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Report on Work Package A of the ETSAP Project "Integrating policy instruments into the TIMES Model"

Theoretical background on the modelling of policy instruments in energy system models

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1. Introduction

Over the last decade, energy policy has grown more and more complex. Ambitious and, in some cases, conflicting targets have been established regarding climate change, energy security, the affordability of energy and the deregulation of energy markets. Especially in the area of climate policy, the introduction of a large variety of new policy instruments on different regional levels could be observed in recent years. In this process, policy makers need to have the possibility to back up their decisions with comprehensive and scientifically sound research on ex-ante and ex-post evaluation of policy instruments in order to arrive at a consistent and efficient policy framework.

Technology-rich bottom-up energy system models have been used for a long time to represent and analyse complex systems in an understandable form. Thus, it can be assumed that this modelling approach possesses great value when it comes to the evaluation of the long-term impact of different types of climate policy instruments on the energy system and the interaction between various policy tools. Traditionally, engineering energy system models have been mainly applied to assess technological potentials for cost-efficient energy savings. In contrast, the aim of this study consists in exploring the usefulness of bottom-up models for ex-ante policy evaluation.

This report is organized as follows. The next chapter lays the theoretical groundwork on environmental policy instruments. In order to be able to explicitly model policy measures in energy system models, it is essential to have extensive background knowledge on what types of policy instruments exist, what their most important characteristics are and how they can be evaluated and contrasted. Therefore, Chapter 2 first of all gives an overview on the assessment criteria that are usually invoked in neoclassical environmental economics. These are then applied to the different types of conventional environmental policy instruments which will be outlined in Chapter 2.3. A separate chapter is devoted to policy tools that focus on the promotion of specific technologies, as these have gained substantial importance in climate policy during the last years. While environmental economic theory generally looks at different instruments separately and compares their features, in reality usually a policy mix is implemented to deal with environmental problems. That is why Chapter 2 also addresses the implications of using multiple policy instruments and the issue of policy interaction.

In Chapter 3, the focus is then turned to the modelling dimension. After a brief overview on the development of long-term quantitative energy modelling, the attributes of an ideal model for evaluating environmental policy instruments are explored. With the aim of identifying the strengths and weaknesses of engineering energy system models, the two main approaches in energy modelling, top-down and bottom-up, are then contrasted by means of the ideal criteria established in the previous section. On this basis, the last two subchapters concentrate on the two main areas for improvement of bottom-up energy system models when used for policy evaluation: the behavioural dimension and the integration of macroeconomic feedbacks.

2. Theoretical background on policy instruments

2.1. Foundation: need for environmental policy instruments

The beginning of modern environmental policy is usually dated to the late sixties and early seventies where it mainly concentrated on air pollutants, water quality and solid waste disposal. At the same time, in the seventies the most important theoretical work in the area of neoclassical environmental economics was established, which is still used today for a theoretical understanding of environmental policy instruments (cf. Rogall 2008, p. 27ff). The basic idea of environmental economics consists in integrating the environment, represented through environmental commodities like “clean air”, into the economic system. Just like regular commodities, these environmental commodities affect the well-being of society, are perceived as scarce and should therefore be taken into account in the economic resource allocation problem.

The aim is then to determine the most efficient way to allocate the scarce resources to the various production processes and the produced goods to the consumers. In neoclassical theory, the *Pareto Criterion* is used as the main indicator to assess the efficiency or optimality of a given allocation of resources, defining an efficient allocation as the one compared to which no other allocation is feasible that increases the utility of at least one individual without decreases the utility of any other (cf. Breyer 2011, p. 199). In welfare economics it is argued that, under the assumption of competitive markets[1], free markets automatically ensure a Pareto efficient allocation of resources. The market mechanism functions through decentralized economic decisions based on a price system which serves to indicate the scarcity of the different commodities and to equate demand with supply in equilibrium such that marginal utility of consumption equals marginal cost of production. Hence, on competitive markets a misallocation or overexploitation of resources and goods can be ruled out (cf. Endres 2011, p. 9ff).

The simple allocation mechanism through markets does, however, not work in the case of environmental commodities. According to environmental economic theory, this can be attributed to two main issues: the characterization of environmental commodities as public goods and the existence of external effects. In contrast to private goods, pure public goods can be described by means of the two properties non-excludability and non-rivalry. Using the example of the environmental commodity clean air, this means that nobody can be excluded from benefiting from this commodity and total available supply is not reduced substantially when it is consumed by one individual. Sometimes it is also argued that only so far many environmental commodities are treated as public goods, while in reality the exploitation of their functions is subject to rivalry, e.g. the supply of clean air is diminished when a large

[1] In detail, the first theorem of welfare economics states that under the assumption of complete markets with price-taking behaviour and perfect information, in the absence of externalities and transaction costs and with locally non-satiated preferences every market equilibrium is Pareto efficient (cf. Beyer 2011, p. 211)

group of consumers uses it as a dump for pollutants. Thus, instead of pure public goods, environmental commodities can also be considered as common goods where only the criterion of non-excludability but not of non-rivalry applies (cf. Rogall 2008, p. 62). As a consequence of their characterization as public or common goods, no regular markets for most environmental commodities exist and their consumption entails external effects that are not accounted for in the market system. Externalities can be formally described as follows:

"An externality exists when the consumption or production choices of one person or firm enters the utility or production function of another entity without that entity's permission or compensation." (Kolstad 2011, p. 92)

An example for a negative production externality, as it might arise in any production process involving the emission of greenhouse gases or other pollutants, is shown in Figure 2-1. Externalities lead to a divergence between private and social costs. In this case of an externality on the production side, the producer only takes into account the costs of production he incurs, i.e. the private costs, when setting the price. Consequently, the resulting price p_p is lower than the Pareto optimal one p_s (including all social costs, both private and external ones) and demand (given as marginal utility of consumption) is too high. Hence, in the presence of negative external effects, the market equilibrium is inefficient with an over-usage of environmental commodities. These effects have often been characterized as "market failure", while the fundamental problem consists in a non-existence of markets for environmental commodities (cf. Wiesmeth 2012, p. 66).

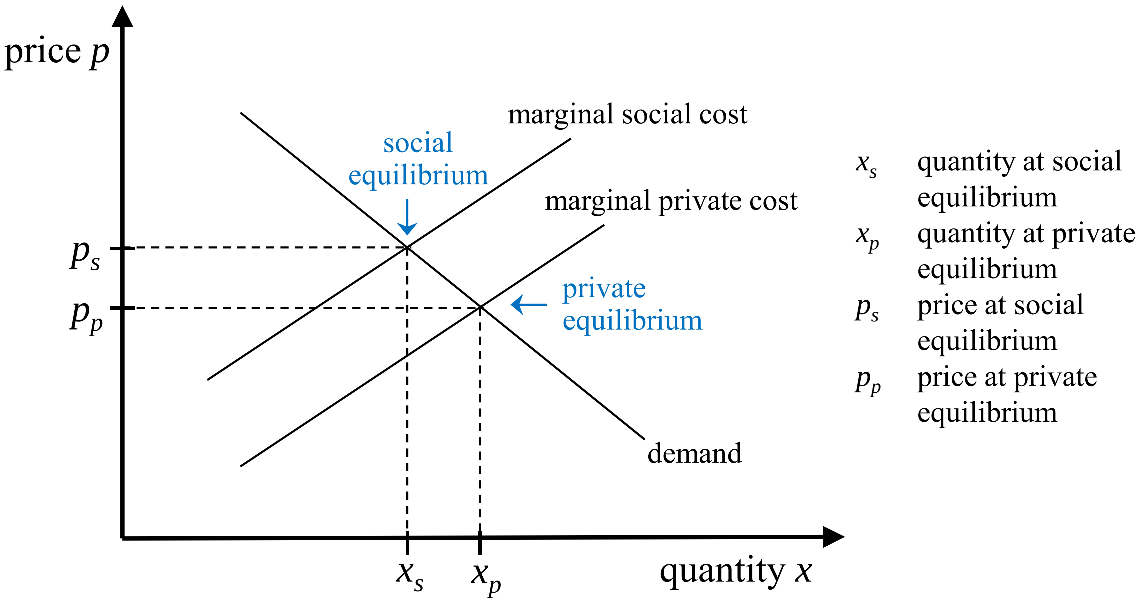

Figure 2-1: Negative production externality (own illustration based on Tietenberg and Lewis 2012, p. 26)

On the basis of these "missing markets" for environmental commodities and the consequent overexploitation, political intervention is justified. The aim of environmental policy instruments is therefore to internalize external effects in order to reestablish the optimality of the resource allocation in equilibrium. Closely related to the concept of internalizing all externalities of production and consumption is the polluter-pays-principle in environmental policy, stating that whoever is responsible for the pollution should bear the costs to the extent of either the damage done to society (“strong” principle) or the exceeding of an acceptable level of pollution (“weak” principle) (cf. OECD 2007a).

