learning effects. They rather benefit from learning experienced by producers of renewable energy technologies, which sell their technology to operators. Due to learning, the production costs of a unit of a renewable energy technology decline as a function of the cumulative number of units produced in the first period. Spillovers are modelled by making the degree of learning dependent on an individual producer's own output and total output of all producers. Bläsi and Requate find that an optimal policy mix includes an emissions policy as well as an output subsidy in addition to the output price for producers (not operators) of renewable energy technologies. Moreover, they allow for free firm entry to the producers' market and show that with learning spillovers an additional policy is necessary to attain a socially optimal outcome. When deciding about market entry, firms do not consider that their entry produces a benefit to other market participants in terms of learning. Market entry will be suboptimal in the absence of regulation and needs to be stimulated by an entry premium.

Both studies provide the theoretical basis for combining an emissions policy and a support scheme for renewable energy technologies in the presence of a pollution externality and learning spillovers. The previous section has revealed that learning spillovers related to renewable energy technologies are likely to be significant. Consequently, the discussion of a policy mix to address both types of failures of private governance structures simultaneously is not only theoretically interesting but also empirically important. It provides a clear argument in favour of implementing an emissions trading scheme combined with feed-in tariffs for renewable electricity generation –

a policy combination that can be observed in Germany (see Sections 2.3.1 and 2.3.2) and many other industrialized countries.

Nevertheless, existing theoretical studies are subject to major constraints. An important limitation of the studies cited above consists in their very simplistic representation of policies. This restriction can be clearly shown by comparing the theoretical policy design with that observed in Germany. Four deviations between theory and the German case study are particularly striking (see Table 4-1): *First of all*, in contrast to theoretical studies, the output subsidy in Germany's feed-in tariff system is not paid in addition to the output price, i.e. the price of electricity. Rather, operators of renewable energy plants receive a fixed remuneration per unit of electricity produced which amounts to the feed-in tariff only and which is irrespective of the prevailing electricity price. *Secondly*, the modes of funding the output subsidy vary significantly between the theoretical analyses and the German case. In the theoretical studies cited above it is assumed that funding is exogenous to the electricity sector, e.g. by general tax revenues. In contrast, the German feed-in tariff system is designed to be revenue-neutral. Feed-in tariffs are funded by an endogenous add-on to the electricity price. A *third* deviation refers to the recipients of the output subsidy. In Germany, feed-in tariffs are paid to the operators of renewable energy installations as in Fischer and Newell (2008) – but not to producers of these plants as proposed by Bläsi and Requate (2007). *Finally*, the design of the emissions policy assumed in theory is not identical to that implemented in Germany. The theoretical studies cited assume a fixed emissions price – as it is set by an emissions tax. In Germany, however, the ETS has been implemented. The emissions price, i.e. the allowance price, arising under the ETS is not fixed. In fact, it may vary depending on variables that are exogenous as well as endogenous to a partial equilibrium model of the electricity sector.

Taking these four deviations into account, the cumulative incentives set by the German policy mix vary from those assumed in theoretical models. This implies that existing theoretical studies are very restricted in their capacity to evaluate an actually existing policy mix as that in Germany and to derive appropriate policy recommendations. They do not provide an answer to the questions whether the complex design of an existing policy mix still allows for addressing a pollution externality and learning spillovers efficiently.

Table 4-1: Deviations in policy mix design between theoretical studies and the German case

	Theory	Germany
Technology subsidy		
Production incentives for learning sector	output subsidy and output market price	output subsidy (feed-in tariff) only
Funding of subsidy	exogenous to the electricity sector	endogenous to the electricity sector
Recipient of subsidy	producers of renewable technologies (Bläsi and Requate 2007)	operators of renewable technologies
Emissions policy		
Emissions price	fixed (emissions tax)	variable (emissions trading)

The analysis in this chapter contributes to filling this important gap in the economic theory of a policy mix. The discussion particularly addresses the first two limitations cited above (see light-grey-shaded rows in Table 4-1). For this purpose, the simple model of Fischer and Newell (2008) is advanced in the following section. The simplistic assumptions about policy design are substituted by the more complex characteristics of the policy mix that can be found in Germany. Firstly, it is assumed that production decisions in the learning renewable energy sector depend on the feed-in tariff only and are irrespective of the prevailing market price of electricity. Secondly, the feed-in tariff is not funded by general tax revenues but by an add-on to the electricity price which is endogenous to the partial equilibrium model of the electricity sector. Given these design specifications, the analysis in the subsequent section examines the efficiency properties of the policy mix and compares them with those of the simplistic policy mix studies available so far.

The theoretical discussion in the subsequent section does not address distortions that arise from the fact the operators instead of producers of renewable energy installations are supported by a feed-in tariff (third deviation highlighted above). This is because such comparison would be of little empirical relevance. As Bläsi and Requate (2007, p. 12) themselves state, direct subsidies to producers of renewable energy installations would be ruled out by rules of the World Trade Organization – especially if installations are traded internationally. Therefore, it is more worthwhile to study and compare the performance of different schemes of feed-in tariffs paid to operators instead of producers of renewable energy technologies. This can be done by advancing the relatively simple model used by Fischer and Newell (2008), which neglects the vertical structure of real electricity markets and assumes operators of renewable energy technologies to experience learning effects.

The analytical determination of equilibrium effects of using a variable rather than a fixed emissions price in a policy mix, i.e. an emissions trading scheme instead of an emissions tax (fourth deviation listed above), is left for future research. However, possible consequences will be discussed in the context of the comprehensive empirical evaluation of the German case study provided in Chapter 6.

4.3 Model

This section encompasses the formal analytical discussion of the policy mix of an emissions policy and a support scheme for renewable energy technologies. First of all, the model is introduced (Section 4.3.1) and the conditions for the social optimum are derived (Section 4.3.2). On this basis, the simple policy mix to address a pollution externality and learning spillovers is presented, as proposed by Fischer and Newell (2008) (Section 4.3.3). Subsequently, the assumptions of Fischer and Newell are modified to account for the more complex design of a feed-in tariff as it is found in Germany. Based on this modification, the efficiency properties of the policy mix are derived (Section 4.3.4).

4.3.1 Model Assumptions

The analysis is based on a stylized partial equilibrium model of the electricity sector. The assumptions regarding the characteristics of the electricity sector mostly correspond to those made by Fischer and Newell (2008).[52] The electricity sector encompasses two subsectors. One sector uses fossil fuels to generate electricity and produces emissions of carbon dioxide as a by-product. The other sector employs renewable energy sources, such as wind or solar, which are carbon-free. Both sectors are perfectly competitive and produce an identical output, electricity.

[52] In their model, Fischer and Newell (2008) do not only consider a pollution externality and learning spillovers but also R&D spillovers. The latter assumption is skipped to focus the analysis on policies to address the first two failures of private governance structures.

Any electricity generated from renewable energy sources substitutes marginal fossil-fuel production.[53]

The model has two periods. Electricity generation, consumption and emission occur in both periods. Firms take the electricity price as given not only in the first period but also in the second period. Moreover, they are assumed to have perfect foresight regarding the price in period two. There is discounting at rate δ between periods, but not within each period. Social and private discounting rates are assumed to be identical.

The fossil-fuel sector may choose between a technology using a carbon-intensive fossil fuel x, e.g. coal, and a technology using a low-carbon fuel y, such as natural gas. The former technology supplies base-load while the latter is the marginal technology producing peak-load electricity. The total output of electricity produced in the emitting sector in period t is $f_t = x_t + y_t$. Emissions with each technology i are assumed to be fixed at rate μ_i. This assumption reflects that fuel switching is the major means to reduce emissions in the emitting sector. Other measures, such as improvements in generation efficiency or carbon capture and sequestration, are of minor importance at the moment because they have limited emission reduction potentials or are in very early stages of technological development. Total emissions from the emitting sector in period t are $E_t = \mu_x x_t + \mu_y y_t$. Production costs of each technology in each period, $c_i^t(x_t)$ and $c_i^t(y_t)$, are assumed to be increasing in output and strictly convex, i.e. $c_i^{t\prime} > 0$ and $c_i^{t\prime\prime} > 0$.

The renewable sector consists of n identical firms, each of which produces an electricity output q_t in period t.[54] The production costs are given as in Bläsi and Requate (2005; 2007). Their approach is preferred to that used by Fischer and Newell (2008) since Bläsi and Requate only consider the impact of learning effects on production costs. In contrast, Fischer and Newell apply a broader approach analyzing the impact of a knowledge stock which depends not only on

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[53] This model abstracts from nuclear and hydro as important further energy sources currently used. However, these sources are carbon-free and employed to generate base load electricity. Their output can be assumed to be fixed in the presence of emission policies and support schemes for renewable energy sources. Thus, integrating them into the model would not change the analytical results.

[54] The number of firms in the renewable sector is assumed to be constant in this model. As mentioned in Section 4.2.2, Bläsi and Requate (2005; 2007, p. 166) allow for firm entry and show that with learning spillovers an entry premium is necessary in addition to an emissions policy and an output subsidy to the learning industry to attain a socially optimal outcome.

learning but on R&D investments as well. This assumption would go beyond the scope of this analysis.

Production cost in period one is $g^1(q_1)$. Production cost in period two is a function of output in period two and the total level of learning (or experience) L in period one, i.e. $g^2(q_2, L)$.[55] Total learning depends on the output of the firm under consideration (private or internal learning) and the output of all other identical firms in the sector (learning spillovers or external learning): $L = q_1 + \rho(n-1)q_1$. The spillover rate ρ indicates to which extent a firm can benefit from the experience made by other firms. Production costs in each period are increasing and convex in output, i.e. $g^t_{q_t} > 0$ and $g^t_{q_t q_t} > 0$. Production cost in period two is declining and convex in learning: $g^2_L < 0$ and $g^2_{LL} > 0$. This assumption reflects the empirical findings on the typical form of the learning curve presented in equation (4.1) in Section 4.2.1.1. Learning also reduces marginal production cost in period two, i.e. $g^2_{q_2 L} < 0$. Moreover, production cost in period two is assumed to be convex overall, which requires that $g^2_{LL} g^2_{q_2 q_2} - \left(g^2_{q_2 L}\right)^2 > 0$. Subscripts q_t and L denote partial derivatives with respect to the subscripted variable. Notably, as in Fischer and Newell (2008), learning is assumed for the relatively immature renewable energy technologies but not for the relatively mature fossil-fuel technologies.[56]

Total output of the electricity sector in period t is the sum of electricity generated in the fossil-fuel sector and the renewables sector: $Q_t = f_t + nq_t$. In equilibrium, electricity output equals electricity demand. The inverse demand function can then be given by $p_t = p_t(Q_t)$, where p_t is the market price for electricity in period t. This function is downward sloping, i.e. $p_t'(Q_t) < 0$.

[55] As stated in Section 4.2.2, the renewables sector in reality exhibits a vertical industry structure consisting of producers and operators of renewable energy installations. Producers rather than operators – as assumed in this analysis – experience learning effects. The resulting distortions of paying the subsidy to the operator rather than the producers are discussed in Section 6.2.1.2.

[56] Learning effects (and related spillovers) in the fossil-fuel sector are neglected because they are relatively less important and would unnecessarily complicate the analysis. In fact, such enhanced analysis would reveal that an additional policy is needed to promote learning in the presence of spillovers in the fossil-fuel sector. Such policy may be required in particular once new promising but immature technologies – such as carbon capture and storage – are to be adopted in the fossil fuel sector.

Carbon dioxide emitted by the fossil-fuel sector in period t produces damage to society, which depends on the overall level of emissions: $D_t(E_t)$. Damage is assumed to be increasing and convex in emissions, i.e. $D_t' > 0$ and $D_t'' \geq 0$.

Social welfare W over the two periods under consideration is given by:

$$W = \int_0^{Q_1} p_1(Q)dQ - c_x^1(x_1) - c_y^1(y_1) - ng^1(q_1) - D_1(E_1)$$

$$+ \delta\left[\int_0^{Q_2} p_2(Q)dQ - c_x^2(x_2) - c_y^2(y_2) - ng^2(q_2, L) - D_2(E_2)\right] \tag{4.2}$$

Thus, social welfare computes as the sum of consumer surplus and firm profits net of production costs with coal, natural gas and renewable energy sources and environmental damage from emissions in the first-period and the same values discounted for period two.

4.3.2 The Social Optimum

The social planner maximizes welfare with respect to electricity generation from coal, natural gas and renewable energy sources in both periods, x_t, y_t and q_t. The resulting first-order conditions are:

$$p_1(Q_1) = c_x'^1(x_1) + D_1'(E_1)\mu_x \tag{4.3}$$

$$p_2(Q_2) = c_x'^2(x_2) + D_2'(E_2)\mu_x \tag{4.4}$$

$$p_1(Q_1) = c_y'^1(y_1) + D_1'(E_1)\mu_y \tag{4.5}$$

$$p_2(Q_2) = c_y'^2(y_2) + D_2'(E_2)\mu_y \tag{4.6}$$

$$p_1(Q_1) = g_{q_1}^1(q_1) + \delta\left[g_L^2(q_2, L) + g_L^2(q_2, L)\rho(n-1)\right] \tag{4.7}$$

$$p_2(Q_2) = g_{q_2}^2(q_2, L) \tag{4.8}$$

These equations represent the conditions for a socially optimal allocation of resources. Conditions (4.3) to (4.6) require that the emitting fossil-fuel sector generate electricity from coal and natural gas in period one and two until the sum of marginal production cost and marginal environmental damage of either technology equals the willingness to pay for another unit of electricity (represented by the market price). Thus, when deciding about its output, the fossil-fuel sector should consider the private and social costs of its electricity generation. Condition (4.7) implies that firms in the renewable sector should produce electricity in period one until the marginal willingness to pay equals the sum of marginal production costs in period one and the discounted marginal reduction of production costs in period two due to learning experienced by all firms in period one. Thus, when deciding about their output in period one, firms in the renewable sector should not only consider learning at the private level but also learning spillovers. Finally, condition (4.8) highlights that firms in the renewable sector should produce electricity in period two until their marginal production costs equals the marginal willingness to pay.

4.3.3 Emissions Price and Output Subsidy: The Optimal Policy Mix

It is shown in this section, that in the presence of two externalities – a pollution externality and learning spillovers – the socially optimal levels of output in the fossil-fuel and the renewable sector can be attained by two simple policies: a fixed emissions price and a subsidy to output of the renewables sector. The analytical results derived in this section correspond to those presented by Fischer and Newell (2008). The results provide the benchmark for the evaluation of a more complex policy mix in the subsequent Section 4.3.4.

A fixed price τ_t per unit of emissions is imposed on the fossil-fuel sector. Consequently, this sector faces the following maximization problem:

$$\max_{x_t, y_t} \pi^F = p_1(x_1 + y_1) - c_x^1(x_1) - c_y^1(y_1) - \tau_1(\mu_x x_1 + \mu_y y_1)$$

$$+ \delta\left[p_2(x_2 + y_2) - c_x^2(x_2) - c_y^2(y_2) - \tau_2(\mu_x x_2 + \mu_y y_2)\right] \tag{4.9}$$

Superscript F denotes the fossil-fuel sector. The resulting first-order conditions for optimal electricity generation from coal and natural gas in period one and two are:

$$p_1 = c_x'^1(x_1) + \tau_1 \mu_x \tag{4.10}$$

$$p_2 = c_x'^2(x_2) + \tau_2 \mu_x \tag{4.11}$$

$$p_1 = c_y'^1(y_1) + \tau_1 \mu_y \tag{4.12}$$

$$p_2 = c_y'^2(y_2) + \tau_2 \mu_y \tag{4.13}$$

Conditions (4.10) to (4.13) can be interpreted as the inverse supply curves for electricity from the fossil-fuel sector. It will produce electricity from coal and natural gas until the sum of marginal production costs and marginal emissions costs with either technology is equal to the market price of electricity. As in Fischer and Newell (2008, p. 146), an interior solution is assumed, i.e. no fuel is completely driven out of the market. In the absence of an emissions policy, an increased electricity supply from renewable energy sources reduces the electricity production from coal and natural gas according to the respective supply curves. An increase in the stringency of the emissions policy will result in a larger reduction of production from emission-intensive coal generation than from low-emission natural gas generation.

Firms in the non-emitting renewables sector receive a subsidy s per unit of output to foster learning. Thus, they get remuneration per unit of output which consists of a variable component (the electricity price) and a fixed component (the output subsidy). Their maximization problem writes as follows:

$$\max_{q_t} \pi^R = (p_1 + s_1)q_1 - g^1(q_1) + \delta\left[(p_2 + s_2)q_2 - g^2(q_2, L)\right] \tag{4.14}$$

Superscript R denotes the renewable sector. Note that, by assumption, firms only consider the effect of private learning on production costs in period two but not that of learning spillovers to other firms, i.e. $L = q_1$ at the private level (Bläsi and Requate 2005, p. 8). The resulting first-order conditions for firms in the renewables sector are:

$$g^1_{q_1}(q_1) = p_1 + s_1 - \delta g^2_L(q_2, L) \tag{4.15}$$

$$g^2_{q_2}(q_2, L) = p_2 + s_2 \tag{4.16}$$

Condition (4.15) implies that firms produce electricity from renewable energy sources in period one until the marginal production cost in period one is equal to the marginal benefits of production to the firm. These include the market price for electricity, the output subsidy and the discounted reduction of production cost in period two which can be appropriated by the firm (note that term $-\delta g^2_L(q_2, L)$ is overall positive). In period two, there are no learning effects by assumption. Thus, firms produce until marginal production cost in period two equal the sum of the electricity price and the output subsidy (condition (4.16)).

The optimal policy levels can be obtained by comparing the first-order conditions for maximum social welfare derived in Section 4.3.2 with those for maximum firm profits. Equating conditions (4.3) and (4.5) with conditions (4.10) and (4.12) yields the optimal emissions price in period one:

$$\tau_1 = D_1'(E_1) \tag{4.17}$$

The optimal emissions price in period one has to be set equal to the marginal damage from emission, i.e. at the Pigovian level. Similarly, equating conditions (4.4) and (4.6) with conditions (4.11) and (4.13) gives the optimal emission price in period two:

$$\tau_2 = D_2'(E_2) \tag{4.18}$$

Thus, the optimal emissions price is uniform for coal- and natural gas-based electricity generators. Given that marginal environmental damages from emission are constant, the emissions price is also fixed over time.

Equating conditions (4.7) and (4.15) gives the optimal output subsidy to electricity produced in the renewables sector:

$$s_1 = -\delta g_L^2(q_2, L)\rho(n-1) \tag{4.19}$$

The optimal subsidy is positive since the marginal effect of learning on production, $g_L^2(q_2, L)$, is negative. The subsidy equals the gains from learning not considered by the firms in the renewable sector. This implies that if learning is purely private, an output subsidy in period one cannot be justified on efficiency grounds. Comparing conditions (4.8) and (4.16) reveals that no subsidy to the output of the renewable sector is needed in period two, i.e. $s_2 = 0$. This is straightforward since no learning and, consequently, no learning spillovers are assumed to occur in the model in period two.

Equation (4.19) reveals that the optimal output subsidy to renewable electricity has to be adapted over time – in contrast to the optimal emissions price. The subsidy has to be decreased as the marginal effect of learning on period two production costs declines, i.e. as renewable energy technologies become more mature. Moreover, the subsidy has to be increased as the number of firms in the renewable sector increases. This is because with more adopters of renewable energy technologies, the overall extent of learning increases. Consequently, learning spillovers will also be higher.

Moreover, equation (4.19) shows that the optimal subsidy does not necessarily have to be uniform across all types of renewable energy technologies. If renewable technologies differ with respect to marginal gains from learning in period two in absolute terms, $g_L^2(q_2, L)$, a differentiation of the subsidy can be justified on efficiency grounds. This case arises when technologies are characterized by different levels of maturity. The subsidy should be lower for relatively mature technologies, such as wind power, than for relatively immature technologies, such as photovoltaics. Moreover, differences between technologies with respect to the spillover rate, ρ , or the number of adopting firms, $n-1$, may also call for a differentiated set of subsidies.

4.3.4 Emissions Price and Feed-in Tariff

This section revises the analysis of the previous section in order to consider the more complex features of a feed-in tariff system as it exists in Germany (see Section 2.3.2). In this respect, the examination in this section goes beyond the analysis of Fischer and Newell (2008) and approaches policy mix theory to reality.

It is now assumed that producers of renewable electricity receive a fixed feed-in tariff σ_1 per unit of electricity produced in period one irrespective of the electricity price.[57] This is in contrast to the analysis in the previous section where an output subsidy was paid to renewable electricity generators in addition to the electricity price. The feed-in tariff includes an implicit subsidy which amounts to the difference between the feed-in tariff and the electricity price. Unlike traditional output subsidies, this implicit subsidy is not exogenously funded by the government – or general tax revenues. Instead, grid operators buying renewable electricity are allowed to spread the difference between the feed-in tariffs paid and the electricity price across all electricity customers. This results in a uniform add-on φ_1 to the electricity price in period one. Thus, within the partial-equilibrium model of the electricity market, the feed-in tariff is revenue-neutral. Revenues from raising the add-on always have to equal the expenditures on tariffs paid for electricity from renewable energy sources net of the prevailing electricity price, i.e. $\varphi_1 Q_1 = (\sigma_1 - p_1)nq_1$. Thus, the add-on to the electricity price computes as the difference of the feed-in tariff and the electricity price times the share of renewable electricity in total electricity supply:

$$\varphi_1 = (\sigma_1 - p_1)\frac{nq_1}{Q_1} \tag{4.20}$$

In a competitive market, electricity producers are assumed to take this add-on as given (similar to an output tax). However, the add-on only affects production choices of fossil-fuel generators. This is because renewable generators benefits from the obligation of grid operators to purchase renewable electricity preferentially. This implies that any reductions in demand for electricity resulting from a price increase induced by the add-on have to be borne by fossil-fuel generators only. In period two, electricity from renewable energy sources is assumed to compete with electricity from fossil-fuels at the market price.

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[57] Recall that since no learning occurs in period two, no promotion of renewable electricity is required in period two.

The profit maximization problem for firms in the fossil-fuel sector can then be rewritten:

$$\max_{x_t, y_t} \pi^F = p_1(x_1 + y_1) - c_x^1(x_1) - c_y^1(y_1) - \varphi_1(x_1 + y_1) - \tau_1(\mu_x x_1 + \mu_y y_1)$$

$$+ \delta\left[p_2(x_2 + y_2) - c_x^2(x_2) - c_y^2(y_2) - \tau_2(\mu_x x_2 + \mu_y y_2)\right] \tag{4.21}$$

The resulting first-order conditions for optimal electricity generation from coal and natural gas in both periods are:

$$p_1 = c_x'^1(x_1) + \varphi_1 + \tau_1 \mu_x \tag{4.22}$$

$$p_2 = c_x'^2(x_2) + \tau_2 \mu_x \tag{4.23}$$

$$p_1 = c_y'^1(y_1) + \varphi_1 + \tau_1 \mu_y \tag{4.24}$$

$$p_2 = c_y'^2(y_2) + \tau_2 \mu_y \tag{4.25}$$

Thus, when deciding about its output in period one, the fossil-fuel sector now produces until the sum of marginal production costs, the add-on per unit of output produced and the emission costs per unit of output equal the market price of electricity (conditions (4.22) and (4.24)). Conditions (4.23) and (4.25) for optimal output in period two are identical to those derived above.

The profit maximization problem for firms in the renewables sector is:

$$\max_{q_t} \pi^R = \sigma_1 q_1 - g^1(q_1) + \delta\left[p_2 q_2 - g^2(q_2, L)\right] \tag{4.26}$$

The corresponding first-order conditions for optimal output in the renewables sector are:

$$g_{q_1}^1(q_1) = \sigma_1 - \delta g_L^2(q_2, L) \tag{4.27}$$

$$g_{q_2}^2(q_2, L) = p_2 \tag{4.28}$$

In period one, firms in the renewables sector produce until their marginal costs of electricity generation equal the sum of the feed-in tariff and the discounted reduction in production costs in period two due to private learning. Optimal production in period two is only determined by the market price of electricity.

Comparing the above first-order conditions with those providing optimal social welfare reveals the optimal policy design. Comparing conditions (4.22) to (4.25) with conditions (4.3) to (4.6) gives the optimal set of emissions prices:

$$\tau_{1x} = D_1' - \frac{\varphi_1}{\mu_x} \tag{4.29}$$

$$\tau_{1y} = D_1' - \frac{\varphi_1}{\mu_y} \tag{4.30}$$

$$\tau_2 = D_2' \tag{4.31}$$

The optimal emissions price in period two has to be set at the Pigovian level, i.e. equal to marginal damages, since no additional policy affects the behaviour of the fossil-fuel sector in that period. This result corresponds to that made in the previous section. However, the modification of policy design made in this section has two major implications on the optimal level of the emissions price in period one: (1) The optimal emissions price in period one is below the marginal damage of emissions. (2) The optimal emissions price has to differentiate between fossil fuels. Emissions from electricity generation with an emission-intensive fuel (coal) have to be subject to a higher price than emissions from combusting a low-emission fuel (natural gas). The explanations are as follows: (1) The add-on to the electricity price results in a reduction of electricity consumption and production. Consequently, emissions from electricity generation are reduced as well. This implies that the emissions price that is required to attain the optimal level of output and emissions must be lower than in the absence of the add-on. (2) Emission pricing has to be differentiated because the add-on reduces electricity output from fossil fuels irrespectively of the emissions related to different types of fuels. It incorporates a higher implicit emissions price, φ_1/μ_i, on low-emission fuels than on emission-intensive fuels. Consequently, the optimal emissions price in the presence of the add-on has to be lower for the low-emission fuel than for the emission-intensive fuel. This result implies that any emissions price imposed uniformly on all fuels (even if it is reduced below the marginal damage from emissions) in the presence of the add-on will bring about inefficiency in the fossil-fuel sector in period one. Electricity generation from the low-emission fuel will be below the optimal level, while that from emission-intensive fuels will be higher than optimal.

The optimal feed-in tariff in period one can be derived by equating conditions (4.27) and (4.7):

$$\sigma_1 = p_1 - \delta g_L^2(q_2, L)\rho(n-1) \tag{4.32}$$

The optimal feed-in tariff has to be set equal to the sum of the market price for electricity and the discounted marginal gains from period-one learning spillovers in period two. The optimal feed-in tariff differs from the optimal output subsidy derived in the previous section (equation (4.19)) in that it includes the electricity price. This can be explained easily. Under the output subsidy, firms' production incentives are determined by the sum of the subsidy and the electricity price. In contrast, with a feed-in tariff, incentives are only driven by the fixed tariff. This deviation has an important implication for designing the optimal feed-in tariff. The market price of electricity is a function of a variety of variables that are exogenous to the partial equilibrium model, such as the prices of crude oil and coal. Variations in these exogenous variables and the electricity therefore require a continuous adaptation of the feed-in tariff. A fixed feed-in tariff that is set with respect to some historic electricity price may therefore bring about an inefficient level of output in the renewable sector. If the current electricity price is higher (lower), electricity generation from renewable energy sources will be too low (high). Moreover, equation (4.32) reveals that conclusions made for the simple output subsidy in Section 4.3.3 also hold for the feed-in tariff. The tariff as to be adapted as technologies become more mature and the number of renewable generators increases. In addition, a differentiation of the feed-in tariff may be justified when renewable technologies differ with respect to marginal gains from learning in period two in absolute terms, $g_L^2(q_2, L)$, the spillover rate, ρ , and/or the number of adopting firms, $n-1$.

Substituting equation (4.32) into (4.29) using (4.20), the optimal emissions price in period one for fossil-fueled electricity generators with high emissions, τ_{1x}, can be rewritten:

$$\tau_{1x} = D_1' - \frac{\left[-\delta g_L^2(q_2, L)\rho(n-1)\right]\frac{nq_1}{Q_1}}{\mu_x} \tag{4.33}$$

The optimal emissions price in period one for fossil-fueled electricity generators with low emissions, τ_{1y}, writes accordingly. Equation (4.33) reveals funding the feed-in tariff by an add-on to the electricity price has another important implication for the optimal level of the emissions price. Designing the emissions price is now a much more complex issue than with an exogenously funded output subsidy for renewable electricity (see equations (4.17) and (4.18) in the previous section). The emissions price does not only have to be below the Pigovian level and differentiated across fuels. Moreover, funding the feed-in tariff by the add-on to the electricity price requires that the emissions price be adapted continuously. Adaptation of emissions price may be necessary in three respects: (1) The emissions price has to be increased as the marginal cost reduction in period 2 due to learning declines, i.e. as technologies become more mature. (2)

The emissions price has to be increased as the number of firms which may generate and benefit from learning spillovers in the renewable sector increases. These effects are owed to the design of the feed-in tariff. (3) Moreover, the optimal emissions price has to be reduced as the share of renewable electricity in total electricity increases. This adaptation is necessary due to the design of the add-on. Notably, however, the optimal emissions price is irrespective of the prevailing electricity price. This is because an increase in the electricity price results in an increase of the optimal feed-in tariff, but a decrease of the add-on. Both effects cancel out.

4.4 Summary, Discussion and Restrictions of Model Results

There is substantial empirical evidence that climate change is attributed to two reinforcing failures of private governance structures in the electricity sector: a pollution externality produced by GHG emissions and learning spillovers related to renewable energy technologies. Seminal theoretical studies show that in this case, a simple first-best policy mix includes an emissions price and an output subsidy paid to renewable electricity generators. However, the applicability of these studies for empirical climate policy problems is quite limited. One of their major restrictions consists in the too simplistic modelling of policies, which can hardly explain the effects of complex real-world policy design. This restriction impedes the use of existing theoretical results to evaluate an actually existing policy mix – such as that in Germany – and to develop to corresponding policy recommendations.

This chapter has made a significant contribution to overcoming this limitation of available policy mix studies. It has substituted the simplistic representation of climate policies by the much more complex design features as they characterize German climate policies, for example. Thereby, the economic analysis in this chapter has improved the theoretical understanding of a policy mix. More importantly, the theoretical advances achieved in this chapter help to increase to relevance of economic theory for real policy-making.

The particular innovation of economic analysis conducted in this chapter consists in two substantial modifications of existing policy mix models. These modifications have been inspired by the example of the climate policy mix implemented for the German electricity sector. First of all, the adoption of renewable energy technologies is not promoted by an output subsidy to the electricity price – as it is assumed in available studies – but by a fixed remuneration with a feed-in tariff which is irrespective of the electricity price. Secondly, the feed-in tariff is not funded by general tax revenues – as it is assumed for the output subsidy in available studies – but by an add-on to the electricity price.

The analysis in this chapter shows for the first time that a policy mix of an emissions price and the feed-in tariff system can also be designed efficiently. However, the theoretical results regarding how to design optimal policies deviate significantly from those made in previous studies. In contrast to a policy mix with a simple output subsidy, the optimal feed-in tariff does not only depend on the maturity of renewable energy technologies and the number of adopters. It also has to be adapted continuously as the electricity price changes. Moreover, funding the feed-in tariff by the add-on has three important implications for designing the emissions price efficiently. All of these implications challenge the classical concept of uniform Pigovian emissions pricing, which is also promoted by previous policy mix studies. (1) The optimal emissions price has to be below the Pigovian level. (2) The optimal emissions price has to be differentiated across fossil fuels. (3) The optimal emissions price has to be adapted on a continuous basis as renewable energy technologies become more mature, the number of adopters increases and the share of renewable electricity rises.

The requirements with respect to optimal policy design reveal that the efficient implementation of a policy mix of an emissions policy and a feed-in tariff may be much more cumbersome for regulators than using a policy mix with a simple output subsidy. Typically, changes of environmental policies cannot be realized ad hoc but rather have to be approved in a tedious political process. This implies that the society may have to incur significant transaction costs in implementing and continuously adapting an efficient policy mix with feed-in tariffs. Drawing on the conclusions made in Chapter 3, high transaction costs may challenge the superiority of the policy mix compared to single policies. If policy makers aim at implementing efficient environmental policies that are easy to administer, they may therefore want to choose a relatively simple policy mix of an emissions policy and an output subsidy for renewable energy technologies. However, such an output subsidy would have to be funded by general tax revenues (rather than a sector-specific add-on to electricity price). A final judgement on the efficiency of this policy mix would therefore also have to consider possible distortions of raising these tax revenues in a general equilibrium model.

Several research questions remain unanswered in this chapter and may provide avenues for further research. First of all, the chapter has analyzed the policy mix from the perspective of the first-best. It may be worthwhile to analyze whether a policy mix of an emissions price and a feed-in tariff does better than a single emissions price in the presence of two market failures – even though the policy mix is not designed optimally. Such analysis would help to reveal whether implementing a feed-in tariff nevertheless increases efficiency although it does not attain a first-best solution. Moreover, the chapter assumed a fixed emissions price, which would correspond to an emissions tax. However, given an emissions trading scheme, the emissions price may vary. Feed-in tariffs are likely to result in a rising market share of renewable energy sources in the electricity market and the substitution of fossil fuels for electricity generation. Consequently, less emission allowances are demanded by the electricity sector and the allowance price decreases. Other participants of the emissions trading scheme outside the electricity sector may benefit from lower allowance prices and increase their emissions. These distortions may raise overall costs of emission abatement and should also be considered in the evaluation of the policy mix. This issue calls for a further modification of existing policy mix models to allow for variable emissions prices. Possible implications will be discussed on a non-formal basis in the context of the evaluation of the German policy mix example in Chapter 6, Section 6.2.1.8.

Its theoretical character and limitations notwithstanding, the economic analysis of a policy mix conducted in this chapter is not only of high theoretical interest. It provides a unique departure from abstract policy analysis to a more applied perspective on policy evaluation. Thereby, the theoretical analysis in this chapter also provides very valuable insight for the evaluation of actually existing policy mixes – such as the EU ETS and the German EEG (see Sections 2.3.1 and 2.3.2). Nevertheless, a comprehensive and appropriate discussion has to employ an even broader perspective. It has to reflect on the actual policy design in every detail necessary and take the characteristics of the institutional environment into account. This even more applied discussion of an actually existing policy mix will be carried out in Chapter 6 when it comes to the comprehensive evaluation of the German climate policy mix in the electricity sector.

5 Emissions Trading and Taxes on Emissions and Output

5.1 Introduction

This chapter analyzes the combination of an emissions trading scheme with a tax. An emissions trading scheme sets a cap on total emissions and requires emitters to hold a sufficient amount of emissions allowances to cover their emissions. It creates a market where these allowances can be traded. The resulting allowance price guides abatement decisions of firms (see Section 2.3.1 for the German example). The emissions trading scheme is supplemented by a tax, which also has an impact on firms' abatement. In the context of this chapter, it is considered that the tax may be imposed either on emissions or on output. A prominent example of an output tax is the German electricity tax (see Section 2.3.3). The most important assumption underlying the analysis in this chapter is that firms have to comply with both policies, i.e. hold an allowance and pay the tax. Thus, incentives set by the two policies are additive. This constellation can be observed in the German case, but also in many other OECD countries. The striking question addressed in this chapter is whether such policy mix is desirable from an economic perspective.

A consultation of the results derived in Chapter 3 reveals that none of the economic rationales can be employed to support a policy mix of emissions trading and taxation with additive incentives. This implies that the combination of both policies to cope with climate change cannot be justified on efficiency grounds. This conclusion is specified by several studies (see, e.g., Heilmann 2005, pp. 11-17; Ziesing 2007b, pp. 274-280; Böhringer et al. 2008). It is argued that a policy mix is particularly inefficient in controlling emissions if an emissions tax is heterogeneous across participants in emissions trading. The straightforward policy recommendation derived by these studies is that participants in emissions trading should be exempt from taxation. Given this general finding, studies abstain from a more in-depth analysis of the policy mix.

Available studies on combining emissions trading and taxes are very abstract and restrictive in their assumptions about policy design, though. In particular, they assume that the tax is only meant to address climate change efficiently. As the German example in Section 2.3.3 illustrates, this assumption may be very far from reality. A tax may be meant to address multiple objectives – not only climate change – and to meet several criteria – not only efficiency. Moreover, the implementation of a tax may be driven by political economy considerations. If this complexity of an actually existing policy mix is taken into account, the available studies loose their theoretical appeal. It turns out that their relevance for actual policy making is quite limited. In fact, abolishing the tax when an emissions trading scheme is in place may be neither economically desirable nor politically feasible. The inefficiency in reducing GHG emissions may be traded off by benefits from attaining other policy objectives pursued by the tax. In order to derive reasonable and useful policy recommendations, a more applied policy mix analysis is therefore necessary.

This chapter aims at increasing the relevance of economic policy mix research by changing the perspective of analysis. It renounces the narrow first-best approach applied by existing studies. Instead, it assumes that a tax cannot be abolished. The tax then represents a constraint under which policy choices to mitigate GHG emissions have to be optimized. This change in perspective reveals a whole new set of important research questions, which have not been addressed so far. It calls for a more in-depth analysis of the actual extent of inefficiency as well as its drivers. It also requires a discussion of alternative policy designs to minimize the inefficiency of the policy mix. By identifying and answering these new research questions, the chapter makes a substantial contribution to improving the economic understanding of a policy mix.

The chapter is organized as follows. Section 5.2 discusses in detail why a policy mix of emissions trading and taxes with additive incentives cannot be justified on the basis of economic rationales. Moreover, the shortcomings of existing research related to this policy mix as well as the resulting research questions are highlighted. Section 5.3 introduces a simple model with emissions not depending on output. This model is used to discuss the efficiency properties of an emissions trading scheme combined with an emissions tax and to answer the open research questions related to this policy mix. Section 5.4 develops a more complex model with emissions depending on output. Using this model, further insight is gained on the performance of a policy mix encompassing emissions trading and an emissions tax. Moreover, the model is also used to analyze a policy mix encompassing an emissions trading scheme and an output tax. Section 5.5 summarizes the major findings.

5.2 Applicability of Existing Theoretical Insight

The review carried out in Chapter 3 revealed that the combination of an emissions trading scheme and a tax may increase the efficiency of pollution control under certain circumstances. This policy mix may be justified when a pollution externality cannot be corrected efficiently by a single policy because the optimal design and implementation of the policy is hampered by high transaction costs. On the one hand, transaction costs may be high because marginal pollution abatement costs are heterogeneous. In this case, designing an emissions trading scheme optimally may be cumbersome, if not impossible, for the regulator. To avoid inefficiencies in abatement, polluters should be allowed to choose between participating in the emissions trading scheme and paying a tax on emissions. The tax establishes a cap on the allowance price and prevents excessive increases of allowance prices and abatement costs (safety valve, see Section 3.4.2). On the other hand, it may be very costly for the polluter to monitor actual emissions of the participants in an emissions trading scheme. Imposing a tax on emissions which are not covered by allowances may then increase the incentives for polluters to comply with emissions trading. Consequently, the regulator may be able to reduce his monitoring efforts and costs (see Section 3.4.3). These recommendations for using a policy imply that the incentives set out by the emissions trading scheme and the emissions tax should be substitutive. That is, a polluter may choose between participating in the emissions trading scheme and paying a tax on his emissions.

However, the applicability of these theoretical studies to actually existing policy mixes may be quite limited. In many cases, real policy design may appear to be much more complex than assumed in the studies. This can be illustrated clearly by comparing the theoretical policy assumptions made above with those found for the German policy mix of the EU ETS and the electricity tax. Three deviations are striking (see Table 5-1): Firstly and most importantly, the incentives set out by the German policies are not substitutive but additive. Polluters cannot choose between obeying the rules of one policy or another. Particularly electricity generators are subject to both policies. They have to participate in the emissions trading scheme and are affected by the tax on electricity output. It is straightforward to see that such a policy mix of an emissions trading scheme and a tax does not allow the regulator to reduce transaction costs. The tax neither establishes a cap on the allowances price, nor does it set incentives to comply with the emissions trading scheme. Secondly, the tax proposed in the theoretical literature is imposed homogeneously on all participants in the emissions trading scheme. In contrast, the German electricity tax is heterogeneous. It is imposed on electricity generators but not on other sectors participating in the emissions trading scheme. Finally, theoretical analyses assume that the tax has to be paid on emissions. However, in the German context, the tax has to be paid on the electricity output. This brief comparison shows that none of the economic rationales derived in Chapter 3 can be applied to a policy mix of an emissions trading scheme and a tax which exhibits the properties of the German example. This implies that in this case, the combination of both policies to reduce GHG emissions cannot be justified on efficiency grounds. The policy mix produces a redundancy at best or an inefficiency at worst with respect to combating climate change.

Table 5-1: Deviations in designing a policy mix of emissions trading and taxation between theoretical analyses and the German case study

	Theory	Germany
Relation of policy incentives	substitutive	additive
Tax rate	homogeneous for participants in emissions trading	heterogeneous for participants in emissions trading
Tax base	emissions	output

The redundancy or inefficiency of a policy mix encompassing an emissions trading scheme and a tax with additive incentives has often been stated in economic analyses. Several authors argue that the policy mix represents a redundancy only when the emissions trading scheme is complemented by a homogeneous tax on emissions (Graichen and Requate 2003, p. 14; Heilmann 2005, pp. 11-15; Böhringer et al. 2006, p. 5; Ziesing 2007b, pp. 274-275). The allowances price is reduced by the tax rate. Compared to a situation with emissions trading only, the overall incentives for firms to abate emissions remain unchanged. Thus, the policy mix may theoretically provide for an efficient solution. However, the same result would be achieved with an emissions trading scheme alone.

Several authors also address the case of an emissions trading scheme combined with a heterogeneous emissions tax, which is only imposed on some of the participants in the emissions trading scheme (Bader 2000, pp. 221-223; Heilmann 2005, pp. 15-17; Sijm 2005, pp. 82-83; Ziesing 2007b, pp. 278-280). It is argued that the sectors subject to both policies will abate too much and too costly while other sectors facing only the emissions trading scheme will not employ relatively cheap abatement options. Consequently, the policy mix increases abatement costs compared to a single emissions trading scheme. However, the overall emissions level remains unchanged since emissions are only shifted from one sector to another. Böhringer et al. (2006; 2008) analyze the case that several countries implement a common emissions trading scheme and only some of these countries adopt emissions taxes in addition – as it has occurred in the EU. Using a numerical partial equilibrium model of the EU carbon market, they estimate that a unilateral emissions tax of five Euros per ton CO_2 would increase abatement costs under the EU ETS by 0.5 percent. Eichner and Pethig (2007) similarly assume a multi-national emissions trading scheme partly overlapping with national emissions taxes. Introducing international trade theory into the analysis, they find that for large countries national emission taxes can serve as a surrogate for import tariffs. National governments may have an incentive to use an emissions tax to manipulate the terms of allowance trade in their favor. Eichner and Pethig (2007) conclude that individual countries' choices then do not provide for an efficient allocation from the perspective of the entire group of countries. All studies clearly suggest refraining from taxing emissions that are covered by an emissions trading scheme.

However, the economic studies presented so far are subject to an important restriction: They assume that the exclusive motivation for policy-making with an emissions trading scheme and a tax is to address climate change efficiently. In doing so, available studies abstract significantly from the complexity of real-world policy-making. Most strikingly, taxes are usually not implemented one-dimensionally. As Sorrell and Sijm (2003, pp. 427-428) emphasize, taxes may also be implemented to attain other goals than climate protection and to meet other criteria than efficiency. Apart from mitigating climate change, taxes may pursue additional environmental goals such as resource conservation or the reduction of air pollution. Moreover, taxes are usually employed to raise fiscal revenues. This objective may be particularly important when an emissions trading scheme does not auction allowances but rather allocates them to polluters free of charge. Tax revenues may be needed to pursue further policy goals, such as economic development. This is definitely the case with the German electricity tax, whose revenues are used to fund pension insurances and to reduce the tax burden on labor. What is more, a regulator may be driven by equity and distributional concerns when implementing a tax. For example, imposing a tax on top of an emissions trading scheme may help to implement the polluter-pays-principle even though allowances are allocated for free – and capture the windfall profits of firms. Last but not least, the existence and maintenance of a tax may be attributed to the political economy of policy-making. Decisions may be subject to the influence of a variety of stakeholder groups. They may also be characterized by multi-level decision-making of possibly competing jurisdictions (see, e.g., Goulder and Parry 2008, p. 170). Moreover, the political process may exhibit path dependencies, which hampers the abolition of policies such as tax once they have been in place for a certain period of time (see, e.g., Woerdman 2004b). If this multiplicity of objectives, criteria and constraints for policy-making is taken into account, it becomes obvious that the focus of existing policy mix studies on the efficient mitigation of climate change is much too narrow. Consequently, their results and policy recommendations should be treated carefully. In fact, their relevance for actual policy problems may be very limited: The abolition of a tax may not be desirable from an economic perspective. The inefficiency in the policy mix with respect to reducing greenhouse gas emissions may be accepted in order to achieve further goals and to satisfy additional policy criteria and constraints. Moreover, the abolition of a tax may be ruled out because it would not be politically feasible. In order to allow for a more appropriate economic evaluation of the policy mix of emissions trading and taxation, it is therefore necessary to approach the analysis to reality.