Furthermore, the question on the optimal balance between environmental protection and environmental use needs to be addressed in the framework of environmental policy. When no political measures are taken, high environmental damage costs arise. At the same time, the implementation of environmental protection measures also results in additional costs. According to neoclassical theory, an efficient allocation is obtained when marginal environmental avoidance cost equal marginal environmental damage cost, as illustrated in Figure 2-2.

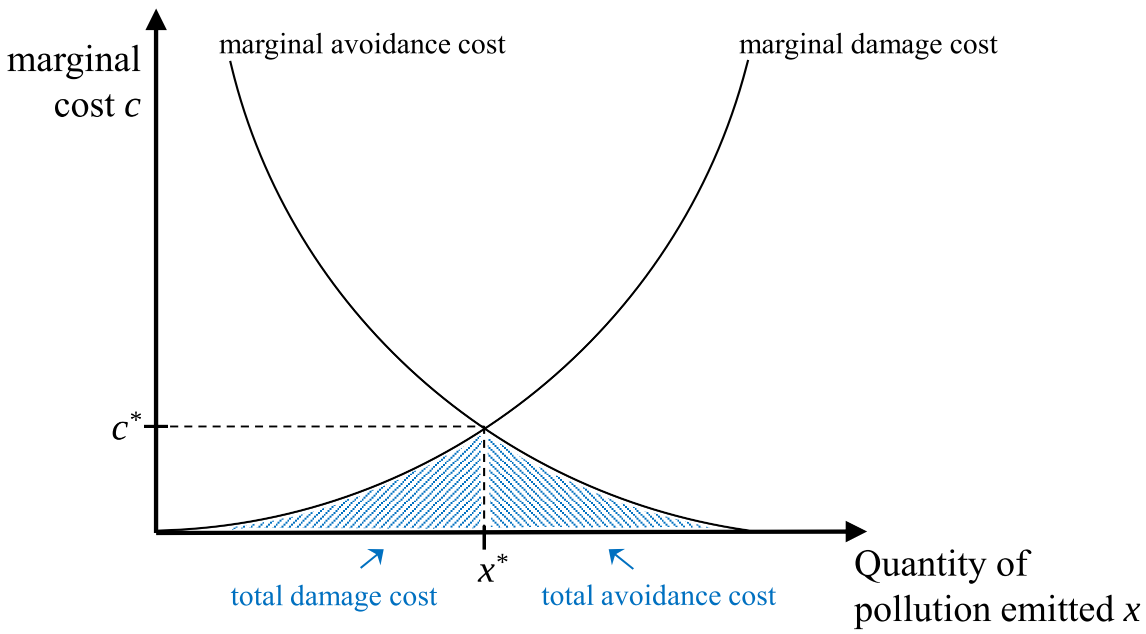

Figure 2-2: Determination of the efficient balance between environmental protection and environmental use (own illustration based on Tietenberg and Lewis 2012, p. 364)

It is obvious that in order to determine this optimal allocation a considerable amount of information is required which is apparently not available in practice (cf. Endres 2011, p. 22ff). First of all, the utility any environmental commodity has (or the perceived damage which is associated with pollution) depends on each consumer’s preferences which are not known to the policy maker, as no markets for environmental commodities where preferences are revealed through the price system exist, and are subject to changes over time. Avoidance costs, on the other hand, depend on the technological state of the art and its future development. Thus, both the marginal avoidance and the marginal damage costs may shift over the course of time affecting also the efficient allocation. An additional problem arises when taking into account future environmental damages which need to be assessed with the help of an appropriate discount rate.

Thus, in reality environmental policy has to deal with information deficits and will therefore not be able to internalize all external effects correctly. Consequently, theoretical solution approaches for the internalization of externalities, like the Pigovian tax and the Coase theorem[2], can only be used as a guideline when designing environmental policy instruments. In practice, the desired environmental quality or standard is generally set exogenously by a team of experts based on scientific, technical, medical and economic criteria, which, however, is not likely to meet exactly the Pareto efficient allocation. Policy instruments will then be constructed with the principal aim of reaching this environmental standard (e.g. a certain emission reduction target) in the most cost efficient manner (cf. Böhringer 1999).

2.2. Evaluation criteria

Having determined the necessity for environmental policy, the crucial question consists in choosing the appropriate policy instruments in order to reach the given target. To facilitate the decision making, a number of evaluation criteria are used, which are outlined in the following.

Ecological precision

Ecological precision, or ecological efficiency, describes the capacity of a policy instrument to fulfil a predefined emission standard or target (in a given region) precisely. Thus, this principle is only focused on attaining a certain goal with the highest possible probability without looking at the cost impact (i.e. economic efficiency). It is particularly crucial in the case of environmental crisis or highly hazardous pollutants. Under this criterion, one can also assess the adjustment time which is needed under a given policy regime to reach the target (cf. Endres 2011, p. 141ff).

Cost efficiency

In most studies, cost efficiency, sometimes also referred to as cost effectiveness, constitutes the central principle against which environmental policy instruments are measured. It represents the capacity of an instrument to fulfil a predefined emission standard or target at the lowest possible cost (cf. Endres 2011, p.121). This is achievable by equating the marginal abatement costs across all available abatement channels (increased energy efficiency, fuel substitution, etc.) and all agents (firms and facilities of all production sectors, households, etc.) such that all economic actors are confronted with a common price on their emissions at the margin (cf. Baumol and Oates 1971). In order to apply the criterion, a clear definition of the considered cost factors is needed (cf. Böhringer 1999). The narrow definition of cost efficiency usually only comprises the compliance costs of the targeted agents and sectors. If, however, the goal is to evaluate the economy-wide cost effects of a certain policy regime,

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2 Arthur Pigou introduced the idea of a tax on activities which entail negative externalities in order to correct the market equilibrium already in the 1920s (cf. Pigou 1962). Ronald Coase stated in his theorem that an efficient allocation in the presence of externalities can be achieved through direct negotiations between the affected parties if property rights are assigned beforehand and no transaction costs occur (cf. Wiesmeth 2012, p.100).

additional costs have to be taken into account, mainly administrative costs (i.e. the costs to implement, monitor and enforce a policy) and macroeconomic impacts (i.e. cost effects on sectors outside of the policy regime, especially fiscal interactions with pre-existing instruments) (cf. Goulder and Parry 2008).

Dynamic efficiency

With the concept of cost efficiency, policy instruments are assessed from a static point of view, assuming that abatement options and technologies are given and unchanging. It has, however, often been argued in environmental economics that in the long term, technological change will play the most prominent role in achieving environmental goals. For example, according to Kneese and Schultz (1978) *"over the long haul, perhaps the single most important criterion on which to judge environmental policies is the extent to which they spur new technology towards the efficient conservation of environmental quality"*. Thus, the criterion of dynamic efficiency or dynamic incentive effect is applied to evaluate the capacity of an instrument to induce the development and deployment of new technologies which reduce the cost of emission mitigation (cf. Endres 2011, p. 130). Accordingly, while the static cost efficiency is concentrated on the minimization of abatement costs in the short term, the principle of dynamic efficiency seeks to minimize emission reduction cost over a longer time period. When analysing the dynamic efficiency of an environmental policy instrument, a differentiation is often made between the potential to encourage the adoption of new, yet existing technologies and the potential to incentivize R&D activities for future technologies (cf. Requate 2005).