To provide a more applied analysis of a policy mix of emissions trading, this chapter promotes a change of perspective. When existing studies analyze the efficiency of the policy mix in mitigating climate change, they assume that policy-makers are free to choose any policy option available. In contrast, it is assumed in the remainder of this chapter that policy-makers are restricted in their, choice of climate policies. In particular, it is assumed that a tax has to be implemented – or cannot be abolished – because it is needed to address other objectives and criteria of policy making. Thus, the climate policy strategy has to be optimized under the constraint of a tax being implemented. With this change in perspective, three important research questions emerge, which have been neglected so far in existing studies. Firstly, it has to be clarified which factors determine the extent of inefficiency of the policy mix in reducing GHG emissions. This insight is necessary to compare welfare losses from combining emissions trading and taxation with respect to mitigating climate change with gains from addressing other objectives and criteria. It has to be analyzed which variables in the models actually increase or reduce welfare losses. Existing studies have not been very specific on this issue. Light has to be shed also on the efficiency implications of output taxation instead of emissions taxation. Only the latter design option has been considered by available policy mix analyses. However, the consideration of an output tax is particularly important since it is in place for electricity, for example, in many countries such as Germany (see Section 2.3.3). Secondly, the question arises whether the implementation of an emissions trading scheme on top of a tax is economically desirable at all when the latter cannot be abolished. Therefore, the performance of the policy mix has to be compared to that of single (possibly inefficient) tax. Existing studies have only been occupied with comparing the policy mix with the unrealistic ideal of an efficient single emissions trading scheme. However, this may be a rather theoretical point of reference. Thirdly, it has to be analyzed whether modifications in the design of the emission trading scheme or the tax may help to reduce the inefficiency of the policy mix. In particular, it seems to be worth investigating whether an emissions trading can be modified in order to compensate for the inefficiency of the tax. This issue has neither been addressed in policy mix studies so far, which have straightly opted for a single-policy strategy. By answering these three research questions, this chapter aims at advancing the economic analysis of a policy mix substantially. It is expected to deliver an evaluation that is far more down to earth and can be used to derive reasonable policy recommendations.

In order to answer these open questions – and to illustrate those findings already mentioned above – two partial equilibrium models are developed in this chapter. Partial equilibrium models have the appeal that they allow evaluating cause-and-effect relationships analytically. This approach is appropriate to understand the interactions existing in a policy mix in detail. It has to be born in mind, however, that this analysis does not provide an overall economic evaluation of the policy mix. Such analysis would require the application of general equilibrium models. The first model assumes that emissions are independent of output. Such model has been used by de Muizon and Glachant (2004) and Sterner and Höglund Isaksson (2006). De Muizon and Glachant employ the model to analyze interactions between relative and absolute emission targets while Sterner and Höglund Isaksson examine the combination of an emissions tax with a refunding scheme. Their assumptions about policy design are modified in this chapter in order to represent an emissions trading scheme and a tax. This model helps to answer most of the questions raised above with respect to the combination of an emissions trading scheme and an emissions tax. This analysis is carried out in the subsequent Section 5.3. The second model considers the case of emissions depending on output. This assumption makes modeling more complex but also more realistic. A similar model can be found in Fischer (2003) and Gersbach and Requate (2004). Fischer (2003) uses the model to study interactions between rate-based and cap-and-trade emissions policies. Gersbach and Requate (2004) analyze a policy mix encompassing an emissions tax and a refunding scheme. Their assumptions about policy mix design are again adapted in order to correspond to the characteristics of an emissions trading scheme and a tax. The model is used in this chapter to reveal some additional insight about the efficiency properties of the policy mix of an emissions trading scheme and an emissions tax. More importantly, the model allows modeling the welfare effects of an emissions trading scheme combined with an output tax instead of an emissions tax. This analysis can be found in Section 5.4.

5.3 Model with Emissions Not Depending on Output

This section analyzes the interaction of an emissions trading scheme with an emissions tax. The analysis is based on a relatively simple model assuming emissions to be independent of output. The model assumptions and the conditions for the social optimum are introduced in Sections 5.3.1 and 5.3.2. Subsequently, the well-known efficiency properties of a single emissions trading scheme and a single emissions tax are outlined in Section 5.3.3. They set the benchmark with which the policy mix is compared in the following. Section 5.3.4 reproduces analytically the findings which have been made by available studies for the combination of emissions trading and homogeneous emissions taxation (policy mix A). In addition to available studies, it provides a formal analysis of the problem as well as a graphical illustration. Section 5.3.5 finally analyzes the combination of an emissions trading scheme with a heterogeneous tax (policy mix B) and develops the major advances in understanding the policy mix economically.

5.3.1 Model Assumptions

The performance of a policy mix of emissions trading and an emissions tax is analyzed in a partial equilibrium model. The model supposes an industry with n identical firms and two sectors i. Sector one encompasses αn firms and sector two $(1-\alpha)n$ firms (with $0 \leq \alpha \leq 1$).

The representative firm in each sector produces an output q_i of a good which it can sell on the sector's output market at price p_i. Firms are assumed to be price takers in the output market. Output markets of both sectors are assumed to be independent from each other. This implies that goods are neither complements nor substitutes. Moreover, neither good is used as an input for the production of the other good. To produce its output, firms in both sectors incur production costs $c_i(q_i)$. Production costs are positive, increasing in output and convex: $c_i > 0$, $c_i' > 0$ and $c_i'' > 0$.

Along with the production of output, firms in both sectors generate emissions of a pollutant e_i. Total emissions E of the entire industry are the sum of each sector's emissions E_i. Sectoral emissions compute as the product of representative firm's emissions times the number of firms in the sector. Thus, total emissions can be written as:

$$E = E_1 + E_2 = \alpha n e_1 + (1-\alpha) n e_2 \tag{5.1}$$

Emissions produce a pollution externality, which is measured as environmental damage D. Damage is assumed to be independent of the location and timing of an emission. This implies that the pollutant is uniformly mixing in the atmosphere. Moreover, damage does not depend on past emissions, i.e. the pollutant does not accumulate. Given these assumptions, damage is a simple function of the total emissions of both sectors: $D = D(E)$. For the sake of simplicity, emissions of each firm are assumed to be independent of the firm's output, i.e. pollution and production are fully separable activities. Firms can reduce their emissions rate by abatement with cleaner technologies or cleaner inputs. To reduce their emissions, firms have to incur abatement costs $g_i(e_i)$. In order to be able to solve the model analytically, the abatement cost function is assumed to have the following quadratic form:

$$g_i(e_i)=\frac{a}{2}e_i^2-be_i+\frac{b^2}{2a}\text{ with } e_i\in\left[0;\frac{b}{a}\right] \tag{5.2}$$

Parameters a and b are assumed to be positive. The upper limit of the interval for emissions, b/a, represents the unique global minimum as well as the unique zero of the quadratic function. It can be interpreted as the "natural" emissions level which is observed when firms undertake no abatement at all. Within the interval given above, abatement costs are positive, decreasing in emissions and convex. The marginal abatement cost function for firms in both sectors, $g_i'(e_i)$ can be derived from equation (5.2). It is linear and writes as:

$$g_i'(e_i)=ae_i-b\text{ with } e_i\in\left[0;\frac{b}{a}\right] \tag{5.3}$$

Rearranging equation (5.3) gives the emissions level chosen by firms in both sectors:

$$e_i=\frac{g_i'(e_i)+b}{a} \tag{5.4}$$

Using equation (5.4), total emissions of the entire industry given in (5.1) can be rewritten as:

$$E=\alpha n\frac{g_1'(e_1)+b}{a}+(1-\alpha)n\frac{g_2'(e_2)+b}{a} \tag{5.5}$$

The regulator intends to maximize social welfare W. Welfare is a function of profits from production in both sectors – i.e. each sector's revenues net of production and abatement costs – minus the environmental damage due to emissions generated with production in both sectors. Consequently, the regulator's maximization problem writes as:

$$\max_{q,e} W=\alpha n[p_1q_1-c_1(q_1)-g_1(e_1)]+(1-\alpha)n[p_2q_2-c_2(q_2)-g_2(e_2)]-D(E) \tag{5.6}$$

5.3.2 The Social Optimum

Differentiating equation (5.6) with respect to output q_i and emissions e_i yields the necessary first-order conditions for firms in both sectors to produce optimal welfare:

$$p_i=c_i'(q_i) \tag{5.7}$$

$$-g_i'(e_i)=D'(E) \tag{5.8}$$

The first condition implies that firms should extend their production until their marginal production costs equal the output price. The second condition implies that the inverse marginal abatement costs, $-g_i'(e_i)$, should equal the marginal damage of emissions. That is, firms should employ measures to abate emissions until the cost of the last unit of emission reduced equals the marginal damage produced by this unit of emission.

5.3.3 Single Policies

This section briefly analyzes an emissions trading scheme and an emissions tax implemented in isolation. For each policy, the well-known efficiency properties are outlined. Understanding these properties is important. They provide the baseline against which the performance of the policy mix will be assessed subsequently.

5.3.3.1 Emissions Trading

An emissions trading scheme sets a cap $\overline{E}$ on total emissions of the entire industry. The regulator is assumed to issue a corresponding amount of allowances and to allocate a fixed number of $\overline{a}_i$ allowances free of charge to firms in both sectors.[58] Each firm is obliged to hold sufficient allowances to cover its emissions. Subject to this constraint, each firm is free to choose whatever combination of production and abatement. Moreover, firms may buy extra allowances from or sell excess allowances to other firms. The emissions trading scheme thus establishes a common emission allowance market for both sectors. In this market, allowances are traded at the equilibrium allowance price t. These assumptions correspond to the design of the EU ETS outlined in Section 2.3.1. Firms are assumed to take this price as given. Therefore, the emissions trading scheme results in firms to incur net allowance costs amounting to $t(e_i - \overline{a}_i)$. In fact, if actual emissions are below the allowance allocation, allowance costs will be negative. In this case, firms will realize additional revenues as they are able to sell excess allowances. Since firms in both sectors are identical, each sector's share in total number of firms, α and $(1-\alpha)$, can also be interpreted as the respective shares of sector one and two in the emissions allowance market.

Firms in both sectors maximize their individual profit π_i. Profit computes as revenues less production and abatement costs and net allowance costs:

$$\max_{q,e} \pi_i = p_i q_i - c_i(q_i) - g_i(e_i) - t(e_i - \overline{a}_i) \tag{5.9}$$

[58] With auctioned allowances, firms do not receive allowances free of charge but have to buy all allowances, i.e. $\overline{a}_i = 0$. This does not change the first-order conditions though because $\overline{a}_i$ is a constant that cancels out with derivation.

The corresponding first-order conditions are:

$$p_i^* = c_i'(q_i^*) \tag{5.10}$$

$$-g_i'(e_i^*) = t^* \tag{5.11}$$

where an asterisk indicates equilibrium values.

In the competitive equilibrium, the output price in each sector will equal the marginal cost of producing an additional unit of output. Moreover, firms will reduce their emissions by abatement until the marginal abatement costs equal the allowance price. If condition (5.11) holds for all firms in both sectors and the cap is binding, i.e. $E_1^* + E_2^* = \overline{E}$, equation (5.5) can be rewritten as:

$$\overline{E} = n\frac{b-t^*}{a} \tag{5.12}$$

Reorganizing equation (5.12) yields the equilibrium allowance price:

$$t^* = b - \frac{a}{n}\overline{E} \tag{5.13}$$

Corresponding to economic intuition, the allowance price increases with a decreasing emissions cap $\overline{E}$, i.e. with a decreasing supply of allowances. The allowance price also rises with an increasing number n of firms demanding allowances. What is more, the equilibrium price will be the higher, the flatter the marginal abatement cost curve is, i.e. the smaller is a. Since marginal abatement costs are assumed to be identical for firms in both sectors, the allowances price is independent of the share of either sector in the emissions market.

Inserting equations (5.11) and (5.13) into equation (5.4) gives the individual emissions levels chosen by firms in both sectors:

$$e_i^* = \frac{\overline{E}}{n} \tag{5.14}$$

Thus, the emissions cap is equally distributed among all firms in the industry.

Condition (5.11) implies that all firms will face the same marginal abatement costs. Thus, marginal abatement costs are equalized across the industry. The basic requirement for cost-effectiveness is therefore met, i.e. the emissions cap is attained at least cost. In order for the emissions trading scheme to be efficient, the regulator has to choose the emissions cap $\overline{E}$ such that condition (5.8) is met. Comparing equations (5.8) and (5.11) reveals that this will be the case when the resulting allowance price equals the marginal damage from emissions: $t^* = D'(\overline{E})$. Substituting t^* in equation (5.12) correspondingly gives the optimal emissions cap: $\overline{E} = n[(b - D'(\overline{E}))/a]$. If the emissions cap is chosen optimally, the emissions trading scheme is not only cost-effective then but efficient as well. Thus, it correctly internalizes the pollution externality generated from emissions. In the presence of a pollution externality only and given that the emissions trading scheme can be implemented without transaction costs, i.e. if none of the rationales derived in Chapter 3 applies, no policy mix is needed.

5.3.3.2 Emissions Tax

The regulator may also choose to impose a tax on firms in each sector at rate τ_i for each unit of emission generated by them. The tax rate may be homogeneous, i.e. $\tau_1 = \tau_2$, or heterogeneous, i.e. $\tau_1 \neq \tau_2$. The maximization problem of firms in both sectors then turns out to be:

$$\max_{q,e} \pi_i = p_i q_i - c_i(q_i) - g_i(e_i) - \tau_i e_i \tag{5.15}$$

This yields the following first-order conditions:

$$p_i^{\tau_i} = c_i'(q_i^{\tau_i}) \tag{5.16}$$

$$-g_i'(e_i^{\tau_i}) = \tau_i \tag{5.17}$$

where superscript τ_i indicates equilibrium values under the emissions tax. As under emissions trading, firms in both sectors will determine their output levels such that the equilibrium output price reflects marginal production costs. Moreover, firms reduce their emissions until marginal abatement costs equal the sector's tax rate.

If the tax rate is homogeneous across both sectors, i.e. $\tau_1 = \tau_2$, the emissions tax provides for an emissions level to be attained cost-effectively. In this case, all firms in both sectors have the same incentive to abate, and marginal abatement costs are equalized across the industry. In order for the tax to be efficient, condition (5.8) of the social optimum has to be met as well. Comparing equations (5.8) and (5.17) reveals that the efficient tax rate in both sectors has to equal marginal damages of emissions: $\tau_1 = \tau_2 = D'(E)$. This corresponds to the concept of a corrective tax on emissions as it was proposed by Pigou (1932). Such a tax provides for an emissions level which equals the emissions cap under an efficient emissions trading scheme, and sets out a tax rate equal to the equilibrium allowance price – as they were derived in the previous section. Thus, as a single emissions trading scheme, a single homogeneous emissions tax may efficiently correct a pollution externality. Again it may be emphasized that no policy mix is needed – given that not further failures of private governance structures exist and that the tax can be implemented at zero transaction costs (otherwise, see the rationales for using a policy mix developed in Chapter 3).

However, for the subsequent discussion of a policy mix with an emissions trading scheme and a heterogeneous emissions tax, the efficiency properties of a single heterogeneous emissions tax are of particular interest as well. This case is usually not considered in the literature. The formal analysis as well as graphical illustration of the problem are developed in the following. If the tax rate is heterogeneous across sectors, i.e. $\tau_1 \neq \tau_2$, the emissions tax cannot provide a cost-effective and efficient allocation of abatement measures since abatement costs are not equalized across sectors. The overall emissions level of the industry in the presence of a heterogeneous emissions tax can be derived by inserting equation (5.17) into equation (5.5):

$$E^{\tau} = n\frac{b - \alpha\tau_1 - (1-\alpha)\tau_2}{a} \tag{5.18}$$

The same emissions level could be attained cost-effectively by a tax rate τ^* that is homogeneous across sectors. Substituting τ_1 and τ_2 by τ^* and reorganizing equation (5.18) yields:

$$\tau^* = b - \frac{a}{n}E^{\tau} \tag{5.19}$$

Combining equations (5.18) and (5.19) gives the optimal homogeneous tax rate depending on the heterogeneous sectoral tax rates:

$$\tau^* = \alpha\tau_1 + (1-\alpha)\tau_2 \tag{5.20}$$

Thus, the optimal tax rate computes as the simple mean average of the heterogeneous tax rates weighted with the shares of the respective sectors in the total number of firms. If sector 2 is exempt from taxation, i.e. $\tau_2 = 0$, the optimal tax rate computes as the product of the tax rate for sector 1 multiplied by the share of sector 1 in the total number of firms: $\tau^* = \alpha\tau_1$. If, in addition, both sectors are assumed to have the same size, i.e. $\alpha = 0.5$, the optimal tax rate is half the tax rate in sector one: $\tau^* = \tau_1/2$.

If taxation is heterogeneous, firms in the sector with the higher tax rate have a too strong incentive to abate and incur too high abatement costs compared to homogeneous taxation. In contrast, firms in the sector with the lower tax rate abate too little and are facing too low abatement costs. Analytically, the resulting welfare loss with heterogeneous taxation can be computed as the absolute value of the sum of the definite integrals of the marginal abatement cost functions for both sectors on the interval $\left[e_i^{\tau_i}, e_i^{\tau^*}\right]$ multiplied by the number of firms in each sector:

$$\Delta W = \left| \alpha n \int_{e_1^{\tau_1}}^{e_1^{\tau^*}} (g_1'(e_1)) de_1 + (1-\alpha) n \int_{e_2^{\tau_2}}^{e_2^{\tau^*}} (g_2'(e_2)) de_2 \right| \tag{5.21}$$

The lower limit of integration, $e_i^{\tau_i}$, corresponds to the emissions level chosen in each sector given heterogeneous taxation. The upper limit of integration, $e_i^{\tau^*}$, represents the emissions level chosen in the presence of the optimal homogeneous tax providing the same overall emissions level. Equation (5.21) can be rewritten using the antiderivatives of the marginal abatement cost functions for both sectors, which correspond to abatement cost function given in equation (5.2):

$$\Delta W = \left| \frac{\alpha n a}{2} \left[\left(e_1^{\tau^*}\right)^2 - \left(e_1^{\tau_1}\right)^2 \right] + \frac{(1-\alpha) n a}{2} \left[\left(e_2^{\tau^*}\right)^2 - \left(e_2^{\tau_1}\right)^2 \right] \right.$$
$$\left. - b\left[\alpha n e_1^{\tau^*} + (1-\alpha) n e_2^{\tau^*} - \left(\alpha n e_1^{\tau_1} + (1-\alpha) n e_2^{\tau_2} \right) \right] \right| \tag{5.22}$$

The last term on the right hand side of equation (5.22), $b[*]$, cancels out since overall emissions under taxation are assumed to be the same under heterogeneous and homogeneous taxation: $E^{\tau} = \alpha n e_1^{\tau^*} + (1-\alpha) n e_2^{\tau^*} = \alpha n e_1^{\tau_1} + (1-\alpha) n e_2^{\tau_2}$. The lower and upper limits of integration, $e_i^{\tau_i}$ and $e_i^{\tau^*}$, can be determined using equation (5.4). Assuming that firms optimize such that marginal abatement costs equal the tax rate in either case (see equation (5.17)), the lower and upper limits of integration write as:

$$e_i^{\tau_i} = \frac{b - \tau_i}{a} \tag{5.23}$$

$$e_i^{\tau^*} = \frac{b - \tau^*}{a} \tag{5.24}$$

Inserting equations (5.23) and (5.24) into equation (5.22) and substituting the optimal emissions tax rate τ^* by equation (5.19) yields the welfare loss under heterogeneous taxation compared to homogeneous taxation:

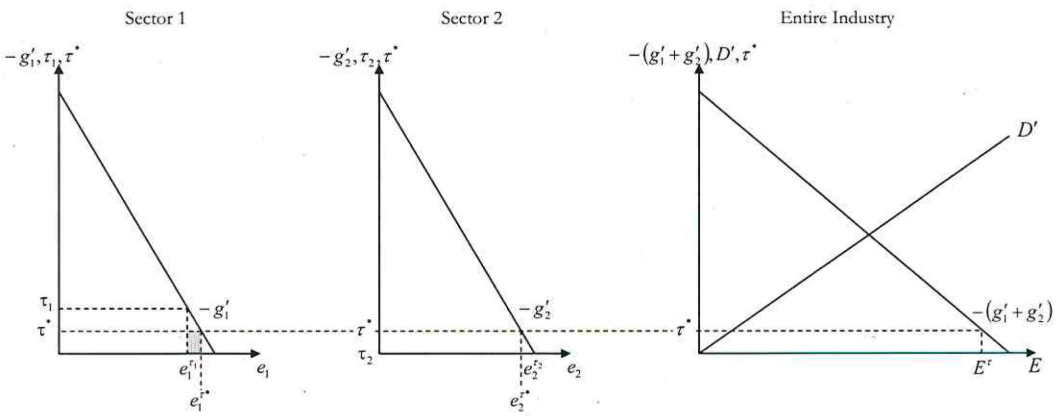
$$\Delta W = \frac{\alpha(1-\alpha) n (\tau_1 - \tau_2)^2}{2a} \tag{5.25}$$

This analytical result will be interpreted extensively in Section 5.3.5.

Figure 5-1 illustrates the results derived for heterogeneous taxation graphically. It refers to the case with both sectors composed of one firm each. Thus, it represents a situation in which both sectors have the same share in the total number of firms, i.e. $\alpha = 0.5$. Figure 5-1 depicts the inverse marginal abatement costs function for sector 1 and 2, $-g_i'(e_i)$, on the left hand side as well as the aggregated inverse marginal abatement cost curve for the entire industry, $-[g_1'(e_1)+g_2'(e_2)]$, on the right hand side. It is assumed that sector 1 faces a positive tax rate while the tax rate is zero for sector 2. Facing the tax rate τ_1, firms in sector 1 will reduce emissions until inverse marginal abatement costs equal the tax rate. This corresponds to emissions level $e_1^{\tau_1}$ in Figure 5-1. In contrast, sector-2 firms do not have any incentive to reduce emissions and will produce the natural level of emissions $e_2^{\tau_2}$, where inverse marginal abatement costs are zero. However, the overall level of emissions $E^{\tau} = e_1^{\tau_1} + e_2^{\tau_2}$ could be attained at least cost by imposing a homogeneous tax rate τ^* on both sectors. Sector 1 would then raise its emissions to $e_1^{\tau^*}$ and save abatement costs. Sector 2 would bring down its emissions level to $e_2^{\tau^*}$ and incur additional abatement costs. The difference between the cost decline in sector 1 and the cost increase in sector 2 corresponds to the welfare gain from homogeneous taxation – or the welfare loss due to heterogeneous taxation. This loss is depicted in Figure 5-1 as the light-shaded rectangle.

Figure 5-1: Heterogeneous emissions tax and optimal homogeneous emissions tax

Source: Own Figure



5.3.4 Policy Mix A: Emissions Trading and Homogeneous Taxation of Emissions

5.3.4.1 Firms' Choices

To start the analysis, light is shed on the case of an emissions trading scheme combined with an emissions tax that is homogeneous across sectors. Suppose all firms in the industry participate in an emissions trading scheme. In addition, all firms of both sectors are required to pay the same tax rate per unit of emission, i.e. $\tau_1 = \tau_2 = \tau$. The profit maximization problem for the representative firms in each in sector is as follows:

$$\max_{q,e} \pi_i = p_i q_i - c_i(q_i) - g_i(e_i) - t(e_i - \bar{a}_i) - \tau e_i \tag{5.26}$$

The first-order conditions yielding optimal profits for firms then compute as:

$$p_i^A = c_i'(q_i^A) \tag{5.27}$$

$$-g_i'(e_i^A) = t^A + \tau \tag{5.28}$$

where superscript A denotes equilibrium values for the policy mix of emissions trading and a homogeneous emissions tax.

The firms' optimizing behavior regarding output remains unchanged compared to a situation with one policy only. However, when deciding about the optimal emissions level, firms now consider the incentives set out by both policies. They will reduce their emissions until the marginal abatement cost falls below the sum of the allowance price and the tax rate.

5.3.4.2 The Policy Mix Compared with Emissions Trading

To start the analysis, it is assumed that the emissions cap is binding under the policy mix as well. Substituting $-g_i'(e_i)$ using equation (5.28) and setting total emissions equal to the emissions cap, equation (5.5) can be rewritten as:

$$\bar{E} = n\frac{b - t^A - \tau}{a} \tag{5.29}$$

Reorganizing equation (5.29) yields the equilibrium allowance price under the policy mix of emissions trading and a homogeneous emissions tax:

$$t^A = b - \frac{a}{n}\bar{E} - \tau \tag{5.30}$$

Comparing equation (5.30) with equation (5.13) reveals that the new equilibrium allowance price under the policy mix is just the equilibrium allowance price under a single emissions trading scheme reduced by the tax rate. In other words, the sum of the tax rate and the equilibrium allowance price emerging under the policy mix equals the equilibrium allowance price arising if only an emissions trading scheme is in place:

$$t^{*} = t^{A} + \tau \qquad . \tag{5.31}$$

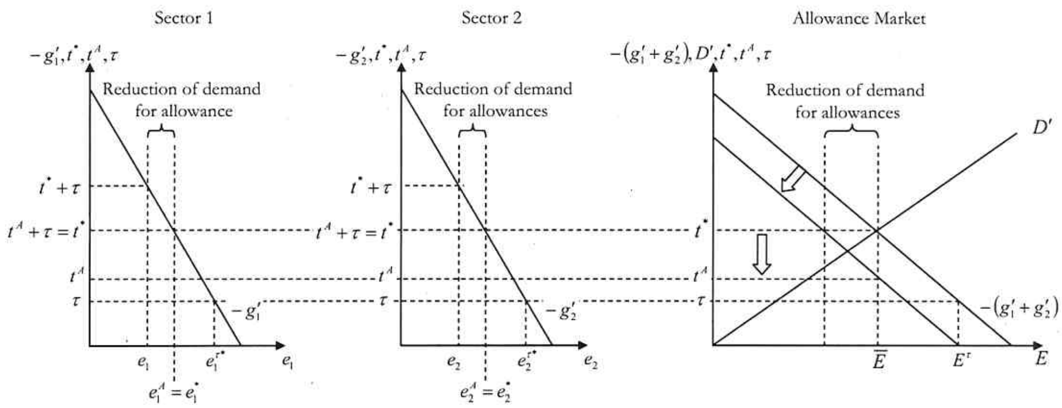
Thus, total abatement incentives eventually do not change with the implementation of a homogeneous emissions tax in addition to emissions trading. Firms choose their emissions levels as under a single emissions trading scheme. The emissions cap is attained cost-effectively since firms in both sectors face the same incentives to abate, and marginal abatement costs are equalized across the entire industry. Moreover, the policy mix will be efficient if the emissions cap is selected optimally. Thus, the efficiency properties of emissions trading – as outlined in Section 5.3.3.1 – prevail under the policy mix of the trading scheme with a homogeneous emissions tax.

Figure 5-2 displays the analytical results graphically. As in Figure 5-1, it is assumed that each sector is composed of one firm. Each firm's willingness to pay for the right to produce an additional unit of emission is limited to the marginal costs of abating this unit. If the firm had to pay more it would rather abate. The inverse marginal abatement cost curves for sectors 1 and 2 may therefore be interpreted as sector-specific demand functions for allowances. The aggregated curve of inverse marginal abatement costs of the industry then represents the demand function for the entire allowance market. If only an emissions trading scheme is in place, t^{*} is the resulting equilibrium allowance price. If an emissions tax is implemented on top of the emissions trading scheme, firms in both sectors face the additive incentives of the allowance price t^{*} and the tax rate τ. They are initially stimulated to reduce the sectors' emissions from e_i^{*}, the optimal choice in the presence of emissions trading only, to e_i. However, e_i is not the equilibrium emissions value. Due to the decreasing demand for allowances, the demand curve for the entire allowance market shifts to the left. The new equilibrium allowance price for the emissions cap $\overline{E}$ is t^{A}. For each sector, the sum of this allowance price t^{A} and the tax rate τ now equals the preexisting allowance price t^{*}. Put another way, if a firm is obliged to pay a tax for an additional unit of emission anyway and has to hold an allowance in addition, its maximum willingness to pay for an allowance is exactly reduced to the difference between the marginal abatement costs and the tax rate. Since all firms in the allowance market pursue this strategy, the allowance price declines by the tax rate.

Indeed, the preceding analysis is based on the assumption that the emissions trading scheme is the more stringent policy. This assumption implies that the emissions trading scheme induces more emissions reduction than the emissions tax. That is, the cap set out by the emissions trading scheme $\overline{E}$ is lower than the emissions level E^{τ} which would result in the presence of the emissions tax only. Consequently, the resulting equilibrium allowance price has to be higher than the tax rate: $t^{*} > \tau$ (this situation is depicted in Figure 5-2). Since the allowance price cannot be negative, i.e. $t^{A} \geq 0$, a dominant emissions trading scheme is the prerequisite for equation (5.31) to hold. In the extreme, if a tax rate is implemented that equals the preexisting allowance price, no trading will occur and the allowance price will be reduced to zero. This will be the case, for example, when both policies are designed efficiently – and the allowance price under isolated emissions trading equals the Pigovian tax rate.

Figure 5-2: Combining emissions trading and a homogeneous emissions tax

Source: Own figure



Things change, however, if the emissions tax is the more stringent policy. In this case, the emissions level induced by the tax is below the cap of the emissions trading scheme. The corresponding tax rate is higher than the allowance price preexisting under emissions trading only: $t^{*} < \tau$. Since the allowance price cannot be negative, equation (5.31) does not hold. The equilibrium allowance price t^{A} will be zero and $t < t^{A} + \tau$. Consequently, the emissions trading scheme becomes ineffective. The emissions tax is the dominant policy and its efficiency properties prevail. Since $t^A = 0$, firms make their abatement and production choices as if they face the emissions tax only. Since $t^* < \tau$, the incentive to abate will be higher under the policy mix than with emissions trading only. Consequently, the emissions level under the policy mix will also be lower than with emissions trading only: $\alpha ne_1^A + (1-\alpha)ne_2^A < \alpha ne_1^* + (1-\alpha)ne_2^* = \overline{E}$. As has been highlighted for the case of a single homogeneous emissions tax in Section 5.3.3.2, this policy mix is cost-effective since it equalizes marginal abatement costs across firms. It will be efficient if the tax has been chosen to equal marginal damages. Implementing an emissions tax on top of a preexisting emissions trading scheme may then increase efficiency if the emissions cap under the trading scheme has not been set tight enough from a social point of view.

5.3.4.3 The Policy Mix Compared with an Emissions Tax

It has been outlined in Section 5.2, the policy mix should not only be compared with a single first-best emissions trading scheme – as it has only been done in existing economic analyses so far. If the tax cannot be abolished, it is also worthwhile to study how the policy mix compares with a single tax. Thereby, it can be clarified whether implementing an emissions trading scheme on top of an emissions tax is economically desirable. The comparison of the policy mix with a single emissions tax yields results that are analogous to the comparison of the policy mix with a single emissions trading scheme in the previous section. It is first supposed that the emissions tax is the more stringent policy, i.e. the tax rate is higher than the equilibrium allowance price of the emissions trading scheme implemented in isolation: $\tau > t^*$. Consequently, the emissions level induced by the tax will be lower than the emissions cap under emissions trading: $E^\tau < \overline{E}$. As has been shown above, implementing an emissions trading scheme on top of an emissions tax is ineffective in this case. The allowance price will be zero. Firms in both sectors only face the tax rate as an incentive to reduce their emissions. The overall emissions level is not affected by the implementation of an emissions trading scheme in addition to the emissions tax, i.e. $E_1^A + E_2^A = E^\tau$. The overall efficiency properties of the single tax will prevail. The policy mix will be cost-effective. Moreover, it will be efficient if the tax rate is chosen to equal marginal damages from emissions.

Things change if an emissions trading scheme is implemented which is more stringent than the emissions tax, i.e. $\tau < t^*$ and $E^\tau > \overline{E}$. It has been shown above that in this case firms will face the additive incentives of a positive allowance price and the emissions tax. These incentives are higher than those under the emissions tax only: $\tau < t^* = t^A + \tau$. Consequently, the overall emissions level will be lower under the policy mix than under the emissions tax only: $E_1^A + E_2^A < E^\tau$. The efficiency properties of the emissions trading scheme prevail. The policy mix is cost-effective. Moreover, it will be efficient if the emissions cap is chosen optimally. Given a pre-existing tax rate which is below the Pigovian tax rate, the implementation of the emissions trading scheme on top of the tax may increase efficiency compared to the case of the tax alone.

Thus, combining emissions trading with a homogeneous emissions tax may provide for an efficient reduction of emissions in a partial equilibrium model. The efficiency of the policy mix crucially depends on the efficiency of the more stringent policy in the mix. A policy mix may increase efficiency in abatement if an economically too lax emissions trading scheme or emissions tax is supplemented by a more stringent version of the respective other policy. However, the same outcome can be attained by either emissions trading or an emissions tax alone – depending on which is the more stringent policy. The policy mix creates a redundancy but not necessarily an efficiency loss. For the further analysis, it is assumed that the emissions trading scheme is the more stringent policy. Thus, the emissions cap holds but some of the emission reductions stimulated by the emissions trading scheme are substituted by emission reductions under the tax.

5.3.5 Policy Mix B: Emissions Trading and Heterogeneous Taxation of Emissions

5.3.5.1 Firms' Choices

As in the previous section, a common emissions trading scheme is supposed to exist for the entire industry. In contrast to the previous section, it is now assumed that an emissions tax, which is implemented in addition, is heterogeneous across sectors. Sector 1 faces the tax rate τ_1. Sector 2 is subject to the tax rate τ_2. Consequently, the maximization problems of firms in sector 1 and 2 differ with respect to the tax burden incurred:

$$\max_{q,e} \pi_i = p_i q_i - c_i(q_i) - g(e_i) - t(e_i - \overline{a}_i) - \tau_i e_i \tag{5.32}$$

The first-order conditions for firms in both sectors are as follows:

$$p_i^B = c_i'(q_i^B) \tag{5.33}$$

$$-g_i'(e_i^B) = t^B + \tau_i \tag{5.34}$$

where superscript B denotes equilibrium values for the policy mix of emissions trading and a heterogeneous emissions tax. As under homogeneous taxation, output decisions are not affected by taxation but only by the market price for output. In contrast to homogeneous taxation, the incentives to abate emissions now vary between sectors due to different tax rates.

5.3.5.2 The Policy Mix Compared with Emissions Trading

This section is devoted to the identification of drivers of the inefficiency in the policy mix, a challenge that has not been addressed properly in existing policy mix studies so far. For this purpose, the policy mix is compared with a single optimal emissions trading scheme. A formal expression is derived for the difference in welfare between both policy scenarios. This expression allows identifying the most important variables determining the extent of welfare losses under a policy mix.

Substituting inverse marginal abatement costs of each sector by using equation (5.34) and setting total emissions equal to the emissions cap, equation (5.5) can be rewritten as:

$$\overline{E} = \alpha n \frac{b - t^B - \tau_1}{a} + (1-\alpha) n \frac{b - t^B - \tau_2}{a} \tag{5.35}$$

Reorganizing equation (5.35) gives the equilibrium allowance price under an emissions trading scheme combined with a heterogeneous emissions tax:

$$t^B = b - \frac{a}{n}\overline{E} - \alpha\tau_1 - (1-\alpha)\tau_2 \tag{5.36}$$

Comparing equation (5.36) with equation (5.13) reveals that the new equilibrium allowance price computes as the equilibrium allowance price under the single emissions trading scheme reduced by the average tax rate. The average tax rate is the sum of the sectors' heterogeneous tax rates weighted with sectors' shares in the allowance market. Thus, it can be concluded that:

$$t^B = t^* - \alpha\tau_1 - (1-\alpha)\tau_2 \tag{5.37}$$

If the tax rates are identical, i.e. $\tau_1 = \tau_2 = \tau$, the equilibrium allowance price under the policy mix is just the allowance price under the single emissions trading scheme reduced by the tax rate: $t^B = t^* - \tau$. This is identical to finding made above in Section 5.3.4.2. Thus, homogeneous taxation can be considered a special case of heterogeneous taxation. If firms in sector 2 are exempt from taxation, i.e. $\tau_2 = 0$, the equilibrium allowance price equals the difference of the allowance price under emissions trading only and the tax rate for sector 1 multiplied by its emissions market share: $t^B = t^* - \alpha\tau_1$. If, in addition, sectors 1 and 2 are assumed to have the same share in the emissions market, i.e. $\alpha = 0.5$, the equilibrium allowance price under the policy mix is by half the tax rate below the allowance price under emissions trading only: $t^B = t^* - \tau_1/2$.

Inserting equations (5.34) and (5.36) into (5.4) gives the individual emissions levels chosen by firms in both sectors in equilibrium:

$$e_1^B = \frac{\overline{E}}{n} + \frac{(1-\alpha)(\tau_2 - \tau_1)}{a} \tag{5.38}$$

$$e_2^B = \frac{\overline{E}}{n} + \frac{\alpha(\tau_1 - \tau_2)}{a} \tag{5.39}$$

The first part of the right hand side of equations (5.38) and (5.39) corresponds to the emissions level chose by firms under a single emissions trading scheme, as given in equation (5.14). The second part represents the distortion resulting from heterogeneous taxation of emissions. This term will be negative if the tax rate for the sector under consideration is higher than the tax rate for the other sector. Otherwise, the term will be positive. This finding implies that firms in the sector with higher taxation will reduce their emissions under the policy compared to a single emissions trading scheme. In contrast, firms subject to the lower tax rate will choose a higher emissions level. However, the overall level of emissions is determined by the emissions cap and remains unchanged. Thus, the implementation of heterogeneous emissions tax on top of an emissions trading scheme only results in a shift of emissions between sectors.

The shift of emissions comes at increasing overall costs of abatement. Condition (5.34) reveals that marginal abatement costs vary between sectors when tax rates are heterogeneous. Thus, with the implementation of a heterogeneous tax in addition to the emissions trading scheme, the emissions cap cannot be attained cost-effectively. Firms in the sector with the higher tax rate abate relatively costly while firms in the other sector abstain from relatively cheap options for abatement. Compared to a situation with emissions trading only, the higher-taxed sector incurs higher overall abatement costs while the lower-taxed sector faces lower abatement costs. The difference between the cost increase in one sector and the cost decrease in the other sector is the efficiency loss due to the policy mix. Analytically, the efficiency loss ΔW due to the policy mix can be understood as the absolute value of the sum of the definite integrals of the inverse marginal abatement costs functions for firms in both sectors on the interval $[e_i^B, e_i^*]$, multiplied by number of firms in each sector:

$$\Delta W = \left| \alpha n \int_{e_i^B}^{e_1^*} (g_1'(e_1)) de_1 + (1-\alpha) n \int_{e_2^B}^{e_2^*} (g_2'(e_2)) de_2 \right| \tag{5.40}$$

Rewriting equation (5.40) using the antiderivatives of the marginal abatement cost functions – i.e. the abatement cost functions provided in equation (5.2) – gives:

$$\Delta W = \left| \frac{\alpha n a}{2}\left[(e_1^*)^2 - (e_1^B)^2\right] + \frac{(1-\alpha)na}{2}\left[(e_2^*)^2 - (e_2^B)^2\right] \right.$$

$$\left. - b\left[\alpha n e_1^* + (1-\alpha)ne_2^* - \left(\alpha n e_1^B + (1-\alpha)ne_2^B\right)\right] \right| \tag{5.41}$$

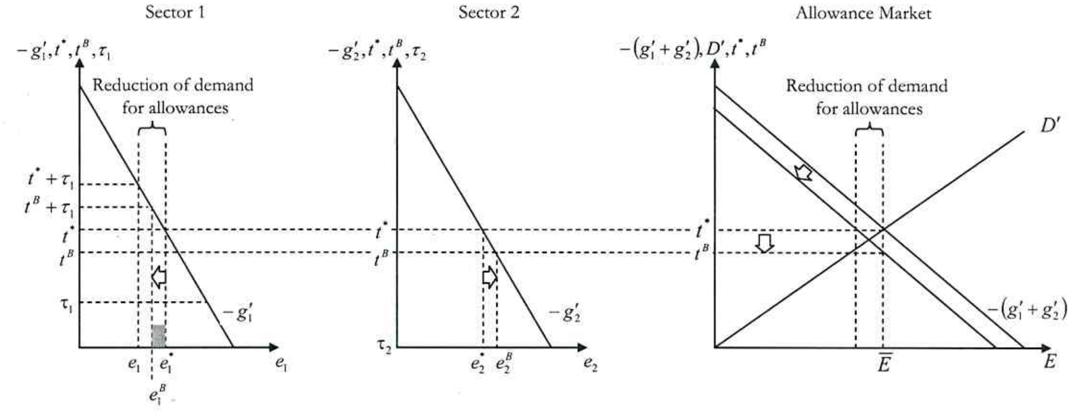
The last term on the right hand side of equation (5.41), $b[*]$, cancels out since overall emissions under the single emissions trading scheme equal those under the policy mix, i.e. $\overline{E} = \alpha n e_1^* + (1-\alpha)ne_2^* = \alpha n e_1^B + (1-\alpha)ne_2^B$. Inserting equations (5.14), (5.38) and (5.39) into equation (5.41) yields the welfare loss under a policy mix of emissions trading and a heterogeneous emissions tax compared to the case of a single emissions trading scheme:

$$\Delta W = \frac{\alpha(1-\alpha)n(\tau_1 - \tau_2)^2}{2a} \tag{5.42}$$

This analytical result can be illustrated and explained again by using the simple example with both sectors composed of one firm and an emissions tax imposed on the sector-1 firm only. This case is depicted in Figure 5-3. It is assumed that the emissions tax has been implemented on top of a preexisting emissions trading scheme. Facing the preexisting allowance price t^* and the tax rate τ_1, the firm in sector 1 is initially stimulated to reduce its emissions from e_1^* to e_1. The firm in sector 2 does initially not have an incentive to deviate from its emissions level e_2^*. However, both emissions levels do not represent equilibrium values. Due to a declining demand for emission allowances in sector 1, the demand function of the entire allowance market is shifted to the left. Since the emissions cap $\overline{E}$ will always be met, the equilibrium allowance price declines to t^B. However, in contrast to the case of emissions trading with homogeneous taxation, the reduction in the allowance price will be less than the tax rate. This is because the firm in sector 2, which are not subject to the tax, will take advantage of the allowance price reduction, demand more allowances and therefore compensate some of declining demand of the firm in sector 1. The equilibrium allowance price with emissions trading and heterogeneous taxation t^B will therefore lie somewhere between the equilibrium price with emissions trading and homogeneous taxation and the equilibrium price in the absence of taxation, i.e. $t^A < t^B < t$.

Figure 5-3: Welfare effects of combing emissions trading and a heterogeneous emissions tax compared to a single emissions trading scheme

Source: Bader (2000, p. 222)



Consequently, firms in sector 1 will have higher incentives to reduce their emissions than with emissions trading only: $t^B + \tau > t^*$ and $e_1^B < e_1^*$. In turn, firms in sector 2 will have fewer incentives to reduce their emissions rates than under a single emissions trading scheme since they only face the reduced allowance price. Therefore, emissions of firms in sector 2 will increase, i.e. $e_2^B > e_2^*$. Since the emissions cap is binding, overall emissions do not change. The emission reduction in sector 1 exactly corresponds to the emissions extension in sector 2. Thus, the implementation of the heterogeneous emissions tax on top of the emissions trading scheme results in emissions being shifted from sector 1 to sector 2 but not in an overall reduction of emissions. However, overall abatement costs increase. Compared to a single emissions trading scheme, sector 1 incurs higher abatement costs since emissions are reduced. Sector 2 bears lower total abatement costs since emissions are extended. The difference between both effects reflects the change in social welfare. Under the policy mix, this difference is negative implying a welfare loss. This loss is depicted as the dark-shaded rectangle in Figure 5-3.

Equation (5.42) allows answering the research question identified in Section 5.2. It reveals the most important factors that actually determine the welfare loss under a policy mix of emissions trading and heterogeneous taxation. Four variables are decisive:

First of all, the welfare loss increases with the heterogeneity of taxation. The larger the absolute value of the difference between both tax rates is, $|\tau_1 - \tau_2|$, the larger the inefficiency under the policy mix will be. As equations (5.38) and (5.39) illustrate, firms subject to the higher tax rate will have a stronger incentive to over-abate emissions with increasing heterogeneity of taxation. At the same time, firms facing the lower tax rate have a stronger incentive to under-abate emissions. Notably, the welfare loss does not increase linearly but exponentially in heterogeneity of taxation. Heterogeneity of taxation may therefore be considered the most important driving factor of welfare losses in the policy mix.

Secondly, the characteristics of the marginal abatement cost curve have an impact on the welfare loss. The inefficient distortion of abatement is the worse, the flatter the slope of the marginal abatement cost curve is, i.e. the smaller a is. This is because a fixed price distortion – as it is produced by the heterogeneous tax – will have a stronger impact on emissions with a flat marginal abatement cost curve compared to function with a steeper slope.

Thirdly, the inefficiency becomes more severe with an increasing total number n of firms. This is straightforward since an increase in the total number of firms implies a rise in total as well as sectoral emissions. Consequently, the total of distorted emissions abatement and emissions inefficiently shifted from one sector to another will increase as well.

Fourthly, the sectors' shares in the total number of firms – and thus their share in the emission allowance market – influence the inefficiency of the policy mix. Assuming all other variables to be constant, the welfare loss will be largest when both sectors have the same share in the emission allowance market. This finding can be proven analytically since the function $f(\alpha)=\alpha(1-\alpha)$ has a global maximum at $\alpha=0.5$. Thus, the larger the difference in market shares is, the smaller the inefficiency of the policy mix will be. This can be illustrated for the example of a tax imposed on sector 1 only. It will first be assumed that sector 1 is very big and sector 2 is very small. As has been explained above, taxing sector 1 reduces emissions in this sector and provides for a reduction of demand for allowances. Firms in sector 2 can benefit from the resulting decrease of the allowance price and extend their emissions. However, if sector 2 is very small, this increase in demand for allowances will hardly compensate the reduction brought about by sector 1. Consequently, the reduction of the allowance price will be close to that arising in the presence of emissions trading and homogeneous taxation. Firms in sector 2, which are not subject to the tax, will under-abate due to the decrease allowance price. However, the inefficiency of the policy mix is small, since most firms (those in sector 1) face nearly efficient abatement incentives set out by the reduced allowance price and the tax rate. Analogous results can be derived if one assumes that sector 1 is very small and sector 2 is very big. In this case, the implementation of a sectoral emissions tax on top of the emissions trading scheme will result in additional abatement in sector 1. However, the resulting decrease of demand for allowances will hardly affect the allowance price since sector 1 is small. The allowance price will therefore be close to the level arising in the presence of emissions trading only. Firms in sector 1 will over-abate since they are subject to the high emissions price and the emissions tax. Yet, the policy mix is almost efficient since most firms (those in sector 2) face the nearly efficient incentives of the emissions trading scheme.

5.3.5.3 The Policy Mix Compared with an Emissions Tax

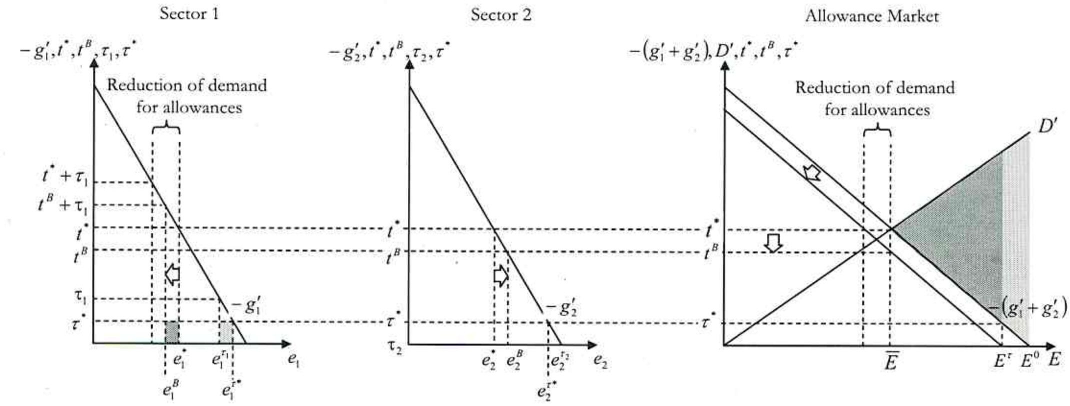
This section addresses the research question of how the policy mix of an emissions trading scheme and a heterogeneous emissions tax compares with a single heterogeneous emissions tax. As has been pointed out in Section 5.2, answering this question is of particular importance when an existing tax cannot be abolished. It has to be clarified then whether the implementation of an emissions trading scheme on top of this tax is desirable from an economic point of view. Section 5.3.3.2 has revealed that a heterogeneous emissions tax implemented in isolation results in an inefficient allocation of abatement measures. It will be analyzed in the following whether implementing an emissions trading scheme on top of the preexisting tax will reduce or deteriorate this inefficiency.

First insight can be gained easily by comparing analytic results derived for the policy mix of emissions trading and a heterogeneous emissions tax and the single heterogeneous emissions tax. Comparing equation (5.42) with equation (5.25) reveals that the welfare loss under the policy mix equals the welfare loss under the single heterogeneous tax in absolute terms. This is illustrated in Figure 5-4 which combines Figure 5-1 and Figure 5-3. As above, it is assumed in Figure 5-4 that both sectors are composed of one firm each and that only the firm in sector 1 faces an emissions tax. The dark-shaded rectangle depicting the loss under the policy mix in the graph for sector 1 equals the light-shaded rectangle symbolizing the loss under the single heterogeneous tax. Thus, implementing an emissions trading scheme on top of a heterogeneous emissions tax does not affect the absolute value of the welfare loss. In this respect, the welfare loss is not attributed to interactions in the policy mix but to the inefficient design of the tax. The inefficiency of the policy mix is just the inefficiency of the heterogeneous emissions tax.

However, the efficiency loss of heterogeneous taxation may become less important in relative terms when an emissions trading scheme is introduced in addition. This is illustrated in Figure 5-4 as well. In the absence of any policy, firms do not undertake any abatement measures. Total emissions amount to the natural level E^0, depicted in the graph for the allowance market. The heterogeneous emissions tax (set below the Pigovian level) provides for total emissions to decline to E^{τ}. In this case, the welfare loss from heterogeneous taxation compares with net gains from internalization depicted as the light-shaded trapezoid. If a more stringent emissions trading scheme is implemented on top of the tax, total emissions will be further reduced to the emissions cap $\overline{E}$. In Figure 5-4, it is assumed that the emissions cap is chosen optimally. The net gains from internalization under the policy mix then amount to sum of the dark-shaded triangle and the light-shaded trapezoid. Thus, the implementation of an emissions trading scheme on top of the heterogeneous emissions tax may be desirable from an efficiency point of view. It may increase the net gains from internalization while leaving the welfare loss from heterogeneous taxation unaffected. In the model under consideration, this will always be the case under two conditions. First of all, the emissions tax has to be set below the Pigovian level. Secondly, the emissions trading scheme has to be more stringent than the tax but the emissions cap must not be set below the efficient level, which is represented by $\overline{E}$ in Figure 5-4.

Figure 5-4: Welfare effects of combining emissions trading and a heterogeneous emissions tax compared with a single heterogeneous tax: The case of linear marginal abatement costs

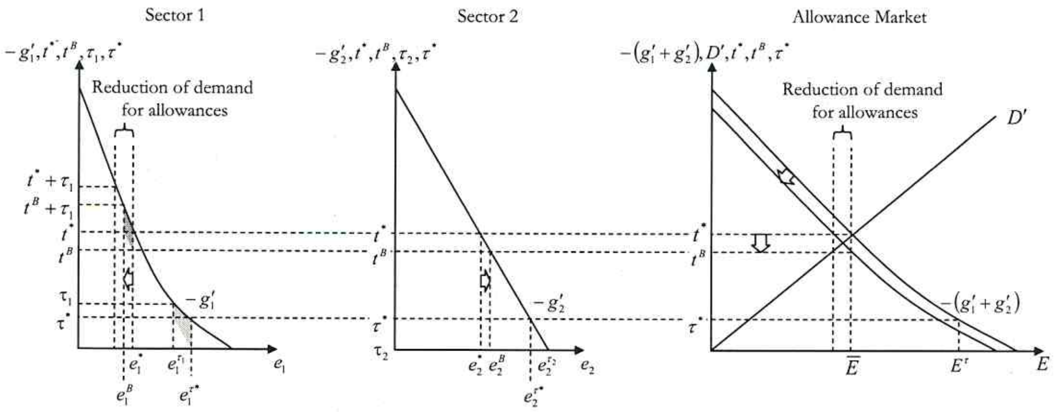
Source: Own figure



The discussion so far has been based on the assumption of linear marginal abatement costs. In reality, however, marginal abatement costs may well turn out to be non-linear. In this case, the above conclusions have to be qualified. With respect to greenhouse gas emissions, marginal abatement costs are often found to be convex (see, e.g., Ellerman and Decaux 1998; Criqui et al. 1999; Klepper and Peterson 2006). That is, marginal abatement costs are increasing at an increasing rate with abatement (or decreasing at a decreasing rate with emissions). In this case, the implementation of an emissions trading scheme on top of a heterogeneous emissions tax may even reduce the welfare loss from heterogeneous taxation. This case is illustrated in Figure 5-5, which is a modification of Figure 5-4. Inverse marginal abatement costs are now assumed to be convex for the representative firm in sector 1. For simplicity reasons, the function is still assumed to be linear for sector 2. If only the tax for sector 1 is in place, the emissions level of firms in sector 1 and 2 will be relatively high. The inefficient reduction of emissions in sector 1 due to the emissions tax occurs in the relatively flat section of the inverse marginal abatement cost curve. In this section, the tax rate has a relatively strong impact on the emissions level chosen. The resulting welfare loss – depicted as the light-shaded area in Figure 5-5 – will be relatively high. If an emissions trading scheme is implemented in addition to the emissions tax, emissions will be reduced in both sectors compared to the situation with the tax only. This implies that the emission distortion caused by the tax rate now occurs in the relatively steep section of sector 1's inverse marginal abatement cost curve. In this section, the same tax rate will have a smaller effect on emissions than in the flat section. The resulting welfare loss – depicted as the dark-shaded area in Figure 5-5 – will be less important than under the single heterogeneous emissions tax. In contrast to the linear case, the emissions cap now has an impact on the welfare loss stemming from heterogeneous taxation under the policy mix. Supposed that the tax rate is below the Pigovian level, the welfare loss will be the smaller, the closer the emissions cap is to the optimal emissions cap, i.e. $\overline{E}$ in Figure 5-5. Reducing the emissions cap (if it is set above the optimal level) will shift the inefficient distortion of the tax towards a steeper section of the inverse marginal abatement cost function and reduce the extent of over-abatement in sector 1.

Figure 5-5: Welfare effects of combining emissions trading and a heterogeneous emissions tax compared to a single heterogeneous tax: The case of convex marginal abatement costs

Source: Own figure



However, marginal abatement cost curves can also be assumed to be concave (see, e.g., Morris et al. 2008). Concavity implies that marginal abatement costs increase at a decreasing rate with abatement (or decrease at an increasing rate with emissions). With concave marginal abatement costs, the findings for convex marginal abatement costs have to be reversed. With a heterogeneous emissions tax only, the inefficient distortion arises at a relatively high emissions level where the marginal abatement cost curve is relatively steep. In this section, the impact of price distortion will be relatively small. Implementing the emissions trading scheme on top of the emissions tax reduces the emissions level. Consequently, the inefficient price distortion now occurs in the relatively flat section of the marginal abatement cost curve. In this section, the same tax rate will have a relatively strong effect on emissions. Given concave marginal abatement costs, the policy mix will increase the welfare loss stemming from heterogeneous taxation. Whether or not the policy mix is superior to the tax depends on how the increasing welfare losses due to heterogeneous taxation compare with the welfare gains due to overall larger emission reductions.

It can therefore be concluded that the implementation of an emissions trading scheme on top of an existing heterogeneous emissions tax may increase the efficiency of abatement if marginal abatement costs are linear or convex. In case marginal abatement costs turn out to be concave, the result is ambiguous.

5.3.5.4 Overcoming Inefficiencies in the Policy Mix

The third striking research question identified in the beginning of this chapter has been whether a policy mix of an emissions trading scheme and heterogeneous emissions tax can be designed such that welfare losses due to policy interactions are minimized. The reasoning behind this question has been that the tax cannot be abolished easily since it is meant to address multiple policy goals and criteria. Therefore light has to be shed on whether modifications of the emissions trading scheme may reduce the inefficiency of the policy mix. It has been shown that the inefficiency under the policy mix arises because emissions are inefficiently shifted from the high-taxed to the low-taxed sector. An intuitive solution to this problem is to restrict emissions trading between sectors. This implies that the policy mix will in fact consist of three policies: an emissions trading scheme for each sector and a heterogeneous tax. The emissions trading schemes may be perfectly separated or linked to a certain extent only. This policy mix will be analyzed in the following.

Finding the optimal policy solution is simple if marginal abatement cost functions are assumed to be identical across all sectors – as has been assumed so far. In this case, trades across sectors do not result in welfare gains. An optimal policy mix then encompasses two perfectly separated emissions trading schemes for sector 1 and 2. No trade should be allowed across sectors. For each sector, an individual equilibrium allowance price t_i emerges on the sector's emission market. In addition, firms in both sectors face a tax which is heterogeneous across sectors but homogeneous within each sector. According to equation (5.28), the first-order conditions for optimal abatement in each sector than write as:

$$-g_i'(e_i) = t_i + \tau_i \tag{5.43}$$

The overall emissions cap, $\overline{E}$, is divided into sectoral caps, $\overline{E}_i$. The respective equilibrium allowance prices in sectors 1 and 2 write similarly to equation (5.30):

$$t_1 = b - \frac{a}{\alpha n}\overline{E}_1 - \tau_1 \tag{5.44}$$

$$t_2 = b - \frac{a}{(1-\alpha)n}\overline{E}_2 - \tau_2 \tag{5.45}$$

In contrast to equation (5.30), the total number of firms in both sectors, n, is replaced by the respective number of firms in each sector in equations (5.44) and (5.45). Using these equations, the first-order conditions (5.43) can be rewritten:

$$-g_1'(e_1) = b - \frac{a}{\alpha n}\overline{E}_1 \tag{5.46}$$

$$-g_2'(e_2) = b - \frac{a}{(1-\alpha)n}\overline{E}_2 \tag{5.47}$$

For the policy mix to be cost-effective, marginal abatement costs have to be equal across both sectors. A cost-effective solution can be attained by choosing the optimal allocation of allowances for each sector. This allocation can be determined analytically by equalizing equations (5.46) and (5.47) and considering that sectoral emissions caps have to sum up to the overall emissions cap, i.e. $\overline{E}_1 + \overline{E}_2 = \overline{E}$. The result shows that each sector should receive a share of the overall emissions cap which corresponds to its share in the total number of firms, i.e. its share in overall emissions:

$$\overline{E}_1 = \alpha \overline{E} \tag{5.48}$$

$$\overline{E}_2 = (1-\alpha)\overline{E} \tag{5.49}$$

The allocation of allowances to both sectors thus becomes decisive for the cost-effectiveness of the policy mix if the emissions market is split up into independent sector markets. This finding is quite similar to that made for restricting emissions trading by zoning (see Section 3.4.1). Any deviation from the allocation derived above will result in inefficiency. The sector receiving more allowances will abate too little, while the other sector employs abatement options that are relatively too costly. This is in contrast to a common single emissions trading scheme for both sectors where the allocation of allowances does not matter. The level and heterogeneity of taxation does not affect cost-effectiveness. Moreover, the allocation within each sector does not have an impact on cost-effectiveness. Trading will always provide for marginal abatement costs to be equalized within each sector. Designing such a policy mix is easy for the regulator. He does not have to obtain any knowledge about the marginal abatement cost functions of the sectors.

Designing cost-effective emissions markets in the presence of a policy mix is more tedious when marginal abatement cost functions are heterogeneous across sectors and firms. This has also been brought forward for the similar case of zonal emissions trading schemes with several separate allowance markets (see Section 3.4.1). In theory, an emissions trading scheme with separated markets for both sectors may provide for a cost-effective solution in this case as well. As above, the regulator has to determine the sector's emissions caps such that the marginal abatement costs are equal for all firms in both sectors. With heterogeneous marginal abatement costs, however, the optimal sectoral emissions cap will depend on the characteristics of firms' marginal abatement costs functions. Yet, the regulator may only have a rough understanding of these functions. This may be due to the high transaction costs of gathering information on marginal abatement cost curves. Consequently, he may not be able to identify the optimal allocation of allowances to sectors 1 and 2. It is then likely that one sector receives too many and the other too few allowances from an efficiency point of view. The sector subject too over-allocation will abate too little. The sector subject to under-allocation will employ abatement measures that are too costly. Thus, the allocation of allowances to each sector will determine the inefficiency of the policy mix in this case. Recall, however, that the level of the tax rate does not have an effect on the efficiency of the policy mix. Since taxation is homogeneous within each sector, the sector's allowance price will be reduced exactly by the tax rate compared to a situation with emissions trading only.

In order to provide for an equalization of marginal abatement costs, the regulator may decide to allow trading of allowances between both sectors. However, trading on a one-to-one basis does neither allow for a cost-effective solution. This policy mix would correspond to a combination of a common emissions trading scheme and a heterogeneous tax. A uniform allowance price would emerge for both sectors. The sector with the higher tax rate would face higher abatement incentives under the policy mix than the sector with the lower tax rate. In this situation, the inefficiency would be driven by the heterogeneity of taxation but not by the allocation of allowances to each sector. Instead of an exchange on a one-to-one basis, the regulator may allow trading of allowances at a certain ratio only. In this case, the efficiency of the policy mix depends on the heterogeneity of taxation as well as the level of over- or under-allocation of allowances to each sector. These arguments will be illustrated for two possible scenarios: allowances may be over-allocated either to the sector with the higher tax rate or to the sector with the lower tax rate.

First of all, it will be assumed that sector 1 faces a higher tax rate, i.e. $\tau_1 > \tau_2$, and receives an over-allocation of allowances. Prohibiting any trades between sectors will result in abatement incentives to be too low for sector 1 and too high for sector 2: $t_1 + \tau_1 < t_2 + \tau_2$. In order to attain an efficient level of abatement, allowances (and thus emissions) should be traded from sector 1 to sector 2. However, an emissions trading scheme based on one-to-one trades can neither provide an efficient outcome. In fact, the results are reversed. Since allowance prices are equalized across both sectors, sector 1 now has higher incentives to abate than sector 2: $t_1 + \tau_1 > t_2 + \tau_2$. This implies that allowing unrestricted trading between sectors would result in too many allowances to be traded from sector 1 to sector 2. This inefficiency arises because due to heterogeneous taxation, sector-1 and sector-2 firms do not have the same incentive to abate if they hold one sector-1 allowance. In fact, this incentive is lower in sector 2 than in sector 1. Consequently, the incentive to buy a sector-1 allowance is higher for sector-2 firms than for sector-1 firms. This deficiency can be overcome by introducing a trading ratio β. The ratio determines how many sector-1 allowances the firms in sector 2 have to hold in order to cover one unit of their emissions. The first-order condition for optimal abatement for sector 1 then writes as in equation (5.43). The first-order condition for sector 2 is:

$$-g_2'(e_2) = \min[\beta t_1 + \tau_2, t_2 + \tau_2] \tag{5.50}$$

The ratio has to provide for firms in sector 1 and 2 to have the same incentives to abate if they buy an allowance on the allowance market of sector 1:

$$t_1 + \tau_1 = \beta t_1 + \tau_2 \tag{5.51}$$

Reorganizing equation (5.51) yields the optimal trading ratio:

$$\beta = 1 + \frac{\tau_1 - \tau_2}{t_1} \tag{5.52}$$

The trading ratio provides that firms in sector 2 implicitly pay the sector-1 allowance price and the difference by which the sector-1 tax rate is higher than the sector-2 tax rate. Since the sector-1 tax rate exceeds the sector-2 tax rate, the trading ratio will be larger than one. Consequently, firms in sector 2 have to buy more than one sector-1 allowance to cover one unit of their emissions. They will do as long as $t_1 + \tau_1 = \beta t_1 + \tau_2 < t_2 + \tau_2$. Gains from trade vanish when $t_1 + \tau_1 = \beta t_1 + \tau_2 = t_2 + \tau_2$. In this case, marginal abatement costs will be equal across sectors. Thus, the trading ratio provides for cost-effective abatement in the entire industry by restricting the amount of allowances traded from sector 1 to 2.

The preceding discussion has been based on the assumption that the sector with the higher tax receives an over-allocation. In contrast, it is now assumed that the low-taxed sector receives excess allowances. It is supposed that the tax rate is lower in sector 1 than in sector 2: $\tau_1 < \tau_2$. Thus, sector 1 receives an over-allocation of allowances. Without trading between sectors, abatement incentives will be lower in sector 1 than in sector 2: $t_1 + \tau_1 < t_2 + \tau_2$. As in the first case, allowances (and thus emissions) should be transferred from sector 1 to sector 2. However, allowing trades from sector 1 to sector 2 on a one-to-one basis will not yield a cost-effective outcome. The explanation is different to the case in which the high-taxed sector receives the over-allocation. If the allowance price is lower in sector 1 than in sector 2, i.e. $t_1 < t_2$, trades of allowances from sector 1 to sector 2 will equalize the allowance prices in both sectors. In this case, however, firms in sector 2 still have higher incentives to abate than sector one due to the higher tax rate. Thus, they buy too few allowances from sector 1 from a cost-effectiveness point of view. There may be even cases where the allowance price in sector 1 is higher than in sector 2, i.e. $t_1 > t_2$, despite the over-allocation of allowances to sector 1. This situation arises when the tax rate for sector 2 is significantly higher than that for sector 1: $\tau_1 << \tau_2$. Firms in sector 2 then do not have any incentive to buy allowances from sector 1. In contrast, sector-1 firms have an incentive to buy allowances from sector 2. The resulting uniform allowance price will be lower than that existing in absence of trading between the sectors. The resulting overall incentives to abate are even lower for sector 1 after trading is allowed. Consequently, emissions are shifted from sector 2 to sector 1. This is contrary to what would be necessary from an efficiency point of view. The implementation of a trading ratio can only overcome this imperfection if the difference of the tax rate in sector 2 (the higher tax rate) and that in sector 1 (the lower tax rate) is below the allowance price which prevails in sector 1 before inter-sectoral trading is allowed, i.e. if $\tau_2 - \tau_1 > t_1$. Under this condition, the trading ratio given in equation (5.52) will be positive and between zero and one. This implies that firms in sector 2 have to hold less than one sector-1 allowance to cover its emissions. Sector-2 firms have a stronger incentive to buy sector-1 allowances than firms in sector 1. Consequently, more emissions will be shifted from sector 1 to sector 2 than with one-to-one trades. Equation (5.52) reveals that the ratio would become negative if the difference between the sectors' tax rates is sufficiently high. In fact, this will be the case if the difference of the tax rate in sector 2 (the higher tax rate) and that in sector 1 (the lower tax rate) exceeds the allowance price which prevails in sector 1 before inter-sectoral trading is allowed, i.e. if $\tau_2 - \tau_1 > t_1$. However, a negative trading ratio cannot emerge on an allowance market. It would imply that sector-2 firms have to pay a negative price – or rather receive remuneration – for each sector-1 allowance they purchase. Yet, sector-1 firms do not have any incentive to sell their allowances at a negative price. The trading ratio thus falls to zero but not below. Consequently, sector-2 firms will choose not to buy any allowances from sector 1 – but neither sector-2 allowances. The allowance price in sector 2 falls to zero. Eventually, firms in sector 2 only face the emissions tax. However, this tax rate is still higher than the sum of the allowance price and the tax rate in sector 1. Therefore, firms in sector 2 still abate too much and too costly compared to the social optimum.

In summary, there are indeed options to modify an emissions tradings scheme such that the welfare loss under a combination with a heterogeneous emissions tax is reduced. The regulator should establish two sectoral emissions trading schemes. If the regulator has perfect knowledge about the marginal abatement costs, he may simply determine the optimal emissions caps for each sector and prohibit any trades between sectors. Otherwise, a more pragmatic approach may be based on allowing the transfer of allowances between sectors at a certain trading ratio only. This approach assures a cost-effective outcome if the sector with the higher tax rate receives an over-allocation of allowances. If the regulator has at least a rough idea about marginal abatement costs in each sector, he may deliberately allocate excess allowances to the sector with the higher tax rate. Over-allocating allowances to the sector with the lower tax rate only yields a cost-effective outcome if the difference between the tax rates is sufficiently low. Whether modifications of the emissions trading scheme are not only cost-effective but also efficient, eventually depends on the overall emissions cap chosen for both sectors – just under a single emissions trading scheme.

5.4 Model with Emissions Depending on Output

The previous section has delivered valuable new insight on the interaction of an emissions trading scheme and an emissions tax. The analysis has been based on a simple model with emissions *not depending on output*. This approach has been very helpful in addressing a couple of open research questions. However, such a model is obviously also subject to restrictions since the assumption of emissions not being related to output is very simplistic. In order to further deepen the understanding of the policy mix, a more complex and realistic model is developed in this section. This model assumes that emissions are *depending on output*. Compared to the simpler model developed and applied in the previous section, this approach provides two additional innovations: First of all, it allows disentangling the multiple effects of an emissions trading scheme and an emissions tax with respect to the level of output and the emissions rate. The output reduction effect of the policy mix can be separated from the substitution effect (see Section 3.4.3.1 for a more in-depth explanation of these effects). Thereby, the analysis specifies how and where distortions under the policy mix occur in detail. Secondly, implications of an output tax – such as the German electricity tax – implemented on top of emissions trading can be investigated. Such analysis could not be conducted with the simple model assuming emissions to be independent from output. In this framework, an output tax would not have had any impact on emissions. Thus, it would not have affected the efficiency of abatement. Within the more complex setting of model developed in this section, the welfare effects of an output tax combined with an emissions trading scheme can be understood in detail for the first time.

In the following, the model assumptions as well as the conditions for the social optimum are laid out first of all (Sections 5.4.1 and 5.4.2). Subsequently, the efficiency properties of single policies

– an emissions trading scheme as well as an emissions and an output tax – are outlined (Section 5.4.3). The fourth section (Section 5.4.4) is devoted to analyzing the efficiency of a policy mix including an emissions trading scheme and a homogeneous or heterogeneous emissions tax (policy mix A and B, Sections 5.4.4 and 5.4.5). Finally and most importantly, the model is used to analyze interactions between an emissions trading scheme and a homogeneous or heterogeneous output tax (policy mix C and D, Sections 5.4.6 and 5.4.7). In either case, the focus is on comparing the policy mix with a single efficient emissions trading scheme. The discussion does not shed light on how the policy mix compares with a single tax since this issue has been tackled to a certain extent already in the preceding section. The emissions cap chosen for the emissions trading scheme is assumed to be constant for all types of policy mixes under consideration. Moreover, the analysis is based on the assumption that the emissions trading scheme is more stringent than the tax. That is, emissions trading brings about a larger emissions reduction than the tax. Deviations from this assumption have also been analyzed in the preceding section.

5.4.1 Model Assumptions

As in the preceding section, the performance of a policy mix of emissions trading and a tax is analyzed in a simple partial equilibrium model. The industry is assumed to have two sectors i. In contrast to the preceding model, both sectors are assumed to have the same size. Each sector consists of one representative firm. Firms are assumed to be price takers in both product and allowance markets. Trade between firms may occur in the allowance market (if existent) but not in the product market. Each sector is assumed to face exogenous demand functions for its output. Consumer demand in each sector q_i is assumed to be a simple, declining function of the output price in each sector p_i. Thus, demand in each sector is independent from demand in the other sector. The resulting inverse demand function for each sector is $p_i = p_i(q_i)$. This function is downward sloping as well, i.e. $p_i'(q_i) < 0$.

The representative firm i in each sector produces a level of output which, in equilibrium, equals demand q_i. Along with the production of output, firms generate emissions of a pollutant. For simplicity, it is assumed that the pollutant is uniformly mixed and does not accumulate. Thus, the environmental damage D is a function of total emissions of the industry E: $D = D(E)$. Total emissions are the sum of each sector's emissions e_i. In contrast to the model in the preceding section, it is now assumed that sectoral emissions compute as the product of an emissions rate μ_i and the sector's output q_i, i.e. $e_i = \mu_i q_i$. Thus, firms can reduce their emissions by bringing down their output and/or their emissions rate. The emissions rate can be reduced by abatement with cleaner technologies or cleaner inputs. For the purpose of this model, c_i is defined as the marginal production costs of firm i. Marginal production costs are assumed to be positive, constant in output, decreasing in the emissions rate and strictly convex: $c_i(\mu_i) > 0$, $c_i'(\mu_i) < 0$, and $c_i''(\mu_i) > 0$.

The regulator intends to maximize social welfare W. His maximization problem writes as:

$$\max_{q,\mu} W = \sum_{i=1}^{2}\left[\int_{0}^{q_i} p_i(q_i)dq\right] - \sum_{i=1}^{2}\left[c_i(\mu_i)q_i\right] - D\left(\sum_{i=1}^{2}\mu_i q_i\right) \tag{5.53}$$

In contrast to the model with emissions not depending on output, the regulator now has to consider consumer surplus as well. This is because emission policies will also have an effect on the output price (see below) and, thus, the welfare of consumers. Total welfare computes as the sum of consumer surplus and firm profits in each sector (represented by $\int_{0}^{q_i} p_i(q_i)dq$) less the corresponding production costs as well as the environmental damages stemming from production emissions.

5.4.2 The Social Optimum

Unlike in the model considered in the preceding section, the efficiency of emissions abatement now depends on firms' choices with respect to the level of output and the emissions rate. Differentiating equation (5.53) with respect to output and the emissions rate yields the necessary first-order conditions for the representative firm in sector i to produce optimal welfare:

$$p_i(q_i) = c_i(\mu_i) + D_q'\mu_i \tag{5.54}$$

$$-c_i'(\mu_i) = D_\mu' \tag{5.55}$$

Subscripts q and μ denote partial derivatives. In contrast to a model assuming emissions to be independent from output, the market price should now reflect not only marginal production costs incurred by the firm but also the marginal environmental damage produced by an additional unit of output. Thus, compared to a situation where emissions are independent of output – or where there is no pollution externality – the optimal price has to be higher. Consequently, the optimal output will be lower. The second condition implies that firms should employ measures to abate emissions until their marginal abatement costs equal the marginal damage resulting from an additional unit of emission.

5.4.3 Single Policies

5.4.3.1 Emissions Trading

As hitherto, it is supposed that the emissions trading scheme sets a cap $\overline{E}$ on total emissions of the entire industry. The regulator issues a corresponding amount of allowances and allocates a fixed number of $\overline{a}_i$ allowances to each firm free of charge. Firm i maximizes its individual profit π_i which computes as revenues less production costs and net allowance costs:

$$\max_{q,\mu} \pi_i = p_i q_i - c_i(\mu_i)q_i - t(\mu_i q_i - \overline{a}_i) \tag{5.56}$$

The corresponding first-order conditions are:

$$p_i^* = c_i(\mu_i^*) + t^* \mu_i^* \tag{5.57}$$

$$-c_i'(\mu_i^*) = t^* \tag{5.58}$$

where an asterisk indicates equilibrium values under the emissions trading scheme.

In contrast to the model with emissions not depending on output, the emissions trading scheme now has an impact on output as well. In equilibrium, firms in both sectors choose their output such that the sum of marginal production costs and marginal allowance costs for producing an additional unit of output equal the output price. Moreover, firms will reduce their emissions rates by abatement until marginal abatement costs equal the allowance price. The equilibrium allowance price t^* provides for conditions (5.57) and (5.58) to hold and the cap to bind: $\sum_i \mu_i^* q_i^* = \overline{E}$. The emissions trading scheme is cost-effective since marginal abatement costs are equalized across both sectors. It is efficient if the emissions cap is chosen optimally, i.e. such that $t^* = D_\mu'$.

5.4.3.2 Emissions Tax

The regulator may also choose to impose a tax on firms at rate τ for each unit of emission generated by them. The maximization problem for firms in both sectors then turns out to be:

$$\max_{q,\mu} \pi_i = p_i q_i - c_i(\mu_i)q_i - \tau \mu_i q_i \tag{5.59}$$

This yields the following first-order conditions:

$$p_i^\tau = c_i(\mu_i^\tau) + \tau\mu_i^\tau \tag{5.60}$$

$$-c_i'(\mu_i^\tau) = \tau \tag{5.61}$$

where superscript τ denotes equilibrium values in the presence of the emissions tax.

Similarly to emissions trading, the equilibrium output price reflects marginal production costs and the marginal tax burden and firms reduce their emissions rates until marginal abatement costs equal the tax rate. Thus, the tax also provides for an emissions target to be attained cost-effectively. The emissions tax is efficient if it equals marginal damages from emissions, i.e. $\tau = D_\mu'$.

5.4.3.3 Output Tax

In order to bring down emissions, the regulator may also decide to tax output. In this model, output is a complement to emissions. Reducing output therefore also decreases emissions. If a unit of output is taxed at rate ρ, firms in each sector maximize profit as in:

$$\max_{q,\mu} \pi_i = p_i q_i - c_i(\mu_i)q_i - \rho q_i \tag{5.62}$$

The corresponding first-order conditions write as:

$$p_i^\rho = c_i(\mu_i^\rho) + \rho \tag{5.63}$$

$$-c_i'(\mu_i^\rho) = 0 \tag{5.64}$$

where superscript ρ symbolizes equilibrium values under the output tax.

As an emissions trading scheme and an emissions tax, an output tax raises the output price and thus reduces output and consumption compared to the unregulated state. However, it may bring about inefficiencies in internalizing damages from emissions since it does not provide incentives to bring down the emissions per unit of output. Condition (5.64) reveals that firms will choose an emissions rate where marginal abatement costs are zero – which actually corresponds to the natural emissions rate prevailing in the absence of any regulation. Thus, an output tax sets incentives to reduce emissions by decreasing output but not by switching from dirty inputs and production processes to cleaner ones. The same reduction would be achieved at less cost if firms produced more output but reduced their emissions rate at the same time.

Only when emissions are fixed with output, i.e. when emissions can only be reduced by reducing output, can an output tax be efficient in correcting an environmental externality. In this case, condition (5.55) becomes obsolete. Comparing condition (5.63) with (5.54) yields that the optimal rate for an output tax equals the marginal environmental damage resulting from producing an additional unit of output: $\rho = D'_q \mu_i$. If the emissions rate varies across firms or sectors, the output tax has to be differentiated accordingly to be efficient.

5.4.4 Policy Mix A: Emissions Trading and Homogeneous Taxation of Emissions

5.4.4.1 Firms' Choices

It is first supposed that all firms in the industry participate in an emissions trading scheme and have to pay a homogeneous emissions tax in addition. Thus, the profit maximization problem for the representative firm in each sector is as follows:

$$\max_{q,\mu} \pi_i = p_i q_i - c_i(\mu_i) q_i - t(\mu_i q_i - \overline{a}_i) - \tau \mu_i q_i \tag{5.65}$$

The first-order conditions yielding optimal profits for firms then compute as:

$$p_i^A = c_i(\mu_i^A) + (t^A + \tau)\mu_i^A \tag{5.66}$$

$$-c_i'(\mu_i^A) = t^A + \tau \tag{5.67}$$

where superscript A denotes equilibrium values for the policy mix of emissions trading and a homogeneous emissions tax.

Now, firms in both sectors will increase their output until the market price equals the sum of marginal production costs and the policy-induced cost per unit of output. The policy-induced cost consists of the tax burden as well as the allowance cost per unit of output. Moreover, firms have an incentive to reduce their emissions rate until marginal abatement costs are equal to the sum of the allowance price and the tax rate.

5.4.4.2 The Policy Mix Compared with Emissions Trading

Compared to a single emissions trading scheme, firms now face a double incentive to reduce their emissions. However, the equilibrium allowance price under the policy mix, t^A, is not identical to the equilibrium allowance price with emissions trading only, t^*. This can be derived analytically as follows. First of all, it has to be considered that the allowance price will provide for the emissions cap to be met. That is, total emissions are not reduced by the implementation of the emissions tax on top of the emissions trading scheme. Second, conditions (5.66) and (5.67) reveal that firms in both sectors have the same incentive to produce output and to reduce their emissions rate. If the emissions cap is binding and incentives are homogeneous across sectors, output prices and marginal abatement costs must correspond to those under a single emissions trading scheme. Combining conditions (5.66) and (5.67) with conditions (5.57) and (5.58) then yields that the equilibrium allowance price arising if only an emissions trading scheme is in place equals the sum of the tax rate and the equilibrium allowance price emerging under the policy mix: $t = t^A + \tau$. This implies that the equilibrium allowance price is reduced exactly by the tax rate if a homogeneous emissions tax is implemented in addition to emissions trading. As outlined for the model with emissions being independent of output, this is because the tax equally reduces the marginal willingness to pay for an emissions allowance for all firms in the industry. Consequently, the efficiency properties are identical to those of a single emissions trading scheme. Firms produce the same level of output and choose the same emissions rate: $q_i^A = q_i^*$ and $\mu_i^A = \mu_i^*$. Thus, sectoral emissions are neither affected by the implementation of the tax: $e_i^A = e_i^*$. The policy mix attains the emissions cap cost-effectively since abatement incentives are equal for firms in both sectors. The policy mix is efficient if the emissions cap is chosen optimally. Yet, the same result would be attained by an emissions trading scheme (or an homogeneous emissions tax) implemented in isolation. The policy mix is redundant. Consequently, the results in this model confirm those derived under the model with emissions not depending on output.

5.4.5 Policy Mix B: Emissions Trading and Heterogeneous Taxation of Emissions

5.4.5.1 Firms' Choices

It is now assumed that a heterogeneous emissions tax is implemented in addition to a common emissions trading scheme. For simplicity reasons, an extreme scenario is considered. Sector 1 is taxed at the full rate. This tax rate is equal to the homogeneous tax rate in the previous section. Sector 2 is entirely exempt from the tax. The representative firm in sector 1 maximizes its profits subject to the incentives set out by the emissions trading scheme and the tax:

$$p_1^B = c_1\left(\mu_1^B\right) + \left(t^B + \tau\right)\mu_1^B \tag{5.68}$$

$$-c_1'\left(\mu_1^B\right) = t^B + \tau \tag{5.69}$$

where superscript B denotes equilibrium values for the policy mix of emissions trading and a heterogeneous emissions tax.

The first-order conditions for the representative firm in sector 2 write similarly to those for the case of a single emissions trading scheme:

$$p_2^B = c_2\left(\mu_2^B\right) + t^B \mu_2^B \tag{5.70}$$

$$-c_2'\left(\mu_2^B\right) = t^B \tag{5.71}$$

Consequently, the first-order conditions of the representative firms in sector 1 and 2 deviate from each other. When deciding about its output and emissions rate, the sector-1 firm considers the allowance price and the tax rate – quite similar to the case of an emissions trading scheme with a homogeneous tax. In contrast, the firm in sector 2 only takes the allowance cost into account.

5.4.5.2 The Policy Mix Compared with Emissions Trading

The implementation of a heterogeneous tax in addition to an emissions trading scheme will again have an effect on the equilibrium allowance price. Facing the preexisting allowance price t^* and the tax rate τ, firm 1 is stimulated to reduce its emissions compared to the level e_1^* chosen under the emissions trading scheme. Since the emissions cap $\overline{E}$ will always be met, the equilibrium allowance price declines. However, in contrast to the case of emissions trading with homogeneous taxation, the reduction in the allowance price will be less than the tax rate. This is because firms in sector 2, which are not subject to the tax, will take advantage of the allowance price reduction, demand more allowances and therefore compensate some of declining demand of firms in sector 1. Therefore, the equilibrium allowance price with emissions trading and heterogeneous taxation, t^B, will lie somewhere between the equilibrium price with emissions trading and homogeneous taxation and the equilibrium price in the absence of taxation, i.e. $t^A < t^B < t^*$.

The incentives firms face in the presence of emissions trading and a heterogeneous emissions tax deviate from those under emissions trading only. Since $t^A < t^B < t^*$ and $t^* = t^A + \tau$, the policy burden per unit of emission will be higher for the sector-1 firm: $t^B + \tau > t^*$. Consequently, it will reduce its emissions rate further than with emissions trading only, i.e. $\mu_1^B < \mu_1^*$. Determining the effect on the equilibrium output price is more tedious. With a lower emissions rate, the marginal production costs of sector 1 $c_1\left(\mu_1^B\right)$ in condition (5.68) are higher than under emissions trading only. In addition, the burden per unit of emission, $t^B + \tau$, is higher. Indeed, the latter effect may be (partly) offset by the reduction in the emissions rate. However, the reduction of the emissions rate must be far beyond the increase of the burden per unit of emission in order to also offset the increase of marginal production costs. Therefore, it may be assumed that the output price in sector 1 increases and the output level decreases due to declining demand if a tax is imposed on firms in this sector: $q_1^B < q_1^*$. With lower output and lower emissions rate, the emissions level in sector 1 will be lower under the policy mix of emissions trading and a heterogeneous tax than in the case of a single emissions trading scheme, i.e. $\mu_1^B q_1^B = e_1^B < \mu_1^* q_1^* = e_1^*$.

Since $t^A < t^B < t$, the firm in sector 2 will have fewer incentives to reduce its emissions rate than under a single emissions trading scheme, i.e. $\mu_2^B > \mu_2^*$. The equilibrium output price in sector 2 is likely to decrease due to the implementation of the heterogeneous emissions tax. On the one hand, the increased optimal emissions rate provides for a decline of marginal production costs. On the other hand, the allowance price that has to be paid per unit of emission is lower. This is outweighed to a certain extent by the higher emissions rate. With decreasing output prices, demand rises and output of sector 2 increases: $q_2^B > q_2^*$. Given a higher level of output and a higher emissions rate, the emissions in sector 2 will be higher with a policy mix of emissions trading and a tax on sector 1 than with emissions trading only: $\mu_2^B q_2^B = e_2^B > \mu_2^* q_2^* = e_2^*$.

Since the emissions cap is binding, emission reductions in sector 1 are perfectly compensated by an increase of emissions in sector 2. Thus, the policy mix of an emissions trading scheme and a heterogeneous emissions tax shifts emissions from the sector with the high tax rate to the sector with the lower (or zero) tax rate. The tax does not have an impact on total emissions. However, due to heterogeneous taxation, the emissions cap is not attained cost-effectively anymore. Comparing condition (5.69) with condition (5.71) reveals that marginal abatement costs are higher in sector 1 than in sector 2. Thus, firms in sector 1 abate relatively costly while firms in sector 2 abstain from relatively cheap options for abatement. Given this insight, an emissions trading scheme combined with a heterogeneous tax cannot provide for an efficient outcome even if the overall emissions cap was chosen optimally. These findings again confirm the results derived before in the simpler model with emissions being independent of output.