Additionally: Political feasibility, distributional equity, flexibility

While ecological precision as well as static and dynamic cost efficiency are surely the most important criteria to evaluate environmental policy instruments, additional aspects have gained importance in recent analyses. The political feasibility and social acceptance of environmental instruments can in practice turn out to be crucial for the decision-making process (cf. Feess 2007, p. 50). Here it has to be noted, that political feasibility does not only depend on the attributes of an instrument but is closely related to the specific political setting with varying constraints, institutional structures, traditions, advocacy groups, etc. (cf. Green and Yatchew 2012). Closely related to the issue of feasibility are distributional impacts of different policy measures, especially between polluting enterprises and other economic agents or across household income groups. Moreover, policy flexibility, i.e. the ability of an instrument to adjust to new information in a flexible and quick manner, has to be taken into consideration (cf. Goulder and Parry 2008).

2.3. Types of environmental policy instruments

The growing endeavours to control global greenhouse gas emissions have assigned additional significance to the issue of instrument choice. The following chapter provides an overview over the most important types of policy instruments which are currently in the centre of the scientific and political discussion when it comes to emission mitigation strategies. The aforementioned evaluation criteria will be applied to highlight the advantages and disadvantages of each instrument.

Command-and-control policies

Command-and-control instruments have dominated environmental policy for a long time. They consist of mandatory regulations where the government directly intervenes in the activities of individual firms by prescribing or forbidding certain activities (cf. Rogall 2008, p. 240). A differentiation is made between technology-based standards, where compliance is only achieved by adopting a certain technology or equipment, and performance-based standards, which stipulate uniform emission ceilings on the firm-level leaving the technology choice to the firm (cf. Hackett 2011, p. 223). Examples of currently valid command-and-control instruments comprise the requirement to install catalytic converters and other legislation to reduce atmospheric pollution as well as the international ban on CFCs. In many cases, regulations do not directly focus on emissions but specify measures which will eventually lead to an emission reduction, as for example minimum energy efficiency standards for buildings or obligations to cover a certain percentage of energy consumption (e.g. of residential buildings) through renewable energies.

The major advantage of command-and-control instruments consists in their high ecological precision making them particularly beneficial to control highly hazardous pollutants and for cases were the spatial distribution of emission is of significance. Taking a closer look, however, it becomes obvious, that regulations usually only target the emission level of an individual firm, such that total emissions which are subject to economic growth are less easy to control (cf. Endres 2011, p. 141). Moreover, an additional drawback arises when looking at the time needed to reach a certain environmental goal. Command-and-control policies usually specify less ambitious requirements for old plants than for new ones thus giving inadvertently an incentive to use the old, less eco-friendly plants for a longer period of time (cf. Endres 2011, p. 144). In the light of behavioural barriers to investments in energy efficiency, though, additional relevance is attached to regulatory mandates as they enforce the realization of in many cases highly cost efficient measures (e.g. in the building sector) which are otherwise not carried out voluntarily (cf. Hackett 2011, p. 224).

With respect to cost efficiency, a differentiation must be made between technology- and performance-based standards. While with regulations setting a specific emission target an individual firm still has the incentive to search for the most cost efficient manner to reach this target, this is not the case for regulations prescribing the use of a certain technology. When looking at the totality of emitting firms, though, both types of command-and-control instruments exhibit considerable disadvantages, as regulations usually set uniform standards not taking into account the heterogeneity of individual polluters. A cost efficient distribution of the emission reduction burden will, however, only be achieved, when the contribution of each firm is determined depending on their individual abatement cost curve. This would require the regulator to have information on the cost situation of each polluter such that a cost efficient differentiation of command-and-control policies is practically impossible (cf. Endres 2011, p. 121ff). Furthermore, an additional weakness of mandatory regulations consists in the fact that, contrary to emissions taxes, under a command-and-control policy framework firms are not charged for the remaining pollution ("weak" form of the polluter-pays-principle), leading to a lower output price and a less pronounced decline in the demand for environmentally harmful goods. Thus, in order to reach the overall emission reduction target, the options of fuel substitution and increased energy efficiency would have to be used above the optimal level, while the output-reduction option is neglected further compromising the cost efficiency of the policy outcome (cf. Cansier 1996, p. 206; Goulder and Parry 2008).

A further shortcoming of command-and-control instruments results from their insufficient stimulation of technological progress. Performance-based standards can spur some efforts to find new processes which make it possible to fulfil the emission targets at lower cost. In general, however, mandatory regulations provide no inducement to develop and introduce technologies that entail emission reductions beyond the standards fixed by the government. In order to increase dynamic efficiency, attempts have been made to increase the flexibility by means of gradually tightening the standards according to the current technological state of art. It has been observed, however, that this approach often tends to have either no or even a dampening effect on innovation and to cause lock-ins into certain technologies, because under such a policy regime polluters have an incentive not to unfold any possibilities of technological improvement and political revision processes are often time consuming and lag behind the actual technological progress (cf. Endres 2011, p. 131ff).

One of the reasons why regulatory mandates have long experienced widespread use is their relatively high acceptance in society (cf. Rogall 2008, p. 243). The fact that polluting firms are only burdened with the cost of pollution going beyond the standard generally facilitates the implementation of such regulations. It has also been argued that (technology-based) regulatory approaches have the advantage of comparatively low monitoring costs, especially when a large number of individual, point-source emissions have to be controlled. Thus, according to Cole and Grossman (1999) if not only compliance but also monitoring costs are taken into account, in some cases, where abatement costs are relatively low and monitoring costs relatively high, the comparative advantage of market-based instruments like emission taxes or tradable permits in terms of cost efficiency might be even offset.

Market-based instruments (1): Emissions taxes

In environmental policy, command-and-control instruments are usually contrasted with so called market-based instruments that try to influence behaviour through market signals instead of setting explicit directives with respect to environmental quality (cf. Stavins 2001). Market-based instruments function either as price-based, i.e. by assigning prices to environmental commodities, most importantly emissions taxes, or as quantity-based, i.e. by assigning property rights and creating markets for environmental commodities, most importantly emission trading systems (cf. OECD 2007a). These types of instruments have gained in importance in environmental policy since the 1990s, their role being highlighted in several official documents like the *Agenda 21* (UNCED 1992), the *Green Paper on market-based instruments for environment and related policy purposes* of the European Commission (2007) or the *OECD Environmental Outlook to 2050* (OECD 2012).

Accordingly, instead of directly limiting environmentally hazardous activities, with emission taxes such activities are made more expensive by putting a charge on the emitted quantity of a pollutant. The approach goes back to the works of Arthur Pigou in the 1920s, but rather than trying to strictly internalize all external effects, the idea today is to reach a certain predetermined environmental standard with the help of imposing a price on pollution. In literature, this is often referred to as the "price-standard approach" introduced by Baumol and Oates (1971) (cf. Endres 2011, p. 109). It has to be noted, however, that the taxes and fees associated with environmental issues which are implemented in practice deviate considerably from the theoretical concept of an emissions tax. As it is often difficult to measure emissions directly, the consumption of input commodities, produced goods or services related to emissions is used as tax base, like for example taxes on gasoline, electricity or motor vehicles (cf. Goulder and Parry 2008). In the European Union, environmental taxes are defined with respect to the tax base which needs to be *"a physical unit (or a proxy of it) of something that has a proven, specific negative impact on the environment"* (Eurostat 2001, p. 9), including all taxes on energy and transport. Moreover, a differentiation can be made regarding the intent of the tax. While fiscal taxes aim at raising revenue, the rational of pure environmental taxes is to influence behavior so as to reduce pollution. Hence, an environmental tax fulfills its objective when revenues are comparatively small (cf. Wiesmeth 2012, p. 185).