5.4.5.3 Implications of Output Market Interactions

So far, it has been assumed that output markets of both sectors are independent from each other. That is, demand in one sector does not depend on the output market price in the other sector. In reality, however, there may be trade interactions between both sectors on the output market. In a model with emissions depending on output, these interactions may also have in impact on the performance of the policy mix of emissions trading and a heterogeneous emissions tax. In this respect, the results derived in this model may deviate from those derived in the model with emissions being independent of output. Analyzing trade interactions in output markets comprehensively would actually call for a general equilibrium model. However, the partial equilibrium model employed so far may already help to understand the character of these interactions.

In the presence of trade interactions, sectoral output does not only depend on the market price in the respective sector but also on the price prevailing in the other sector: $q_1(p_1, p_2)$ and $q_2(p_2, p_1)$. The change in output of sector 1 can then be given as

$$\frac{dq_1}{q_1} = \eta_{11}\frac{dp_1}{p_1} + \eta_{12}\frac{dp_2}{p_2} \tag{5.72}$$

where η_{11} is the own-price elasticity of demand and η_{12} is the cross-price elasticity of demand. A change of output in sector 2 can be explained analogously. The own-price elasticity represents the percentage change in demand and output in one sector resulting from a one-percent change of the output price in the same sector. The own-price elasticity is typically negative. That is, an increase in the output price usually reduces demand and output. The cross-price elasticity indicates the percentage change in demand and output in one sector due to a one-percent change in the output price of the other sector. The sign of this elasticity depends on the characteristics of the goods produced in both sectors. On the one hand, goods may be *complements*. In this case, the cross-price elasticity is negative. Demand and output in one sector decrease as the price in the other sector increases. This case may occur when one sector produces an input to the other sector. For example, electricity generation provides an input to the steel industry. An increase in electricity prices increases production costs of the steel industry and, thus, reduces demand and output. In turn, an increase in steel prices reduces the output of steel and, thus, also the demand for the input electricity. Likewise, complementarity exists when both sectors provide an input to a third sector, such as electricity and steel to the manufacturing industry. On the other hand, goods produced in both sectors may be *substitutes*. In this case, the cross-price elasticity is positive. An increase of the output price in one sector brings about an increase in demand in the other sector. This situation may arise, for example, when both sectors produce the same good but use different types of input. Thus, output prices may be driven by different factors. For example, electricity generators may be divided into a sector producing electricity from coal and another producing electricity from natural gas. If the output price of coal-generated electricity increases consumers may switch to suppliers using natural gas. A similar case may occur when sectors produce similar goods in different countries.

Given this insight, the interactions under the policy mix revealed in the previous section have to be reconsidered. As hitherto, the analysis supposes that a common emissions trading scheme is implemented for both sectors and only sector 1 faces an emissions tax in addition. The discussion in the previous section revealed that the implementation of a heterogeneous tax on top of an emissions trading scheme resulted in an increase of the equilibrium output price in sector 1, i.e. $dp_1/p_1 > 0$. At the same time, the price in sector 2 declined, i.e. $dp_2/p_2 < 0$. Price changes caused an unambiguous decrease of output in sector 1 as well as an increase of output in sector 2. These results may be qualified when trade interactions between the output markets are taken into account.

First of all, it is assumed that goods produced in both sectors are complements ($\eta_{12} < 0, \eta_{21} < 0$). In this case, trade interactions on the output market may counteract output distortions resulting from the policy mix. This can be shown using equation (5.72). In sector 1, the negative output effect of an increase in the sector-one price ($\eta_{11}(dp_1/p_1) < 0$) may be compensated by the positive output effect of a price decline in sector 2 ($\eta_{12}(dp_2/p_2) > 0$). Thus, output in sector 1 will be higher compared to a situation with not interactions on the output market. A higher output will bring about higher emissions in sector 1. In turn, output and emissions in sector 2 will be lower. Consequently, the inefficient distortion of emission abatement under the policy mix (too much abatement in sector 1, too little in sector 2) may be ameliorated when trade interactions on the output market are taken into account. This argument can be illustrated using the case of an electricity sector and a steel sector participating in emissions trading. If the electricity sector is taxed, the electricity price will increase and output and demand will decline. The electricity sector will abate too much. In the absence of trade interactions on the output market, the steel sector will benefit from a decreasing allowance price. The output price will decline and output and demand will increase. This sector will abate too little. If electricity is considered as an important input to steel production, i.e. trade interactions on the output market exist, two effects may arise: On the one hand, the rising demand for steel may also foster the demand and output of electricity.[59] On the other hand, the increased electricity price may raise steel production costs and curb the increase in steel demand. Thus, there may be a positive effect

[59] However, steel producers may also have the option to generate electricity themselves. Increasing electricity price may thus result in a modification of steel producers' make-or-buy decisions.

on output and emissions in the electricity sector and/or a negative effect on output and emissions in the steel sector. The inefficient shift of emissions from the electricity sector to the steel sector is reduced.

Alternatively, it may be assumed that goods produced in both sectors are substitutes ($\eta_{12} > 0, \eta_{21} > 0$). In this case, interactions on the output market may in fact deteriorate the output distortions stemming from the policy mix. Again, this can be demonstrated analytically using equation (5.72). In sector 1, the negative output effect of an increase in the sector-1 output price ($\eta_{11}(dp_1/p_1) < 0$) may be even aggravated by the negative effect of a price decrease in sector 2 ($\eta_{12}(dp_2/p_2) < 0$). Output reduction in sector 1 will be even larger under the policy mix compared to a case without any trade interactions on the output market. A lower output results in lower emissions in sector 1. In contrast, output and emissions in sector 2 will increase. Thus, the shift of emissions from sector 1 to sector 2 and the resulting welfare losses in the policy mix will be even more severe with trade interactions in output markets. It may be considered, for example, that all fossil-fueled electricity generation is covered by emissions trading and only coal-fired generation is subject to an emissions tax in addition. In the absence of trade interactions, the coal sector will then abate too much and the natural gas sector too little. This distortion will be deteriorated when electricity consumers may switch from coal-fueled to natural gas-fueled electricity generation.

Therefore, it may be concluded that trade interactions in the output market have to be taken into account when the performance of the policy mix is under consideration. If sectors produce complementary goods, the welfare loss under the policy mix may be mitigated. If goods are substitutes, the inefficiency may be increased. Nevertheless, the basic argument regarding the policy mix of emissions trading and emissions taxes still holds: The policy mix will exhibit a redundancy at best and an inefficiency at worst. For simplicity reasons, trade interactions will not be considered in the remainder of the section.

5.4.6 Policy Mix C: Emissions Trading and Homogeneous Taxation of Output

5.4.6.1 Firms' Choices

It is now supposed that emissions trading is not supplemented by a tax on emissions but by a tax on output. Both policies are assumed to cover the entire industry. The representative firm in sector i then faces the following profit maximization problem:

$$\max_{q,\mu} \pi_i = p_i q_i - c_i(\mu_i)q_i - t(\mu_i q_i - \bar{a}_i) - \rho q_i \tag{5.73}$$

Compared to the case with homogeneous taxation of emissions, the tax burden is irrespective of the emissions rate. The corresponding first-order conditions for maximum profit are:

$$p_i^C = c_i\left(\mu_i^C\right) + t^C \mu_i^C + \rho \tag{5.74}$$

$$-c_i'\left(\mu_i^C\right) = t^C \tag{5.75}$$

where superscript C labels the equilibrium values under a policy mix of emissions trading and a homogeneous tax on output. Firms will choose an output level where the sum of marginal production costs, allowances costs and the output tax rate equal the market price for output. In contrast, the choice of the emissions rate is only dependent on the allowance price.

5.4.6.2 The Policy Mix Compared with Emissions Trading

Equation (5.75) shows that the implementation of the output tax on top of the emissions trading scheme does not have a direct effect on the emissions rate. At first, firms still consider the preexisting allowance price t^* only and choose the same emissions rate as under emissions trading. Equation (5.74) reveals, however, that the output tax has a direct impact on output. In the presence of the preexisting allowance price and the preexisting emissions rate, the output price under the policy mix exceeds the equilibrium output price under a single emissions trading by the tax rate ρ. Consequently, demand and output are lower in the presence of the policy mix than with emissions trading only. Given the same emissions rate and lower output, overall emissions under the policy mix fall below the emissions cap. Of course, extra allowances which were not needed to cover actual emissions are offered for sale bringing down the equilibrium allowance price. Therefore, the equilibrium allowance price decreases with the implementation of an output tax in addition to emissions trading: $t^C < t^*$. Consequently, firms have fewer incentives to reduce the emissions rate than under emissions trading only (conditions (5.75) and (5.58)). That is, the equilibrium emissions rate increases: $\mu^C > \mu^*$. Thus, the homogeneous taxation of output does not have a direct effect on the emissions rate but an important indirect impact. In order for overall emissions under the policy mix to satisfy the emissions cap, a higher emissions rate is compensated by a lower output level. In equilibrium, the output price will therefore still be higher and the output level will still be lower than with emissions trading only, i.e. $q^C < q^*$. Thus, supplementing emissions trading by an output tax does not alter the overall emissions level. Yet, it substitutes emission reductions due to abatement by reductions due to a decrease of output.

The policy mix of emissions trading and a homogeneous output tax cannot be efficient even though the emissions cap was chosen correctly. It has to be recalled that the allowance price under a single emissions trading scheme provides for the basic first-order conditions (5.54) and (5.55) to hold as long as only the emissions cap is determined efficiently. With emissions trading only, the output price reflects marginal production costs and marginal damage costs of an additional unit of output. Marginal abatement costs equal marginal damage costs of an additional unit of emissions. Since the equilibrium allowance price under the policy mix, t^C, is lower than that under emissions trading only, the basic first-order conditions are not satisfied. Marginal abatement costs corresponding to the equilibrium emissions rate are lower than marginal damage costs. In contrast, the equilibrium output price is higher than the sum of marginal production costs and marginal damage costs. Thus, overall welfare would be increased by reducing the emissions rate (and increasing the marginal abatement costs) and producing more output at the same time. Similarly to a single output tax, the policy mix leads results in too little production and too high emissions per unit of output in both sectors.

5.4.7 Policy Mix D: Emissions Trading and Heterogeneous Taxation of Output

5.4.7.1 Firms' Choices

Finally, the regulator may choose to combine both deviations from the ideal of a homogeneous tax on emissions, which have been discussed so far. It is supposed that both industry sectors participate in the emissions trading scheme. In addition, the extreme case of a heterogeneous output tax is considered. Firm 1 faces the full output tax rate. This rate is identical to the homogeneous output tax rate considered in the previous section. Firm 2 is entirely exempt from the output tax. The first-order conditions for firm 1 are quite similar to those given for the case of emissions trading and a homogeneous output tax:

$$p_1^D = c_1\left(\mu_1^D\right) + t^D \mu_1^D + \rho \tag{5.76}$$

$$-c_1'\left(\mu_1^D\right) = t^D \tag{5.77}$$

where superscript D denotes the equilibrium values for the policy mix of emissions trading and a heterogeneous output tax. The first-order conditions for firm 2 write as for the case of emissions trading only:

$$p_2^D = c_2\left(\mu_2^D\right) + t^D \mu_2^D \tag{5.78}$$

$$-c_2'\left(\mu_2^D\right) = t^D \tag{5.79}$$

The first-order conditions differ only slightly between the sectors. Both sectors consider the allowance price when determining the optimal emissions rate. This is unlike the case of a policy mix with heterogeneous taxation of emissions. Only the conditions for the equilibrium output prices vary with the sector. The output price in sector 2 reflects marginal production and allowance costs while that in sector 1 includes the tax rate in addition.

5.4.7.2 The Policy Mix Compared with Emissions Trading

For illustrative reasons, it is supposed at first that the allowance price t^*, which emerged under the single emissions trading scheme, prevails after the output tax is introduced for sector 1. The first-order conditions then remain unchanged for the sector-2 firm. It chooses the same emissions rate and output level and produce the same amount of emissions. Likewise, firm 1 reduces the emissions rate as under emissions trading only. However, the output price in sector 1 is higher by the tax rate ρ. Consequently, output and overall emissions in sector 1 are lower than with emissions trading only. Thus, a heterogeneous output tax supplementing emissions trading frees up allowances in sector 1. Therefore, the equilibrium allowance price under policy mix is eventually lower than with emissions trading only: $t^D < t^*$. With a lower allowance price, the equilibrium emissions rate is higher in both sectors than with emissions trading: $\mu_i^D > \mu_i^*$. As a result, the output price in sector 2 is likely to decline. Marginal production costs are lower compared to a single emissions trading scheme due to a higher emissions rate. The allowance price is lower as well - although this effect is (partly) offset by the higher emissions rate. With a lower output price, demand and output increase in equilibrium in sector 2: $q_2^D < q_2^*$. Since the emissions rate as well as the output level in sector 1 rise due to the implementation of the tax, the emission level in sector 2 is also higher than with emissions trading only: $e_2^D > e_2^*$. Given a fixed emissions cap, this finding implies that emissions in sector 1 must be lower than with emissions trading only: $e_1^D < e_1^*$. Since the emissions rate in sector 1 is found to be higher than under a single emissions trading scheme, the emission reduction is brought about by a significant reduction of output in sector 1, i.e. $q_1^D < q_1^*$.

Obviously, the implementation of an output tax on only some of the participants of an emissions trading scheme brings about inefficiency – even if the emissions cap was chosen correctly. In both sectors, the marginal abatement costs will always be below the marginal damage costs since the equilibrium allowance price will always be too low. The output price in sector 2 will fall short of the optimal price while the output price in sector 1 will exceed the optimal one. Consequently, output in sector 1 (sector 2) will be to low (high). The emissions rate is too high in both sectors.

Thus, the policy mix of an emissions trading scheme and a heterogeneous output tax combines the two distortions which have been analyzed in isolation in the previous sections. Due to the heterogeneity of taxation, emissions are inefficiently shifted from the taxed sector to the sector exempt from taxation. Due to the taxation of output rather than emissions, emission reduction by bringing down the emissions rate is inefficiently substituted by reducing output. Consequently, both the rise of emissions in the non-taxed sector as well as the increase of the emissions rate in the taxed sector are compensated by a reduction of output in the taxed sector. Overall welfare could be increased if firms in both sectors reduced their emissions rate and if firms in the taxed sector produced more while those in the other sector reduced their production.

5.5 Summary

The chapter has analyzed theoretically the combination of an emissions trading scheme and a tax with additive incentives. The assumption of additive incentives implies that firms have to hold allowances for their emissions and pay a tax in addition, which is directly or indirectly related to emissions as well. This analysis is of high political relevance since such a policy mix can be found in many countries. A striking example is Germany where the EU ETS is combined with an electricity tax (see Sections 2.3.1 and 2.3.3).

Existing studies argue that such a policy mix is redundant with respect to mitigating climate change when all participants in the emissions trading scheme are subject to a homogeneous emissions tax. It is also shown that deviations from homogeneous taxation result in an inefficient "policy mess", which makes GHG emission abatement more costly than necessary. The straightforward recommendation usually made is to abolish taxation for participants in the emissions trading scheme. Given this finding, not much effort has been invested so far in understanding interactions in the policy mix in detail. However, this existent discussion of the policy mix is fairly theoretical. It is based on the very restricted assumption that emissions trading as well as the tax are only meant to mitigate greenhouse gases efficiently. The characteristics of a real-world policy mix may be much more complex, though. In particular, the tax is usually meant to meet a multiplicity of goals and criteria, which have been neglected in the available theoretical studies. This empirical insight may reveal that abolishing the tax may not be a desirable or feasible option. Consequently, the suitability of available theoretical analyses to derive useful policy recommendations seems to be quite limited.

In order to strengthen the relevance of economic thinking for real-world policy-making, an important change in perspective has been suggested in this chapter. It has been assumed that the tax cannot be abolished because it also addresses other policy objectives than mitigating climate change. Under this constraint, it has been found necessary to understand interactions in the policy mix in much more detail than so far. In fact, a whole new set of open research questions has been identified: What are drivers of the inefficiency in the policy mix compared to the social optimum? Is it at all desirable to implement an emissions trading scheme in addition to a tax if the latter cannot be abolished? Are there means to reduce the inefficiency of the policy mix? By addressing these questions, the analysis in this chapter has made important contributions to the economic understanding of a policy mix encompassing emissions trading and a tax. These contributions are of theoretical as well as of political interest. In order to answer the open questions properly, partial equilibrium models have been developed and applied in this chapter. Similar models had been used to study other policy combinations. These models have been modified and extended in this chapter to analyze interactions between an emissions trading scheme and a tax.

As a *first* result of the theoretical analysis, it has been clarified which factors actually determine the extent of inefficiency in the policy mix. In a simple model with emissions being independent of output, four important drivers have been identified: (1) It has been shown that the heterogeneity of taxation across participants is the major determinant of welfare losses in the policy mix. The higher is the difference in tax rates, the more emissions are inefficiently shifted from the high-taxed to the low-taxed sector, both of which are subject to the emissions trading scheme. Three further factors have also been found to be decisive in this model: (2) the slope of the marginal abatement cost curves, (3) the total number of firms (and corresponding emissions) and (4) the shares of each sector in the emissions market. Additional insight has been provided by the more complex model with emissions depending on output. In this model, the important distorting effect of heterogeneous taxation has been confirmed. It has been shown that with heterogeneous taxation, output level and emissions rate of the high-taxed sector are relatively too low while the reverse is true for the low-taxed sector. Moreover, the model has served to identify two additional drivers of inefficiency in the policy mix: (5) It has revealed that welfare under the policy mix may also be affected by output market interactions. If goods produced in the high-taxed and low-taxed sectors are complements, the inefficiency of the policy mix may be mitigated by trade interactions in the output markets. If goods are substitutes, the inefficiency may be even deteriorated. (6) Finally, it has been demonstrated in the complex model that output taxation – instead of emission taxation – may be a further important driver of inefficiency in the policy mix. If output is uniformly taxed for all participants in emissions trading, firms will engage too much in reducing output and too little in bringing down the emissions rate. Emissions abatement is distorted. If an emissions trading scheme is combined with a heterogeneous output tax on one sector only, the inefficient distortions resulting from heterogeneous taxation and output taxation combine. Emission abatement is inefficiently shifted from the non-taxed sector to the taxed sector. Moreover, emission reduction by bringing down the emissions rate is inefficiently substituted by reducing output in the taxed sector. Table 5-2 summarizes the major findings derived under the model with emissions depending on output for different types of policy mix design. The profound understanding of the six drivers identified helps to determine the actual extent of inefficiency under a policy mix with emissions trading and taxation. This knowledge is very valuable when policies are found to pursue multiple objectives and criteria. It allows comparing the welfare losses regarding the mitigation of climate change with benefits from attaining other policy goals. Sound policy recommendations can only be derived on the basis of this comparison.

Table 5-2: Efficiency of firm decisions under different policy mixes of emissions trading and taxes

	Emissions trading and homogeneous emissions tax	Emissions trading and heterogeneous emissions tax (tax on sector 1 only)	Emissions trading and homogeneous output tax	Emissions trading and heterogeneous output tax (tax on sector 1 only)
Output sector 1	Optimal	Too low	Too low	Too low
Emissions rate sector 1	Optimal	Too low	Too high	Too high
Output sector 2	Optimal	Too high	Too low	Too high
Emissions rate sector 2	Optimal	Too high	Too high	Too high

The *second* innovative result has been derived from the comparison of the policy mix with a single tax. This analysis has been conducted in the framework of the simple model with emissions not depending on output. The comparison revealed that whether or not the welfare loss of a policy mix should be blamed on interactions in the policy mix depends crucially on the point of reference. As has been shown above, the policy mix is clearly inferior to a single emissions trading scheme. Thus, implementing a tax on top of an emissions trading scheme impairs the efficiency of emissions abatement. However, the policy mix may be superior to a heterogeneous emissions tax implemented in isolation in terms of efficiency. Thus, the inefficiency should be attributed to the suboptimal design of the tax rather than to interaction in the policy mix. It has been found that if a heterogeneous tax cannot be abolished, implementing an emissions trading scheme in addition to it may in fact reduce the inefficiency in emissions abatement. This is particularly so when marginal abatement costs are found to be convex.

As a *third* important contribution, an innovative approach to reduce the inefficiency of the policy mix of emissions trading and a tax has been identified and studied in this chapter for the first time. It has been demonstrated that welfare losses can be brought down by modifying the design of the emissions trading scheme. A pragmatic solution may be to establish separate allowance markets for each sector, over-allocate allowances to the sector with the higher tax rate and allow for the exchange of allowances between sectors at a certain trading ratio only. This new approach may help to overcome the inefficiency resulting from heterogeneous taxation. However, it has to be born in mind that it does not correct the distortion from output taxation.

The analysis carried out in this chapter has made substantial contributions to approaching economic policy mix theory to reality. Therefore, it has established an important basis for evaluating an actually existing policy mix. The new insight can be used to evaluate the combination of the EU ETS and the electricity tax found in Germany. Nevertheless, deriving explicit policy recommendations for the German case on the basis of theoretical considerations only is difficult. An appropriate discussion of possible policy recommendations has to employ a broader approach considering the details of policy design, the characteristics of the allowance and output markets as well as other features of the institutional environment. This discussion will be provided subsequently in Chapter 6.

6 Evaluation of the Climate Policy Mix in the German Electricity Sector

6.1 Introduction

The previous chapters have been devoted to the theoretical analysis of a policy mix. The analyses conducted in these chapters have provided substantial advances in the economic understanding of a policy mix compared to pre-existing studies. A clear set of economic rationales for using a policy mix for pollution control has been identified. Moreover, the analysis of selected combinations of climate policies has been approached to reality. These contributions have delivered a lot of new and valuable insight on the welfare effects of a policy mix. The results promise to be of significantly higher relevance for the evaluation and guidance of real-world policy-making than those findings that were available before.

It is the aim of this chapter to interpret the theoretical results and make them fruitful for the evaluation of an actually existing combination of policies: the climate policy mix implemented in the German electricity sector as it has been presented in Chapter 2.. The evaluation is based on a less formal but more applied analysis. It encompasses two tasks: First of all, the key factors which have been found in the theoretical analyses to determine the efficiency (or inefficiency) of the policy mix are specified for the German case. It is verified for each policy combination whether or not a rationale for using a policy mix is actually applicable to a certain policy combination. If no rationale applies, the extent of important drivers of welfare losses revealed in the theoretical analyses is determined. One example may be the specification of the marginal abatement cost curve. This task is performed using existing studies analysing these factors empirically for the German context. Secondly, the framework of the theoretical analyses is broadened. Their advances notwithstanding, analyses in previous chapters have still accounted for a certain degree of abstraction, which has impeded the derivation of policy recommendations for the specific German case. In order to allow for reasonable evaluation results, the characteristics of policy design are taken into account in even more detail in this chapter. This includes the objectives (and a possible multiplicity of objectives), the actual scope as well as the specific rules of each policy. The analysis also reflects on the complex characteristics of the CO_2 allowance market and the electricity market. Moreover, it takes into account the institutional environment into which the policies are embedded. In this respect, it sheds light – where appropriate – on transaction cost and political economy issues.

The analysis provided in this chapter can claim to be by far the most extensive, theoretically guided evaluation of an actually existing climate policy mix – not only in Germany but also worldwide. It does not only address the three policies – emissions trading, feed-in tariffs for renewable electricity generation and taxation – that have been subject to closer investigation in Chapters 4 and 5. The analysis includes all policies that have been introduced in Chapter 2 as part of the German policy mix. The focus is on combinations of the ETS with each of the other policies. The restriction to two policies is for simplicity reasons. It allows understanding and identifying the interactions between these policies in detail. The focus on the ETS is due to the fact that the eventual emissions level of the ETS participants is determined by the ETS cap – as has been shown several times throughout the preceding chapters. Additional policies do not bring about any additional emissions abatement. However, they may have a positive or negative effect on the cost of abatement. These effects are specified in the following for the selected German policy combinations.

The subsequent sections are devoted to evaluating the combination of the EU ETS with the feed-in tariff for renewable electricity generation (Section 6.2), the electricity tax (Section 6.3), the CHP bonus (Section 6.4), the low-interest loans for technology innovation and diffusion (Section 6.5), the eco-design standards (Section 6.6), the energy labelling for household appliances (Section 6.7) and the voluntary agreement (Section 6.8).

6.2 ETS and Feed-in Tariffs for Renewable Electricity Generation

The applied evaluation of the policy mix of the ETS and the feed-in tariffs for electricity generation from renewable energy sources is conducted in three steps: As a first step, it is assumed that both policies are only meant to address two related failures of private governance structures which contribute to climate change: a pollution externality which is reinforced by spillovers from technological learning-by-doing (Section 6.2.1). This assumption corresponds to that made in the theoretical discussion of the policy mix in Chapter 4. The discussion has revealed that in this case a combination of an emissions policy and a technology policy may provide for an efficient outcome. In this section, it is analysed whether the theoretical efficiency properties of the policy mix actually hold when the specific characteristics of German policy design are taken into account. The analysis discusses in more detail the characteristics which have already been addressed in the theoretical chapter – but also goes beyond that analysis. As a second step, the analytical frame is further broadened. It is investigated whether a feed-in tariff may also be a useful complement to an emissions policy when other barriers to technological change, apart from learning spillovers, reinforce the pollution externality (Section 6.2.2). As a third step, the analysis is even further widened. It is analyzed whether a feed-in tariff may be justified as a means to address other policy objectives than climate change, such as resource conservation, security of energy supply or economic development (Section 6.2.3). Based on these three steps, the most important policy recommendations are highlighted and summarized (Section 6.2.4).

6.2.1 Addressing Climate Change: Overcoming a Pollution Externality and Learning Spillovers

6.2.1.1 Theoretical Rationale for Using the Policy Mix

In Section 3.2.4.1, it has been highlighted that a pollution problem – such as climate change – may be attributed to multiple, reinforcing failures of private governance structures: a pollution externality and spillovers related to learning-by-doing. Learning spillovers may hamper the development of abatement technologies. Thus, a pollution externality may be worse and reducing it may be more costly in the presence of learning spillovers. In order to address climate change efficiently, it may then be necessary to implement one policy for each failure.

There is strong evidence that climate change and its impacts can be attributed to a pollution externality (see Section 1.2). Economic actors in Germany, and worldwide, emit GHGs. These emissions are likely to cause global warming and the related damages. These costs are not taken into account by polluters. There is also some indication that spillover effects related to technological learning-by-doing play a role (see Section 4.2.1). Technologies to use renewable energy sources have been found to exhibit significant cost reductions through learning-by-doing. So far, there is hardly any empirical evidence for renewable energy technologies that this knowledge may in fact spill over from the learning actor to other actors. However, substantial spillover effects have been observed for other technologies. Therefore, one may conclude that such effects are also likely to occur for renewable energy technologies. Yet, this cannot be taken for granted. It is particularly unclear to what extent these spillover effects may occur. Since the existence of spillovers is an important requirement for having a policy mix, improving the understanding of these effects is a major challenge for future research.

It has been pointed out in Section 3.3.1 and more extensively in Chapter 4 that, if a pollution externality and learning spillovers are both existent, the ETS and the feed-in tariff for renewable electricity generation may be complementary policies. The ETS aims at reducing GHG emissions and correcting the pollution externality. The feed-in tariff can be understood as a means to promote technological development and to overcome spillovers related to learning-by-doing in the industry producing renewable energy technologies. At this point, it is worth emphasizing again that the presence of only internal (or private) learning-by-doing without spillovers does not provide a rationale for having a corrective policy. The analysis in Section 4.3.3 revealed that, in this case, a feed-in tariff would result in an inefficiently high investment in renewable energy technologies. This finding is in contrast to some authors' statements that the promotion of renewable energy technologies in Germany can be justified *per se* by learning curve effects (see. e.g., Cames et al. 2001, p. 133; Diekmann and Kemfert 2005, p. 445).

However, the presence of multiple reinforcing failures of private governance structures is a necessary but not a sufficient condition for implementing a policy mix. As outlined in Section 3.2.4.1, a policy mix should only be implemented if the difference of the net value of reducing an additional unit of a failure and the transaction costs of implementing a corresponding policy to bring this reduction about is positive for each failure.[60] Moreover, the difference must be larger for regulation than for any private governance structure. A comprehensive discussion – or even quantification – of the net value of reducing a failure and the related transaction costs for regulation or alternative private governance structures would be challenging and go beyond the scope of this book. However, it is worth discussing certain aspects of the German policy mix design which may have an effect on the net value of reducing a failure as well as transaction costs. The theoretically suggested optimal policy mix includes a policy which sets a fixed emissions price and an output subsidy which is paid to the producers of renewable energy technologies (see Section 4.2.2). The German policy mix of the ETS and the feed-in tariff varies from the theoretical suggestion in several respects. The implications of these deviations with respect to efficiency will be discussed in the remainder of this section. The discussion follows up on the theoretical analysis of the policy mix in Chapter 4. It elaborates on the issues raised in more detail. Moreover, it also sheds light on debates which have not been addressed so far.

6.2.1.2 Issue 1: Recipients of the Feed-in Tariff

In Germany, operators of renewable energy technologies receive a subsidy in the form of a feed-in tariff per unit of electricity generated. A feed-in tariff would be an efficient means to overcome learning spillovers if operators of renewable energy plants also produced the necessary technologies and experienced the spillovers themselves. However, the renewables sector in Germany and elsewhere typically exhibits a vertically disintegrated industry structure. For

[60] It is worth recalling that the net value of reducing an additional unit of a failure of private governance structures is the balance of the corresponding welfare gains and the related costs. In this concept, costs refer, for example, to abatement costs and exclude transaction costs.

example, some firms produce wind turbines while other actors buy and operate them to produce electricity. Learning effects and spillovers arise mainly with respect to the production of these technologies. Consequently, producers rather than operators should receive the output subsidy (Bläsi and Requate 2007).

However, subsidizing the producers of renewable energy technologies would violate the rules of the World Trade Organization (Bläsi and Requate 2007, p. 12). Technologies, such as wind turbines or solar cells, are traded internationally. The subsidy might then put German producers at an undue advantage. If subsidizing producers of renewable energy technologies is not a feasible option, supporting operators of these installations may be a second-best solution. In fact, output produced by operators, i.e. electricity, is a proxy for output produced by producers, i.e. renewable energy installations. A higher production of electricity from renewable energy sources can usually only be realized by a higher production and installation of renewable energy technologies. In this sense, promoting electricity generation also fosters the production of renewable energy technologies. Yet, the quantity of electricity generated does not only depend on the technology employed. It may also be attributed to other factors, such as weather and site characteristics. Therefore, a feed-in tariff paid to operators cannot perfectly mimic the incentives of an output subsidy to the producers. It may result in inefficient distortions. Under certain conditions, a feed-in tariff may reward operators for an increase in electricity although they have not invested in new technologies.

If the feed-in tariff was only meant to correct learning spillovers, it should be differentiated, e.g., with respect to weather and site conditions. Operators facing adverse conditions should receive a higher tariff. The differentiation would result in all operators having the same incentive to invest in a technology. However, differentiating the feed-in tariff in such a manner would bring about high transaction costs for the regulator. More importantly, it would impair the efficiency in correcting the pollution externality related to climate change. Renewable energy sources are one means to abate GHG emissions from electricity generation. In this respect, the goal is to minimize abatement costs with renewable electricity generation. This implies that operators should be rewarded for generating the highest possible amount of electricity with one unit of a technology – e.g., by taking advantage of favourable weather and site conditions. From this point of view, for example, it would be more desirable to set up wind turbines along the shore than inland. This provides a strong argument against differentiating the subsidy with respect to weather or site conditions.[61] This reveals a first trade-off between reducing a pollution externality and correcting learning spillovers (see Section 6.2.1.8 for a more in-depth discussion of how a feed-in tariff affects the efficiency of GHG abatement). Taking these findings into account, a feed-in tariff which is paid to operators of renewable energy technology and which is uniform for certain types of technology may be the best possible solution. Such a feed-in tariff is unlikely to realize the optimal net value from correcting the negative effects of learning spillovers. Nevertheless, it may be the superior approach when institutional constraints, transaction costs and trade-offs with respect to the efficiency of GHG abatement are taken into account.

6.2.1.3 Issue 2: Scope of the Feed-in Tariff

The Scientific Advisory Council of the Federal Ministry of Economics criticizes that support under the German feed-in tariff is restricted to a selection of certain technologies only (BMWA 2004, p. 16). The implicit statement is that this selection has emerged from political negotiations and is unlikely to be optimal. This argument may be correct in many instances. Nevertheless, it has to be highlighted that the feed-in tariff covers the entire range of renewable energy sources, from hydropower and wind energy to biomass and solar radiation. Moreover, the basic feed-in tariff may depend on the capacity of a plant in some case – but it is usually independent of the specific technology used to generate electricity from a specific renewable energy source. Thus, it leaves it to the investor to choose the appropriate technology. The feed-in tariff sets incentives for the diffusion of a broad variety of technologies. In this respect, the feed-in tariff can be assumed to provide for a high net value of reducing learning spillovers in the field of renewable energy technologies.

The Scientific Advisory Council is right in pointing out that learning-by-doing – and related spillovers – may not only occur with renewable energy technologies (BMWA 2004, p. 16). However, this finding does not provide an argument against promoting the diffusion of renewable energy sources. It rather provides a rationale for promoting the development of other technologies as well if it exhibits learning spillovers. Regarding the abatement of GHG emissions from electricity generation, promising technologies may also include highly efficient gas and steam power plants, carbon capture and sequestration or nuclear fusion. Promoting these technologies may also reduce abatement costs in the long term and thus contribute to correcting the pollution externality more efficiently. Thus, these technologies may contribute to increasing the net value of internalization under the ETS.

[61] Moreover, a feed-in tariff differentiated by weather and site conditions would probably not be legally feasible.

6.2.1.4 Issue 3: Differentiation of the Feed-in Tariff with respect to Technologies

The German feed-in tariff is differentiated across renewable energy sources. Moreover, apart from the basic feed-in tariff, operators receive additional bonuses for certain technologies in some cases. Frondel and Schmidt (2006, p. 10) strongly criticize this differentiation. They point out that differentiated feed-in tariffs reduce the incentive to innovate and to bring down the costs of renewable energy technologies. They argue in favour of a feed-in tariff which is uniform across all renewable energy sources and technologies. In this case, the market would decide which renewable energy technology is able to compete with fossil energy technologies in the long term.

Frondel and Schmidt's reasoning is somewhat peculiar. The same argument could be put forward to dismiss the promotion of renewable electricity generation in general. One could argue that the incentive for innovation is highest when renewable generation only faces the common market price of electricity – just as fossil generation. The market would then decide about the energy technology to prevail in the long term. However, the main point to be put forward in favour of feed-in tariffs is not the promotion of innovation but the support for the adoption and diffusion of new technologies. More importantly, regulation is needed because technology markets are not working properly. Imperfections may vary between markets for fossil fuel technologies and renewable energy technologies – but also between markets for different types of renewable energy technologies. In this case, the differentiation of the feed-in tariff can be justified theoretically. The analysis in Section 4.3.4 has revealed three factors which should determine the level of a feed-in tariff – and which may well differ across renewable energy sources and technologies. The level of the optimal feed-in tariff depends on the technology's maturity (i.e. the marginal cost reduction from learning), the corresponding spillover rate as well as the number of adopters. Therefore, it may be efficient to grant a higher feed-in tariff to a relatively immature technology, such as photovolatics, than to relatively mature technologies, such as hydropower or wind energy. In this respect, the differentiation of the feed-in tariff may contribute to increasing the net value from reducing learning spillovers. It may be questioned, though, to what extent spillover effects actually occur and vary across renewable energy technologies. Consequently, the level and differentiation of the German feed-in tariff system should be revised critically and continuously. To the German regulator's credit, it should be emphasized that this revision has been undertaken every four years so far.

In fact, the differentiation may impair the efficiency of abating GHG emissions. In order to minimize abatement costs with renewable energy technologies, operators with the lowest generation cost should have the highest incentive to produce electricity. Renewable electricity should be generated using hydropower or wind energy rather than solar radiation. A differentiated tariff does not provide this incentive. Rather a uniform feed-in tariff would be needed – as demanded by Frondel and Schmidt (2006, p. 10). Thus, the differentiation of the feed-in tariff is another example where the reduction of spillovers may be (partly) traded off by an inefficiency in emission abatement. However, a judgement against the differentiation cannot be made on a general level. Rather it is necessary to carefully balance the net value of reducing spillovers and the net value of internalizing the pollution externality.

6.2.1.5 Issue 4: Degression of the Feed-in Tariff

Feed-in tariffs in Germany are subject to an annual degression of up to ten percent. Frondel and Schmidt (2006, p. 10) highlight that this continuous reduction may theoretically stimulate innovation. However, they find that, in reality, the feed-in tariff rather creates incentives to install existent technologies as soon as possible. Regarding their analysis, it is worth mentioning again that the feed-in tariff should be understood as a means to promote the diffusion, not necessarily the innovation of technologies. Taking this reasoning into account, promoting the increased installation of already existent technologies should exactly be the goal of the feed-in tariff. Thereby, output levels of technology production which are inefficiently low due to learning spillovers are raised to higher level. The degression of the tariff can be justified using the model employed in Chapter 4. As cumulative output increases, technologies become more mature and the marginal benefits from learning decline. Accordingly, the necessary output subsidy should also be reduced continuously. It is also reasonable to differentiate the rate of degression across technologies. Relatively immature technologies experience learning effects in the steep part of the learning curve. In contrast, production costs of relatively mature technologies are already located in the flat part of the learning curve. Consequently, the same increase of cumulative output will result in a stronger learning effect for immature than mature technologies. For this reason, rates of degression which are higher for photovoltaics than for hydropower can be justified economically. Whether or not the degression and its differentiation have been chosen appropriately may be debated. Nevertheless, a differentiated degression of feed-in tariffs is necessary to provide for a high net value of reducing learning spillovers.

6.2.1.6 Issue 5: Feed-in Tariff Irrespective of Electricity Price

The German feed-in tariff scheme is designed such that producers of renewable electricity receive a fixed remuneration per unit of electricity. Thus, operators are not granted an output subsidy in addition to the market price – but instead of the market price. The analysis in Section 4.3.4

revealed that the regulator can theoretically determine a first-best feed-in tariff. Notably, the level of the optimal tariff should depend on the electricity price. Electricity prices may be very volatile, though. Monthly averages of spot market prices at the EEX in Leipzig varied between 50 and 90 Euro per MWh for base-load electricity and 70 and 120 Euro per MWh for peak-load electricity in 2008 (RWE 2008, p. 45). Thus, the regulator would be required to adapt the feed-in tariffs continuously as the electricity price changes. Of course, such dynamic adaptation process would result in significant transaction costs to be incurred by the regulator. This may be one reason why such a system has not been implemented in Germany. The feed-in tariff paid to operators is irrespective of the currently prevailing electricity price. Without continuous adaptation of the tariff, however, the net subsidy paid to the operator, i.e. the feed-in tariff less the prevailing electricity price, is unlikely to be optimal from the point of view of efficiency. It may be either too high when electricity prices are low – or too low when electricity prices are high. As a consequence, the incentive to install and produce renewable energy technologies may be either too high or too low. Both effects may bring down the net value of overcoming learning spillovers. With a fixed feed-in tariff, the regulator thus has to choose between high transaction costs (when the tariff is adapted continuously) or a suboptimal net value of reducing the spillover (when the tariff is not adapted continuously).

The analysis in Section 4.3.3 highlighted that an output subsidy which is paid in addition to the electricity price could be determined efficiently irrespectively of the prevailing market price. Thus, it would not have to be adapted continuously as electricity prices change. The total remuneration paid to producer of renewable electricity then has two components: the fixed output subsidy depending on the spillover effect and the variable electricity price driven by the electricity market. This approach can provide the same net value of reducing learning spillovers as a fixed feed-in tariff that is continuously adapted over time – but at significantly reduced transaction costs.

An output subsidy paid in addition to the electricity price had been implemented in Germany before the EEG was introduced in 2000. The approach was replaced because it was found not to set sufficient incentives for technology adoption when the liberalization of the electricity market became effective. As wholesale electricity prices declined in a liberalized market, the overall remuneration for renewable electricity generation was also reduced (Diekmann and Kemfert 2005, p. 442). The above discussion reveals, however, that this development does not necessarily call for fixed feed-in tariff which is irrespective of the electricity price. If the value of electricity declines, a reduction in the overall remuneration for renewable electricity is desirable on efficiency grounds. If the eventual level of remuneration is found to be too low to overcome learning spillovers, an appropriate solution would be to raise the output subsidy – but not to replace it by a fixed feed-in tariff. Admittedly, the decline of electricity prices was not the only rationale for implementing fixed feed-in tariffs. Further reasons will be discussed in Section 6.2.2 below.

6.2.1.7 Issue 6: Funding of the Feed-in Tariff Endogenous to the Electricity Sector

The German feed-in tariff system has been designed such that it is funded by a uniform add-on to the electricity price. In this respect, feed-in tariffs differ from conventional subsidies which are paid by the government from general tax revenues. As demonstrated in Section 4.3.4, the add-on has an impact on the optimal level of the emissions policy implemented in parallel. The add-on can be interpreted as an output tax on electricity. In this respect, the analysis of combining an emissions policy and a feed-in tariff exhibits similarities to that of an emissions trading and an tax conducted in Chapter 5. The add-on increases the output price of electricity. As a consequence, electricity generation as well as related emissions are reduced. An inefficient distortion of emission reduction arises since the output tax does not directly address emissions. The reduction will be relatively too large for costly but low-emission generation, based on natural gas, for example, and too small for cheap but emission-intensive generation, using coal, for example. These effects have three implications for the optimal design of the emissions policy:

Firstly, the optimal emissions price has to be below the Pigovian level. This is because some emission reduction is already brought about by the add-on. This requirement should not pose too much of a problem with respect to the current design of the ETS, though. The Pigovian emissions price can be roughly assumed to be around 70 Euro per ton of CO_2. This level corresponds to the best estimates for the marginal damage of one ton of CO_2 emitted (Downing et al. 2005).[62] Since 2005, emission allowances prices for the second commitment period under the ETS (2008 to 2012) have ranged between 10 and 30 Euro per ton of CO_2 (see Section 2.3.1). Thus, the existing emissions price is significantly lower than the Pigovian price. Moreover, emissions trading is characterized by an advantage compared to an emissions tax which appears to be useful in this respect: Since the add-on eventually reduces emissions from the electricity sector, it will also bring down the allowance price to a certain extent. The overall effect the feed-in tariff system including the add-on has on the allowance price will be discussed comprehensively in the subsequent Section 6.2.1.8.

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[62] The estimation of marginal damages from CO_2 emissions is subject to tremendous uncertainty. Downing et al. (2005) find that estimates may range from 14 Euro per ton to 284 Euro per ton of CO_2. The IPCC (2007d, p. 17) points out that values may run from US$-10 per ton of CO_2 up to US$95 per ton of CO_2.

Secondly, the emissions price has to be adapted as the add-on changes. Changes in the add-on are attributed to changes in the overall volume of feed-in tariffs paid. The analysis in Section 4.3.4 has highlighted three factors driving this volume. First of all, feed-in tariffs should decrease as technologies become more mature and the marginal benefits of learning declines. Secondly, feed-in tariffs have to be increased if the number adopters, and thus the level of spillovers, increases. Thirdly, the add-on has to be raised as the share of renewable electricity in total electricity increases. In recent years, the add-on has increased from 0.2 Cent per kWh in 2000 to 1.2 Cent per kWh in 2008 (see Section 2.3.2). As the add-on rises, the emissions price has to be reduced continuously. Under an emissions tax, this requirement may result in additional transaction costs for the regulator. Modifying the emissions policy may be less cumbersome under the ETS, though. An increase of the add-on automatically results in a decline of the allowance price (see again Section 6.2.1.8 below for a more comprehensive discussion of allowance price effects of the feed-in tariff).

Thirdly, the add-on calls for a differentiation of the emissions price. This differentiation is necessary since the add-on implicitly causes a heterogeneous taxation of emissions. The implicit tax rate is higher for low-carbon fuels, such as natural gas, and lower for emission-intensive fuels such as coal (for an extensive explanation, see Section 4.3.4). To compensate for the resulting distortion, the emissions price set out by the emissions policy should be lower for low-emission generation than for high-emission generation. This can be illustrated using a simple numerical example. It is assumed that the Pigovian emissions price is 70 Euro per ton of CO_2. The add-on amounts to 1.2 Cent per kWh – as it was in Germany in 2008. Emissions rates of lignite, hard coal and natural gas combusted in modern gas and steam power plants amount to 873, 780 and 386 grams per kWh, respectively (Krewitt and Schlomann 2006, p. 35).[63] The optimal emissions price for lignite generation can be calculated easily using equation (4.29) in Section 4.3.4:

$$70\ Euro/t - \frac{0.012\ Euro/kWh}{0.000873\ t/kWh} \approx 56\ Euro/t$$

The corresponding prices for hard coal and natural gas generation are roughly 55 Euro per ton of CO_2 and 39 Euro per ton of CO_2, respectively. The difference between these prices and the Pigovian emissions price reflects the emissions tax that is implicit to the add-on. The theoretical

[63] Emissions refer to life-cycle emissions with the respective fuel and technology. The assumed effectiveness of the gas and steam power plant is 48 percent, 46 percent and 58 percent for lignite, hard coal and natural gas, respectively (Krewitt and Schlomann 2006, p. 35).

analysis in Chapter 4 assumed the emissions rate to be fixed with the fuel. In reality, however, the emissions rate will also depend on the technology employed. Emissions from conventional steam power plants using lignite are higher than those from a modern gas and steam power plant. For the former technology, emissions amount to 1054 gram per kWh (Krewitt and Schlomann 2006, p. 35). Consequently, the emissions price also has to be differentiated with respect to the technologies employed. With lignite-fired generation, it amounts to 56 Euro per ton of CO_2 for emissions from gas and steam power plants and 59 Euro per ton of CO_2 for emissions from conventional steam power plants.

Differentiating the emissions price may be a relatively easy task for the regulator when an emissions tax is in place. It is far more challenging, however, with an emission trading scheme. Due to its very nature, trading schemes result in a uniform emissions price. From a transaction cost perspective, this mechanism is desirable. The regulator can attain a certain emissions level cost-effectively without having information on the marginal abatement costs of the polluters.[64] In order to provide for a differentiation of the emissions price, the emissions market would have to be split up into several markets. This challenge is quite similar to that faced by US regulators when they tried to establish an allowance market for air pollutants (see Section 3.4.1). To account for the heterogeneity of pollution damages, they considered separating the emissions market spatially. Regarding the CO_2 emissions market, it would be necessary to separate the market with respect to fuels and technologies used for electricity generation in the presence of the add-on. Another market would be needed for non-electricity emitters of CO_2. The discussion of the US example revealed, however, that market separation is very challenging for the regulator. He has to determine the amount of allowances allocated to each market as well as possible trading ratios for trades across markets. In order to solve this problem cost-effectively, the regulator needs information on marginal abatement costs. Consequently, transaction costs under such a scheme with separate emissions markets are much higher than under a single emissions trading scheme. The increase in the net value of internalizing the pollution externality under a system with several markets is counteracted by an increase in transaction costs. Market separation is therefore likely to be ruled out as a policy option.

The extent of the inefficiency in the absence of differentiated emissions prices is difficult to determine. First of all, it is worth mentioning that not all electricity consumption is subject to the

64 The regulator would need information on marginal abatement costs, though, if he wanted to determine the emissions cap efficiently.

full add-on to the electricity price. As pointed out in Section 2.3.2, some consumers are eligible for a reduced add-on of 0.05 Cent per kWh. In 2007, roughly 15 percent of the total electricity consumed in Germany qualified for this reduction (BMU 2008b, p. 29). Moreover, it has to be taken into account that the add-on accounts only for a minor share in the entire retail price of electricity. Next to the add-on, the retail price reflects the wholesale electricity price, the grid system usage fee, the value-added tax (household customers only) and several other policy-induced price components. The share of the add-on in the retail price is about five percent for household customers. The share is higher for industry customers – eight to eleven percent – since these do not have to pay the value-added tax on electricity. For industry customers that are eligible for the reduced add-on, however, less then one percent of the electricity bill are attributed to the add-on (BMU 2008a, p. 60). A twenty percent rise of the add-on – as it occurred from 2007 to 2008 – therefore only results in a one percent increase of electricity prices for household customers. Other effects of the feed-in tariff system on the electricity price will be discussed in the subsequent Section 6.2.1.8. It may be concluded, though, that the sole effect of the add-on on electricity demand may be small. Consequently, the reduction of emissions and related distortions should be limited as well. Nevertheless, it has to be emphasized that funding the feed-in tariff by an add-on to the electricity price will either decrease the net value of internalizing the pollution externality or bring about high transaction costs.

If the add-on is found to be inefficient, it is worth reviewing possible alternatives. Instead of the add-on, one could also consider funding feed-in tariffs by general tax revenues. In this respect, light has to be shed on two issues: First of all, taxes – imposed, for example, on labour income or capital – reduce the consumers' and producers' rents in a general equilibrium setting. These so-called deadweight losses (or excess burden) of taxation have to be compared with those losses resulting in the electricity market due to the add-on. In fact, general equilibrium distortions of taxes can be substantial (see, e.g., Browning 1976; 1987). This compares with relatively modest distortions in the electricity sector due to the add-on. One may therefore assume that deadweight losses would probably exceed the limited efficiency losses from an add-on. Secondly, funding feed-in tariffs by general tax revenues would convert them into ordinary subsidies. Due to the revenue-neutral design implemented so far, feed-in tariffs have not been considered as state aid under European law (European Court 2001). Funding feed-in tariffs by tax revenues could result in a review process to be enacted under European state aid law. One result of this process could be that feed-in tariffs represent an undue subsidy distorting competition in the EU. These drawbacks provide arguments against replacing the existing add-on.

6.2.1.8 Issue 7: Partial Overlap of ETS and Feed-in Tariff

In the theoretical analysis of the policy mix in Chapter 4, it has been assumed that the emissions price is fixed – i.e. an emissions tax is in place. Moreover, all actors subject to the emissions policy were also paid the feed-in tariff. Thus, the emissions policy was restricted to the electricity sector. Consequently, this analysis did not reveal possible implications for efficiency arising when the emissions price is variable – as for emissions trading – and the emissions policy and the feed-in tariff do not overlap perfectly. The resulting interactions in the policy mix are briefly discussed in the following. They are quite similar to those identified for the combination of an emissions trading scheme and a heterogeneous emissions tax in Section 5.3.5.

Interactions were described first for the ETS and the German feed-in tariff by the Scientific Advisory Council of the Federal Ministry of Economics (BMWA 2004, p. 8). Renewable electricity generation promoted by the feed-in tariff substitutes fossil-fueled generation. The electricity sector emits less and demands fewer allowances (or offers more allowances for sale). The allowance price declines. Non-electricity sectors in Germany as well as sectors in other European countries participating in the ETS benefit from a reduced price, buy allowances and emit more. Thus, the implementation of a feed-in tariff on top of the ETS results in emissions shifted from the German electricity sector to other ETS sectors. The overall emissions level remains unaffected by the feed-in tariff. However, costs of attaining this level increase. The electricity sector abates too much and too costly. Other ETS sectors emit too much and do not employ relatively cheap abatement options. In fact, German electricity consumers, which eventually have to bear abatement costs, subsidize emissions in German industry sectors as well as in other EU Member States. In this respect, the policy mix of the ETS and the feed-in tariffs exhibits a static inefficiency. It reduces the net value of internalizing the pollution externality compared to a single emissions trading scheme. Determining the extent of this inefficiency is a challenging task. It crucially depends on the characteristics of the CO_2 allowances market and the electricity market as well as interactions between these markets. The most important drivers of the inefficiency are discussed in the following.[65]

[65] Interactions between an emissions trading scheme and a support scheme may be even more complex when a quota system for renewable electricity is implemented instead of feed-in tariffs. In this case, the remuneration for renewable electricity is not fixed. The overall efficiency of the policy mix then depends on an even more complex set of interactions between the CO_2 allowances market, the green certificates market and the electricity market. Several authors shed light on these interactions (Boots et al. 2001; Morthorst 2001; Baron and Serret 2002; Jensen and Skytte

First of all, the inefficiency of the policy mix is driven by the quantity of renewable electricity generated, the corresponding amount of CO_2 emissions abated as well as the abatement costs. In 2007, 67,010 gigawatt-hours of electricity from renewable energy sources were fed into the grid and remunerated at the feed-in tariffs (BMU 2008b, p. 29). The corresponding reduction of CO_2 emissions in the electricity sector depends on the fuels and technologies that are substituted by renewable generation (Diekmann and Horn 2008, p. 15). This substitution process is determined by the characteristics of electricity supply in the electricity market. Electricity supply is represented by the so-called merit order curve. It ranks available power plants in order of their marginal costs of generation (including the allowance costs induced by the ETS). According to this order, plants generate electricity until electricity demand is satisfied, i.e. the cheapest plants produce first. Electricity generation from renewable energy sources has low marginal costs and crowds expensive marginal fossil-fueled power plants out of the market. This crowding-out effect depends on the total amount of renewable electricity as well as the demand for electricity, which varies significantly during a day (Bode and Groscurth 2006, pp. 12-13). Emission abatement from renewable electricity generation corresponds to the emissions the crowded-out power plants would have produced. The Federal Ministry of the Environment estimates that some 57 million tons of CO_2 were abated by renewable electricity generation promoted by the feed-in tariff in 2007 (BMU 2008b, p. 20). This quantity compared with the additional cost of renewable electricity generation, i.e. feed-in tariffs paid net of the average cost of non-renewable generation. It amounted to 4.3 billon Euro in 2007 (BMU 2008b, p. 30). In addition, an estimated cost of 0.3 to 0.6 billion Euro was incurred by society due to higher controlling needs with renewable electricity fed into the grid (BMU 2008b, p. 34). These figures would translate into marginal CO_2 abatement costs with renewable energies of 86 Euro per ton of CO_2. However, calculating marginal abatement costs on the basis of feed-in tariffs paid is somewhat flawed. Tariffs may include windfall profits to operators which have costs below the tariff level. This consideration implies that actual marginal abatement costs may be even lower. A study by the Federal Ministry of the Environment estimates that average abatement costs with renewable electricity are 92 Euro per ton of CO_2 – but only 66 Euro per ton of CO_2 when photovoltaics is excluded (BMU 2008a, p. 66). Another study yields higher costs – between 95 and 168 Euro per ton of CO_2 – for abatement with wind energy in Germany (Dena 2005, p. 309). Their uncertainty and range notwithstanding, these values are generally higher than the prevailing prices in the ETS

2002; Hindsberger et al. 2003; Jensen and Skytte 2003; Morthorst 2003; Azuma-Dicke et al. 2004; Unger and Ahlgren 2005).

allowances market. This confirms that the promotion of renewable energies by the feed-in tariff in fact increases the abatement costs under the ETS. A McKinsey (2007, pp. 32-35) study reveals that cheap abatement options are available with fossil-fueled electricity generation as well as industry emissions. Efficiency improvements in lignite power plants may reduce CO_2 emissions at 15 Euro per ton. The implementation of CHP technologies for hard coal power plants may only cost 20 Euro per ton of CO_2 abated. A shift from hard coal to gas is supposed to cost 28 Euro per ton of CO_2. In the German industry sector, some 40 million tons of CO_2 can be abated by improving energy efficiency at costs below 20 Euro per ton. If the feed-in tariff drives down the allowances price, some of these abatement options may be foregone although they would be cheaper than the use of renewable energy sources for electricity generation.

The extent by which the feed-in tariff actually results in an inefficient shift of emissions abatement between ETS sectors also depends on the characteristics of the CO_2 allowance market. In this respect, some of the theoretical findings made in Section 5.3.5.2 regarding interactions between an emissions trading scheme and a heterogeneous emissions tax can be transferred to the case of the policy mix of the ETS and the feed-in tariff. An important driver is the aggregated ("European") marginal abatement cost curve of the ETS participants (Diekmann and Horn 2008, p. 21). It determines the demand curve for allowances – and thus the effect a decreasing allowance demand from the electricity sector has on the allowances price. The flatter the marginal abatement cost curve is – or the more inelastic the demand for allowances is – the larger will be the inefficient distortion of abatement under the ETS resulting from the feed-in tariff. Marginal abatement costs in Europe have mostly been found to be convex (Ellerman and Decaux 1998; Criqui et al. 1999; Klepper and Peterson 2006). So far, overall emissions reductions have been limited. Thus, a change of emissions abatement is likely to occur in the relatively flat section of the marginal abatement cost curve. Based on this insight, one may assume that a shift of emissions abatement may indeed have a significant effect on abatement costs. As revealed in Section 5.3.5.2, the overall inefficiency of the policy mix also depends on the share of the sector subject to overlapping policies in the entire allowance market. With its annual allocation of 253.81 million allowances (see Section 2.3.1), the German energy sector has a share of roughly 12 percent in the European allowances market, which accounts for a total of 2,086.50 million allowances (European Commission 2008g, p. 14). In this respect, the allowance price effect of reducing emissions by 57 million tons of CO_2 in the electricity sector due to the feed-in tariff may be limited. However, policy decisions of other EU Member States are also important. Apart from Germany, many other Member States have also implemented policy schemes to promote electricity generation from renewable energy sources (Ziesing 2007b, p. 301). Therefore, there may indeed by a substantial and inefficient shift of emissions from the electricity sector to other ETS sectors at the European level. However, it also has to be taken into account that allowance prices may also be driven by other political as well as psychological factors (Bode and Groscurth 2006, p. 14).

Finally, the distortions in the allowance market may be influenced by price effects induced by the feed-in tariff. A higher price would reduce electricity demand and production as well as related emissions. Consequently, the inefficient shift of emissions abatement from the electricity sector to other ETS sectors would be deteriorated. In turn, a price decline would ameliorate the inefficient distortion. Observed price effects of the feed-in tariff are ambiguous. Four effects are worth mentioning. Firstly, the add-on funding the feed-in tariff increases retail electricity prices. However, this effect may be limited for industry customers paying the reduced add-on only (see Section 6.2.1.7). Secondly, the promotion of renewable electricity generation is likely to increase the grid system usage fee – another component of the retail electricity price. This is because grid system operators have to adopt costly measures to equalize variations in the supply of renewable electricity generation (see, e.g., Zander and Nailis 2004; Neubarth et al. 2006, pp. 43-44). Moreover, these costs may increase when renewable electricity generation requires an extension of existing grids. Thirdly, the so-called merit-order effect results in a reduction of the electricity price. The wholesale electricity price always equals the marginal generation costs of the marginal power plant, i.e. the last and most expensive plant needed to satisfy electricity demand. As has been described above, the feed-in of renewable electricity with zero marginal costs crowds out the marginal power plant. In other terms, the supply curve is shifted to the right. The new marginal power plant has lower marginal costs than the previous one. The wholesale electricity price – and, thus, the retail electricity price declines. Some studies provide an indication of the merit-order effect. Using an econometric analysis, Neubarth et al. (2006, p. 43) estimate that a 1,000 megawatt increase in installed wind energy capacity results in an average price decline of 1.9 Euro per MWh. Bode and Groscurth (2006, p. 17) apply a model of a hypothetical electricity market. They find a lower reduction of the electricity price of 0.61 Euro per MWh for each 1,000 megawatt increase in installed capacity. Assuming an annual electricity generation from renewable energy sources of 36,714 gigawatt-hours, this would correspond to a 2.40 Euro price reduction per MWh. Taking into account a higher level of renewable electricity generation of 52,200 gigawatt-hours in 2006, Sensfuß and Ragwitz (2007, p. 10) estimate a higher merit order effect of

7.83 Euro per MWh.[66] Fourthly, the electricity price may decline with decreasing allowance prices. It is generally assumed that electricity prices reflect the (opportunity) costs of allowances (Sijm and van Dril 2003, pp. 55-58; Sijm et al. 2005; Sijm et al. 2006; Ziesing 2007b, pp. 305-334). Since the promotion of renewables brings down the allowance price, it may also contribute to declining electricity prices in this respect. Given the numerous factors determining the allowance price, determining this effect quantitatively is tedious. Diekmann and Horn (2008, p. 13) provide the rough estimate that a decline in the allowance price by one Euro per ton of CO_2 results in a decrease of the electricity price of 0.6 Euro per MWh. The overall effect the feed-in tariff has on the electricity price is difficult to assess. Nevertheless, there seems to be some consensus that the effect is negative, i.e. price-decreasing for industry customers eligible for the reduced add-on, and positive for other customers, such as households (Bode and Groscurth 2006, p. 17; Diekmann and Horn 2008, p. 21). Since the share of privileged electricity consumption subject to a lower add-on in total electricity assumption is relatively low (see Section 6.2.1.7), the feed-in tariff can be assumed to reduce electricity demand and to deteriorate the inefficiency of the policy mix. However, this effect also depends on the price elasticity of electricity demand. Industry customers, many of which face a price decrease under the feed-in tariff, are often found to have a relatively high price elasticity of electricity demand (see, e.g., Bjørner et al. 2001; Kamerschen and Porter 2004).[67] In contrast, many of the customers subject to an increased electricity price – particularly households – are characterized by a low elasticity of demand (see, e.g., Taylor 1975; Kamerschen and Porter 2004; Reiss and White 2005; Narayan et al. 2007).[68] Moreover, price increases due to the feed-in tariff may be counteracted by a general price decline in liberalized electricity markets (Sorrell 2003, p. 20). Thus, the price effect of the feed-in tariff as well as its contribution to the inefficiency of the policy mix may be assumed to be limited.

Given the complex interaction within and between the CO_2 allowances market and the electricity market, it is challenging to provide an overall evaluation of the static inefficiency resulting from

[66] These estimates still neglect dynamic adaptations of the composition of power plant facilities in response to public policies such as the ETS or the feed-in tariff. These adaptations may have an effect on the merit order curve as well (Wissen and Nicolosi 2008).

[67] Nevertheless, estimates of the price elasticity of industrial electricity demand still reveal an inelastic demand, i.e. the elasticity is below one.

[68] However, in the long term, household electricity demand – as well as industrial demand – may also turn out to be elastic with respect to prices (for a comparison of short-term and long-term elasticities, see Taylor 1975; Bohi and Zimmerman 1984).

the partial overlap of the ETS and the feed-in tariff. Using a simplified theoretical model, Rathmann (2007, p. 347) estimates that the allowances price was 27 percent lower than it would have been without the feed-in tariff in 2005. This figure would have corresponded to a price reduction by 7 Euro per ton of CO_2. Diekmann and Horn (2008, pp. 21-22) emphasize, however, that this estimate is subject to a high uncertainty, particularly with respect to characteristics of the marginal abatement curve. Kemfert and Diekmann (2009, p. 172) point out that the price effect of the feed-in tariff may have been as low as one Euro – or even zero – in the first ETS trading period from 2005 to 2007. Prices were very volatile and in the end, many allowances were not used at all. Therefore, they doubt that the feed-in tariff actually resulted in inefficient shifts between ETS sectors. Nevertheless, the partial overlap between the ETS and the feed-in tariff may provide for inefficient distortions in the current and future trading periods. The emissions cap has been tightened and market participants have learned about the functioning of the market. Therefore, the allowance price effect of the feed-in tariff may be of more significance.

A simple solution to the inefficiency of the policy mix, which is often suggested, is to reduce the emissions cap of the ETS by the expected emissions reduction due to the feed-in tariff (Böhringer et al. 2005, p. 44; Diekmann and Kemfert 2005, p. 445; Diekmann and Horn 2008, p. 16). The total emissions reduction thus corresponds to the sum of the reductions brought about by the ETS and the feed-in tariff implemented in isolation. In this respect, the reduction of the emissions cap improves the effectiveness of the policy mix. Yet, it will still exhibit the inefficiency described above. The increase in the allowance price resulting from the reduction of the cap compensates the price decrease resulting from the promotion of renewable electricity generation. Therefore, the non-electricity sector abates as much as under a single emissions trading scheme with a higher cap. The reduction of the cap thus provides that no emissions are shifted from the electricity sector to the industry sector. However, this implies in turn that the reduction of the cap is only born by the electricity sector. From an efficiency point of view, it would be superior if at least some of this reduction would be realized by the non-electricity sector as well. The same overall emissions level could be attained at less cost if the electricity sector abated less and the industry sector abated more. Thus, the inefficiency of the policy mix prevails despite the modification of the emissions cap – but at an overall lower emissions level. If the emissions cap is set suboptimally high – as can be assumed for the ETS – this modification is desirable because it allows realizing additional benefits from internalization. Since it has not been implemented in Germany during the first and second commitment period of the ETS, a policy recommendation would be to consider the feed-in tariff effect on emissions when designing future national allocation plans (Diekmann and Horn 2008, pp. 31-33; Kemfert and Diekmann 2009, p. 172). It should be born in mind, however, that this approach will still exhibit inefficient interactions between the ETS and the feed-in tariff.

6.2.2 Addressing Climate Change: Overcoming a Pollution Externality and Other Barriers to Technological Change

So far, it has been assumed that the feed-in tariff is needed to overcome learning spillovers hampering the diffusion of renewable energy technologies. Yet, the development of these technologies may also be suboptimal because of other barriers to technological change. These barriers may contribute to path dependencies in the field of energy technologies. Renewable energy technologies may be locked out. As learning spillovers, other barriers may reinforce the pollution externality. Overcoming these barriers may therefore also contribute to increasing the net value of internalizing the pollution externality. Consequently, the existence of these barriers may provide another rationale for having a technology support policy as part of the climate policy mix. In each case, it is necessary to analyse, however, whether the net value of overcoming the barrier by a certain regulation is positive and superior to other solutions. Particular light has to be shed on the issue whether a feed-in tariff is an efficient means to overcome these barriers. Four barriers merit particular attention and discussion – although the list of possible barriers may be much longer (for an extensive discussion, see Unruh 2000; Unruh 2002; Neuhoff 2005).

First of all, the development of renewable energy technologies may also be hampered by externalities related to research and development (R&D) (see Section 3.2.2.1). It has been discussed that the presence of an R&D externality next to a pollution externality provides a rationale for a policy mix (see Section 3.3.1.2). In this respect it is sometimes argued that feed-in tariffs may also provide a useful means to overcome these failures of private governance structures (Sorrell 2003, p. 24; Sorrell and Sijm 2003, p. 429; Böhringer et al. 2005, p. 42). The actual impact of promoting electricity generation with renewable energy technologies on R&D related to these technologies is unclear. It has already been outlined in Section 6.2.1.2 that the promotion of renewable electricity generation imperfectly mimics the promotion of producing renewable energy technologies. Moreover, the uniform promotion of technology production is only an efficient means to foster R&D when all producers can be assumed to have the same amount of R&D expenditures per unit of output and the same degree of R&D externalities. This is a very unlikely case, though. It will be therefore more efficient to pay producers a direct subsidy per unit of R&D expenditure depending on the extent of the existent R&D (Frondel and Schmidt 2006, p. 10). This subsidy is likely to provide a higher net value of internalizing the R&D externality than an indirect feed-in tariff. Only if an R&D subsidy is not a feasible policy option –

for example, because it would violate WTO or European competition rules – feed-in tariffs should be considered as a second-best alternative.

A second important barrier to the development and employment of renewable energy technologies is the character of fossil-fuel technology investments. Coal mines, oil and gas fields as well as fossil-fueled power plants are large-scale investments which last for decades (Neuhoff 2005, p. 98). Many of these investment costs are sunk. Moreover, many of these investments were realized decades ago and have already amortized. Therefore, the opportunity costs of changing from a fossil-fueled to a low-carbon technology may be high (Grubb et al. 1995, p. 422). Moreover, investments in fossil-fuel technologies can be embedded in an extensive network of existing infrastructures, suppliers relationships and consumer outlets, in which they can benefit from positive network externalities (Grubb 1997, pp. 162-163). The feed-in tariff may be a means to improve the competitive situation of renewable energy technologies compared to fossil-fuel technologies. The net value of overcoming these barriers may be negative in the short run but positive in the long term.

Thirdly, the capital market may induce additional barriers to the innovation and diffusion of renewable energy technologies. Technology-related investments are usually funded by loans. The cost of these loans depends on the uncertainties and risks related to the returns on investment. With respect to energy technologies, risks arise from the lacking knowledge about future prices of inputs to the production of technologies, such as steel, prices of inputs to electricity generation, such as coal or natural gas, as well as electricity prices. These risks and uncertainties are reflected in the risk premium banks raise for energy-related investments in general. In this respect, two issues may put renewable energy technologies at a systematic disadvantage compared to fossil-fuel technologies: Firstly, the risk premium may be higher for renewable energy technologies. This may be due to the uncertainty about technical aspects as well as the future development of the market for relatively young technologies (Isoard and Soria 2001, p. 631). Moreover, renewable technologies may be subject to additional exogenous risks related, for example, to wind speed or sunshine duration. Moreover, the premium may be higher because investors in renewable energy technologies are often small and new market actors which can provide less security for loans than large producers and adopters of fossil-fuel technologies. In addition, they cannot rely on long-lived relationship with banks (Walz 2005, p. 265). Secondly, the relative importance of the risk premium is higher for renewable energy technologies. This is because these technologies are more capital-intensive than fossil-fuel technologies. Production costs depend on investment costs but hardly on variable input costs, such as fuels. Therefore, investors in liberalized electricity markets prefer the least capital-intensive technologies, and investment in renewable technologies is suboptimal (Menanteau et al. 2003, p. 801; Neuhoff 2005, p. 95). Moreover, the transaction costs of risk-management instruments may be relatively high for small-scale renewable energy projects. To overcome these barriers it may therefore be necessary to improve the security of renewable energy investments. In this respect, the current design of the German feed-in tariff has proven to be very helpful. It provides secure revenues from electricity generation – irrespectively of the prevailing electricity price – for a period of twenty years. It reduces the uncertainty surrounding the investment, and probably also the risk premium (Diekmann and Kemfert 2005, p. 446).[69] Implementing a feed-in tariff that is irrespective of the electricity price may thus produce a positive net value from overcoming investment uncertainties. This welfare gain trades off the potential losses such policy design may bring about with respect to the net value of internalizing the pollution externality (see Section 6.2.1.6).

Finally, renewable energy technologies usually operate in an uneven political playing field. Most importantly, the electricity price, at which renewable generation would compete with fossil-fuel generation in the absence of a feed-in tariff, may be below the actual costs of fossil-fuel generation. Conventional energy sources have benefited from direct and indirect subsidies in many cases (Neuhoff 2005, p. 93). In Germany, subsidies to hard coal mining are most noteworthy. They amounted to 2.285 billion Euro in 2006. To a smaller extent, lignite coal production has also received subsidies. Nuclear-based electricity generation benefited directly from R&D subsidies and implicitly from relaxed liability rules in the event of a nuclear accident (UBA 2008c). Moreover, despite the ETS, electricity prices are unlikely to reflect the full external costs of fossil-fuel electricity generation (Isoard and Soria 2001, p. 631). The windfall profits generated by grandfathering allowances under the ETS can also be interpreted as a subsidy – amounting to about 2.5 billion Euro in 2006 – which is granted to large fossil-fuel based electricity generators (UBA 2008c, p. 16). Finally, land development plans traditionally envisage specific zones for industrial development. These tend to favour centralized fossil-fuel electricity generation over numerous disperse renewable energy plants (Neuhoff 2005, p. 96). Feed-in tariffs can be interpreted as a means to overcome this inequality in regulating the energy sector. If the mitigation of climate change is under consideration, however, the first best approach would be to abolish these adverse subsidies. Only if this is not a desirable or feasible option – for example, because subsidies are also meant to address goals of economic development or employment – the implementation of a feed-in tariff would be desirable from an efficiency point of view.

[69] This can also be seen as a major advantage of feed-in tariffs over quotas for renewable electricity (Madlener and Stagl 2005, p. 156).

6.2.3 Addressing Climate Change and Other Policy Objectives

So far it has been assumed that the ETS as well as the feed-in tariff are only meant to address climate change. It has been shown that their combination may improve efficiency when climate change is attributed to two reinforcing failures of private governance structures: a pollution externality and barriers to the development of renewable energy technologies. The unidirectional objective of mitigating climate change efficiently can be assumed for the ETS. The German feed-in tariff, however, is characterized by multiple objectives. Apart from climate change, it also aims at environmental protection in general, improving the security of energy supply and fostering economic development (see Section 2.3.2). This multiplicity of policy objectives should be considered as part of a comprehensive discussion of the policy mix of the ETS and the feed-in tariff. Under certain conditions, welfare losses with respect to addressing climate change may be traded off by attaining other goals. In the following, the most important contributions of the feed-in tariff to achieving these other policy objectives are discussed.

The substitution of fossil-fuels by renewable energy sources for electricity generation may help to reduce externalities of air pollution, such as negative health effects. The Federal Ministry of the Environment estimates that the feed-in tariff has contributed significantly to reducing air pollution. In 2007, some 45,000 tons of SO_2 emissions and roughly 13,000 tons of NO_X emissions were abated by renewable electricity generation (BMU 2008b, p. 18). Moreover, the promotion of renewable energy sources contributes to the conservation of fossil fuels. It thus also reduces externalities of excessive resource exploitation. 39.1 million tons of lignite, 14.2 million tons of hard coal and 8.78 billion cubic metres of natural gas were saved in 2007 by using renewable energy sources (BMU 2008b, p. 25).[70] On efficiency grounds, one may argue that externalities of air pollution and resource exploitation can be reduced more efficiently by increasing the price for fossil fuels correspondingly. As for externalities of CO_2 emissions, it can be assumed, though, that these externalities are also reinforced by barriers to technological development. Therefore, a feed-in tariff to promote renewable energy technologies would be even necessary if the price of fossil resources reflected all external costs. It is worth highlighting, however, that the actual effects of the feed-in tariff on air pollution control and resource conservation may be smaller under the policy mix than the estimates provided by the Ministry of the Environment. Just as the feed-in tariff results in a shift of emissions from the electricity sector to other sectors, a similar shift may also arise with respect to air pollution and resource

[70] These figures refer to electricity as well as heat generation from renewable energy sources.

use. Benefits of the feed-in tariff may be offset by opposite effects in other ETS sectors. ETS participants outside the electricity sector may take advantage of the allowance price reduction induced by the feed-in tariff and produce more. Consequently, their emissions of air pollutants and resource consumption are also likely to increase. Even though the net effect on CO_2 emissions is zero, this is not necessarily the case for air pollution and resource consumption. The relationship between CO_2 emissions and air pollution and resource consumption may well differ across as well as within sectors. The overall effect on air pollution and resource use thus depends on the characteristics of the production processes which are extended at the cost of reduced electricity generation. The effect cannot be determined on a qualitative basis. It may be positive, negative or zero.

Moreover, the promotion of renewable energy sources reduces externalities of fossil fuels with respect to the security of energy supply. Some fossil fuels, particularly oil and natural gas, are imported from countries with an instable political environment. Moreover, the overall volume of fossil-fuel resources is limited. If electricity generation is based primarily on fossil fuels, the interruption or termination of their delivery may produce significant costs to society. The use of renewable energy sources can improve the security of energy supply. They increase the variety of energy sources and are domestic resources. Due to using renewable energy sources, fossil fuel imports in the amount of 1.0 billion Euro were saved in 2007. Thereby, renewable electricity generation reduces the vulnerability of the energy system to exogenous shocks, such as abrupt interruptions of fuel delivery. What is more, renewable energy sources are available at a virtually unlimited time scale (Sorrell and Sijm 2003, p. 430; Neuhoff 2005, p. 94). However, renewable electricity generation also exhibits an important drawback with respect to supply security. Particularly, wind energy and solar radiation cannot generate electricity on a continuous basis (Frondel and Schmidt 2006, p. 2). In general, it could be argued again that it is more efficient to internalize the externalities of fossil fuels with respect to supply security by raising the resource price correspondingly. Nevertheless, a feed-in tariff may be a useful complement to such price policy if the externalities of supply security are reinforced by barriers to technological development.

Finally, the promotion of electricity generation from renewable energy sources has contributed to economic development. It has fostered the growth of industry branches producing renewable energy plants, such as wind turbines of photovoltaic cells. Their products are not only sold in Germany but marketed worldwide (Kemfert and Diekmann 2009, p. 169). The Federal Ministry of the Environment estimates a sales volume of 25.5 billion Euro for renewable energy technologies produced in Germany in 2007. This figure corresponded to an increase by 155 percent from 2003 to 2007 (BMU 2008b, p. 27). Moreover, the Ministry highlights that the renewable energy industry had roughly 250,000 employees in 2007. This implied a 55 percent increase since 2004. About 60 percent of this employment effect can be attributed to the promotion of renewable electricity generation by the feed-in tariff. The Ministry therefore concludes that the overall effect of the feed-in tariff on economic development is positive. This conclusion is drawn into question by some economists, though. Frondel and Schmidt (2006, pp. 6-8) point out that the Ministry's estimates do not consider possibly adverse effects of the feed-in tariff in other industry sectors. The tariff is funded by higher electricity prices. This price increase results in a reduction of industry investments and household consumption. Consequently, the feed-in tariff may in fact have negative effects on employment in sectors which do not benefit from the subsidy. Frondel and Schmidt conclude that the overall effect the feed-in tariff has on employment may therefore be zero or even negative. Their results are confirmed by studies which also find zero (Henrich et al. 2004), or negative employment effects (BEI 2003; EWI et al. 2004). The efficiency of the feed-in tariff with respect to promoting economic development may therefore be debatable.

6.2.4 Policy Recommendation

The previous evaluation of the policy mix of the ETS and the German feed-in tariff has revealed a complex set of interactions between both policies. Some of these interactions may improve the efficiency of the policy mix while others may reduce it. From an economic point of view, an unambiguous judgement of the policy mix cannot be derived on a general level. The pros and cons of the policy mix have to be balanced carefully.

The major argument in favour of having a policy mix of the ETS and the feed-in tariff is the presence of multiple, reinforcing failures of private governance structures. A pollution externality is deteriorated by barriers to the development of environmentally friendly technologies. Learning spillovers but also other barriers such as R&D externalities, capital market restrictions, an uneven political playing field or sunk investments in fossil fuel technologies may result in suboptimal investment in renewable energy technologies. In order to correct these failures efficiently, at least two policies are necessary: The ETS is the main policy to overcome pollution externalities related to climate change. It is likely to produce a positive net value of internalization. This value is higher, though, when parallel barriers to technological development are also reduced. For this purpose, the feed-in tariff should be implemented. It produces a positive net value of overcoming barriers to technological change. It is thus worth emphasizing that the tariff should be understood first of all as a technology policy rather than as an emissions policy. In a long-term dynamic perspective, however, the promotion of renewable electricity generation may then also help to increase the net value of internalizing the pollution externality as barriers to technological development are overcome. Moreover, the feed-in tariff may be possibly employed to address other policy objectives next to mitigating climate change, such as controlling air pollution, resource conservation and improving the security of energy supply.

In a short term static perspective, however, the implementation of the feed-in tariff in addition to the ETS may actually reduce the net value of internalizing the pollution externality. Compared to a single ETS, the policy mix provides the same level of emissions abatement. However, the costs of attaining the abatement level increase due to interactions in the policy mix. Most notably, the partial overlap of the ETS and the feed-in tariff results in an inefficient shift of emissions abatement from the non-electricity sectors to the electricity sector. Costly emissions abatement by renewable electricity generation substitutes cheaper abatement measures available for fossil-fuel electricity generation and other industry sectors. The resulting welfare loss is deteriorated by other characteristics of the feed-in tariff. Since the feed-in tariff is paid to technology operators rather than producers, irrespective of the prevailing electricity price and differentiated by technologies, the costs of abating emissions with renewable electricity generation are not minimized. What is more, the add-on to the electricity price funding the feed-in tariff additionally distorts the emissions abatement with fossil-fueled generation.

Consequently, the analysis in this section has shown that dynamic welfare gains of addressing climate change (and possibly other policy objectives as well) with a policy mix of emissions trading and a feed-in tariff are traded off by static welfare losses. Thus a clear point can be made at the end of this section: A comprehensive evaluation of interactions between the ETS and the feed-in tariff has to integrate the static and the dynamic perspective. In this respect, the section has significantly gone beyond existing studies focussing either on the dynamic or the static perspective. Simple dynamic analyses, which argue in favour of implementing the policy mix to cope with multiple failures of private governance structures, are likely to neglect possible distortions arising in the short run. In contrast, static analyses emphasizing only the inefficiency of the policy mix in combating climate change disregard positive long-term effects of promoting renewable energy sources.

What is more, the extensive qualitative analysis of combining emissions trading and a feed-in tariff conducted in this section also allows for a policy recommendation for the German context. In fact the discussion has revealed that actual extent of the static inefficiency may be quite limited. This is due to the complex characteristics of the CO_2 and electricity markets. Thus, the qualitative analysis has provided some indication that dynamic welfare gains under the policy mix may be larger than static welfare losses. This analytic result discloses that the German policy mix of an emissions trading scheme and a feed-in tariff may be desirable from an economic point of view. Nevertheless, the qualitative results call for a confirmation by a quantitative estimation of dynamic welfare gains and static welfare losses. It is necessary to determine the actual level of spillovers related to learning-by-doing as well as the extent of other barriers to technological development empirically. Likewise, the static welfare losses, which are driven by the complex characteristics of the CO_2 allowance markets and the electricity market as well as interactions between these markets, have to be quantified.

Even though the policy mix appears to be desirable overall, it may be useful to examine whether modifications of the emissions policy and the feed-in tariff may improve the efficiency of the policy mix. Modifications may possibly help to increase dynamic welfare gains and to reduce static welfare losses under the policy mix. However, for the feed-in tariff, the thorough evaluation has yielded that a substantial modification of the tariff design cannot be recommended. Some features of the feed-in tariff system may not be changed for institutional constraints (e.g., remunerating technology operators rather than producers, funding the feed-in tariff by an add-on to the electricity price). Other design features may have to be maintained because alternative solutions would only increase the net value of internalization at the cost of higher transaction costs (e.g. remunerating operators irrespective of weather and site conditions, fixed feed-in tariff irrespective of electricity price). In other cases, an increase in the net value of internalizing the pollution externality in the short term due to adapting policy design may be traded off by a reduced net value from overcoming barriers to technological development, i.e. a smaller net value of internalization in the long run (e.g., differentiation of tariffs, fixed tariffs irrespective of electricity price). In addition, design features producing inefficiency in a partial-equilibrium model may be the superior choice from a general equilibrium perspective (e.g. funding the feed-in tariff by an add-on). What is more, the inefficient distortions resulting from some of these design features have been found to be limited (e.g. funding the feed-in tariff by an add-on). With respect to the feed-in tariff one may therefore conclude that static inefficiency is the "price" of dynamic welfare gains. In general, however, it can be put forward that the level as well as the differentiation of the feed-in tariff should be revised continuously. This is necessary to account for possible advances in learning-by-doing – and a consequent reduction in related spillovers.

Recommendations can also be made with respect to the design of the emissions policy. First of all, it is necessary to consider emissions effects of the feed-in tariff when designing the ETS. This implies that the emissions cap should be reduced by the expected emissions reduction brought about by renewable electricity generation. Consequently, the overall emissions reduction corresponds to the sum of abatement which would be produced under the ETS and the feed-in tariff if these were implemented in isolation. Under this condition, the feed-in tariff would provide for emissions reductions that are additional to those under the emissions trading scheme. Nevertheless, it should be born in mind that despite this modification, the policy mix will still be inefficient in abating emissions. Incentives to abate under the policy mix are the same as under a single ETS for non-electricity sectors. However, the additional emissions reduction attributable to the feed-in tariff would be produced cheaper if it was realized by fossil-fueled electricity generators or the non-electricity sector. Secondly, it is worth to mention that some of the inefficient interactions under the policy could be avoided by implementing an emissions tax instead of the emissions trading scheme. Under an emission tax the emissions price is fixed. Abatement incentives set by the tax are not affected by the feed-in tariff. The implementation of a feed-in tariff in addition to the tax does therefore not result in an inefficient shift of emissions between the electricity sector and other sectors subject to the tax. Emissions reductions under the feed-in tariff are in addition to those brought about by the tax. Moreover, differentiating the emissions price to compensate inefficient distortions of the add-on to the electricity price produces less transaction costs under the tax than under the trading scheme. The discussion of the policy mix has thus revealed advantages of an emission tax over an emissions trading scheme in a policy mix. However, shift from an emissions trading scheme to an emissions tax could not be made by Germany only. Rather, a fundamental reform of the European climate strategy would be necessary.

Finally, it has to be emphasized that implementing a feed-in tariff in addition to the emissions policy may not be sufficient to overcome all existing barriers to technological change. In order to address climate change efficiently, further policies may be necessary. First of all, learning spillovers may not only occur with renewable energy technologies but also with other low-carbon technologies for electricity production. It may be necessary, for example, to subsidize the production of carbon capture and sequestration or nuclear fusion technologies as well. Secondly, technological change may also be hampered by R&D externalities in the field of renewable energy technologies. These cannot be overcome efficiently by a feed-in tariff. Rather a more targeted support for R&D is necessary. Thirdly, it is desirable to reduce environmentally harmful subsidies which put conventional electricity generation at an inefficient advantage compared to renewable generation.

6.3 ETS and Electricity Tax

This section is devoted to the applied analysis of the ETS and the electricity tax. As a first step, it is assumed that both policies are only meant to address the pollution externality related to climate change. That is, they aim directly at reducing GHG emissions. The discussion in Chapter 5 has revealed that the policy mix is inefficient in reducing a pollution externality. Section 6.3.1 analyzes to what extent this inefficiency actually arises in the German context. The analysis is based on the theoretical discussion but also goes beyond it. It takes into account the detailed design of the ETS and the electricity tax as well as the characteristics of the CO_2 allowance market and the electricity market. Secondly, Section 6.3.2 will broaden the discussion of the policy mix. It is still assumed that both policies are to address climate change. It is investigated, however, whether an electricity tax is a useful complement to the ETS to overcome barriers to efficient energy generation and consumption which may reinforce the pollution externality. However, in Chapter 5, it has been highlighted that it may be a very restricted perspective to assume that the emissions trading scheme as well as the tax are only meant to mitigate climate change efficiently. As a third step, Section 6.3.3 therefore sheds light on the fact that the electricity tax does not only aim at achieving GHG emission reductions but other policy objectives as well. It is discussed to what extent the attainment of multiple policy objectives may justify the inefficiency of the policy mix with respect to mitigating climate change. Based on these considerations, Section 6.3.4 derives policy recommendations.

6.3.1 Addressing Climate Change: Overcoming a Pollution Externality

In the theoretical discussion of Chapter 5, it is has been demonstrated that the combination of an emissions trading scheme and a tax may produce a redundancy with respect to correcting a pollution externality. If emissions of all participants in the emissions trading scheme are taxed homogeneously, the policy mix may provide for an efficient reduction of GHG emissions. The same result may be attained, though, by a single emissions trading scheme or a single emissions tax – whichever policy is more stringent. Thus, the policy mix produces the same net value of internalization as the single policy. However, transaction costs of the policy mix are very likely to be higher than those of either policy implemented in isolation. Therefore, the policy mix should be rejected as a means to control GHG emissions, even under the assumption of homogeneous taxation of emissions.

If the assumption of homogeneous taxation of emissions is relaxed, the policy mix will result in a suboptimal net value of internalization. In the context of the German policy mix encompassing the ETS and the electricity tax, three deviations are noteworthy: First of all, taxation is heterogeneous across ETS sectors. The electricity tax only affects the electricity sector. Thus, emissions abatement is distorted across ETS sectors. Secondly, the electricity tax is not paid on emissions but on output of the electricity sector. This results in emissions abatement being distorted across electricity generators using different fuels and technologies. Both types of distortions have already been analyzed theoretically in Chapter 5. A third distortion, which was not considered in the theoretical discussion, arises due to the heterogeneity of the electricity tax rate. This implies that emissions abatement is also distorted across electricity consumers. These three issues will be illuminated in further detail in the following.