The major advantage of using taxes for emission control lies in their high cost efficiency, both on the level of the individual firm and for the totality of polluters. With a price on emissions, a polluting firm will undertake abatement activities as long as their individual marginal abatement costs are below the tax rate using all possible options of emission mitigation. From this it follows that with a uniform emission tax the most cost efficient manner of fulfilling a certain reduction target will also be reached on the aggregate level, because all affected polluters will reduce emissions until their marginal abatement costs equal the tax rate leading to a single emission price on all sources. Thus, in equilibrium marginal abatement costs will be the same for all affected firms, whereas their contribution to emission mitigation will vary (cf. Endres 2011, p. 122ff). The superiority of emission taxes regarding cost efficiency as compared to mandatory regulations is graphically illustrated in Figure 2-3. For simplicity reasons, only two firms are taken into account which both emit the pollutant *E*. The graphs show the marginal abatement cost curves for each firm (MAC_1 and MAC_2) as well as the (horizontally) aggregated curve adding the reduction potentials at different cost levels for both firms (MAC_{1+2}). It follows that emission mitigation becomes more costly as reduction levels get more ambitious. Moreover, it is assumed that the abatement cost curve of *Polluter 2* is steeper than the one of *Polluter 1* leading to higher marginal (and absolute) abatement costs for the same reduction level. At the outset, with no measures in place, emissions sum up to E^* in total with both firms emitting the same amount (E_1^* and E_2^*). In the case of a undifferentiated command-and control policy both firms are obliged to cut their emissions by half, resulting in emission levels of $E_1^*/2$ and $E_2^*/2$ as well as relatively high marginal abatement costs for *Polluter 2* (MAC_2^*) and lower costs for *Polluter 1* (MAC_1^*). The aggregated abatement costs curve MAC_{1+2} shows the tax rate $\bar{t}$ which would be required if the halving of total emission was to be achieved through a uniform tax rate. In this case, each polluter will abate its individual emissions up to the point where marginal abatement costs equate the tax rate. Consequently, *Polluter 1* lowers its emissions additionally to $\overline{E}_1$ associated with slightly higher marginal abatement costs $\overline{MAC_1}$, while *Polluter 2* decreases its mitigation efforts with the result of a higher emission level $\overline{E}_2$ and the same marginal abatement costs ($\overline{MAC_2} = \overline{MAC_1}$). Due to the fact that the marginal abatement cost curve of *Polluter 2* is steeper than the one of *Polluter 1*, the drop to $\overline{MAC_2}$ is more pronounced as the increase to $\overline{MAC_1}$. Accordingly, absolute abatement costs (given as the area under the marginal abatement cost curve) will be lower under the tax regime than with the mandatory standard, as in Figure 2-3 for *Polluter 1* mitigation costs only rise by area *a*, while mitigation costs for *Polluter 2* diminish by the larger area *b*. Thus it becomes obvious, that the uniform emission tax leads automatically to a cost efficient allocation of emission reduction without the regulator having to know the abatement cost curves of each affected polluter.

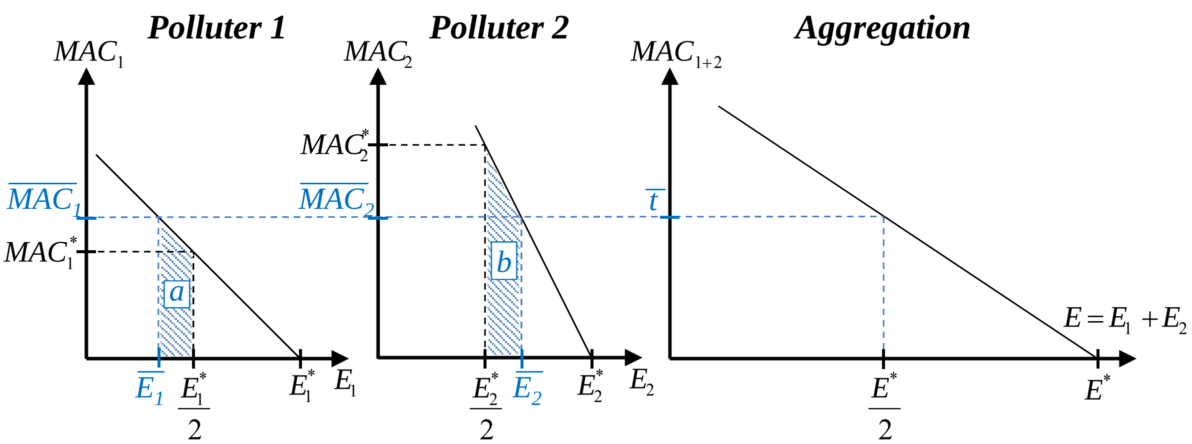

Figure 2-3: Graphical depiction of the cost efficiency of an emission tax compared to a uniform mandatory standard (own illustration based on Endres 2011, p. 125)

An additional remark on cost efficiency has to be made, however, with respect to the design of many environmental taxes currently in existence. A deviation from using emissions as the tax base often entails efficiency losses as not all reduction options may be activated equally.

If, for example, a tax is put on electricity consumption, the only avoidance strategy consists in lowering electricity demand, whereas no incentives for higher efficiency or fuel substitution in electricity generation are generated (cf. Goulder and Parry 2008).

Apart from the high cost efficiency, emission taxes stand out by their strong incentive to introduce new technologies which will reduce abatement costs. Whereas command-and-control instruments only create an inducement to reach a certain standard at minimal cost, tax systems generate constant pressure to realize abatement cost savings regardless of the reduction level already reached. The difference between the two policy approaches stems from the fact that with taxes, emitters do not only pay for the avoided but also for the remaining emissions, such that each additional unit of abated emissions brings cost savings in the form of lower tax payments (OECD 2001, pp. 23f). This distinctive incentive structure is further highlighted in Figure 2-4. It shows the marginal abatement cost curve of a firm before ($\overline{MAC}_{old}$) and after ($\overline{MAC}_{new}$) the implementation of a new, emission-saving technology. The emission level $\overline{E}_{old}$ is assumed to be the standard prescribed in the case of a command-and-control approach and at the same time the level which is reached with the help of an emission tax t before the introduction of the new technology. After the innovation, marginal abatement costs drop to $\overline{MAC}_{new}$, such that for both policy instruments, abatement cost savings in the amount of area *a* are realized. While the emission level remains the same ($\overline{E}_{old}$) with the mandatory standard, in the case of the emission tax, firms have an incentive to lower emissions to $\overline{E}_{new}$ thereby saving tax payments (represented by areas *c* and *d* in Figure 2-4). Thus, in total the polluter is able to obtain cost savings that amount to areas *c* and *a* (as area *d* accounts for additional abatement costs) and therefore has a stronger incentive to introduce an innovative technology than in the case of a fixed emission standard where potential savings are limited to area *a*.

When looking at the ability to reach a given emission target precisely, emissions taxes clearly exhibit disadvantages. It is an intrinsic feature of an environmental tax not to be aimed at the quantity of the commodity in question, but to change behavior through pricing signals. Hence, in order to set the tax rate such as to arrive at a predetermined target, the regulator would require information on the adjustment behavior of all affected polluters, i.e. would have to know their marginal abatement cost curves. Hence, in reality a certain emission standard could only be reached through a stepwise trial-and-error process until the appropriate tax rate is found. Given the lengthiness of political decision-making and the adverse impacts on planning security such a process would have, this does not seem to be a viable approach. Moving away from the static view, additional difficulties with respect to ecological precision arise, as important economic parameters like economic growth, technological progress or inflation rates, which change over time, influence the polluters' response to the tax system. Moreover, it has to be mentioned that the applicability of emission taxes reaches a limit in the case of non-uniformly mixed pollutants where the regional distribution matters and the risk of local "hot spots" needs to be averted (cf. Cansier 1996, pp. 174ff).