6.3.1.1 Issue 1: Heterogeneous Taxation of ETS Participants

It has been outlined in Chapter 5 that emissions abatement is distorted when a tax is implemented in addition to an emissions trading scheme and tax rates are heterogeneous. This section focuses on this inefficient distortion. The analysis neglects further distortions resulting from output rather than emissions taxation and the differentiation of electricity tax rates. These are discussed in subsequent sections.

Virtually all electricity generation in Germany is covered by the ETS (Heilmann 2005, p. 26). Thus, the ETS almost perfectly overlaps the electricity tax. In contrast, the German electricity tax affects only electricity generators. Other ETS participants are not covered by the tax.[71] The electricity tax results in an increase of the electricity price. Electricity demand and output decline. The electricity sector emits less and demands fewer CO_2 allowances. Consequently, the allowance price declines. ETS participants outside the electricity sector take advantage of this price reduction. Compared to the case with an absent taxation of electricity, they purchase more allowances and emit more (or abate less). The overall level of emissions is fixed at the emissions cap. The electricity tax therefore only results in a shift of emissions from the electricity sector to other ETS sectors. This shift increases the overall abatement costs incurred to achieve the emissions cap. From an economic point of view, the electricity sector abates too much and too costly. In contrast, other sectors do not employ relatively cheap abatement measures.

It is difficult to determine the actual extent of the inefficient distortion of abatement resulting from heterogeneous taxation. It is first of all necessary to study the effect the electricity tax has on emissions. For this purpose, the behaviour of the three largest electricity-consuming sectors –

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[71] In fact, ETS participants are likely to use electricity as an input to their production. The implication of the output market interaction is discussed later on.

households, trade and commerce, and industry – has to be analyzed. These sectors accounted for 95 percent of Germany's electricity consumption in 2007 (BDEW 2008a, p. 9). Households as well as the trade and commerce sector have to pay the full electricity tax rate. For them, the electricity tax has a relatively minor share of 12 percent in the total retail electricity price (Bode and Groscurth 2006, p. 6). Moreover, the price elasticity of these sectors is low (see, e.g., Taylor 1975; Kamerschen and Porter 2004; Reiss and White 2005; Narayan et al. 2007). Therefore, the electricity tax can be assumed to hardly result in a decline of electricity consumption in these sectors (Ziesing 2007b, p. 282). In contrast, the price elasticity of electricity demand is usually found to be higher for industrial customers (see, e.g., Bjørner et al. 2001; Kamerschen and Porter 2004). The industry sector consumes roughly 45 percent of the total amount of electricity marketed in Germany. Under the electricity tax, however, this sector benefits from a 40 percent reduction of the tax rate. In addition, energy-intensive industry companies are eligible for a tax cap. This cap limits the marginal tax burden to three percent of the electricity tax rate, i.e. 0.615 Euro per MWh. For these companies, the share of the tax rate in the marginal retail electricity price is around one percent (Bode and Groscurth 2006, p. 6). Therefore, the electricity tax is unlikely to reduce electricity consumption of energy-intensive industry consumers. Only industry customers whose production is less energy-intensive and which do not qualify for the tax cap are significantly affected by the electricity tax. The electricity tax accounts for 22 percent of their electricity bills. It may therefore in fact result in a substantial reduction of electricity consumption by these sectors (Ziesing 2007b, p. 282). Nevertheless, the overall impact of the electricity tax on electricity demand and output may be limited.

To what extent a possible decline of electricity generation brings down emissions depends on the characteristics of the merit order curve of electricity supply. These determine which fuels and technologies are crowded out of the market when electricity demand declines (see Section 6.2.1.8). There is no study available which determines the specific CO_2 reduction effects of the electricity tax. Bach et al. (2002, p. 806) estimate that the entire German ecological tax reform – including the electricity tax but also other taxes on fossil fuels – results in an reduction of CO_2 emissions by 20 to 25 million tons per year. This is less than half of the emission reduction attributable to the feed-in tariff for renewable electricity generation (see Section 6.2.1.8). Most probably, the emissions reduction mainly stems from abatement activities undertaken outside the ETS sectors, e.g. by reducing fuel consumption for transportation or household heating (for an overview, see BMU 2004b). The actual effect of the electricity tax on CO_2 emissions can therefore be assumed to be far below the above figure.

To what extent the emissions reduction induced by the electricity tax then provides for an inefficient distortion of abatement depends on the characteristics of the CO_2 allowance market. First of all, the slope of the marginal abatement cost curves in the electricity sector as well as in the other ETS sectors has an impact on the inefficiency. The theoretical analysis in Section 5.3.5.2 highlighted that the flatter the curves are, the larger will be the inefficient distortion from heterogeneous taxation. Marginal abatement cost curves in Germany can be assumed to be convex. Since emissions levels are still high, emissions reductions are likely to occur in the flat section of the curve. Thus, the inefficient distortion may indeed be high (see also the discussion in Section 6.2.1.8). Moreover, the market share of the taxed sector in the allowances market drives the level of welfare losses under the policy mix. The number of allocated allowances can be used as a proxy for the possible market share. The German energy sector received 12 percent of the total number of allowances allocated in the EU (European Commission 2008g, p. 14). In this respect, the impact of the sectoral electricity tax on the allowance price and abatement in other sectors may be limited. However, there is an EU Directive mandating electricity taxation throughout the EU (see Section 2.3.3), and several EU Member States have implemented regulations that are beyond these requirements (BMU 2003, p. 277). Consequently, the distortions resulting from electricity taxation at the EU level may be more substantial.

The inefficient shift of emissions from the electricity sector to other ETS sectors may be mitigated by three issues. Firstly, other ETS sectors may also face taxes in addition to the ETS. Fossil fuels, such as mineral oil, natural gas or coal, which may serve as an input to industry production, are also subject to energy taxation. However, at least in Germany, the energy-intensive sectors participating in the ETS are largely exempt from these taxes (BMU 2004b). Secondly, output market interactions may reduce the distortion of abatement (see Section 5.4.5.3). Electricity is an important input to other ETS sectors. Thus, electricity and output in other sectors are complements. The implementation of an electricity tax does therefore not only reduce electricity generation but also production and emissions in other ETS sectors. However, since ETS participants are energy-intensive, they are eligible for a significant reduction of the electricity tax rate. Thirdly, it has to be taken into account that the allowance price is not only driven by actual allowance demand. In the first trading period of the ETS from 2005 to 2007, political and psychological factors were at least equally important determinants of the allowance price (Bode and Groscurth 2006, p. 14; Kemfert and Diekmann 2009, p. 172).

It may be concluded that the heterogeneous taxation of electricity is in fact likely to result in inefficient abatement. The extent of this inefficiency, however, is a function of a variety of variables, and therefore hard to determine quantitatively. A very rough calculation using the theoretical framework of Section 5.3 may at least provide an idea of the effect the electricity tax has on the allowance price. It is assumed that the allowance market is restricted to Germany and that marginal abatement cost curves are identical across ETS sectors. Average specific emissions related to electricity generation amount to roughly 600 grams per kWh (UBA 2008b). Based on this finding, the electricity tax of 2.05 Cent per kWh translates into an emissions tax of 34 Euro per ton of CO_2 imposed on the electricity sector. Equation (5.37) in Section 5.3.5.2 has pointed out that the equilibrium allowance price is reduced by the sectoral emissions tax times the sector's share in the allowance market. The share of the energy sector in the German allowance market is 56 percent (see Sector 2.3.1). Given these assumptions and figures, the electricity tax reduces the equilibrium allowances price by 19 Euro per ton of CO_2. Given allowance prices ranging between 10 and 30 Euro per ton of CO_2, this price effect would be substantial. Yet, this figure neglects many of the considerations mentioned above. It should probably be understood as an upper bound of a possible price effect. The qualitative examination has shown that the actual effect may be far smaller.

As has been highlighted in Section 5.3.5.3, whether or not the possible inefficiency can actually be blamed on the policy mix depends on the point of reference. The policy mix is inferior to a single emissions trading scheme. However, the comparison of the policy mix with a single heterogeneous tax reveals that the welfare loss can in fact be attributed to the suboptimal design of the tax. From this point of view, the inefficiency of the policy mix is not produced by an interaction between the ETS and the electricity tax. Quite the reverse is true. The combination of the heterogeneous tax with the emissions trading scheme may even result in a better outcome in terms of efficiency than the single tax. The policy mix may increase the net value of internalization in two respects: (1) The policy mix attains a higher level of emissions abatement and related benefits. (2) The policy mix may even reduce the welfare loss produced by the tax when the marginal abatement cost curve is convex – as condition that has been confirmed for Germany and other EU Member States (Ellerman and Decaux 1998; Criqui et al. 1999; Klepper and Peterson 2006). This perspective seems to be more appropriate to evaluate Germany's policy mix. When the ETS was implemented throughout the EU in 2005, Germany's electricity tax had already been in place. Therefore, the policy mix should not be condemned *per se*. The implementation of the ETS on top of the electricity tax is likely to have reduced the welfare loss resulting from heterogeneous taxation in absolute as well as in relative terms.

This reasoning notwithstanding, it should be considered how the welfare loss under the policy mix can be further reduced. Many authors have argued for the German context that participants of the ETS should be exempt from taxation (Graichen and Requate 2003, p. 14; Frondel and

Hillebrandt 2004, p. 332; Böhringer et al. 2005, p. 45; Heilmann 2005, p. 26). Taking into account that in some cases – as under the electricity tax – emissions are not addressed directly but rather indirectly via outputs or inputs, this conclusion should be generalized. If mitigating climate change is the only policy objective, all processes and products whose emissions are covered by the ETS should be excluded from any additional taxation. Since the ETS covers virtually all CO_2 emissions from electricity generation, the electricity tax cannot be justified as a means to combat climate change efficiently. From this perspective, the electricity tax should be abolished entirely.

Institutional constraints may prevent the entire abolition of the electricity tax, though. The EU mandates a minimum of electricity taxation of 1.00 Euro per MWh for non-business uses and 0.50 Euro per MWh for business uses. Further tax breaks are only allowed for energy-intensive industries and firms subject to a voluntary agreement or an emissions trading scheme. EU regulation thus provides for an entire exemption of ETS participants from taxation but not for a general abolition of the electricity tax. From a national perspective, it would therefore be advisable to reduce German tax rates to EU minimum requirements. Since current tax rates in Germany are beyond these requirements, a reduction would already reduce inefficient distortions significantly. From a European perspective, it would be necessary to allow a tax exemption not only for ETS participants but more broadly also for products whose emissions are covered by the ETS.

If the abolition of the electricity tax is not a feasible policy option – because it addresses other objectives than climate change as well (see Section 6.3.3 below) – a separation of the allowance market may be considered, as it has been analyzed in Chapter 5.3.5.4. Separate allowance markets may be established for the electricity sector and for other ETS participants. Under such a system with several markets, the allocation of allowances to each sector is decisive for the efficiency of the entire system. Determining an efficient allocation may bring about significant transaction costs for the regulator since he has to gain knowledge about marginal abatement costs. These transaction costs are likely to exceed the increase in the net value of internalization expected from the separation of the allowance market. This is particularly likely in the German case where the inefficient distortion resulting from the electricity tax can be assumed to be limited. It may then be a superior solution to deliberately over-allocate allowances to the taxed sector, i.e. the electricity sector. This over-allocation could be determined if the regulator has at least a rough understanding of the marginal abatement costs. If the regulator had no knowledge at all, he could even decide to allocate all allowances to the electricity sector. Given this over-allocation, trades should be allowed between the electricity sector and the other ETS sectors on the basis of a trading ratio. Non-electricity firms would have to buy more than one allowance from the electricity sector to cover one unit of their emissions. The exact trading ratio would depend on the average emissions tax implicit to the electricity tax as well as the allowance price in the electricity sector. Yet, such a modification of the ETS could not be implemented unilaterally by Germany. Rather it would call for an adaptation of the EU ETS Directive. Moreover, despite its theoretical appeal, this solution may affect the attainment of other policy objectives next to climate protection negatively. Possible issues in this respect are discussed in Section 6.3.3.

The inefficiency stemming from the heterogeneity of taxation under the ETS may be deteriorated by two further design features of the electricity tax: the taxation of output rather than emissions and the differentiation of the tax across electricity consumers.

6.3.1.2 Issue 2: Taxation of Output

As has been demonstrated in Section 5.4.7, output taxation even aggravates the inefficiency of a heterogeneous tax. To understand this argument, it is necessary to differentiate between the effects emissions policies have on output and the emissions rate, i.e. the specific emissions per unit of output. An emissions trading scheme sets incentives to efficiently reduce the level of both variables. If a sectoral output tax is implemented on top of the emissions trading scheme, however, these incentives are distorted. In the German case, the electricity tax results in a decline of electricity output. The resulting decline of the allowance price induces electricity generators as well as other ETS participants to increase their emissions rate – compared to a single trading scheme. Moreover, non-electricity sectors also increase their output as the allowances price decreases. Consequently, emissions are shifted from the electricity sector to other ETS sectors – as has been described in the previous section. However, the electricity tax brings about an additional distortion in the electricity sector. Not only are overall emissions reduced to an inefficiently low level. More importantly, the taxation provides for emissions abatement by reducing the emissions rate to be substituted by emissions abatement through output reduction. The output reduction in the electricity sector has to compensate the increase of the emissions rate in the electricity sector and the increase of the emissions rate and the output level in other ETS sectors. In order to provide for an efficient correction of the pollution externality, the emissions rate in the electricity sector should be lower and output should be higher, while both variables should be lower in other ETS sectors. The taxation of output rather than emissions thus additionally decreases the net value of internalization under the policy mix.

The inefficiently high emissions rate in the electricity sector can be attributed to the fact that electricity taxation distorts the use of fuels and technologies for electricity generation. The electricity tax incorporates an implicitly higher tax on cleaner fuels and technologies. This is analogous to the effects of the add-on to the electricity price discussed in Section 6.2.1.7. The emissions rates provided in that section can also be used to compute the implicit emissions tax rates under the electricity tax (for the emissions rates, see Krewitt and Schlomann 2006, p. 35). A tax rate of 20.50 Euro per MWh implies an emissions tax of 23 Euro per ton of CO_2 for lignite-fueled generation and of 53 Euro per ton of CO_2 for natural gas-fired generation. Both values refer to electricity generation in modern, efficient gas and steam power plants. In contrast, lignite combustion in a conventional steam power plant is subject to in implicit emissions tax of only 19 Euro per ton of CO_2. The electricity tax thus results in a substitution of clean fuels and technologies by dirtier ones. Therefore, the electricity tax reduces electricity output but increases the average emissions rate of electricity generation at the same time. This effect is even deteriorated by the specific characteristics of electricity supply. A reduction of electricity output due to the tax does not affect all types of fuels for generating electricity equally. According to the merit order curve, marginal, relatively costly power plants are taken from the grid first. These plants are typically natural gas-fired (see, e.g., Bode and Groscurth 2006, p. 10). This implies that emissions from marginal (natural gas-fueled) plants which are shut down are taxed at an amount that impedes further economic operation. In contrast, other plants remaining in operation are in fact not taxed at all. The extent of the inefficiency due to the taxation of output eventually depends on the effect the electricity tax has on emissions and the allowance price (see the discussion in the previous section). Thus, actual welfare losses due to output taxation may also turn out to be limited eventually.

If the correction of pollution externalities related to climate change is the only policy objective, the taxation of output rather than emissions provides another rationale for abolishing the electricity tax. If the abolition of the tax is not a politically feasible option, two modifications of policy design could be considered. On the one hand side, the electricity tax could be differentiated with respect to emissions produced during generation. This would require mechanisms to relate each unit of the homogeneous good electricity to a specific generation process. This would bring about high transaction costs – if it was technically feasible at all – which would probably exceed the resulting increase in the net value of internalizing the pollution externality. On the other hand, the ETS could be modified. In Section 6.2.1.7, it has been highlighted that the emissions price has to be differentiated in the presence of an add-on to the electricity price (or an output tax) in order to provide for efficient emissions abatement. It has also been shown, though, that such a differentiation may be a challenging task bringing about (prohibitively) high transaction costs for the regulator. Modifications of policy design are therefore not helpful from an economic point of view to reduce the inefficiency of the policy mix. In this respect, the abolition of the tax also seems to be the only desirable policy alternative.

6.3.1.3 Issue 3: Heterogeneous Taxation of Electricity Consumers

A third distortion of abatement under the ETS arises from the heterogeneity of the electricity across electricity consumers. The tax rate is highest for households and the trade and commerce sector amounting to 20.50 Euro per MWh. Industry companies that are not energy-intensive as well as agricultural enterprises only pay 12.30 Euro per MWh. The tax rate is lowest for energy-intensive industry companies, which are subject to a marginal tax rate of only 0.615 Euro per MWh. Consequently, households and the trade and commerce sector have a higher incentive to reduce electricity demand and to abate emissions than industry consumers. From an efficiency perspective, this policy design contributes to the fact that the abatement incentives are not equalized across all economic actors. Households abate too much and too costly compared to the industry sector. In this respect, the electricity tax does not only distort abatement between ETS sectors and different types of electricity generation but also between consumers of electricity. This distortion resulting from the heterogeneity of taxation across electricity sectors thus further reduces the net value of internalization under the policy mix.

This distortion could be overcome by implementing a uniform electricity tax rate for all electricity consumers. If this modification implied a significantly positive tax rates for the industry sector, however, it would deteriorate the distortion from the heterogeneous taxation of ETS sectors (see Section 6.3.1.1). Industry companies, which are characterized by a high price elasticity of electricity demand, would have a stronger incentive to reduce their electricity consumption. The emissions reduction in the electricity sector due to the electricity tax would be higher. Consequently, the inefficient shift of emissions from the electricity sector to other ETS sectors would become more severe. The straightforward alternative to implementing a uniform positive electricity tax would be a reduction of the tax rate to zero. Therefore, the heterogeneity of taxation across electricity sectors provides but another reason to abolish the electricity tax.

Again it may be argued that the extent of the distortion is limited – at least in a partial equilibrium model of the electricity sector. Energy-intensive industry consumers with a high price elasticity of electricity demand are largely exempt from taxation. Households and the trade and commerce sector face the full tax rate. However, their demand for electricity is characterized by low price elasticity. Consequently, emissions abatement is hardly distorted between these sectors. A welfare loss may only arise since less energy-intensive industry companies are subject to a significant tax rate. Their electricity demand usually exhibits a relatively high price elasticity. Consequently, these companies are likely to reduce their electricity consumption too much compared to other electricity consumers. A more severe distortion may arise in general equilibrium context, though. There may be a particularly obvious distortion between the energy-intensive industry sector on the one hand side and households and the trade and commerce sector on the other hand side. The relative and absolute tax burden is very low for energy-intensive industry companies. The electricity tax therefore hardly affects their investment decisions. In contrast, the tax burden may be substantial for households and firms in the trade and commerce sector. In this case, the electricity tax may well result in a reduction of these sectors' investment and consumption of other goods than electricity. Consequently, the electricity tax may significantly distort the investment and consumption decisions. From an efficiency point of view, the industry sector may invest too much while households and the trade and commerce sector spend too little of their budget for other goods then electricity.

6.3.2 Addressing Climate Change: Overcoming a Pollution Externality and Barriers to Efficient Energy Generation and Consumption

So far it has been assumed that the ETS as well as the electricity tax are meant to correct a pollution externality related to climate change simultaneously. Some authors argue implicitly, however, that the sectoral tax may also be useful to overcome other barriers to low-carbon electricity generation and energy-efficient electricity consumption (Sorrell et al. 2003, pp. 62-63; Heilmann 2005, pp. 27-28). Their main reasoning is that a tax may induce a higher level of abatement in some sectors or countries (if the European ETS is considered). This extra abatement incentive may be desirable from an economic point of view to initialize the transition away from a long-lived, carbon-intensive capital stock and infrastructure. Thereby, emitters can be brought on track for even larger emissions reductions expected to be necessary in the future. The authors highlight that taxation may be particularly necessary when allowance prices can be expected to be low – for example, due to an overallocation of allowances to Eastern European EU Member States. In this respect, the tax would serve as a back-up providing for a minimum level of sectoral or domestic abatement.

The studies thus presume that the implementation of an emissions trading scheme alone cannot transform an economy efficiently into a low-carbon economy from one day to another. The underlying reason is that, next to a pollution externality, other barriers may impede this shift. These barriers result in a path dependency, i.e. a lock-in of existing fossil-fuel technology systems. These barriers thereby reinforce the pollution externality. It has been argued several times throughout this book that a variety of barriers may hamper efficient emissions abatement in the electricity sector. These include R&D and learning spillovers related to low-emission technologies (Section 3.3.1, Chapter 4 and Section 6.2.1), asymmetric information between market participants and within firms (Section 3.3.2), sunk costs of fossil-fuel technology investments, capital market barriers and an uneven political playing field (Section 6.2.2). In this respect, additional policies to the ETS may also be justified on efficiency grounds.

However, the German electricity tax is a rather rough means to address these barriers. Since it is based on output rather than emissions, it does not set incentives to switch to low-carbon fuels and technologies. On the contrary: It increases the average emissions rate of electricity generation (Section 6.3.1.2). Moreover, the tax reduces electricity consumption irrespectively of consumers' actual potentials to improve energy efficiency and the extent of related barriers. While the first argument would not apply to a tax imposed on emissions, the second argument would also hold for an emissions tax. In addition, it has to be taken into account that promoting additional abatement in one sector by implementing a tax comes at decreasing incentives to reduce emissions in other sectors. The electricity tax may be an (inefficient) means to promote the transition away from a fossil-fueled electricity sector. At the same time, however, it may even foster the lock-in of other ETS sectors into a carbon-intensive economy. Therefore, the suitability of the electricity tax to overcome barriers to low-emission electricity generation and energy-efficient electricity consumption may be drawn into question. A superior alternative would be to complement the ETS by policies addressing these barriers directly. Such policies may include subsidies, information measures, the abolition of environmentally harmful policies, and many other measures more.

6.3.3 Addressing Climate Change and Other Policy Objectives

The policy mix is not only meant to address the problem of climate change. In particular, the electricity tax has multiple objectives. These have to be taken into account when the policy mix is to be evaluated comprehensively. It has to be examined to what extent the inefficiency of the policy mix in addressing climate change is compensated by attaining other policy objectives.

First of all, the electricity tax – as well as the entire ecological taxation reform – has been implemented according to the concept of the double dividend. Revenues are used to reduce other distorting taxes – most notably the burden on labour imposed by contributions to pension insurances. Thereby, the tax is meant to reduce labour costs, promote economic growth – and eventually increase employment (see Section 2.3.3). On a general level, several studies reveal welfare gains from using environmental tax revenues to reduce other distorting taxes – compared to returning them in a lump sum manner (see, e.g., Goulder 1995a; Goulder 1995b; Parry 1997).

However, there are also analyses which only find a small or no double dividend (see, e.g., Bovenberg 1999; Babiker et al. 2003). Welfare gains of reducing contributions to pension insurances have not been quantified for Germany. However, several studies provide evidence that they are small indeed. Bach et al. (2002, pp. 806-807) find that the ecological taxation reform has almost no effect on economic growth. Likewise, the impact on employment is small. Bach et al. estimate that some 250,000 jobs could be created due to lower labour costs in the period from 1999 to 2010. Limited employment effects are also confirmed by Frondel and Hillebrand (2004, p. 331). They conclude that due to the ecological taxation reform only 75,000 jobs are maintained. These numbers are in strong contrast to original estimates by the Federal Ministry of the Environment, which predicted 250,000 jobs to be generated by the ecological taxation reform by 2003 already (BMU 2004b, p. 16). These empirical findings for the German context seem to confirm Bovenberg's (1999, p. 441) conclusion "that the case for environmental taxes should be made primarily on environmental grounds." However, environmental benefits of the electricity tax were found to be non-existent with respect to climate change in the previous section.

Even significantly positive effects on growth and employment would not necessarily justify the electricity tax. Fiscal revenues could not only be raised by taxes but by auctioning allowances under the ETS as well. So far, auction revenues have been small since only ten percent of total allowances are auctioned in the second period of the ETS from 2008 to 2012. If this share was increased, however, the five billion Euro of electricity tax revenues (see Section 2.3.3) could also be provided by auctioning allowances. If the entire quantity of 291.81 million allowances associated with the energy sector (253.81 million allowances allocated free of charge so far and 38 million allowances already auctioned, see Section 2.3.1) was auctioned, an allowance price of 17 Euro would generate equivalent fiscal revenues. Auctioning off all 451.81 million allowances available under the ETS would produce the same revenue at an allowance price of 11 Euro. In this respect, the ETS could produce similar revenues as the electricity tax – without detrimental effects on the efficiency of mitigating climate change. However, using auctioning revenues to fund pension insurances may also be subject to restrictions. It is still unclear whether such arrangement would be in line with the German constitution. Moreover, auction revenues are uncertain due to volatile allowance prices, which may be driven by exogenous factors such as fuel prices. Thus, auctioning allowances may not be suitable to generate steady fiscal revenues (Ziesing 2007b, p. 278).

Possibly, complementing the ETS by the electricity tax may also be desirable to attain further policy objectives, which have not been made explicit in regulations on these policies. Taxing electricity may have other positive environmental effects than climate change mitigation only.

Firstly, a reduction of electricity generation is also likely to reduce air pollution and resource conservation related to the electricity sector. These benefits may be offset by opposite effects in other ETS sectors, though – this effect is quite similar to the one described for the policy mix of the ETS and the feed-in tariff in Section 6.2.3. Under the ETS, the electricity tax may result in a shift of air pollution and resource use from the electricity sector to other sectors. As for the feed-in tariff, the overall effect on air pollution and resource use thus depends on the characteristics of the production processes which are extended at the cost of reduced electricity generation.

Secondly, a policy mix of an emissions trading scheme and a tax may also be chosen for equity concerns. Under the ETS, equity concerns may arise because allowances are allocated free of charge by grandfathering or benchmarking approaches (see Section 2.3.1). Since these allowances can be potentially sold on the allowance market, firms consider their opportunity costs when deciding about abatement and production. As a result, output prices – as that of electricity – increase. Firms receive so-called "windfall profits". These profits are likely to be particularly high for electricity generators. Since electricity demand is inelastic, generators can almost perfectly pass opportunity costs through to electricity customers. Thus, income is transferred from electricity customers to electricity generators (Sijm et al. 2005; Sijm et al. 2006; Ziesing 2007b, pp. 305-334). A tax may then be implemented in order to capture and redistribute windfall profits (Johnstone 2003, pp. 18-19). However, this rationale is based on the assumption that the tax perfectly overlaps the emissions trading scheme and is imposed on emissions. In this case, the allowance price is reduced by the tax, and windfall rents are reduced. In contrast, the German electricity tax mainly affects the electricity sector and is imposed on output. It has been shown that the effect of the electricity tax on the ETS allowance price is rather limited (see Section 6.3.1.1). Consequently, the electricity tax hardly captures any windfall profits under the ETS. To the contrary, it establishes an additional burden on the electricity price. Just as ETS opportunity costs, electricity generators may pass the tax through to electricity customers. As a result, the electricity tax increases the burden on customers. Moreover, the environmental taxation reform itself has uneven distributional effects. Due to its differential design, private households end up being net payers of the environmental taxation reform. Their tax burden exceeds the corresponding reductions in their contributions to pension insurances. In contrast, energy and industry sectors largely benefit from the reform (Bach et al. 2002, p. 808; Frondel and Hillebrandt 2004, p. 331). Equity concerns may also pose an important argument against implementing solutions, which have been discussed in Sections 5.3.5.4 and 6.3.1.1 to overcome the inefficiency of the policy mix. Over-allocating allowances to the electricity sector would even increase windfall profits of generators and further contribute to equity concerns. In summary, the current design of the policy mix of the ETS and the electricity tax – and the related inefficiency – cannot be justified by equity reasons. Designing the policy mix efficiently would rather deteriorate the equity of the policy mix.

Finally, having a policy mix of the ETS and the electricity tax may be desirable for considerations of competitiveness. As has been shown, the electricity tax may shift emissions from the electricity sector to other sectors. Thus, the tax mitigates the ETS burden imposed on sectors such as steel, pulp and paper or cement. Their products are sold in globalized markets. The electricity tax may thus help to minimize adverse effects the ETS has on German firms' competitive position in these markets. In contrast, electricity markets are still dominated by national suppliers. Therefore, the extent to which the electricity tax distorts competition in electricity markets can be assumed to be limited. Concerns about competitiveness of German firms may again provide a rationale against implementing alternative policy designs to overcome the inefficiency of the policy mix. Restricting trade between sectors and over-allocating allowances to the electricity sector would in fact reverse the evaluation of the policy mix with respect to competition. The competitive position of ETS participants outside the electricity sector would be deteriorated. Compared to the current design they would be obliged to buy (more) allowances to cover their emissions. Consequently, the ETS burden imposed on these firms would increase substantially. Thus, efficiency gains would be traded off by losses in the competitiveness of the German industry.

6.3.4 Policy Recommendation

The main conclusion from the preceding evaluation is that the existing policy mix of the ETS and the electricity tax cannot be justified by mitigating climate change. Neither of the two economic rationales for using a policy mix applies. The policies are not meant to address multiple failures of private governance structure. Moreover, the policy mix is not suitable to internalize a pollution externality at less transaction costs compared to a single-policy strategy. Rather, the policy mix of the ETS and the electricity tax reduces the net value of internalizing the pollution externality related to GHG emissions, compared to the case with the ETS only. Since the electricity tax only affects electricity generators, emissions are inefficiently shifted from the electricity sector to other ETS sectors. Imposing the tax on output rather than emissions distorts the choice of output, technologies and fuels in the electricity sector. Differentiating the tax rate across electricity customers leads to inefficient consumption decisions. The policy mix thus increases abatement costs compared to single ETS. At the same time, it does not bring about a larger reduction in CO_2 emissions. The extensive qualitative discussion has revealed, however, that the actual extent of the related welfare losses may be limited due to the various tax breaks provided under the electricity tax and the characteristics of the electricity and the CO_2 allowance market.

What is more, the inefficiency identified cannot be blamed on the policy mix. It is attributed to the suboptimal design of the electricity tax. In fact, the policy mix may be even superior to a single electricity tax in terms of efficiency. Given that the ETS is more stringent than the electricity tax, the policy mix provides for a higher level of internalization. Moreover, given that marginal abatement costs are convex, the absolute level of welfare losses stemming from heterogeneous taxation across ETS sectors is reduced. In this respect, the implementation of the ETS on top of the pre-existing electricity tax in 2005 has actually increased the efficiency of mitigating climate change in Germany.

Both findings – the relatively small extent of inefficiency in mitigation under the policy mix and the superiority of the policy mix over a single tax – may provide arguments in favour of the policy mix if the tax is also meant to address other policy objectives than climate change. The inefficiency in mitigating climate change may then be the "price" of addressing these other objectives in addition. However, the benefits of the electricity tax with respect to other policy objectives than climate change have been found to be rather limited. Using tax revenues to reduce contributions to pension insurances has been found to have only small positive effects on economic growth and employment. Notably, tax revenues could possibly also be replaced by revenues from auctioning allowances under the ETS. Benefits of the tax with respect to pollution control and resource conservation can be assumed to be limited as well. Moreover, it rather deteriorates equity issues than ameliorating them. Positive effects can only be expected with respect to the competitive position of non-electricity ETS sectors in international output markets. Consequently, it may be drawn into question whether the attainment of multiple policy objectives under the policy mix may actually justify welfare losses with respect to mitigating climate change.

Taking these considerations into account, the obvious policy recommendation is to exempt all electricity generation whose emissions are subject to the ETS from the electricity tax. The ETS virtually covers all electricity generated in Germany. Moreover, electricity imports mainly stem from other EU Member States, which have also implemented the ETS. Consequently, the electricity tax should be abolished completely. However, all EU Member States including Germany are subject to the EU Directive on energy taxation mandating minimum tax rates for electricity. In the presence of this restriction, the German electricity tax should at least be reduced to the legally required minimum. The recommendation with respect to the European climate and energy strategy would be to abolish any regulation on minimum tax rates for electricity.

If the abolition of the electricity tax at the German and European level is not politically feasible, theoretical analysis suggests a restriction of allowance trading between electricity and non-electricity sectors. The optimal restriction in the presence of imperfect knowledge on marginal abatement costs is simple. The electricity sector (i.e. the sector subject to the higher tax rate) should be granted an overallocation of allowances and be allowed to sell these allowances at a certain ratio to other sectors. This restriction may be useful to overcome the inefficiency resulting from heterogeneous taxation across ETS sectors. It is not suited, though, to correct distortions arising from heterogeneous output taxation within the electricity sector. Moreover, overallocating allowances to the electricity sector may have adverse effects on equity and competition.

The clear recommendation to abolish the German electricity tax notwithstanding, it is worth highlighting that the analysis should not be misunderstood as a general plea against combining emissions trading and taxes. First of all, both policies may efficiently complement one another when their incentives are substitutive, and not additive as in the German case. That is, firms may in fact choose between holding allowances and paying a tax. This policy mix may bring down transaction costs and/or increase the net value of internalizing the pollution externality with an emissions trading scheme. The tax may be implemented as a cap on the allowances price in the presence of uncertainty about marginal abatement costs ("safety valve"). Alternatively, a punitive tax may help to foster compliance with the emissions trading scheme. Secondly, both policies may be complements in a policy mix when theirs scopes do not overlap. So far, the ETS only covers CO_2 emissions from the energy sector and some industry sectors. In this case, additional policies are required to regulate other GHG emissions as well as emissions from other sectors. In this respect, other taxes, which formed part of the ecological taxation reform in Germany, should be maintained. This refers, for example, to taxes imposed on fuels used for transportation purposes or decentral household heating (for an overview, see BMU 2004b). It has to be emphasized again, however, that these arguments do not hold for the electricity tax. CO_2 emissions of electricity generation are almost completely covered by the ETS. With respect to mitigating climate change, the additional electricity tax is not necessary. It does not bring about further emissions reductions but rather increases the cost of emissions abatement.

6.4 ETS and CHP Bonus

This section evaluates the combination of the ETS and the bonus paid for electricity generated in combined heat and power (CHP) plants. The analysis draws on the theoretically derived rationales for using a policy mix. It also takes into account theoretical findings made with respect to the policy mix of the ETS and the feed-in tariff for renewable electricity generation. Moreover, it considers specific issues arising from the actual design of the CHP bonus in Germany.

The evaluation of the policy mix is divided into three steps: First of all, it is assumed that the policy mix is to address two failures of private governance structures contributing to climate change: a pollution externality related to GHG emissions and learning spillovers related to CHP technologies (Section 6.4.1). Secondly, it is analyzed whether the bonus may also be a useful complement to the ETS to overcome further barriers to technological change, which reinforce the pollution externality (Section 6.4.2). Thirdly, light is shed on the fact that the CHP bonus does not only aim at mitigating climate change but at other policy objectives as well (Section 6.4.3). Finally, the most important policy recommendations are derived (Section 6.4.4).

6.4.1 Addressing Climate Change: Overcoming a Pollution Externality and Learning Spillovers

6.4.1.1 Theoretical Rationale for Using the Policy Mix

Quite similar to the policy mix of the ETS and the feed-in tariff for renewable electricity, the combination of the ETS and the CHP bonus can be justified in the presence of two failures of private governance structures, both of which contribute to climate change. Fossil-fuel electricity generation produces a negative pollution externality by emitting GHGs. This externality may be reinforced by spillovers related to learning-by-doing in the production of CHP technologies. Spillovers hamper the development of cheaper CHP technologies, which are one means to abate GHG emissions. Consequently, the pollution externality is more severe in the presence of spillovers than in its absence. In order to address climate change efficiently, both failures have to be addressed simultaneously. The theoretical analysis in Chapter 4 revealed that an emissions price is needed to correct the pollution externality. In addition, an output subsidy has to be implemented to foster learning-by-doing. It has to be highlighted, though, that the policy mix is only desirable on efficiency ground if the balance of the net value of correcting the failure and the corresponding transaction costs is positive for each failure. Moreover, the balance must be larger for regulation than for any private governance structures. A comprehensive assessment – or even quantification – of the net value of correcting each failure and the related transaction costs is beyond the scope of this book. However, some important aspects are highlighted in the following, which may have an impact on these variables.

The existence of negative pollution externalities from GHG emissions of electricity generation can be assumed with high certainty (see Section 1.2). In addition, learning spillovers may occur with CHP technologies. There seems to be a lack of empirical studies in this respect, though.

Evidence of learning-by-doing and related spillovers has been found for renewable energy technologies and many other technologies as well (see the review in Section 4.2.1). Therefore, learning spillovers are likely to be significant for CHP technologies as well. Consequently, the policy mix of the ETS and the CHP bonus may produce a positive net value of correcting both the pollution externality and learning spillovers. However, several issues may arise due to the fact that the policy mix does not correspond to the theoretically proposed efficient policy design. These issues may have an impact on the net value of correcting the two prevailing failures as well as the transaction costs to bring this correction about. Several issues are quite similar to those arising with the policy mix of the ETS and the feed-in tariff for renewable electricity. These are only briefly mentioned. Interested readers are referred to corresponding discussions in Section 6.2. In some respects, the effects of the CHP bonus in the policy mix may deviate from those of the feed-in tariff. This may be due to differences in policy design. Further deviations may arise because CHP plants may be subject to the ETS while renewable energy plants are generally exempt. The most important issues are discussed in the following.

6.4.1.2 Issue 1: Recipients of the CHP Bonus

As the feed-in tariff for renewable electricity, the CHP bonus is paid to operators of CHP plants per unit of electricity generated. In reality, however, producers of CHP plants – rather than operators of these plants – experience learning-by-doing and related spillovers. Thus, a first-best subsidy should be paid to producers per unit of CHP plant produced. As has been highlighted in Section 6.2.1.2, electricity generation from CHP plants can be assumed as a proxy of the production of CHP plants. In contrast to renewable energy plants, this proxy exhibits smaller distortions with CHP plants. CHP electricity generation is mainly driven by technological features of the CHP plants. It does not depend on exogenous determinants such as weather and site conditions. Therefore, an increase in CHP electricity generation is likely to result in a corresponding rise in the production of CHP plants. Taking these considerations into account, promoting CHP electricity generation instead of CHP plant production hardly reduces the net value of correcting learning spillovers.

6.4.1.3 Issue 2: Degression of the CHP Bonus

As has been discussed in Section 6.2.1.5, the output subsidy to the learning industry has to be reduced continuously. As cumulative output increases, producing firms gain experience. Technologies become more mature and the marginal benefits from learning as well as related spillovers decrease. Consequently, the optimal output subsidy also declines. This result implies that the CHP bonus has to be subject to a steady degression – just as the feed-in tariff for renewable electricity. CHP bonuses paid for electricity from plants commissioned until 2008 were in fact subject to a continuous reduction. However, no such degression has been implemented for CHP plants commissioned or modernized from 2009 on. Operators receive a fixed CHP bonus irrespective of the year in which their plant is commissioned – i.e. irrespective of the increase in cumulative output and learning that is likely to have occurred in preceding years. If the CHP bonus has been chosen efficiently for 2009, it will be too high in subsequent years. That is the incentive to increase CHP electricity output – and the production of CHP plants – will be too high. As result, the net value of correcting learning spillovers related to CHP technologies is reduced.

6.4.1.4 Issue 3: CHP Bonus Paid in Addition to Electricity Price

Unlike the feed-in tariff for renewable electricity generation (see Section 6.2.1.6), the CHP bonus is paid in addition to the electricity price. That is, the incentive to invest in CHP is at least to some extent driven by the electricity market as well. In terms of transaction costs, this difference implies a substantial advantage of the CHP bonus compared to the feed-in tariff. The optimal level of the CHP bonus does not depend on the prevailing electricity price. Therefore, the regulator is not required to adapt the CHP bonus continuously as the electricity price changes. Taken all other determinants of the CHP bonus *ceteris paribus*, a fixed CHP bonus can be implemented. In this respect, the CHP bonus can provide an optimal net value of correcting the learning spillover at relatively low transaction costs. However, further adaptations of the bonus may become necessary as CHP technologies become more mature and the number of adopters of CHP technologies increases (see Sections 4.3.4).

6.4.1.5 Issue 4: Funding of the CHP Bonus Endogenous to the Electricity Sector

As the feed-in tariff for renewable electricity, the CHP bonus is revenue-neutral for the government. It is funded by a uniform CHP add-on to the electricity price. The discussion of the effects of the feed-in tariff system in Section 6.2.1.7 has revealed that such add-on may impair the efficiency of the policy mix. Like an output tax, it drives down electricity demand and distorts the choice of fossil fuels in the electricity sector. In order to allow for efficient CO_2 abatement, these effects have to be compensated by modifying the design of the emissions price. It has to be lower than the Pigovian level and differentiated across fuels. However, establishing an emissions trading scheme with differentiated prices for CO_2 brings about high transaction costs for the regulator. What is more, due to the effects of the CHP add-on, the emissions price has to be adapted continuously as CHP technologies become more mature, the number of CHP adopters increases and the market share of CHP electricity rises. This requirement brings about additional transaction costs which have to be incurred by the regulator. Consequently, there is a trade-off between a high net value of correcting the pollution externality and learning spillovers on the one hand side and high transaction costs on the other hand side. Since the emissions price is fixed under the ETS, transaction costs under the existing policy mix can be assumed to be low. However, this comes at the cost of welfare losses which are incurred due to inefficient CO_2 abatement and suboptimal promotion of CHP electricity output.

The actual extent of this inefficiency may be limited. This is due to the fact that the actual impact of the add-on on electricity demand may be rather small. The full CHP add-on of 0.231 Cent per kWh is only paid by small electricity consumers from the household and trade and commerce sectors (see Section 2.3.4). These households are characterized by a price-inelastic demand for electricity. Moreover, the CHP add-on accounts for only two percent of the retail electricity price of these electricity consumers. In contrast, energy-intensive electricity consumers with a higher price elasticity of electricity demand pay a reduced add-on of 0.05, or even 0.025, Cent per kWh. The burden stemming from the CHP add-on for these electricity consumers is one percent or less of their electricity bills (Bode and Groscurth 2006, pp. 6-7). Consequently, the add-on is unlikely to result in a significant reduction of electricity demand. The inefficiencies in the policy mix can be assumed to be small.

Alternatively to the CHP add-on, the CHP bonus could also be funded by general tax revenues. This approach would avoid distortions resulting from the add-on in the partial equilibrium model of the electricity sector. However – as has been pointed out in Section 6.2.1.7 – general taxes may bring about significant deadweight losses in a general equilibrium setting. These distortions are likely to exceed the only modest welfare losses resulting from the add-on. Whether general tax revenues are an efficient alternative to the CHP add-on for funding the CHP bonus may therefore be doubted.

6.4.1.6 Issue 5: Partial Overlap of ETS and CHP Bonus

Another issue arises from the fact that the ETS and the CHP bonus do not overlap perfectly. As the feed-in tariff for renewable electricity, the CHP bonus is primarily meant to reduce emissions in the electricity sector. As a result, the allocation of abatement under the ETS may be distorted across sectors. However, the actual effect of the CHP bonus on emissions abatement may differ from the effect of the feed-in tariff. As has been pointed out in Section 6.2.1.8, the feed-in tariff unambiguously results in an inefficient shift of emissions from the electricity sector to other ETS sectors. This is not necessarily the case with the CHP bonus. First of all, this difference arises from the fact that CHP plants are covered by the ETS if their production capacity exceeds 20 megawatts and fossil fuels are combusted (Section 2.3.1). Secondly, heat generated in CHP plants covered by the ETS and distributed via district heating networks may substitute decentral heating, e.g. in private households, which is not covered by the ETS. Thus, emissions in the entire energy sector may in fact increase due to the promotion of CHP electricity (Graichen and Requate 2003, p. 15). This effect is reinforced by an additional bonus which is paid for newly constructed district heating systems (see Section 2.3.4).

Taking these considerations into account, Ziesing (2007b, pp. 290-291) distinguishes several possible scenarios: Firstly, it may be assumed that a new CHP plant (or the extension of an existing plant) is subject to the ETS. Moreover, the assumption is made that electricity generation in the CHP plant substitutes electricity generation by another ETS participant using a similarly efficient technology and the same fuel. If CHP heat generation substitutes heat generation outside the ETS, overall emissions in the ETS energy sector increase. Emissions are shifted from non-ETS sectors to the ETS energy sector. If CHP heat generation substitutes heat generation by other ETS participants using a similarly efficient technology and the same fuel, overall emissions in the ETS energy sector decrease. Secondly, the assumption may be relaxed that CHP plants subject to the ETS substitute electricity and heat generation by ETS participants using similarly efficient technologies and the same fuels. In fact, coal-fired CHP plants may substitute low-carbon electricity and/or heat generation with natural gas. Consequently, overall emissions in the energy sector may increase. An opposite effect of gas generation substituting coal generation is unlikely. Gas-fired plants are usually the marginal producers according to the merit order curve of supply, which will be crowded out of the market first. Thirdly, it may be assumed that new or extended CHP plants are not subject to the ETS. These plants usually substitute electricity generation – and possibly also heat generation – covered by the ETS. Consequently, overall emissions may decrease.

If the eventual effect on emissions in the energy sector is negative, emissions are inefficiently shifted from the energy sector to other ETS sectors. The energy sector abates too much and too costly while other sectors do not employ relatively cheap abatement measures. The reverse is true if the overall effect of the CHP bonus on emissions in the ETS energy sector is positive. The non-energy sector abates too much and the energy sector too little.[72] In either case, the policy mix will result in a reduction of the net value of internalizing the pollution externality. The overall sign of the emissions effect as well as the extent of the resulting inefficiency depend on the total quantity of CHP electricity and heat generated as well as the share of CHP plants covered by the ETS. In this respect, it should be considered that support under the CHP bonus is mainly directed at plants that are smaller than two megawatts and therefore not covered by the ETS. This provides some indication that the overall effect on emissions in the energy sector may be negative. Further important drivers include the characteristics of the energy market (e.g., the merit order curves of electricity and heat supply, the elasticity of electricity and heat demand), the characteristics of the CO_2 allowances market (e.g., marginal abatement costs, the share of German energy sector in the EU allowance market, political and psychological drivers) as well as interactions between these markets (e.g., the impact of the CO_2 allowance price on the electricity price). The discussion of these factors for the case of the feed-in tariff for renewable electricity generation (see Section 6.2.1.8) has revealed that the eventual inefficiency of the policy mix may be rather small. Thus, the inefficiency resulting from the partial overlap of the ETS and the CHP bonus may be assumed to limited as well.

6.4.2 Addressing Climate Change: Overcoming a Pollution Externality and Other Barriers to Technological Change

It has been pointed that energy supply is likely to be locked into a fossil-fuel energy system (see Section 6.2.2). Learning spillovers addressed in the previous section are but one of many barriers contributing to path dependencies in energy supply. Other barriers may reinforce the pollution externality related to GHG emissions as well. In this respect, the implementation of the CHP premium in addition to the ETS may also be justified by other barriers to technological change. Most notably there are considerable inertia in the energy system. Conventional fossil-fuel technologies are characterized by large-scale, long-lived investments, which can be considered as sunk. Consequently, changing to a low-carbon energy system with CHP technologies brings about high opportunity costs. This inertia may be even reinforced by positive network

[72] The statement of inefficiently high emissions in the energy sector may have to be qualified under certain conditions. As has been pointed out, the increase of emissions in the energy sector may be attributed to a shift of heating emissions from non-ETS sectors to the ETS energy sector. This shift may in fact be desirable if the entire German economy is considered – not only the ETS sectors. The shift is efficient if abating heating emissions is cheaper in CHP plants subject to the ETS than in small decentral heating systems. However, this reasoning does not impair the statement of too low emissions in the non-energy sectors subject to the ETS.

externalities in conventional energy systems. Moreover, many CHP plants are operated by new and small market participants. Compared to large conventional energy producers, these may face higher difficulties (and costs) in obtaining capital from banks or the capital market. The CHP bonus may contribute to overcoming these barriers and to increasing the net value of internalizing the pollution externality in the long term.

Moreover, R&D spillovers in the field of CHP technologies and an uneven political playing field may provide additional rationales for having the CHP bonus in addition to the ETS (see Section 6.2.2). Nevertheless, the CHP bonus should only be considered as a second-best solution if other policies addressing these barriers directly are not available. If possible, regulators should rather choose direct R&D subsidies and/or abstain from policies treating conventional energy technologies preferentially.

6.4.3 Addressing Climate Change and Other Policy Objectives

While the ETS only aims at reducing CO_2 emissions efficiently, the CHP bonus pursues a multiplicity of policy objectives (Ziesing 2007b, p. 290, see also Chapter 2.3.4). The objective to save energy by promoting CHP technologies is not only driven by concerns about climate change. It may also help to reduce other environmental problems related to energy generation. A reduced output of electricity and heat is also likely to bring down air pollution and resource consumption in the energy sector. However, at least some part of these benefits may be traded off by an increase of emissions and resource use in other ETS sectors. If the CHP bonus results in a decline of the allowance price, other sectors may respond by increasing output and related environmental effects. Moreover, the decrease of energy consumption in general also reduces Germany's dependency on imported fossil fuels. Thus, the CHP bonus may contribute to a higher security of energy supply. Finally, the CHP bonus can also be interpreted as a tool of economic development. On the one hand, it promotes the development and implementation of CHP technologies. Thereby, it may help to establish an innovative industry of technology producers in Germany, which produce income and employment in the long term. On the other hand, the promotion of CHP plants may also foster competition in the energy market. CHP plants are often operated by small and new market participants. Their extension may thus help to break the oligopoly of the large energy suppliers in Germany – and bring down energy prices in the long term. There are no studies available estimating the possible benefits from attaining these objectives. Nevertheless, the existence of these objectives implies that the policy mix of the ETS and the CHP bonus should not only be evaluated with respect to its efficiency in addressing climate change – but also with respect to pursuing multiple policy objectives.

6.4.4 Policy Recommendation

The previous discussion has revealed an important rationale for complementing the ETS by a CHP bonus. Climate change may be attributed to multiple reinforcing failures of private governance structures requiring multiple policies. The ETS is to correct the pollution externality related to GHG emissions from electricity generation. The CHP bonus addresses barriers to technological change towards clean CHP technologies. The bonus may produce a positive net value of overcoming these barriers. It may thus help to reduce the costs of abatement technologies. In a dynamic perspective, the policy mix may thereby contribute to a higher net value of internalizing the pollution externality as well. The overall benefit of the CHP bonus in this respect depends on the actual extent of the market barriers, e.g. the importance of learning spillovers. Moreover, the CHP bonus may produce additional benefits from attaining other policy objectives than mitigating climate change, e.g. promoting the security of energy supply and competition in energy markets.

In a static perspective, however, the policy mix may bring down the net value of internalizing the pollution externality. In particular, the add-on funding the CHP bonus and the partial overlap of the ETS and the CHP bonus may result in an inefficient distortion of abatement. The actual level of this inefficiency depends on a variety of characteristics of the energy and the allowance market. However, a qualitative estimate provides some indication that welfare losses are relatively small. Consequently, the analysis has provided an indication that the dynamic welfare gains brought about by the CHP bonus exceed the expected static welfare losses with respect to mitigating climate change. Thus, the policy mix of the ETS and the CHP bonus can be justified on economic grounds. Of course, this recommendation based on qualitative considerations calls for a confirmation by quantitative studies the actual extent of dynamic welfare gains and static welfare losses.

The analysis in this section has also provided some insight on how welfare under the policy mix may be even increased: Firstly, the net value of internalizing the pollution externality may theoretically be increased by a modification of the emissions policy. The analysis provides reasons to prefer an emissions tax over an emissions trading scheme as it is used in the EU. Under an emission tax, the emissions price is fixed. Consequently, an inefficient shift of emissions as it occurs under the ETS between participating sectors would not arise with an emissions tax. Secondly, the net value of overcoming learning spillovers may be increased by adapting the CHP bonus continuously as the maturity of technologies and the number of producers increase. To compensate for the steady increase in maturity, the CHP bonus should be made subject to an annual degression. Thirdly, further policies, such as R&D subsidies, may be required in order to address other market barriers to the use of CHP technologies.

6.5 ETS and Low-Interest Loans for Technology Innovation and Diffusion

This section is devoted to analyzing the combination of the ETS with low-interest loans promoting technology innovation and diffusion, including the Environmental Innovation Programme, the Environment and Energy Efficiency Programme and the KfW Programme Renewable Energies. The first part of this section reviews theoretical rationales for implementing this policy mix (Section 6.5.1). The second part derives policy recommendations (Section 6.5.2).

6.5.1 Addressing Climate Change: Overcoming a Pollution Externality and Barriers to Technological Change

A variety of support schemes, including the Environmental Innovation Programme, the Environment and Energy Efficiency Programme and the KfW Programme Renewable Energies, provide subsidies to market participants in the form of low-interest loans. These subsidies are either paid for innovation investments in the field of energy generation and consumption (Environmental Innovation Programme) or the diffusion of energy-efficient and renewable energy technologies (Environment and Energy Efficiency Programme and the KfW Programme Renewable Energies). The combination of these policies with the ETS can be justified on the basis of multiple failures of private governance structures. The pollution externality may be reinforced by failures related to the innovation and diffusion of new, low-carbon technologies. Promoting these technologies may help to overcome these barriers – and increase the net value of internalizing the pollution externality in the long term (see Section 3.3.1). More specifically, the Environmental Innovation Programme may reduce the negative effects of R&D spillovers. The Environment and Energy Efficiency Programme and the KfW Programme Renewable Energies may help to overcome positive externalities of learning-by-doing. Moreover, these policies may also be a means to address other barriers to technological change, such as asymmetric information, inertia due to large-scale, long-time investments in fossil-fuel technologies and an uneven political playing field (see Sections 3.3.2 and 6.2.2).

As has been emphasized for the feed-in tariff for renewable electricity and the CHP bonus (Chapters 6.2.1 and 6.4.1), the dynamic welfare gains due to the policy mix may be traded off by a static decrease of the net value of internalization. In the presence, technologies promoted by the diverse support schemes are relatively expensive means of abating CO_2 in the electricity sector. Under the ETS, their promotion may substitute currently cheaper abatement options in the electricity sector and in other ETS sectors. Thus, the policy mix of the ETS and the support schemes may result in an overall increase of abatement costs. The extent of this inefficiency depends on the characteristics of the electricity market and the CO_2 allowance market as well as interactions between these markets. The actual static welfare loss due to the policy mix may be small – as has been argued for the case of the feed-in tariff for renewable electricity. However, these potential losses have to be taken into account when analyzing the policy mix.

6.5.2 Policy Recommendation

In a dynamic perspective, the policy mix of the ETS and different types of low-interest loans technology innovation and diffusion may be justified on economic grounds. It is necessary to overcome a pollution externality that is coupled with barriers to technological change. The dynamic welfare gains are likely to be higher than static welfare losses produced by the policy mix. The analysis in this section has therefore provided a strong argument for maintaining the policy mix of the ETS and low-interest loans. As has been stated in previous sections, it would be worthwhile to verify this qualitative finding by a quantitative study.

6.6 ETS and Eco-Design Standards

In this section the combination of the ETS with eco-design standards is evaluated economically. As in previous sections, the first part is devoted to the question whether the policy mix can be justified by addressing climate change only (Section 6.6.1). The second part integrates additional policy objectives into the analysis of the policy mix (Section 6.6.2). Finally, policy recommendations are derived (Section 6.6.3).

6.6.1 Addressing Climate Change: Overcoming a Pollution Externality and Barriers to Using Energy-Efficient Appliances

The pollution externality associated with GHG emissions may be reinforced by barriers to using energy-efficient appliances, such as asymmetric information, learning and R&D spillovers and inertia in the use of less energy-efficient appliances. It has been argued that in this case, a single ETS cannot overcome these barriers. Consequently, the ETS alone cannot address climate change efficiently. Rather a policy mix of the ETS and measures to promote energy efficiency is required to provide for an optimal net value of internalizing the pollution externality.

The eco-design standards can be considered a policy to promote energy efficiency. Standards exhibit the particular characteristic that they prescribe the use of certain currently energy-efficient technologies. They do not leave it to the economic actor which technology to choose. They directly remove less expensive and less environmentally friendly technologies from the market (Jaffe et al. 2005, p. 172). In this respect, standards produce a net value of overcoming barriers to adopting energy-efficient equipment. However, this does not necessarily translate in an increase of the net value of correcting the pollution externality – neither in a static nor in a dynamic perspective. Since eco-design standards restrict the choice of technologies, they are likely to produce undesirable costs to some customers at least. This is particularly likely due to the heterogeneity in customers' preferences and circumstances, e.g. regarding usage rate, discount rate and qualitative attributes of technologies (Lund 1978, p. 17). Moreover, eco-design standards fix a certain state of technology and do not provide incentives to improve top-end products in the future (Rolfe et al. 1999, p. 10).

As a consequence, eco-design standards are likely to bring down electricity consumption – but may also increase abatement costs in the electricity sector. Obviously, these effects may also impair the efficiency of the policy mix of the ETS and the eco-design standards. As has been found for other policy combinations, overall emissions are fixed at the emissions cap and not affected by the eco-design standard. Due to the standards, however, emissions may be shifted from the electricity sector to the other ETS sectors. As a consequence, overall abatement costs under the ETS may increase since the electricity sector may abate too much and too costly compared to other ETS sectors. In this case, the energy-efficiency standards should be replaced by other policies which address barriers to energy efficiency, but also leave some discretion to customers. Alternative policies may include information provision, a subsidy to energy-efficient technologies or a tax on energy-inefficient technologies.

Nevertheless, the reasoning against eco-design standards may have to be qualified under certain conditions. Firstly, standards may be found to perform only slightly poorer than more flexible policies if the technology mandated by the standard is superior to other technologies in most cases. Despite the heterogeneity in adopters, this is in fact likely for investments in energy-efficient equipment. Standards may significantly bring down energy consumption of appliances. US energy standards, for example, have lowered the energy use of refrigerators by 60 percent (Rolfe et al. 1999, p. 10). Accordingly, abatement costs with energy-efficient investments have been found to be zero or even negative in 90 percent of the cases. Savings may be as high as 350 Euro per ton of CO_2 abated (McKinsey 2007, p. 37). These massive savings may outweigh possible costs for adopters of using energy-efficient technologies (e.g. due to a decrease in qualitative attributes, such as the difference in hue between fluorescent and incandescent lighting) for many customers. Therefore, the net value of internalizing the pollution externality may be higher under the policy mix than under the single ETS. Secondly, standards do not require that every individual undertake a costly information and assessment process in order to weigh purchase and operation costs of different types of technologies (Jaffe et al. 2005, p. 172). Compared to other policies, eco-design standards may therefore bring down transaction costs – particularly on part of the customers.

6.6.2 Addressing Climate Change and Other Policy Objectives

In contrast to the ETS, the eco-design standards are much broader in their policy objectives. Promoting the energy efficiency of household appliances and electronic devices brings down electricity demand. Apart from mitigating climate change, this reduction may also help to address other environmental problems related to electricity generation, such as air pollution and resource use. Moreover, the eco-design standards are meant to reduce environmentally harmful effects of the entire life cycle of energy-using products. Thus, the focus is not only on electricity generation and related environmental effects. Eco-design standards may produce additional welfare gains, for example, in terms of reducing water consumption in production or the use of toxic or hazardous inputs. The attainment of these multiple policy objectives – rather than only that of mitigating climate change – may provide another rationale for having the policy mix of the ETS and the eco-design standards. It has to be revised, though, whether other environmental problems than climate change related to the life cycle of energy-using products could potentially be addressed more efficiently by additional policies.

6.6.3 Policy Recommendation

The coexistence of a pollution externality and barriers to using energy-efficient technologies provides a rationale for combining the ETS and measures to promote energy efficiency. Eco-design standards can be considered as one such measure. However, economic theory reveals that the eco-design standards are not necessarily an efficient means to increase energy efficiency – neither in a static nor in a dynamic perspective. Alternatively, market-based and information policies should be preferred.

Nevertheless, energy-efficiency standards may significantly bring down energy consumption and increase welfare for most customers of energy-using products. Therefore, the actual extent of the inefficiency may be low. Moreover, standards may exhibit advantages in terms of transaction costs compared to alternative policies. In addition, eco-design standards produce benefits with respect to other environmental issues than climate change.

These considerations reveal that the overall welfare balance of the policy mix may be positive. Consequently, the analysis suggests that the combination of the ETS with eco-design standards is desirable from the point of view of efficiency. Again, it is necessary, though, to confirm this qualitative finding by an empirical quantitative study.

6.7 ETS and Energy Labelling for Household Appliances

The following section is devoted to the economic analysis of interactions between the ETS and the requirement to label the energy consumption of household appliances. The analysis draws largely on the theoretical discussion of rationales for using a policy mix in Chapter 3. It also includes some insights gained for other types of policy combinations in previous chapters. In the first part, it is examined whether the policy mix of the ETS and the energy label contributes to efficiency in addressing climate change (Section 6.7.1). Subsequently, it is analyzed whether other policy objectives than mitigating climate change may also justify the policy mix (Section 6.7.2). Finally, policy recommendations are derived (Section 6.7.3).

6.7.1 Addressing Climate Change: Overcoming a Pollution Externality and Asymmetric Information

Section 3.3.2 has outlined that a policy mix of an emissions policy and energy labelling may increase efficiency in the presence of a pollution externality and asymmetric information between market participants. In the case of household appliances, buyers can be assumed to be less informed about the electricity consumption of these appliances than suppliers. In their decision on whether or not to buy an appliance, buyers are therefore unable to perfectly consider the long-term energy costs associated with an appliance. Instead, their focus is on purchase costs. Consequently, buyers tend to purchase appliances with relatively low purchase costs, which may have relatively high long-term operation costs resulting from electricity consumption. Asymmetric information can therefore be considered as one important reason for the so-called "energy-efficiency gap" (Sutherland 1991; Jaffe and Stavins 1994; Sanstad and Howarth 1994; Howarth and Sanstad 1995).

Asymmetric information may reinforce the pollution externality related to GHG emissions. Choosing less energy-efficient appliances increases electricity consumption and corresponding GHG emissions. In the absence of asymmetric information, the pollution externality would be less severe. This is particularly likely since abatement costs associated with installing energy-efficient equipment are zero or even negative in many cases (McKinsey 2007, p. 37). In turn, this finding implies that correcting the pollution externality is cheaper – or more efficient – when asymmetric information is overcome. In this respect, the ETS alone, which aims at reducing the negative pollution externality, does not address climate change efficiently. Indeed, it increases the electricity price and sets incentives to save electricity (Sijm and van Dril 2003, pp. 55-58; Sijm et al. 2005; Sijm et al. 2006; Ziesing 2007b, pp. 305-334). However, electricity consumers are unable to make efficient choices with respect to energy-efficient technologies in the presence of asymmetric information. Asymmetric information thus increases the abatement costs under the ETS. Consequently, the net value of correcting the pollution externality with the ETS decreases.

Therefore, combining the ETS with an information policy provides for a higher net value of correcting the pollution externality. The ETS sets incentives to save electricity. The energy label for household appliances provides electricity consumers with the information. Thus, consumers are able to respond efficiently to higher electricity prices by choosing the most energy-efficient technologies.

As has been stated for the other policy combinations in previous chapters, the policy mix of the ETS and energy labelling is also characterized by interactions in the allowance market. The label is likely to result in a decline of electricity consumption. Although, this effect has not been estimated for the energy label, there is some evidence for other similar information programs (see Section 3.3.2.2). A decrease in electricity consumption brings about fewer emissions in the electricity sector and a reduction of the allowance price. Other ETS sectors take advantage of the declining allowances price and increase their emissions. The overall emissions level is fixed at the ETS emissions cap. Just like other policies in the electricity sector, such as the electricity tax or the feed-in tariff for renewable electricity, the implementation of the energy label on top of the ETS shifts emissions from the electricity sector to other ETS sectors. In contrast to other policy mixes, however, this shift does not bring about a static inefficiency in emissions abatement. On the contrary, the policy mix of the ETS and the energy label reduces the cost of abating CO_2 emissions (Sorrell and Sijm 2003, p. 432). The energy label does not involve mandatory abatement measures. Given better information, electricity consumers choose only those energy-efficient investments which pay off at the prevailing electricity price (including the allowance price). Therefore, abatement costs of measures induced by the energy label do not exceed the allowance price. More likely, they are far below the allowance price. The McKinsey (2007, p. 37) study estimates that 90 percent of energy-efficiency investments in the building sector (including household appliances) have a positive payoff for the investor. The application of energy-efficient household appliances, consumer and communication electronics, office equipment and lighting exhibits an abatement potential of 16 megatons of CO_2 emissions. Savings from these abatement options may range from 25 to 350 Euro per ton of CO_2. The implementation of the energy label may help to use at least a part of this potential. It activates cheap abatement options in the electricity sector. Consequently, the policy mix of the ETS and the energy label always increases the net value of internalizing the pollution externality compared to a single ETS.

The increase in the net value of internalization is brought about, however, by an increase in transaction costs of information provision. These costs are mainly incurred by the suppliers of household appliances, which have to provide information in line with the requirements of EU and German legislation. However, transaction costs can be assumed to be small compared to the increase in the net value of internalization resulting from information provision. Thus, there is strong evidence that the policy mix of the ETS and energy labelling does overall increase the efficiency of GHG emissions abatement.

Nevertheless, the energy label is subject to restrictions. It provides buyers with information on the energy consumption in kilowatt-hours. This value provides some indication of long-term operation costs with an appliance. Nevertheless, buyers may find it difficult to compare monetary values (given directly for purchase costs) and technical values (given as a proxy for operation costs). As a result, buyers may make suboptimal decisions with respect to energy-efficiency. That is, the net value of overcoming asymmetric information is suboptimal. The disclosure of lifecycle costs is discussed as an approach to overcome this issue (see, e.g., Lund 1978). Lifecycle costs represent purchase costs as well as discounted operation costs. They allow for a direct comparison of total costs of different appliances. Indeed, the disclosure of lifecycle costs is also subject to challenges. Lifecycle costs are subject to uncertainty about future energy and fuel prices. Moreover, lifecycle costs vary with customers' preferences and situations. Differences in the usage rate, the discount rate or regional climate may produce different levels of lifecycle costs for the same appliance (Lund 1978, p. 17). These differences may be more significant for some appliances, such as washing machines, than for others, such as fridges. A uniform representation of lifecycle costs as part of a label may therefore also produce a suboptimal net value of correcting asymmetric information and pollution externalities. Individualizing lifecycle costs by more detailed decision support schemes increase transaction costs for buyers of appliances. These concerns should be born in mind when designing labels. However, they do not provide a general rationale against including lifecycle costs. Deutsch (2007) shows that even a relatively simple, uniform disclosure of life-cycle costs for a representative customers may significantly reduce electricity consumption of the chosen appliances.

Finally, it has to be highlighted, though, that asymmetric information is not the only factor causing the energy-efficiency gap. Thus, energy labels cannot overcome all obstacles to using energy-efficient household appliances. Other barriers may include spillovers related to technological development and diffusion, sunk investments in less energy-efficient technologies or the qualitative attributes of new technologies (e.g. the difference in hue between fluorescent and incandescent lighting) (see, e.g., Jaffe and Stavins 1994). In order to overcome these barriers, further policies, such as subsidies, may be necessary.

6.7.2 Addressing Climate Change and Other Policy Objectives

In Chapter 2.3.7, it has pointed out that by labelling energy consumption of household appliances, the regulators pursues the broad goal of reducing energy consumption. As has already been emphasized before, the reduction of energy does not only provide benefits with respect to mitigating climate change. It may also reduce air pollution and resource conservation in the electricity sector – even though some of these effects may be traded off by higher production and emissions in other sectors participating in the ETS. Moreover, a reduction of energy consumption also reduces the dependency on energy imports and increases the security of energy supply. Finally, the energy label explicitly considers water consumption. In this respect, it may also produce benefits in terms of reduced water consumption and wastewater generation. A comprehensive evaluation of the policy mix of the ETS and the energy label has to take these welfare gains into account as well.

6.7.3 Policy Recommendation

The analysis has demonstrated that an emissions trading scheme and information measure should be combined when a pollution externality is reinforced by asymmetric information between market participants. Both types of failures of private governance structures occur in the German electricity sector. Consequently, the existing policy mix of the ETS and the energy label for household appliances can be unambiguously justified on efficiency grounds. The label may help to activate at least a part of the large and mainly economic abatement potential related to improving energy efficiency. The transaction costs related to implementing the energy label can be assumed to be relatively small. Thus, the policy mix may help to overcome the energy-efficiency gap, which deteriorates the efficiency of a single ETS. Nevertheless, the policy mix could be improved by providing additional information. In particular, labels should also include explicit information on lifecycle costs associated with appliances. Moreover, additional policies may be necessary to overcome further barriers contributing to the energy-efficiency gap.

6.8 ETS and Voluntary Agreement

The evaluation of the policy mix of the ETS and the voluntary agreement between the Federal Government and German industry associations is carried out in three steps. Light is first shed on the issue whether the policy mix can efficiently correct a pollution externality associated with GHG emissions (Section 6.8.1). Subsequently, possible other policy objectives than climate change are considered in the evaluation (Section 6.8.2). Finally, the section concludes with policy recommendations (Section 6.8.3).

6.8.1 Addressing Climate Change: Overcoming a Pollution Externality

It is first assumed that the ETS and the voluntary agreement are meant to address a pollution externality. Both policies directly aim at reducing GHG emissions. However, there are two systematic contradictions between the ETS and the voluntary agreement. Firstly, the ETS is characterized by a flexible allocation of abatement measures based on tradable allowances. The eventual allocation is driven by market allowance prices and individual abatement decisions. In contrast, the voluntary agreement determines a fixed allocation of abatement measures between as well as within sectors. In this respect, the agreement is quite similar to classical command-and-control policies (Döring and Ewringmann 2003, p. 19). Secondly, the ETS sets an absolute cap on emissions. By contrast, the voluntary agreement determines specific emissions targets for most sectors. That is, absolute emissions are allowed to increase with output (Walz and Betz 2003, p. 8). Nevertheless, these contradictions do not necessarily imply that the policy mix produces a welfare loss compared to a single ETS. Rather, the efficiency of the policy mix depends on the actual overlap of both policies, the binding character of the agreement and the stringency of the agreement compared to the ETS.

The voluntary agreement on emissions reductions covers all CO_2 emissions of ETS participants. However, the agreement also goes beyond the ETS. It refers to other GHGs than CO_2 as well. It includes industry sectors which are not participating in the ETS, such as the chemical, aluminium, electrical engineering, textile and sugar industries (RWI 2008, pp. 10-11). Moreover, the agreement is also directed at installations of ETS sectors which are exempt from the ETS due to their minor capacity or output. It applies, for example, to energy installations smaller than 20 megawatts. The voluntary agreement on promoting the use of CHP plants only affects the energy sectors participating in the ETS. Thus, the voluntary agreement is overlapping with the ETS. However, it also covers a variety of GHGs and emitting sources which are not subject to the ETS. At least for these gases and sources, it may be desirable to have the agreement in addition to the ETS.

Whether abatement choices of ETS participants are distorted by the voluntary agreement then depends on whether the agreement is actually binding. If this is not the case, the agreement is obviously obsolete. It does not impair the efficiency of the ETS because ETS participants make their decisions on the basis of the allowance price only. In fact, the ETS may produce additional incentives not to obey the rules of the agreement since some firms may find it more profitable to buy allowances than to actually reduce emissions. Walz and Betz (2003, pp. 7-8) point out that it is very unlikely that the voluntary agreement is binding due to lacking sanctions. Moreover, they find that a particular incentive not to reduce emissions arises from the fact that the emissions reduction agreements have been adopted for industry sectors, not for individual firms. This implies that within a sector, firms can free-ride on emissions reductions of other firms. An abating firm has to bear the full costs of emissions reduction and helps other (competing) firms to fulfil the sectoral target. In this respect, the firm produces a positive externality and has too small an incentive to do more than business-as-usual. These arguments may be countered, though, by the fact that the voluntary agreement has a self-enforcing component. If the objectives of the agreement are not met, the Federal Government is allowed to implement stricter domestic policies, such as command-and-control policies or mandatory energy audits (see Section 2.3.8). Thus, the agreement may be reinforced due to the threat of more stringent policies. This is what Sorrell et al. (2003, p. 41) understand as "sequential interaction of policies". In fact, the RWI (2008, p. 18) monitoring report finds that eight out of the 21 agreements to reduce emissions by 2012 had already been fulfilled in 2007. Another 11 agreements had already attained 80 percent of the target level. A remarkable exception in this record is the electricity sector. This sector had only achieved 50 percent of the agreed emissions reductions by 2007. However, the agreement refers to the year 2015 instead of 2012. Nevertheless, there is some doubt whether the electricity sector actually perceives its agreement as binding. This may be particularly due to the fact that agreed emissions reductions are dependent on the Federal Government's promise not to restrict the use of nuclear power. The Government's decision to phase out nuclear power may allow electricity generators to relax their agreement.

Even a binding voluntary agreement does not necessarily bring about inefficiency in emissions abatement. Three cases can be distinguished. Firstly, it can be assumed that the stringency of the agreement is lower than or equal to the ETS for all ETS sectors. Thus, the emissions level of each sector which would be chosen under the ETS is below or equal to the level agreed upon with the Federal Government. In this case, the allowance price is not affected by the voluntary agreement. The overall emissions level corresponds to the ETS cap. The efficiency properties of the ETS prevail. Secondly, the agreement may be more stringent than ETS for some ETS sectors and less stringent for others. In this case, sectors subject to very stringent agreements reduce their emissions more strongly than they would under the ETS only. They demand less allowances, and the allowance price decreases. Other ETS sectors which are subject to less stringent agreements benefit from this price decline and emit more. Consequently, emissions are shifted from ETS sectors with more stringent agreements to those with less stringent agreements. The overall emissions cap still binds. However, abatement costs increase. Thus, the net value of internalizing the pollution externality would be reduced by the policy mix as compared to a single ETS. Thirdly, agreements may be more stringent than the ETS for all ETS sectors. In this case, the allowance price falls to zero. The ETS becomes obsolete. The overall emissions level as well as the costs of attaining it are only determined by the properties of the voluntary agreement.

It is obvious that the latter case has not occurred in Germany since allowance prices have been positive. In fact, there is some evidence that the agreements are less stringent than the ETS in general. The electricity sector agreed to reduce absolute emissions to 264 million tons in 2015. During the second phase of the ETS, however, the theoretical allocation to electricity sector only amounted to 243.59 million tons (or allowances) (see Section 2.3.1).[73] Similar data cannot be derived easily for other ETS sectors. It has to be taken into account, though, that most of the agreements for other sectors refer to specific emissions. Thus, overall emissions increase with economic growth of the sector. This provides at least some indication that the agreements may be less stringent than the ETS. In this case, the voluntary agreement would not impair the inefficiency at all. Otherwise, if some sectors appear to have more stringent agreements, emissions would be shifted inefficiently from these sectors to the electricity sector. As has been stated in previous sections, the actual extent of this inefficiency depends on the characteristics of the CO_2 allowance market (see Section 6.2.1.8 for an extensive discussion).

An additional reduction in the net value of internalization may result from the agreement on CHP generation. An extended installation of CHP plants may reduce emissions in the ETS electricity sector and result in an inefficient shift of emissions from the electricity sector to other ETS sectors. However, identifying the actual extent of the agreement on CHP generation may be

[73] The actual allocation to the electricity sectors was lower. The amount was reduced by 38 million allowances to be auctioned and subject to an additional reduction factor allowing for the cap to be met. Using the allocation of allowances to estimate the level of emissions under the ETS is somewhat flawed, though. Due to allowance trades, actual emissions may well deviate from the initial allocation. Nevertheless, data from the UBA (2009, p. 84) reveal that actual emissions were pretty close to the number of allowances allocated in the first ETS period from 2005 to 2007.

difficult. It has to be separated from the impact of the CHP bonus (see Chapter 6.4).[74] Concluding, it can be stated that for the ETS sectors the voluntary agreement represents a redundancy at best and an inefficient policy mix at worst.

6.8.2 Addressing Climate Change and Other Policy Objectives

As many of the energy and climate policies, the voluntary agreement is not only addressing climate change but also other related issues (see Chapter 2.3.8). A reduction of greenhouse gases is also expected to bring down other types of emissions as well as resource consumption. Moreover, the switch to CHP generation may also help to reduce Germany's dependency on fossil fuel imports, and thus to increase the security of energy supply. Possibly related welfare effects have to be taken into account when the policy mix is to be evaluated. It may be doubted, however, whether the agreement actually results in benefits that are additional to those produced by other policies. Overall CO_2 emissions reductions and related effects are determined by the ETS cap. The switch to CHP technologies can be assumed to be driven mainly by the CHP bonus than by the voluntary CHP agreement. Therefore, only those benefits can be actually attributed to the agreement which are related to the reduction of other GHGs not covered by the ETS.

Moreover, there are also political economy arguments for having the voluntary agreement. The agreement to reduce GHG emissions is coupled to the tax rebates for the manufacturing sector under the electricity tax as well as other taxes of the Ecological Tax Reform (see Section 2.3.3). Full tax rates for the manufacturing sector would not have been politically feasible. Firms were afraid of being put at a competitive disadvantage in world markets. Therefore, the policy mix of the tax and the agreement was to ensure that firms actually reduce emissions without intervening too strongly in their decisions. This deal had been agreed upon before the ETS was implemented in the EU. As has been shown in this section as well as in Section 6.3, however, both the voluntary agreement and the electricity tax become obsolete with the ETS – at least when it comes to CO_2 emissions of ETS participants. The political economy rationale for having the policy mix therefore only holds for CO_2 emitters outside the ETS and emissions of GHGs other than CO_2.

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[74] In fact, the coexistence of the CHP agreement and the ETS could produce additional welfare gains if the pollution externality is reinforced by barriers to the adoption of CHP technologies. This issue is not discussed in this context, since the CHP bonus can be assumed as the major policy to address that issue (see Section 6.4).

Likewise, political economy arguments also played a role in implementing the voluntary agreement to promote CHP. Political resistance from the energy sector prevented a CHP quota from being implemented. Instead, the voluntary CHP agreement was adopted. In order to help its implementation, it was supplemented by the CHP bonus (see Section 6.4) (Ziesing 2007b, p. 288). It can be assumed that the diffusion of CHP technologies is mainly driven by the bonus, rather than by the voluntary agreement. Consequently, there is not really a need to have both policies in place.

6.8.3 Policy Recommendation

The previous analysis has shown that two scenarios are possible with a combination of the ETS and the voluntary agreement: (1) If the voluntary agreement to reduce CO_2 emissions is binding and more stringent than the ETS for at least some participants, it impairs the inefficiency of the ETS. (2) If, however, the emissions reduction levels set under the agreement are below those levels chosen by firms and sectors under the ETS, the agreement does not have any impact on the efficiency of the ETS. It may therefore be concluded that the combination of both policies is redundant at best and inefficient at worst when it comes to CO_2 abatement. The net value of internalizing the externality related to CO_2 emissions under a policy mix is equal to or below the net value with a single ETS. Consequently, there are strong arguments for abolishing the voluntary agreement – at least for those CO_2 emissions and emitters covered by the ETS. Only for emitters outside the ETS and emissions of other GHGs than CO_2, it may be useful to maintain the voluntary agreement. It may be doubted, though, that related benefits of emission reduction are large due to the lack of enforcement of the voluntary agreement. Moreover, it should be revised whether these emitters and emissions can be or are already addressed more efficiently by other policy measures.

6.9 Summary

This chapter has addressed the evaluation of an actually existing policy mix: the combination of climate policies to reduce GHG emissions in the German electricity sector. For this purpose, the results of the theoretical analyses in the preceding chapters have been applied and interpreted for the empirical German case. It has been shown that on this basis, it is possible to derive valuable insight on the performance of the German policy mix and to provide reasonable policy recommendations. The evaluation of the policy mix has been further strengthened by the consideration of the concrete design of the different policies presented in Chapter 2 (objectives, scope, rules) as well as their institutional environment. The result has been the most extensive evaluation of an existing policy mix available so far in the economic literature.

The comprehensive examination of combinations of the ETS with each of the other climate policies implemented in Germany for the electricity sectors has given an ambiguous picture of the German policy mix. On the one hand side, there is a variety of policy combinations which can be justified on efficiency grounds – even if the complex design of the policies is taken into account. Thus, this chapter has again delivered clear evidence that a *per se* rejection of policy mix approaches is flawed. On the other hand, there are also some policy combinations which have been found to be inefficient – even when a broad perspective with multiple possible policy objectives is employed.

Generally, is has been pointed out that the overall CO_2 emissions level of ETS participants is determined by the ETS emissions cap. Additional policies in the electricity sector do not bring about additional emissions reductions. They only have an effect on the allocation of abatement measures across ETS participants. This effect may increase efficiency – or the net value of internalizing the pollution externality related to CO_2 emissions – if one of the rationales for using a policy mix applies (see Chapter 3). If no rationale applies, the policy mix may inefficiently distort abatement. Nevertheless, it has also been shown that an evaluation of an existing policy mix on the basis of theoretical rationales only is not sufficient. A comprehensive policy mix analysis has to take into account the complex characteristics of real-world policy mix design. This complexity makes it a tedious task to derive an overall evaluation of a policy mix and corresponding policy recommendations. This chapter has coped with this challenge and demonstrated that even a qualitative analysis can provide a variety of important indications and conclusions. It may be worthwhile to verify and confirm these in a quantitative setting. Nevertheless, the implications for policy-making are striking. The existing set of policy combinations of the ETS and a second policy can be divided roughly into three groups (see Table 6-1).

Table 6-1: Evaluation of different policy combinations for the German context

Welfare effects of supplementing the ETS by policy X are …		
… clearly positive	… likely to be positive	… likely to be negative
▪ Energy labelling for household appliances ▪ Low-interest loans promoting technology innovation and diffusion	▪ Feed-in tariff for renewable electricity generation ▪ Bonus for combined heat and power generation ▪ Eco-design standards	▪ Electricity tax ▪ Voluntary agreement

Firstly, there are some combinations which clearly increase welfare. This refers to combinations of the ETS with energy labelling and low-interest loans promoting technology innovation and diffusion (Environmental Innovation Programme, the Environment and Energy Efficiency Programme and the KfW Programme Renewable Energies). These combinations can be justified on the basis of multiple failures of private governance structures. In these cases, the specific characteristics of policy design hardly impair the theoretical superiority of the policy mix.

Secondly, there is a group of policy combinations for which there is at least an indication that the policy mix increases welfare. This group includes combinations of the ETS with the feed-in tariff for renewable electricity, the CHP bonus and the eco-design standards. For these types of a policy mix a theoretical rationale for using a policy mix applies. However, the net value of internalizing the pollution externality is reduced because of deviations from the theoretically optimal policy design. These deviations may have different consequences. Dynamic welfare increases may be offset by static welfare losses. Increases in the net value of internalization may be traded off by higher transaction costs. Nevertheless, the analyses carried out in this chapter have revealed some evidence that the overall effect of these policy combinations on welfare is likely to be positive. It may be useful, though, to confirm these qualitative findings by empirical quantitative studies. Moreover, a variety of modifications and additions have been suggested:

- In general, an emissions tax should be preferred to the ETS.
- The expected emissions reductions due feed-in tariff and the CHP bonus have to be considered when the ETS emissions cap is determined.

- The level and differentiation of the feed-in tariff and the CHP bonus have to be revised and adjusted continuously.
- The CHP bonus has to be made subject to annual degression.
- The feed-in tariff and the CHP add-on have to be complemented by R&D subsidies for renewable energy and CHP technologies as well as other new and innovative technologies, such as highly efficient coal- and gas-fired power plants, carbon capture and storage or nuclear fusion.
- Environmentally harmful subsidies and regulations, such as subsidies to hard coal, which hamper the development and adoption of low-carbon technologies, have to be abolished.

Thirdly, some climate policy combinations are likely to reduce welfare compared to a single ETS. The two striking examples are the combinations of the ETS with the electricity tax and the voluntary agreement. In both cases, the policy mix cannot be justified as a means to reduce CO_2 emissions efficiently. Just as the ETS, the electricity tax and the voluntary agreement are meant to correct the pollution externality related to CO_2 emissions directly. Neither do these policies address additional failures of private governance structures which may reinforce the pollution externality. Nor are they a means to reduce transaction costs of the ETS. With respect to CO_2 emissions abatement, the combination of the ETS with the electricity tax and the voluntary agreement consequently represents a redundancy or even an inefficient policy mix. Moreover, it is shown that other policy objectives than mitigating climate change provide an only weak rationale for having the policy mix. Consequently, there is a strong indication that the electricity tax and the voluntary agreement should be abolished for ETS participants.

7 Conclusion: Major Results, Transferability and Avenues for Future Research

Nowadays, pollution control strategies are based on a policy mix in many OECD countries. In order to address a single pollution problem, a set of policies – rather than only one – is employed. Within this mix, policies may complement each other. However, there is also the risk of adverse interactions between policies. Consequently, the question has to be raised whether existing pollution control strategies represent a useful policy mix – or rather an undesirable policy mess.

Answering this question is a particularly important challenge for the case of climate policies, which have been implemented by governments worldwide. Climate change is one of the most outstanding and urgent pollution problems. In order to mitigate climate change, GHG emissions have to be reduced significantly in the near future. Consequently, an elaborate set of measures is needed to actually induce these emission reductions. Climate policies implemented so far have been found to have had only limited success. Therefore, taking a closer look at the interplay of these policies is a primary research task.

In order to enrich climate policy research empirically, it is particularly promising to refer to the case of the policy mix implemented in the German electricity sector. This policy mix is highly representative for most industrialized countries. On the one hand, this is due to the fact that several German policies have been mandated by EU Directives and can therefore be found in other EU Member States as well. On the other hand, many German policies have served as an example, and have been "exported" worldwide. As a consequence, the German policy mix encompasses virtually the entire range of policy types that are currently used in the industrialized world. This implies that findings made for the German context are of large interest for policy decisions taken in other countries as well. Moreover, the analysis of the climate policy mix in the German electricity sector is also highly demanded because of the large importance and urgency of mitigation measures in this sector.

From an economic perspective, it is particularly necessary to examine the efficiency of a policy mix. Does the simultaneous use of multiple policies allow attaining an optimal level of pollution control at least cost? In the past, economists have not invested much effort in addressing that question. Their main focus has been on the analysis and comparison of single policies. Those studies actually analyzing the issue of a policy mix have been subject to important restrictions. Firstly, they have been characterized by a large diversity in backgrounds and assumptions. They have not revealed overarching rationales for using a policy mix. Secondly, they have been carried out on an abstract level assuming simplified policy designs and institutional environments. Consequently, these studies have hardly been able to derive useful guidelines for policy-makers operating under real-world conditions.

In this respect, this book has made three substantial contributions to improving the economic understanding of a policy mix. First of all, it has uncovered a set of general economic rationales for using a policy mix for pollution control. If these rationales apply, a policy mix may be superior to a single policy in terms of efficiency. Secondly, the economic evaluation of selected policy combinations has been approached to reality. Simplistic assumptions about policy mix design have been substituted by more complex and more realistic ones. Both contributions represent substantial advances of economic theory. The book's third contribution has consisted in applying these theoretical advances to the analysis of the German policy mix. Thereby, the book has provided the most extensive evaluation of an actually existing policy mix available.

The book's three major results are summarized in the following Section 7.1. Section 7.2 discusses the transferability of these findings to other policy contexts. Subsequently, avenues for future research in the field of economic policy mix analysis are highlighted in Section 7.3. Finally, Section 7.4 gives a brief outlook.