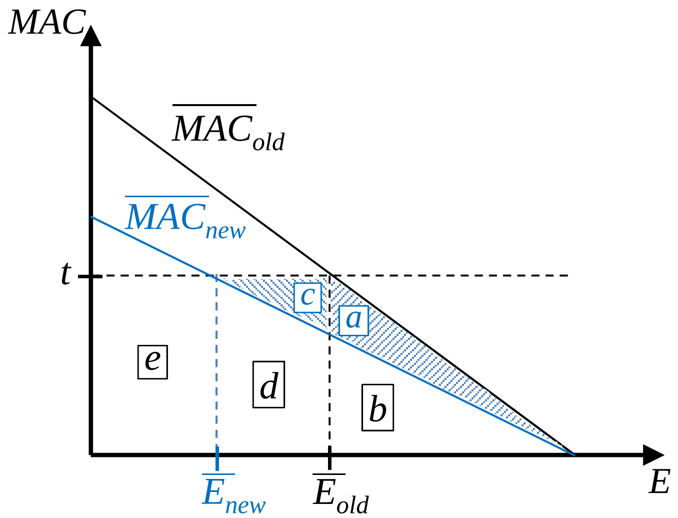

Figure 2-4: Graphical depiction of the dynamic efficiency of an emission tax compared to a uniform mandatory standard (own illustration based on OECD 2001, p. 23)

The size of the administrative costs of environmental taxes depends largely on the tax design, e.g. the complexity of the tax base, the number of specific tax provisions, etc. (cf. OECD 2001, pp. 91ff). One of the main obstacles to the acceptability of effective emission taxes is the additional burden they constitute for the affected economic agents, as, unlike with command-and-control instruments, costs arise not only for the abated emissions but also for the residual ones. With respect to industrial companies, this might raise issues of international competitiveness, whereas in the case of households the income distribution might be impaired. Environmental taxes can have a regressive impact as they often charge goods of basic necessity (especially energy), such that low-income households are more adversely affected (cf. Kosonen and Nicodeme 2009). In this context, additional attention needs to be paid to the question on how the revenues of an environmental taxes are spent, which might lead to an alleviation of the distributional effects. Since the 1990s, the concept of environmental tax reforms (ETR) has taken on greater significance. The idea here is to use the additional environmental tax revenues to lower conventional taxes on production factors, such as labour or capital, i.e. to transfer the tax burden from so called “goods” to “bads” (cf. EEA 2005, pp. 83f). This approach has often been associated with the double dividend hypothesis stating that such a (revenue neutral) tax shift could generate two possible dividends: firstly, welfare gains through the (cost efficient) internalization of environmental externalities (primary welfare gain) and secondly, welfare gains through the reduction of other distortionary taxes (revenue-recycling effect), which could for example, in the case of cutting labour taxes, result in more employment. Theoretical and empirical literature casts, however, doubt on the existence of this double dividend (cf. OECD 2001, pp. 35ff; Böhringer et al. 1997). As Parry and Oates (1998) outline, when analyzing the double dividend hypothesis in a second-best setting with pre-existing factor taxes, a third effect, called the tax-interaction effect, has to be taken into account which works in the opposite direction of the revenue-recycling effect. As environmental taxes will raise the cost of production, after-tax factor return will decrease, intensifying the distortions of the already existing factor taxes. Whether a double dividend can be realized, depends therefore on the magnitude of the revenue-recycling and the tax-interaction effect, with many analytical studies stating that under most conditions the latter tends to outweigh the former (cf. Goulder 1998).

Market-based instruments (2): Tradable allowance systems

Apart from emission taxes, tradable allowance systems for environmental goods represent the most important market-based environmental policy instrument. The approach, originated by Dales (1968), consists of the following steps: (1) the political decision maker (on national or international level) sets a limit on the use of a certain natural resource (e.g. maximum emissions of a pollutant) for a given region and time period; (2) within the specified limit, the overall right to emission is split up into a large number of partial rights permitting the user to emit the proportionate fraction of the total amount; (3) these rights are then transferred to the polluters as tradable emission certificates. Thus, an emissions trading system, also referred to as cap and trade system, is based on the idea of assigning property rights and thus creating artificial markets for environmental commodities (cf. Endres 2011, pp. 110ff). A tradable allowance system is therefore conceptually the mirror image of emission taxes – instead of setting the price of emission and leaving the quantity determination to the market, here the maximum emission level is fixed while the certificate price is market-determined. Consequently, the adjustment behavior of the affected polluters is similar in the sense that abatement activities are undertaken as long as the certificate price on the market exceeds marginal abatement costs, whereas any further emissions are covered through emission allowances purchased on the market (cf. Feess 2007, pp. 123f).

One of the most crucial aspects when implementing an emissions trading system consists in choosing a procedure for the initial allocation of the certificates, with the main differentiation between auctioning and free allocation. The process of auctioning off the certificates is in line with the strong version of the polluter-pays-principle as the participating polluters have to pay both for the avoided and the remaining emissions. In the case of an initial allocation free of charge, difficulties arise regarding the determination of the current emission level of each polluter. Here, the distribution can either be established based on historical emission levels (grandfathering) or on a differentiated reference standard that applies to all installations of a sector (benchmarking) (cf. Möst et al. 2011). Critics have pointed out, however, that grandfathering tends to favour firms with high emission levels, who hitherto have done little for environmental protection, and to discriminate new entrants to the market (cf. Cansier 1996, pp. 193f). Additional issues regarding the design of a cap and trade system include the scope (with respect to the covered pollutants, the regional expansion and the target group), the length of the trading periods as well as the implementation of banking/borrowing (the option to store certificates for a future period or to use certificates of future periods in an earlier one) (cf. Rudolph et al. 2011).

With respect to cost efficiency, the results for emission taxes can be transferred to emission trading as both instruments create the same incentive structure. In the context of a tradable allowance system, the certificate price assumes the role of the tax rate in establishing a uniform price on emission that will ensure a distribution of abatement activities between polluters at minimal cost using all possible mitigation channels. Thus, the graphical representation in Figure 2-3 can also be adopted to illustrate the cost efficiency of emission trading schemes, with the difference that instead of the tax rate here the emission level $E^* / 2$ will be fixed by the regulating authority, resulting in a permit price at level $\bar{t}$ (cf. Endres 2011, pp. 121ff). Furthermore, emission mitigation will be cost efficient with emission trading irrespective of the initial allocation mechanism, as also in the case of free allocation polluters will orient their decision as to whether to abate emissions or buy certificates on the market price. The only prerequisite is the development of well-functioning markets for emission allowances with a large enough number of sellers and buyers (cf. Cansier 1996, p. 195).

Basically, a cap and trade system has the same effect on technological progress as emissions taxes, as polluters can achieve savings in terms of certificate costs with every additional unit of abated emissions realized with the help of an innovative technology. This is also the case with free allocation: even though firms initially did not pay for their permits, these permits could be sold on the market and therefore create opportunity costs representing foregone profits. In the long-run, however, an important difference to emissions taxes is observable. While with a tax system the incentive for innovation remains constant over time given the exogenously fixed tax rate, in an emission trading system the pressure on innovation declines as the certificate price drops with a rising number of firms that have already introduced emission-saving technologies. Hence, it becomes crucial, that the political decision maker takes the anticipated technological progress rates into account when setting the long-term emission caps. Moreover, the regulator has the possibility, to withdraw allowances from the market in the case of a price drop with the additional effect of tightening the emission target (cf. Endres 2011, p. 134). On the whole, while the superiority regarding dynamic efficiency of market-based instruments over command-and-control regimes has clearly been established in theoretical and empirical literature, no clear ranking seems to have emerged in the comparison of emissions taxes and permit trading systems (cf. Jaffe et al. 2002; Requate 2005).

One of the advantages of emission trading systems is their high ecological precision. In such a system, the predetermined emission target will be fulfilled with certainty without the regulator having to know the marginal abatement cost curves of the affected polluters. There is, however, one limitation with respect to non-uniformly mixed pollutants where the regional distribution of emissions needs to be controlled, which is not feasible with a tradable allowance system (cf. Feess 2007, pp. 126f).

Recent studies indicate that permit trading schemes involve comparatively high administrative costs on the government side and high transaction costs for the participating firms, which might reduce the cost efficiency of such instruments. Environmental taxes can usually be relatively easily integrated into the existing tax system and be operated by existing tax authorities, whereas for an emission trading system new structures, institutions, etc. have to be established giving rise to additional administrative costs (cf. Pope and Owen 2009). An analysis by Heindl (2012) shows that especially for smaller emitters the operating costs of an emission trading scheme constitute a high burden. With respect to distributional impacts, it is obvious that allocating the certificates free of charge puts less pressure on the affected polluters. At the same time, one must not forget that under auctioning additional government revenue is generated, which can be used to decrease distortionary taxes or compensate those who have been affected negatively by the cap and trade system (cf. Goulder and Parry 2008). The free allocation mechanism can also be criticized on the grounds of generating high windfall profits, as the affected firms pass on the opportunity costs of the freely allocated permits to the consumer (cf. Sijm et al. 2006). On the other hand, free allocation is sure to increase the acceptability and political feasibility of an emission trading system.

"Soft" policy instruments

With the aim to cover all types of policy tools for environmental goals, so called "soft" policy instruments are introduced here as a third category, in addition to command-and-control and market-based mechanisms. These instruments are based on the cooperation principle and try to induce modifications in the behaviour of economic agents through incentives and the provision of information relying on voluntarism, learning processes and procedural change. Most importantly, information campaigns, voluntary agreements (that are not legally binding), environmental product labelling, public disclosure requirements, best practice dissemination and environmental management systems are counted in this category (cf. Hertin et al. 2004). In order to illustrate the classification of policy instruments, the approach of Bemelmans-Videc et al. (1998) can be utilized distinguishing between *carrots* (i.e. economic instruments which manipulate market incentives), *sticks* (i.e. command-and-control tools that entail a high level of coercion) and *sermons* (i.e. "soft" instruments that imply less constraints and mainly build on persuasion). Environmental subsidies are sometimes placed in the first category – here, however, they are treated as "soft" instruments given their usually limited scope and financial resources (cf. Rogall 2008, p. 244).

"Soft" policy instruments are mostly used to change the attitude of economic agents towards environmental action, provide information on possible emission mitigation options and to overcome barriers to (in many cases) cost efficient investments in energy efficiency. Their main advantage can be found in their high level of acceptability which facilitates their implementation. Moreover, these measures are usually associated with comparatively low administrative costs (cf. Gunningham and Sinclair 2004). On the other hand, one must not forget that the impact of these non-binding measures will hardly be sufficient to accomplish comprehensive environmental goals. Conceptually, they are no longer grounded on the basic ideas of environmental policy, like the polluter-pays-principle and the internalization of environmental externalities. Apart from that, subsidy schemes are often not cost efficient as they are likely to attract free riders. Critics have also expressed concern that a complete reliance on soft, non-binding measures might in the long run encourage a regulatory race to the bottom. Thus, on the whole one can conclude that soft environmentally policy instruments are only useful as complementary measures to precede and accompany more effective tools from the command-and-control or market-based categories (cf. Rogall 2008, pp. 248f).

2.4. Policies promoting environmental technologies

Rationale for technology policies

Economists usually argue that any environmental policy intervention should be clearly focused on the market failure it tries to correct. That means, if the purpose of a policy instrument is to internalize the negative externalities from greenhouse gas (GHG) emissions, it should directly aim at reducing these emissions taking into account all possible mitigation options. Yet, in reality, a large variety of policy instruments can be observed that try to foster the innovation and diffusion of certain environmental technologies that will help to abate GHG emission, especially in the area of renewable energies. Therefore, it needs to be analyzed whether there is a rationale for the implementation of specific instruments that encourage environment-related technological change or if it is sufficient and more cost efficient to concentrate on measures that directly target the reduction of GHG emissions, like emission trading systems or carbon taxes.

Environmental policy and technological change are closely intertwined (cf. Popp 2002). On the one hand, major technological innovations are required if substantial emission reduction targets are to be reached. On the other hand, standard emission abatement policies, like emissions taxes, per se already provide incentives to spur technological innovations. The effect that alterations in relative prices will have themselves on technological progress is referred to as the induced-innovation hypothesis and was first initiated by Sir John Hicks (1932). The question remains, however, whether environmental policies alone are capable of bringing about the socially optimal rate of innovation. If environmental externalities were the only market failure inherent to environmental technologies, no argument to justify any additional, specific measure for technology promotion could be brought forward.

It has, however, often been noted that the two steps of technology development, innovation and diffusion (or adoption), exhibit themselves market failures and external effects. Jaffe et al. (2005) divide these into knowledge externalities, adoption externalities as well as market failures due to imperfect information. Knowledge externalities, associated with the innovation phase, occur when a firm investing in the invention of a new technology is not able to capture all the benefits of this innovation for themselves, as other firm copy or imitate their technology or use the results in their own research. Hence, due to the characterization of new knowledge as a public good, innovators generate a positive external effect for other firms, often referred to as knowledge spill-overs. This results in a private return to innovation considerably lower than the social return. Consequently, R&D activities in the private sector will be less than socially optimal warranting governmental intervention in the form of public sector research, subsidies for private R&D, tax credits, stricter patent rules, etc. With respect to the diffusion phase of a new technology, the existence of additional market failures is less controversial. It has been argued that similar to the innovation phase, early adopters of a technology create positive externalities for later adopters and manufacturers through learning-by-using (on the demand side) and learning-by-doing (on the supply side), thereby giving rise to dynamic increasing returns. If this is the case, government action to stimulate the diffusion of new technologies through subsidies, technology standards, information campaigns, etc. could be justified. Furthermore, imperfections in the capital markets for technology development due to the high uncertainty of returns and the asymmetric distribution of information between developer and investor might lead to bottlenecks in the financing of innovation projects (cf. Jaffe et al. 2005).

It has to be pointed out, however, that all these arguments can only be used as a basis for a general promotion of technology development without concentrating on specific areas, like the environment. Economists have often argued that the government is not very well-suited for "picking winners" and that the market mechanism is more likely to channel funds to the most promising areas (cf. Lundvall and Borrás 2005). Yet, some reasons have been brought forward that could explain the particular focus on environmental technologies. First of all, special attention might be warranted given the public good nature of environmental commodities that renders them an area of government procurement (cf. Jarre et al. 2005). Besides, Goulder and Parry (2008) as well as Matthes (2010) argue that in order to achieve ambitious emission reduction goals extensive technological breakthroughs and the development of backstop technologies will be needed which should be fostered by means of targeted policy tools, especially in the light of the expected dramatic cost reductions due to learning effects. Apart from that, one must not forget the implications of the long timescales both for the formulation of climate change policies and targets and the turnover of energy capital stock such that the uncertainty about future emission prices or tax rates might dampen innovation activities today (cf. Fischer and Newell 2008; Montgomery and Smith 2007). In this context, Jaffe et al. (2005) also argue that in a second-best world where not all environmental externalities from climate change are yet internalized, technology policy might even play a more prominent role as such tools are more easily implemented than emission pricing policies.

On the whole, it seems that against the background of knowledge externalities and imperfect information, policy instruments aimed at supporting environmental R&D activities can be justified, while there is less consensus regarding the need for specific policies promoting the adoption of certain environmental technologies. In reality, though, governmental intervention can be found both in the area of innovation and adoption of environmental technologies. Especially support schemes for the market introduction and diffusion of renewable electricity technologies have gained momentum in recent years and will therefore be discussed in more detail in the following section.

Instruments for the promotion of renewable electricity

Increasing the use of renewable energies is seen as one of the major strategies to combat climate change. This is reflected in the fact that currently all 27 member states of the European Union have some type of support scheme for renewable electricity in place (cf. de Jager et al. 2011, p. 28). In addition to the justifications for technology promotion measures outlined in the previous sections, further arguments for supporting renewable electricity generation are brought forward by the proponents of such instruments. First of all, it is argued that renewable energies can contribute to energy security by a diversification of energy supply and a reduction of import dependency (cf. Olz et al. 2007, pp. 23ff). Yet at the same time, relying more heavily on renewable energies in electricity production might also pose risks to energy security due to the intermittency of important renewable sources like wind and solar and the need for scarce raw materials for some renewable energy technologies (cf. Sathaye et al. 2012, pp. 727f). Moreover, it is claimed that fostering renewable electricity technologies will lead to a creation of viable export industries and additional jobs. This argument is, however, very controversial given the fact that energy generation exhibits relatively high capital to labour ratios and in other areas job opportunities might be lost due to high electricity prices such that the net effect might be negligible or even negative (cf. Green and Yatchew 2012). Because of the decentralized generation structure of many renewable energies, the European Commission names rural development as an additional rationale for the promotion of renewable electricity generation (cf. EC 2009).