7.1 Major Results

7.1.1 Identification of Economic Rationales for Using a Policy Mix for Pollution Control

As a first important contribution to economic theory, this book has clarified under which general conditions a policy mix may be more efficient than a single-policy strategy. For this purpose, the first extensive review of economic literature on using a policy mix has been carried out. The focus has been on analyses which put forward economic arguments for combining policies for pollution control. In order to organize this diffuse body of existing policy mix studies, an innovative analytical framework has been developed. The framework has opened the economic analysis of policies and their combinations for ideas from New Institutional Economics. Based on the seminal work by Coase (1960), the concept of transaction costs has been introduced into the analysis of pollution problems and possible means to overcome these problems. The framework explains the existence of pollution problems by the failure of private governance structures including the market, the firm or bilateral bargains. Thus, the framework goes beyond the classical market failure concept dominating Environmental Economics. Public governance in the form of regulation may be needed to correct not only for a market failure but also for the failure of other private governance structures. It has to be verified, though, that the balance of the net value of correcting a failure and the related transaction costs is positive under regulation. The net value of correcting a failure corresponds to the benefits of correction net of the costs of correction. For the example of a pollution externality, the net value equals the reduction of pollution damages net of abatement costs.

This framework has been very helpful in overcoming the complexity and diversity of existing policy mix studies. Two overarching rationales for using a policy mix for pollution control have been derived. Firstly, a policy mix may be necessary when pollution problems are attributed to multiple reinforcing failures of private governance structures. In fact, a pollution externality may be aggravated by technology spillovers and/or asymmetric information. In this case, one policy may be needed for each failure respectively. However, a basic requirement is that the balance of the net value of correcting each failure and the related transaction costs be positive under the policy mix and larger than for any private governance structure. Secondly, a policy mix may be an efficient strategy when single policies to address a pollution externality would exhibit high transaction costs. Transaction costs of designing a policy efficiently may be high due to heterogeneous marginal pollution damages or heterogeneous marginal abatement costs. Transaction costs of implementing a policy may be high due to monitoring and enforcement needs. In these cases, the implementation of additional policies or a set of alternative policies may bring down transaction costs while producing a similar net value of internalization. However, this cannot be taken for granted. The regulator has to make sure that the balance of the net value of internalization under the policy mix and related transaction costs is actually positive and higher than for a single policy or private governance structures.

With the identification of these general rationales for using a policy mix, this book has provided clear arguments against the perception that a pollution problem can always be addressed efficiently by a single policy – a perception that is widespread among economists. It has been shown that the superiority of a policy mix over a single policy is not an extraordinarily specific and rare case. In fact, a policy mix may be justified in many circumstances. Multiple failures of private governance structures and high transaction costs of single policies can often be observed. If one or several of these rationales apply, this may be a first indication for decision-makers that a policy mix may outperform single-policy strategies in terms of efficiency. However, the book has also made the important point that the concrete design is decisive for the actual performance of the policy mix. This qualification is directly linked to the second major result of the book.

7.1.2 Integration of Design Complexity into Models of a Climate Policy Mix

The second important contribution of the book has consisted in approaching economic analysis of a climate policy mix to real world conditions. For this purpose, partial-equilibrium models for the analysis of a policy mix have been advanced. Most importantly, the simplistic assumptions about policy design have been replaced by the more complex features of actually existing climate policies. These modifications have been inspired by the German case study. That is, they reflect outstanding characteristics of climate policies implemented in the German electricity sector. The analyses have clearly demonstrated that the consideration of complex policy design is crucial for an appropriate economic evaluation of a policy mix. In fact, it has been shown that the consideration of complex real policy design may deliver results and policy recommendations that are contrary to those derived under a simplistic model. This result has demonstrated that the consideration of complex real-world policy design is decisive to increase the relevance of economic policy mix research. The book has focussed on the analysis of two selected policy combinations: an emissions trading scheme supplemented by a feed-in tariff to promote renewable electricity technologies, or a tax on emissions or output.

It has been shown that the combination of an emissions trading scheme and technology support can be justified by multiple failures of private governance structures: if a pollution externality resulting from GHG emissions is reinforced by learning spillovers related to renewable energy technologies. The emissions trading scheme addresses the pollution externality, while the technology support scheme corrects for the spillover effects. Thereby, direct technology support may help to reduce emissions at less cost in the future. Thus, the policy mix may produce dynamic welfare gains compared to a single emissions trading scheme. Existing studies suggest using a simple output subsidy to promote new renewable energy technologies. In reality, however, a more complex feed-in tariff has been implemented, e.g. in Germany. Two deviations between actual feed-in tariff design and the theoretical concept of a simple output subsidy are striking: The tariff is paid to renewable electricity generators instead of the electricity price rather than in addition to it. Moreover, the tariff is not funded by general tax revenues but by an add-on to the electricity price. Both deviations in policy assumptions have been incorporated in a partial-equilibrium model of the electricity sector. The results found for a policy mix with a feed-in tariff differ significantly from those derived in a model with a simple output subsidy. The optimal emissions price is below the Pigovian level and has to be differentiated across fuels. Moreover, the emissions price as well as the feed-in tariff have to be adapted continuously as characteristics of the electricity market and the renewable energy technologies change. These findings imply that an efficient policy mix is theoretically possible with a feed-in tariff. However, its implementation is much more cumbersome for the regulator than with a simple output subsidy. If the regulator tries to bring down transaction costs, e.g. by abstaining from continuous adaptations of the policies, this comes at inefficiency in abatement and technology development. Consequently, actual policy design has been found to challenge the theoretical superiority of the policy mix.

In contrast to this first example, the combination of an emissions trading scheme and a tax on emissions or output with additive incentives cannot be justified by a theoretical economic rationale. With additive incentives, firms are obliged to hold emission allowances and pay a tax in addition which is directly or indirectly related to emissions. Existing studies argue that this second example of a climate policy mix is inefficient when taxation is heterogeneous across emissions trading sectors – and redundant otherwise. Consequently, the abolition of the tax is recommended. This conclusion is subject to the important restriction that both policies are only meant to reduce GHG emissions efficiently. In reality, however, the tax may address a variety of further objectives and criteria, such as revenue generation (e.g. in Germany) and equity concerns. Consequently, the abolition of the tax may not be desirable or feasible in the real world. This book has therefore promoted a change in perspective: The economic analysis of climate policy has been rethought under the constraint that the tax has to remain in place. Three issues, which are neglected by existing studies, have been analyzed in detail. Firstly, the major drivers of inefficient abatement under the policy mix have been identified. This insight is necessary to compare welfare losses with respect to mitigating climate change with benefits from meeting other policy objectives and criteria. Variables that are decisive in this respect include the heterogeneity of taxation across emissions trading sectors, the size of these sectors, the total number of firms in the industry and the shape of the marginal abatement cost curve. In addition, welfare losses have been found to be more severe when output is taxed instead of emissions. Moreover, the inefficiency of the policy mix may be ameliorated or deteriorated by interactions between the output markets of high- and low-taxed emissions trading sectors. Secondly, the policy mix has been compared with a single heterogeneous tax. It has been shown that welfare losses under the policy mix should not be attributed to interactions between policies but to the inefficient design of the tax. In fact, the implementation of an emissions trading scheme on top of an existing tax is likely to mitigate abatement distortions induced by the tax. Thus, the implementation of a policy mix is economically desirable when the tax cannot be abolished. Thirdly, it has been shown that the inefficiency of the policy mix can be reduced by modifications in the design of emissions trading. A simple solution separates allowance markets for sectors with different tax rates, over-allocates allowances to the sector with the higher tax and permits allowance trading across sectors at a certain ratio only.

These results reveal that the assumptions made about policy design are decisive for outcome of economic policy mix analyses. Approaching them to reality is necessary to increase the relevance of corresponding policy recommendations. The integration of complex design features into policy mix analysis qualifies theoretical arguments – both in favour and against using a policy mix – which are made under simplistic models. Policy combinations are likely to exhibit advantages and disadvantages in terms of efficiency which have to be balanced carefully on an empirical basis. In this book, this has been done for the representative case study of the policy mix in the German electricity sector. The related results represent the third important contribution of this book.

7.1.3 Evaluation of the Climate Policy Mix in the German Electricity Sector

The third major contribution of this book has been the extensive evaluation of the policy mix in the German electricity sector. For this purpose, the theoretical insight derived in this book has been applied to the German case study. Moreover, the perspective of the theoretical discussions has been further broadened in order to consider the full complexity of German policy design (objectives, scope and rules) as well as the characteristics of the electricity and CO_2 markets and the institutional environment. The evaluation has been carried out for combinations of the European ETS with all other policies in place in the German electricity sector. The result has been the most comprehensive, theory-guided analysis available for an existing policy mix. Based on this analysis, clear and concrete policy recommendations have been derived for the German case study.

The picture of the German policy mix that has been drawn in this book is ambiguous. The policy mix encompasses examples of both efficient and inefficient policy combinations. Generally, it has been shown that the overall level of GHG emissions is determined by the emissions cap under the ETS. The effect of supplementing the ETS by further policies is either an increase or a decrease in the costs of attaining the emissions cap. Three classes of policy combinations have been distinguished: Firstly, the combinations of the ETS with energy labelling and low-interest loans promoting technology innovation and diffusion have been found to be clearly positive in terms of efficiency. These combinations can be justified by economic rationales, and their performance is not impaired by certain specific features of policy design. Secondly, the combinations of the ETS with the feed-in tariff for renewable electricity generation, the CHP bonus and the eco-design standards are at least likely to produce a welfare gain compared to a single ETS. In these cases, the policy mix can also be justified by economic rationales. However, it has been pointed out that the theoretical superiority of the policy mix is compromised due to some specific characteristics of policy mix design. Modifications in policy design may reduce these detrimental effects. Thirdly, it has been demonstrated that the supplementing the ETS by the electricity tax and the voluntary agreement is likely to produce welfare losses. These examples can be justified by none of the economic rationales for using a policy mix to combat climate change. Moreover, the benefits of the electricity tax and the voluntary agreement from addressing other policy objectives than mitigating climate change have been found to be non-existent or very limited. The evaluation therefore provides a strong indication that these policies should rather be renounced.

In summary, the analysis of the German case study has empirical confirmed the theoretical finding that a policy mix may be superior to single-policy strategies. Moreover, it has once more become obvious that the eventual performance of a policy mix depends on a complex set of characteristics of policy design and the institutional environment. Of course, the empirical analysis has also revealed that not all policies are economically desirable components of a policy mix. This has been clarified for the electricity tax, for example.

7.2 Transferability of Results to Other Policy Contexts

7.2.1 Identification of Economic Rationales for Using a Policy Mix for Pollution Control

In this book, rationales for using a policy mix have been developed for pollution control problems, such as climate change, air and water pollution, or waste disposal. Thus, rationales have been referred to the issue of pollutants being introduced into the environment. These rationales may be transferred to other environmental problems as well. Economic explanations, such as externalities and transaction costs, are also employed to explain problems related to the extraction of natural resources. Consequently, the above rationales can be employed as well to design efficient strategies for controlling the extraction of resources from the environment. In fact, there are already some studies which discuss the use of a policy mix in this respect. It has been examined, for example, for controlling agriculture (Weinberg and Kling 1996), forestry (Koskela et al. 2007), fisheries (Howlett and Rayner 2004), and, above all, land development (Bizer 1997; Hansjürgens and Schröter 2004; Gawel 2005b; Hansjürgens and Schröter-Schlaack 2008; Lehmann and Schröter-Schlaack 2008; Korthals Altes 2009; Nuissl and Schröter-Schlaack 2009). Two examples may illustrate the transferability of rationales for using a policy mix for pollution control to problems of resource extraction (or resource conservation).

An example of multiple reinforcing failures of private governance structures can be taken from agriculture. Farmers adopting environmentally friendly practices are usually perceived to produce a positive externality (Hanley et al. 1998, p. 104).[75] If they restore buffer strips, hedgerows or wetlands they may produce external benefits in terms of higher biodiversity.[76] Since they are unable to appropriate these benefits, they will under-invest in environmentally friendly practices. The adverse effects of the positive externality may be aggravated by asymmetric information in markets for agricultural goods. Supplying farmers are better informed about a good's quality than customers. Some consumers may have a preference of goods produced in an environmentally friendly manner. However, they may be unable to identify those in the market. In turn, it may be too costly for a single farmer to provide credible information about the quality of his goods. As a consequence, consumers are likely to buy cheaper but possibly less environmentally friendly products. In this case, two policies are needed. A subsidy has to be provided to environmentally friendly farmers to address the positive externality. Such subsidies have been implemented in many OECD countries under the label of so-called agri-environmental schemes (see, e.g., Hanley and Oglethorpe 1999; Hanley et al. 1999; Whitby 2000). These schemes allow environmentally friendly farmers to compete with conventional farmers at market prices. However, these schemes cannot overcome asymmetric information and stimulate efficient consumer behaviour. Rather, additional measures are necessary allowing consumers to make optimal choices according to their preferences. One such measure could be a governmentally initiated eco-label for environmentally friendly goods. Yet, as emphasized in the definition of the rationale in Section 3.2.4.1, decision-makers have to revise whether the balance of the net value of correcting the failure and the related transaction costs is positive and larger than with private governance structures. One result of this analysis could be, for example, that governments should rather promote privately initiated labelling systems instead of governmentally initiated ones.

An example of high transaction costs related to a single policy can be taken from land development control. Tradable development rights (TDRs) are increasingly discussed as a tool to

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[75] In fact, only could also put forward that farmers not adopting environmentally friendly practices produce a negative externality. The eventual distinction between positive and negative externalities depends on the society's agreement on the allocation of property rights. If farmers are assumed to hold the complete property rights on land and resources, the adoption of environmentally friendly practices should be understood as a positive externality (Bromley 1997, p. 5)

[76] In this example, the focus is on resource-based benefits, such as producing biodiversity. In fact, environmentally friendly practices also generate positive externalities from bringing down pollution, e.g. due to reduced fertilizer use. This aspect of agriculture corresponds to a classical pollution control problem. The rationales for using a policy mix can be employed analogously.

impose a restriction on land development (see, e.g., Bizer 1997; Cohen 2002; Frece 2005). TDRs set a quantitative cap on land development while still leaving some discretionary power to local developers. These are free to develop as much land as they desire if they hold a sufficient amount of TDRs. However, designing a single TDR scheme efficiently would be very cumbersome for the regulators. This is due to the heterogeneity of damages from land development. Damages mainly result from foregone regulatory functions, such as flood control, filtering of ground water and air, or the provision of natural habitat. These functions may vary significantly between sites. They depend on the properties of the site but also those of adjacent parcels of land (see, e.g., Lehmann and Schröter-Schlaack 2008, pp. 3-4). This heterogeneity implies that TDRs cannot be traded efficiently on the basis of acreage only. Separate markets would have to be established for each regulatory function a site provides. Transaction costs under such system would be high – particularly for developers which would have to participate in separate markets. This finding would be quite similar to those made for controlling air pollution by ambient tradable permits (see Section 3.4.1). Transaction costs can be decreased by using a policy mix (Bizer 1997; Gawel 2005b; Hansjürgens and Schröter-Schlaack 2008; Lehmann and Schröter-Schlaack 2008; Nuissl and Schröter-Schlaack 2009). It has been proposed to integrate a relatively simple TDR scheme based on acreage into the existing planning regulation. TDRs would allow reducing control costs within the restrictions of planning. These restrictions, in turn, would provide for a minimum of regulatory functions to be maintained despite trading.

7.2.2 Integration of Design Complexity into Models of a Climate Policy Mix

In this book, a substantial contribution to economic theory has been made by integrating the complexity of real-world climate policies into the theoretical models of a climate policy mix. The policies considered in the theoretical analyses represent basic concepts: tradable allowances, taxes and subsidies. The use of these policies is not restricted to the mitigation of climate change. These approaches are also widely employed and combined to address other environmental problems – be they of pollution or non-pollution character. The advances in understanding how certain characteristics of policy design affect the overall performance of a policy mix are therefore of a high general interest. For example, the insight that the efficiency of a policy mix may be impaired by differences in the scope of and the overlap between policies is very helpful for the evaluation of environmental protection strategies in general.

It is self-evident that the analyses in this book cannot be used as a blueprint for the evaluation of any environmental policy mix. Attempts to transfer insight from the climate policy case to other environmental fields have to assess to what extent the objectives, scopes and rules of the policies are actually comparable. The characteristics of policies addressing other environmental problems may well deviate from those of climate policies. Nevertheless, even for very different policy mix designs, this book has provided useful insight on how to tackle with policy mix analysis methodologically. It has been shown that and how simple partial equilibrium models can be employed and advanced to understand the complexity of real-world policy mixes.

Of course, it is also important to highlight that not only the detailed policy design but also the characteristics of resources underlying the environmental problem may have an important impact on the performance of a policy mix. The different policy implications of uniformly and non-uniformly mixing pollutants are a striking example (see Section 3.4.1). These restrictions to transferability notwithstanding, the main message of this book remains clear and valid: Understanding and considering the real complexity of a policy mix is indispensable in order to derive reasonable policy recommendations.

7.2.3 Evaluation of the Climate Policy Mix in the German Electricity Sector

This book has delivered an extensive evaluation of the climate policy mix in the German electricity sector. This evaluation has been based on the theoretical insight derived in this book, but also on a thorough study of the specific characteristics of the German policy mix and its environment. Many of the results of this applied analysis can be transferred to the evaluation of climate and energy policies in other EU Member States and worldwide. This is due to the high representativeness of the German case study. Firstly, the German policy mix includes the entire range of basic policy types used in industrialized countries – from allowance trading schemes, taxes and subsidies to command-and-control approaches, information measures and voluntary agreements. Most of the policy combinations evaluated for the German context also exist in other countries. Secondly, climate policy mixes in Germany and other countries do not only exhibit similarities with respect to the basic policy types used but also regarding the specific design of each policy. On the one hand, this is because a lot of German climate policies are mandated and guided by EU Directives and Regulations. Respective policies include the ETS, the feed-in tariff for renewable electricity, the electricity tax, the CHP bonus, eco-design standards and energy labelling. Due to these regulations, other EU Member States have implemented policies whose design corresponds to that of German policies (see, e.g., Boemare and Quirion 2002; Sorrell 2002; Mavrakis et al. 2003; Sijm and van Dril 2003; Sorrell et al. 2003; del Rio and Gual 2007). On the other hand, several European and German policies have also served as examples for policy design outside the EU. Most notably, policies that are similar to the European ETS and the German feed-in tariff for renewable electricity generation can nowadays be found in many non-EU countries as well. Thirdly, certain features of the institutional environment of the German policy mix are not specific to the German case. It has been demonstrated that the efficiency of the German policy mix is significantly determined by the characteristics of the electricity and CO_2 markets. Since these markets are transnational, the same characteristics also affect the performance of policy mixes in other countries. These three arguments clearly indicate that the possible interactions identified for the German policy mix are of high importance and interest for many other country contexts as well. The detailed evaluation of the German policy mix can therefore serve as a guiding analysis for climate policy mixes in general.

Obviously, policy design may also deviate between Germany and other countries with respect to certain details. For example, it is worth highlighting that EU Directives usually leave some discretion to Member States. Under the European ETS, National Allocation Plans have been set up decentrally by the Member States. During the second commitment period from 2008-2012, for example, Member States have been free to choose the level of auctioning – up to the maximum of ten percent. With respect to the taxation of energy, the EU Directive prescribes a minimum level. The actual level and differentiation of the tax rates deviates remarkably from country to country (see, e.g., BMU 2003). Likewise, the promotion of renewable electricity is subject to EU framework regulation. However, some Member States have chosen feed-in tariff systems while other have implemented a quota with tradable green certificates (see, e.g., Morthorst 2003).. Some of these deviations may only affect the extent but not the quality of interactions. Others may be worth consideration and require additional analyses. In any case, this restriction does draw into question the high relevance which the evaluation of the German policy mix has for many countries.

7.3 Avenues for Future Research

Their important advances in the theoretical and empirical understanding of a policy mix notwithstanding, the analyses in this book have also been subject to a couple of limitations. Understanding these restrictions and integrating them into the economic analysis of a policy mix provides avenues for future research.

A *first* important limitation is the inadequate consideration of transaction costs. It has been shown in this book that a proper policy mix analysis has to rest on a thorough understanding of transaction costs. Transaction costs can only be determined on a case-by-case basis taking into account the institutional environment of policy-making. In order to derive a reasonable analysis of a policy mix, a careful empirical evaluation of transaction costs is needed. A sole examination on a theoretical level cannot be sufficient. In this respect, most existing studies exhibit deficiencies as they make only rough assumptions about the level of transaction costs. Studies analyzing a policy mix to correct for multiple failures of private governance structures usually assume sufficiently low (or zero) transaction costs. Likewise, studies referring to high transaction costs of a single policy often assume that transaction costs of a policy mix are sufficiently low. Studies hardly ever employ or refer to an empirical analysis of transaction costs related to a policy mix. This restriction also applies to the theoretical analyses carried out in this book. In the context of the applied analysis of the German policy mix, at least some light has been shed on the issue of transaction costs. However, these considerations have been rather anecdotic. A comprehensive systematic evaluation of transaction costs arising under the German policy mix – and the climate policy mixes existent in other countries – is still lacking. It is also important to highlight that existing analyses as well as this book have hardly invested efforts in understanding why private governance structures fail in correcting pollution externalities, technology spillovers or asymmetric information. In this context, the analysis of transaction costs would be equally important. It may possibly open up (more efficient) alternatives to using a policy mix. Under certain conditions, the abolition of certain institutional constraints which impair the performance of private governance structures may be more desirable than implementing a government policy. These aspects still call for integration into the analysis of existing policy mixes.

A *second* limitation arises from the fact that the analysis of a policy mix has focused on the efficiency criterion to evaluate the climate policy mix. Of course, efficiency is not the only criterion for policy-makers. Most notably, equity may be another important guideline for designing policies. In fact, equity concerns may be another reason for implementing a policy mix. This has been discussed as an excursus for the combination of the ETS and the electricity tax in Germany (see Section 6.3.3). Given the importance of equity concerns for real-world policy-making, it seems to be promising, though, to analyse distributional effects of certain policy combinations more systematically and thoroughly.

Thirdly, the analysis of a policy mix with multiple objectives merits further investigation. Existing policy mix studies often assume that all policies are designed to address a single pollution problem, such as climate change, exclusively. In reality, however, policies may pursue a multiplicity of objectives. These may include other environmental goals as well as non-environmental objectives, such as security of energy supply, raising fiscal revenues or economic development (see, e.g., OECD 2007, p. 165). In this book, the issue of multiple objectives has been addressed in the applied analysis of the German policy mix, where useful. However, these considerations have not been very systematic. Moreover, the existence of multiple policy objectives has significantly influenced the theoretical analysis of the ETS and the electricity tax (see Chapter 5). There, the multiplicity of objectives resulted in a constraint (a tax which cannot be abolished) under which the climate policy mix had to be optimized. This approach implied basically that the regulator only has the choice between abolishing and maintaining the tax. In reality, however, the regulator may have more discretion. For example, he may also be able to decide about the level of the tax rate. It would then be interesting to investigate how the optimal tax rate is influenced when there are positive effects with respect to one policy objective and negative with respect to another. In order to better understand such implications of multiple objectives, they have to be considered more explicitly in policy mix models. This may allow analysing trade-offs between objectives as well as policy effects more thoroughly.

Fourthly, this book has hardly shed light on how (German) policies have actually come about. However, it is important to integrate the political economy of political decision-making into the analysis of a policy mix as well. Political decisions may be driven by a variety of factors apart from the actual policy objectives. They may be subject to the influence of stakeholder groups. They may be characterized by multilevel decision-making with possibly competing legislations (see, e.g., Braun and Santarius 2008). Moreover, the political process may also exhibit path dependencies (see, e.g., Woerdman 2004a; 2004b). There may be constituencies for certain policies within the administration. This may be the case, for example, when a regulating authority has gained experience in administrating an existing policy and therefore objects any changes. This may be a reason why newly implemented policies cannot simply substitute existing ones. Political economy arguments may result in the insight that an existing policy mix – although it is inefficient – is the only politically feasible option. In other words, political economy arguments may provide an additional rationale for using a policy mix. A policy mix may help to buy in stakeholders. For example, the allocation of allowances under an emissions trading scheme may be based on an existing voluntary agreement or command-and-control regulation (Johnstone 2003, p. 27; Sorrell et al. 2003, p. 78). Stakeholders may also be allowed to opt out of an emissions trading scheme if they can demonstrate equivalent efforts by a voluntary agreement (Sorrell et al. 2003, pp. 75-76). Moreover, the revenues of an emissions tax may be refunded to the taxed parties on another basis than emissions (OECD 2007, p. 169). On the one hand, the analysis of the political economy of a policy mix may reveal restrictions to policy choices. On the other hand, it may provide an indication which political economy barriers have to be removed in order for an optimal policy mix to be implemented.

Fifthly, the theoretical analyses of a policy mix in this book have been based on partial equilibrium models of the electricity and CO_2 allowance market. These models have been useful to identify and analyze interactions between policies. This analytical approach cannot claim to be comprehensive, though. In addition, a general equilibrium analysis is needed to assess effects of the policy mix and possible alternatives outside the covered markets, e.g. in labour, input and output markets. For example, a general equilibrium approach could allow addressing the question whether the feed-in tariff should be funded by general tax revenues – or rather by an add-on to the electricity price.

Sixthly, there is a general need for more second-best analyses. This approach may yield useful results as first-best solutions are often unlikely to be implemented perfectly in reality. One such analysis that has been included in the book is the analysis of the policy mix of the ETS and the electricity tax. It has been shown that the policy mix is inferior to a first-best single ETS – but superior to the suboptimal single electricity tax. Thus, the analysis revealed that under certain conditions, the policy mix may be desirable even if it is inefficient. This type of analysis could be applied for other policy combinations as well. It could be investigated, for example, whether the combination of the ETS and the feed-in tariff for renewable electricity, which has been found to produce inefficient distortions, is nevertheless superior to a single ETS in the presence of multiple failures of private governance structures.

Seventhly, the analyses in this book have focussed on combinations of two policies for simplicity reasons. Once the interactions between two policies are understood, the analysis should be extended to a policy mix with more policies implemented in parallel. The striking question would be whether interactions between two policies may be aggravated or mitigated by additional policies.

Eighthly and finally, some of the policy mix discussions presented in this book are also limited by their qualitative character. A qualitative analysis allows identifying the directions of interaction effects but not their extent. When certain interaction effects trade off each other, the derivation of a clear policy recommendation may be difficult on qualitative grounds. An example discussed in this book is the combination of the ETS and the feed-in tariff for renewable electricity generation. It has been shown that dynamic welfare gains come at the cost of static welfare losses. The qualitative analysis has provided some indication that the overall balance of these effects is positive. Nevertheless, this indication calls for verification by a quantitative analysis. In this case, the spillover effects related to learning-by-doing with renewable energy technologies as well as the static increase of abatement costs have to be quantified. Similar analyses are needed for combinations of the ETS with the CHP bonus and the eco-design standards. In these cases, quantification will help to specify evaluation results for a policy mix and deliver more precise policy recommendations.

Consequently, there are a variety of challenges and open questions which are suited to guide future research in the field of policy mixes. Pursuing these avenues is promising and important. It will contribute to a further improvement in the understanding of a policy mix, and increase the relevance of economic policy mix theory for real-world problems even more.

7.4 Outlook

The main message of this book is clear: The classical economic perception that a pollution problem such as climate change can be addressed efficiently by only one policy should no longer be considered as a guideline for policy-making. The existence of a climate policy mix in many countries is not only fact. In many cases, it may be an economic necessity. When deciding about a strategy to mitigate climate change, policy-makers should not only consider one policy but also a set of policies. Nevertheless, this message must not be misunderstood as a universal justification for having policy mix. A policy mix is not always superior to a single policy. A careful theoretical as well as empirical evaluation has to be employed to assess which policy combinations are appropriate means to reduce GHG emissions, and which are not. Such analysis has to take into account the complexity of real-world policy design as well as the institutional environment of a policy mix. This book as provided numerous valuable findings and insights which may assist economists as well as policy-makers in evaluating existing policy mixes and deriving feasible policy recommendations.

Annex

A.1 Activities and Installations Subject to the ETS According to Annex I TEHG

Energy Generation and Conversion		
I	Installations for generating power, steam, hot water, process heat or hot waste gas using fuels in a combustion facility (e.g. power station, heat & power station, heat-only station, gas turbine station, combustion engine installation, other firing installation), including associated steam boiler, with a rated thermal input of 50 MW or more	Energy Installations
II	Installations for generating power, steam, hot water, process heat or hot waste gas using coal, coke, including petroleum coke, coal briquettes, peat briquettes, peat fuel, untreated wood, emulsified natural bitumen, fuel oil, gaseous fuels (notably coke oven gas, mine gas, steel gas, refinery gas, synthesis gas, petroleum gas from tertiary extraction of petroleum, sewage gas, biogas), methanol, ethanol, untreated vegetable oils, vegetable oil methyl esters, untreated natural gas, liquefied gas, gases from public gas utilities or hydrogen with a rated thermal input of more than 20 MW and less than 50 MW in a combustion facility (e.g. power station, heat & power station, heat-only station, gas turbine station, combustion engine installation, other firing installation), including associated steam boiler, except for combustion engine facilities for drilling rigs and emergency power generators	
III	Installations for generating power, steam, hot water, process heat or hot waste gas using solid or liquid fuels other than those listed under No. II in a combustion facility (e.g. power station, heat & power station, heat-only station, gas turbine station, combustion engine installation, other firing installation), including associated steam boiler, with a rated thermal input of more than 20 MW and less than 50 MW	
IV	Combustion engine facilities to power working machinery using fuel oil EL, diesel fuel, methanol, ethanol, untreated vegetable oils, vegetable oil methyl esters or gaseous fuels (notably coke oven gas, mine gas, steel gas, refinery gas, synthesis gas, petroleum gas from tertiary extraction of petroleum, sewage gas, biogas, untreated natural gas, liquefied gas, gases from public gas utilities, hydrogen) with a rated thermal input of more than 20 MW, except for combustion engine	

	installations for drilling rigs with a rated thermal input of more than 20 MW and less than 50 MW	
V	Gas turbine facilities to power working machinery using fuel oil EL, diesel fuel, methanol, ethanol, untreated vegetable oils, vegetable oil methyl esters or gaseous fuels (notably coke oven gas, mine gas, steel gas, refinery gas, synthesis gas, petroleum gas from tertiary extraction of petroleum, sewage gas, biogas, untreated natural gas, liquefied gas, gases from public gas utilities, hydrogen) with a rated thermal input of more than 20 MW, except for closed-cycle installations with a rated thermal input of more than 20 MW and less than 50 MW	
VI	Installations for the distillation or refinement or other downstream processing of petroleum or petroleum products in mineral oil or lubricant refineries	**Energy-intensive Industry installations**
VII	Installations for the dry distillation of black coal or lignite (coke ovens)	
Production and processing of ferrous metals		
VIII	Installations for the roasting, smelting or sintering of iron ores	
IX	Installations for the production or smelting of pig iron or steel including continuous casting, including with the use of concentrates or secondary raw materials, with a smelting capacity of 2.5 tonnes/hour or more, excluding operation in integrated smelting plants	
IXa	Integrated smelting plants (installations for producing pig iron and downstream processing of crude steel where the producing and downstream processing units are located next to each other and linked functionally) with downstream processing units with a rated thermal heat input of more than 20 MW	
IXb	Downstream processing units within integrated smelting plants (installations for the hot rolling of steel, foundries, installations for metalliation) with a rated thermal heat input of more than 20 MW, if not part of an activity under No. IXa	
Mineral Industry		
X	Installations for the production of cement clinkers with an output of more than 500 tonnes/day in rotary kilns or more than 50 tonnes/day in other kilns	
XI	Installations for the calcination of limestone or dolomite with an output of more than 50 tonnes/day quicklime or unslaked dolomite	

XII	Installations for the production of glass, also from recycled glass, including installations for the production of glass fibre, with a melting output of more than 20 tonnes/day	
XIIa	Installations for the melting of mineral compounds, including installations for the production of mineral fibre, with a melting output of more than 20 tonnes/day	
XIII	Installations for the baking of ceramic products with an output of more than 75 tonnes/day, where kiln capacity is at least 4 m^3 and setting density per kiln at least 300 kg/m^3	
Other industrial activities		
XIV	Installations for the production of pulp from timber, straw or similar fibrous materials	
XV	Installations for the production of paper and board with an output of more than 20 tonnes/day	
XVI	Installations for the production of propylene and ethylene with an output of more than 50,000 tonnes/year	
XVII	Installations for the production of carbon black with a rated thermal heat input of more than 20 MW	
XVIII	Installations for the burning of excessive gaseous substances in sea-land transfer stations for mineral oil and gas with a rated thermal heat input of more than 20 MW	

A.2 Emission Factors Applied under the ETS According to Annex 3 ZuG 2012

Installation	**BAT-Benchmark**
Installations for generating power using	
- gaseous fuels	365 g CO_2/kWh net power generation
- other fuels	750 g CO_2/kWh net power generation
Installations for powering working machinery	530 g CO_2/kWh net power generation
Installations for generating heat using	
- gaseous fuels	225 g CO_2/kWh
- other fuels	345 g CO_2/kWh
Installations for the production of cement or cement clinkers in production facilities with	
- three cyclones	845 g CO_2/produced kg cement clinker
- four cyclones	815 g CO_2/produced kg cement clinker
- five or six cyclones	805 g CO_2/produced kg cement clinker
Installations for the production of glass, including	
- container glass	330 g CO_2/produced kg glass
- flat glass	670 g CO_2/produced kg glass
Installations for the baking of ceramic products, including	
- front-wall bricks	115 g CO_2/produced kg brick
- back-wall bricks	68 g CO_2/produced kg brick
- roof tiles (U-cassette)	130 g CO_2/produced kg tile
- roof tiles (H-cassette)	158 g CO_2/produced kg tile

A.3 Standard Utilization Factors Applied under the ETS

The activity-specific standard utilization factor is calculated as the ratio of the annual full utilization hours according to Annex 4 ZuG 2012 and the maximum full utilization hours per year set out in the installation's operating permit. If no such maximum is defined, it is set at 8,760 full utilization hours per year. The maximum of the standard utilization factor is one.

Annual full utilization hours per year according to Annex 4 ZuG 2012

Activity/Installations	Annual Full Utilization Hours per Year
Energy Generation and Conversion: Activities according to Numbers I to V of Annex 1 TEHG	
Condensation power stations	7,500
Condensation power stations for the use of lignite	8,250
Open gas turbine facilities	1,000
Installations for the compression of natural gas for transport purposes	4,200
Installations for the compression of natural gas for underground storage	3,100
Combined heat and power stations for the provision of the paper, mineral oil or chemical industry	8,000
Other combined heat and power stations	7,500
Process heat installations for the provision of the paper, mineral oil and chemical industry and Installations for the production of Bioethanol	8,000
Heat-only stations of the public district heat	2,500
Process-heat installations for the provision of the food and sugar industry	7,500
Heat installations for the provision of the trade, commerce and service sectors, other sectors, and hospitals	7,500
Activities according to numbers VI to XVIII of Annex 1 TEHG	

Installations of the petroleum industry	8,000
Coke ovens	8,300
Installations for sintering	8,300
Installations for the production and processing of ferrous metals	8,300
Installations for the production of cement	7,500
Installations of the lime industry for the production of limestone	7,500
Installations of the sugar industry for the production of limestone	2,500
Installations for the production of glass	8,000
Installations for the baking of ceramic products	7,500
Installations for the production of pulp	8,000
Installations for the production of paper and board	8,000
Installations for the production of ethylene and propylene	8,000
Installations for the production of carbon black	8,000
Installations for the burning of excessive gaseous substances in sea-land transfer stations for mineral oil and gas	500

A.4 Indicators and Energy Efficiency Classes by Appliances Applied for Energy Labelling

	Refrigerators, freezers and their combinations	Washing machines	Driers	Washer-driers	Dishwashers	Electric lamps	Air-conditioners	Electric stoves
	Directives 94/2/EC, 2003/66/EC	Directives 95/12/EC, 96/89/EC	Directive 95/13/EC	Directive 96/60/EC	Directives 97/17/EC, 99/9/EC	Directive 98/11/EC	Directive 2002/31/EC	Directive 2002/40/EC
Energy Efficiency Index	I = annual energy consumption of appliance divided by standard annual energy consumption of appliance depending on type of appliance X 100	C = kWh per kilogram laundry for a standard 60°C cotton cycle	C = kWh per kilogram laundry for a dry cotton cycle,	C = kWh per kilogram laundry for a standard 60°C cotton cycle and a dry cotton drying cycle	E = energy consumption of appliance divided by reference energy consumption depending on capacity of appliance	E = power input W in watts/reference power input for the lumen output Φ of the lamp	EER (energy efficiency ratio) = ratio of cooling output and electrical energy input	energy consumption E in kWh based on standard load
Class A++	I < 30	-	-	-	-			
Class A+	30 ≤ I < 42	-	-	-	-			
Class A	42 ≤ I < 55	C ≤ 0.19	C ≤ 0.51	C ≤ 0.68	E ≤ 0.64	fluorescent lamps without integral ballast if W ≤ 0.15 √Φ + 0.0097 Φ, other types of lamps if W ≤ 0.24 √Φ + 0.0103 Φ	3.20 < EER	E ≤ 0.80
Class B	55 ≤ I < 75	0.19 < C ≤ 0.23	0.51 < C ≤ 0.59	0.68 < C ≤ 0.81	0.64 < E ≤ 0.76	E < 0.60	3.00 < EER ≤ 3.20	0.80 ≤ E < 1.00
Class C	75 ≤ I < 90	0.23 < C ≤ 0.27	0.59 < C ≤ 0.67	0.81 < C ≤ 0.93	0.76 < E ≤ 0.88	0.60 ≤ E < 0.80	2.80 < EER ≤ 3.00	1.00 ≤ E < 1.20
Class D	90 ≤ I < 100	0.27 < C ≤ 0.31	0.67 < C ≤ 0.75	0.93 < C ≤ 1.05	0.88 < E ≤ 1.00	0.80 ≤ E < 0.95	2.60 < EER ≤ 2.80	1.20 ≤ E < 1.40
Class E	100 ≤ I < 110	0.31 < C ≤ 0.35	0.75 < C ≤ 0.83	1.05 < C ≤ 1.17	1.00 < E ≤ 1.12	0.95 ≤ E < 1.10	2.40 < EER ≤ 2.60	1.40 ≤ E < 1.60
Class F	110 ≤ I < 125	0.35 < C ≤ 0.39	0.83 < C ≤ 0.91	1.17 < C ≤ 1.29	1.12 < E ≤ 1.24	1.10 ≤ E < 1.30	2.20 < EER ≤ 2.40	1.60 ≤ E < 1.80

Class G	$125 \leq I$	$0.39 < C$	$0.91 < C$	$1.29 < C$	$1.24 < E$	$1.30 \leq E$	$EER \leq 2.20$	$1.80 \leq E$
			classification differentiated by type of drier (air-vented or condensing), classification provided above for air-vented driers				classification differentiated by type of air conditioning (air-cooled or water-cooled, split/multi-split, packaged or single-duct), classification provided above for split and multi-split air-cooled air-conditioners	classification differentiated with respect to size of electric stove (small, medium, large), classification provided above for medium-sized electric stoves

A.5 Information Required for Labels, Fiches and Mail Order Catalogues by Appliances Applied for Energy Labelling

Refrigerators, freezers and their combinations	Washing machines	Driers	Washer-driers	Dishwashers	Electric lamps	Air-conditioners	Electric stoves
Directives 94/2/EC, 2003/66/EC	Directives 95/12/EC, 96/89/EC	Directive 95/13/EC	Directive 96/60/EC	Directives 97/17/EC, 99/9/EC	Directive 98/11/EC	Directive 2002/31/EC	Directive 2002/40/EC
suppliers name or trade mark	suppliers name or trade mark	suppliers name or trade mark	suppliers name or trade mark	suppliers name or trade mark	energy efficiency class (A most efficient, G least efficient) *	suppliers name or trade mark *	suppliers name or trade mark
suppliers model identifier	suppliers model identifier	suppliers model identifier	suppliers model identifier	suppliers model identifier	luminous flux in lumens *	suppliers model identifier *	suppliers model identifier
energy efficiency class (A++ most efficient, G least efficient) *	energy efficiency class (A most efficient, G least efficient) *	energy efficiency class (A most efficient, G least efficient) *	energy efficiency class (A most efficient, G least efficient) *	energy efficiency class (A most efficient, G least efficient) *	input power in watts *	energy efficiency class (A most efficient, G least efficient) *	energy efficiency class (A most efficient, G least efficient) *
EU Eco-label	EU Eco-label	EU Eco-label	EU Eco-label	EU Eco-label	average rated life in hours *	EU Eco-label *	EU Eco-label
energy consumption in kWh/year *	energy consumption in kWh per cycle using a standard 60°C cotton cycle *	energy consumption in kWh per dry cotton cycle *	energy consumption in kWh per complete washing (washing, spinning, drying) cycle using standard 60°C cotton cycle and "dry cotton" drying cycle *	energy consumption in kWh per standard cycle *		indicative annual energy consumption (total input power X 500 hours per year in cooling mode at full load) *	energy consumption in kWh for the functions (conventional and/or forced air convection and/or hot steam) based on standard load *
sum of net storage volume of all compartments not meriting a star rating *	washing performance class (A highest, G lowest) *	rated capacity of cotton in kg *	energy consumption in kWh per washing (washing and spinning only) cycle using standard 60°C cotton cycle *	cleaning performance class (A highest, G lowest) *		cooling output (cooling capacity in kW in cooling mode at full load) *	usable volume of the cavity in litres *
sum of net storage volume of all frozen food storage compartments which merit a star rating *	spin drying efficiency class (A highest, G lowest) *	type of appliance (air vented or condensing)	washing performance class (A highest, G lowest) *	drying performance class (A highest, G lowest) *		energy efficiency ratio (in cooling mode at full load) *	size of the appliance (small: 12l to 35l, medium: 35l to 65l, large: more than 65 l) *
star rating of frozen food storage compartment *	maximum spin speed attained for 60°C cotton cycle *	noise *	maximum spin speed attained for standard 60°C cotton cycle *	capacity in standard place settings *		type of appliance (cooling only, cooling/heating) *	noise *
noise *	capacity of appliance for standard 60°C cotton cycle *	water consumption per dry cotton cycle *	capacity in kg of appliance for standard 60°C cotton cycle (without drying) *	water consumption in litres per complete standard cycle *		cooling mode (air cooled, water cooled) *	time taken to "cook" standard load
type of appliance (specification of refrigerator/cooler)	water consumption per cycle of washing using standard 60°C cotton cycle *	drying time for dry cotton cycle	capacity in kg of appliance for "dry cotton" drying cycle *	noise *		for appliances with heating capability: heat output (heating capacity in kW in heating mode at full load) *	noise
power cut safe in hours (temperature rise time)	noise *	average annual consumption of energy and water for a four-	water consumption in litres per complete operating cycle using standard 60°C cotton cycle and "dry cotton" drying cycle *	manufacturers name, code or indication for "standard" cycle		for appliances with heating capability: heating mode energy efficiency class (A highest, G lowest) *	declaration of power consumption when no heating function is performed and the oven is in the lowest power

		person household *		*			consuming mode
freezing capacity in kg/24h	water extraction efficiency for standard 60°C cotton washing cycle		noise *	programme time per standard cycle		noise *	area of the largest baking sheet expressed in cm² and determined as "surface area"
climate class	programme time		water extraction efficiency for standard 60°C cotton washing cycle *	estimated annual consumption of energy and water *			
	average annual consumption of energy and water for four-person household *		water consumption in litres for washing and spinning using standard 60°C cotton cycle *				
			consumption of energy and water of a four-person household always/never using a drier *				

Legend

	Information required for labels and fiches		Information required for fiches only	*	Information required for mail order catalogues

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