A variety of policy instruments has been applied to promote the use of renewable energies in electricity production. As with market-based instruments, a differentiation is made between price-based and quantity-based measures (cf. Menanteau et al. 2003). Fixed feed-in tariffs (FIT) form part of the first category as the price for renewable electricity is set exogenously. They consist of an obligation on the side of the energy utilities to purchase electricity from renewable producers at tariffs that have been fixed beforehand by the regulator and usually apply for a period of several years. The additional costs that distributors incur in this system are usually passed through to power consumers by means of a levy on end-use electricity prices. Apart from that, fiscal incentives or investment grants also belong to the price-based measures, but are usually only employed in a complementary way. One of the most important quantity-based instruments are tradable green certificate (TGC) schemes, where electricity utilities have to cover a fixed quota of capacity or generation of electricity by renewable sources. In this system, a utility has the choice between generating the required amount themselves and buying certificates (representing a certain amount of renewable electricity) that renewable generators or other utilities, which exceed their quota, sell. Thus, the quantity of renewable electricity is set by the government, whereas the price is determined on the market. This also applies to the other important quantity-based tool, tendering procedures, where a predefined target of renewable capacity or generation is assigned through a bidding process to the bidder with the lowest price. As with the other instruments, the additional costs can be financed through a surcharge on electricity prices.

A variety of studies has evaluated the advantages and disadvantages of the different support schemes based on model analyses as well as the experience in European countries and the Unites States (cf., amongst others, Ragwitz et al. 2007; IEA 2008; Sawin 2004; Menanteau et al. 2003; Green and Yatchew 2012; Butler and Neuhoff 2005; Schmalensee 2011). It becomes apparent, that one of the distinguishing features between feed-in tariffs and green certificate schemes is the fact that the latter typically offers a uniform level of support to all types of renewables, while in the former system usually technology-specific tariffs are implemented. Consequently, one of the major benefits of uniform TGC schemes consist in their cost-efficiency as always the cheapest options to cover the renewable quota are chosen. Thus, such a system fosters the diffusion of those renewable technologies which are closest to market competitiveness. In a technology-specific FIT system, on the other side, the tariffs are generally fixed as a function of the generation costs of the different technologies. This allows for a higher technology diversification at the expense of a cost efficient realization of renewable electricity targets. Keeping in mind the expected large learning effects, however, it has been argued that in the long-run it might be more cost efficient to foster at once the market penetration of different types of renewable technologies in order to realize those cost reduction potentials (cf. Ragwitz et al. 2007). To reduce the cost burden and the free-rider effects under a FIT scheme, regular adjustments of the tariffs as well as automatic tariff degression mechanisms can be applied. In a tendering scheme, cost efficiency is assured through the bidding process assuming that all types of renewable technologies take part in the system. With respect to dynamic efficiency, all support systems are supposed to have strong incentives on innovation, either because of competition (TGC and tendering schemes) or the possibility to increase the profit margin (FIT).

While cost efficiency is one of the major criteria to evaluate renewable support systems, some studies also pay attention to the resulting public costs, i.e. the transfer costs for consumers or taxpayers. Even though a system features high cost efficiency through a minimization of generation costs, the burden on consumers might still be high if the renewable generation sector is able to generate high profits. Hence, it is argued that in addition to maximizing cost efficiency, support schemes should be formulated in such a way that producer rents are limited (cf. Resch and Ragwitz 2010). Here, again a difference between uniform FIT or TGC schemes and technology-specific tariff systems can be observed. While a uniform remuneration tends to lead to an over-subsidization of less costly technologies resulting in high profit margins for producers, stepped tariffs that reflect disparities in generation costs of renewables can limit the producer surplus and the additional costs for consumers. Figure 2-5 offers a stylized illustration of this distinction.

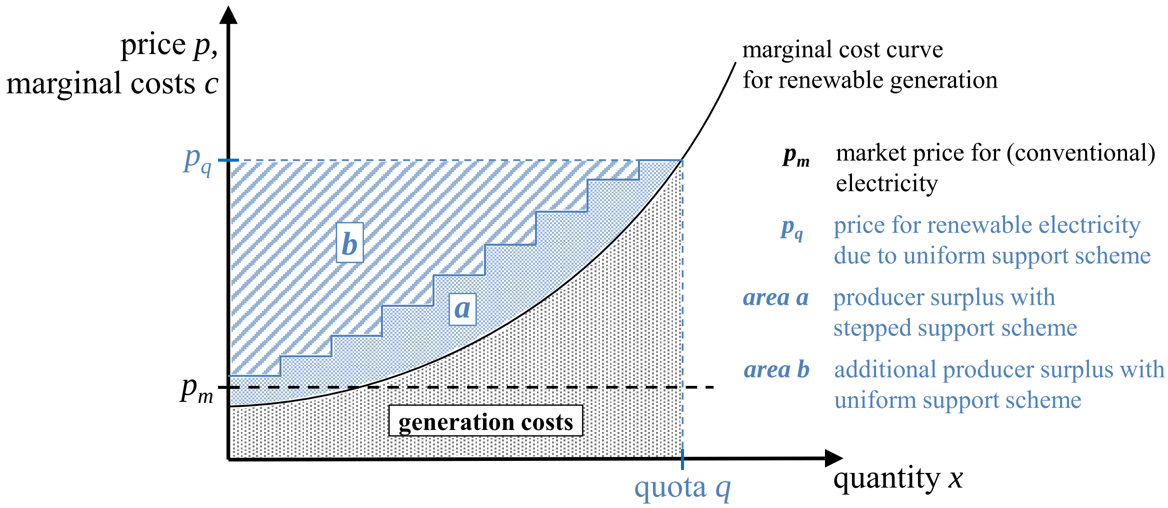

Figure 2-5: Comparison of producer surplus with uniform and stepped support schemes for renewable electricity generation (own illustration based on Ragwitz et al. 2007, p. 101)

One of the arguments in favour of TGC systems is their greater compatibility with conventional power markets such that renewable electricity generators are from the beginning exposed to market forces making the integration of renewable electricity into the electricity market less complicated. As the future amounts of renewable generation are fixed beforehand, TGC schemes can also be more easily coordinated or combined with other policy instruments, in particular emission trading schemes. In order to increase the market orientation of FIT systems, instead of fixed tariffs, which are independent of the market electricity price, premiums for renewable electricity paid on top of the electricity price can be implemented. On the other hand, this undermines one of the major advantages of fixed tariffs schemes, i.e. the planning security for investors. The implementing and monitoring costs on the government side are highly dependent on the complexity of the respective support system. The transaction costs, especially for small producers, tend to be smaller under pricing systems (cf. Sawin 2004).

Altogether, it can be observed that support instruments for renewable electricity have gained increasing significance in recent years, with a clear focus on FIT systems in Europe and a stronger emphasis on TGC schemes in several U.S. states (cf. Schmalensee 2011). Many of these systems have been highly successful in stimulating renewable electricity generation. At the same time, the question remains whether a clear justification for the specific promotion of the adoption of renewable technologies based on environmental externalities from climate change or market failures adherent to technological innovation and diffusion can be found.

2.5. The use of multiple policy instruments and policy interaction

Economics literature on environmental policy usually sees different policy instruments as alternatives and therefore concentrates on comparing the features of different types of policy instruments. In political reality, however, in most cases several policy instruments are implemented to address an environmental externality, like for example climate change, and it is generally stated that such a policy mix is more suitable to achieve environmental targets (cf. for example OECD 2007b; Giljum et al. 2006). According to the Tinbergen rule, the number of policy instruments should match the number of political targets such that each target is covered by a specific measure while at the same time limiting interaction which could arise from an excessive number of instruments (cf. Tinbergen 1952; Knudson 2008). Hence, it needs to be clarified under which circumstances the use of multiple instruments might be justified.

Analyses on the rationale of using a policy mix for environmental externalities are usually set in a second-best world. This means that there exists some kind of constraint in the general equilibrium system making it impossible that one of the conditions of Pareto optimality is reached leading to a situation where the attainment of the other Pareto conditions might no longer be welfare improving (cf. Lipsey and Lancaster 1956). Bennear and Stavins (2007) name the two most important incidences where this might be the case in the context of environmental policy: political constraints and market failures. When evaluating environmental policy instruments, pre-existing political distortions stemming, for example, from the tax system have to be taken into account. This can lead to the result that in combination with the tax system already in place, implementing a revenue-raising instrument, like a pollution tax or an auctioned permit system, has the benefit of providing the possibility to cut other distortionary taxes. The fact that multiple market failures can warrant the use of multiple policy instruments has already been shown for policy tools fostering technological innovation and diffusion.

Moreover, analyses have highlighted that other types of market failures play a role when designing environmental policies. First of all, barriers to more energy-efficient investment due to asymmetric information and behavioural issues can be overcome by additional measures like information campaigns, labelling systems or subsidies for energy audits (cf. Lehmann 2008). Secondly, market failures related to split incentives, most importantly the landlord/tenant dilemma, might need to be addressed by specific instruments promoting, for example, energy-saving measures in rented buildings or heating contracting (cf. OECD 2007b, p. 26). When taking transaction costs into consideration, the first-best solution might no longer be optimal as it causes high administrative and monitoring costs. This is especially the case with non-uniformly mixed pollutants, where a combination of a tradable permits scheme with localized emission standards might be advantageous, or with situations where emissions are difficult to monitor thus making enforcement more challenging. Here, a combination of a tax and a subsidy (e.g. a deposit-refund system) might represent the second-best optimal solution (cf. Lehmann 2008). Furthermore, several recent studies have pointed out the benefits of implementing so called hybrid instruments, where a price-based and a quantity-based market-oriented tool are combined in order to reduce the uncertainty, either regarding the emission level or the price of emission, that arises if one of the instruments is implemented alone (cf. Jacoby and Ellerman 2004; Hepburn 2006; Murray et al. 2009; Philibert 2009; Fankhauser et al. 2011). For example, by complementing an emission trading scheme with an emission tax that can be chosen in the case of high marginal abatement costs, a safety valve in the form of a cap on the certificate price is created.

Hence, it can be demonstrated that under certain conditions the use of multiple policy instruments to address environmental issues can be justified. From this, however, it cannot be concluded that the combinations of policy instruments that are currently in place actually constitute socially optimal mixes. Moreover, attention must be paid to the interactions between different policy instruments in order to create a coordinated and consistent policy mix (cf. Bennear and Stavins 2007).

An extensive study on the nature of policy interactions in climate policy has been conducted within the scope of the INTERACT project (cf. Sorrell et al. 2003). Here, a basic definition of policy interaction is provided as follows:

"Policy interaction exists when the operation of one policy affects the operation or outcomes of another." (Sorrell et al. 2003, p. 27)

In order to evaluate the interaction between policy instruments in a specific case, a theoretical background on the different types of interactions and their potential effects is needed. Policy interactions can be defined along different lines (cf. Oikonomou and Jepma 2008):

- *Internal vs. external*: Two policy instruments can either operate in the same policy area (e.g. two environmental policy instruments), or in different ones (e.g. an environmental policy instrument and a fiscal policy instrument).
- *Horizontal vs. vertical*: Horizontal interactions refer to two instruments that are implemented at the same level of governance (e.g. the EU level), while vertical interactions occur in the case of instruments on different levels of governance (e.g. one instrument on the EU level, one on the national level of a member state).
- *Direct vs. indirect*: An interaction is classified as direct if one specific target group is directly affected by both policy instruments in question. With indirect interactions, on the other hand, at least one of the policy instruments influences the target group only indirectly (e.g. the group is not directly targeted by the policy instrument but impacted by the adjustments that are made by a directly affected target group).

According to Sorrell and Sijm (2003), the analysis of policy interactions in climate policy involves several steps. First of all, it is helpful to define the *scope* of each instrument in order to identify possible overlaps (both in terms of directly and indirectly affected target groups). This is followed by an examination of the *objectives* of each policy tool. In climate policy, all instruments should generally have the same target, i.e. the mitigation of GHG emissions – assessed along the lines of ecological precision, cost efficiency, dynamic efficiency and the other evaluation criteria outlined in Chapter 2.2. When taking a closer look, however, it becomes apparent that environmental instruments often follow additional objectives besides emission reduction such as technology promotion, reduction of import dependency or even broader economic policy goals. This further complicates the evaluation of policy interactions. In the next step, the *operation* of the instruments, e.g. their joint effects on the different target groups, is determined. With respect to a given policy target, the combination of two instruments can have different implications ranging from conflicting over neutral to reinforcing. In this context, Gunningham and Sinclair (2004) speak of "*inherently complementary combinations*" (where the effectiveness and the efficiency of the instruments is increased when used in combination with each other), "*inherently counterproductive combinations*" (where the effectiveness and the efficiency is clearly deteriorated through the interaction), and "*combinations where the outcome will be context-specific*". Other areas that need to be considered in the process of evaluating policy interactions comprise the analysis of the *implementation* of the policy instruments, looking at the potentials for coordination and rationalization of the administrative necessities, and the *timetable* of each instrument.

3. Energy models for policy evaluation

With the background on the different types of policy instruments and the most common evaluation criteria, the following chapter looks at the different modelling approaches that can be applied to analyse the long-term impacts of different environmental policy instruments on the energy system and the economy as a whole. In addition, the ideal requirements for a quantitative modelling framework for policy evaluation are explored and contrasted with the attributes of existing energy models.

3.1. Overview on energy modelling

Major progress in scientific modelling approaches is usually fuelled by changing demands from policy makers. Accordingly, even though the origins of energy modelling go back to the 1960s, the most important quantitative energy model tools used for the long-term evaluation of possible future energy paths were developed in the 1970s. In this decade, the two oil crises gave rise to an increased focus on energy policy concerns and corresponding efforts for the development of energy system models (cf. Hoffman and Jorgenson 1977). This development is also reflected in the creation of the Energy Modeling Forum (EMF) in 1976 with the aim to *"improve the use and usefulness of energy models in the study of important energy issues"* (Sweeney and Weyant 1979, p.1).

Energy system models provide a consistent tool for decision making and planning for complex problems in energy policy or for energy utilities. The aim, therefore, does not consist in predicting the exact future development of the energy system, but to analyse possible trends in energy supply and demand, from which so called "robust steps" can be identified, i.e. decisions and actions that turn out to be necessary and appropriate even when taking a wide uncertainty range of the most significant influencing factors into account (cf. Voß 1982). For that reason, energy modellers usually look at various scenarios in order to evaluate different potential energy futures. Scenarios can be described as *"plausible, challenging and relevant stories about how the future might unfold"* (Raskin et al. 2005), based on a consistent set of assumptions on the most important determinants in the energy system.

In order to assess energy models according to their applicability for policy evaluation it is helpful to look at the different types of modelling approaches that exist today. From the beginning, a differentiation has been made between two broad categories: top-down and bottom-up models. Top-down models look at the energy system "from above", i.e. from a macroeconomic perspective. This entails a high level of aggregation, while at the same time a full equilibrium framework is applied taking into account all repercussions of the energy system on the rest of the economy. Today, the field of top-down energy modelling is dominated by computable general equilibrium (CGE) models, while input-output and macroeconometric models are other important examples (cf. Möst and Fichtner 2008). In these models, the relationships between the different sectors of the economy are represented with the help of aggregate supply and demand functions. Hence, the technical production conditions in the energy sector are modelled on the basis of production functions describing the substitution possibilities between the production factor energy and the other input factors (usually capital and labour). The most important model parameters, which are usually estimated based on historical data, are therefore the elasticities of substitution (ESUB), determining the substitutability between two inputs, and the autonomous energy efficiency index (AEEI), denoting the rate of price-independent progress in energy productivity. From this it also follows that no individual technologies can be represented in top-down models (cf. Jaccard 2009).

While top-down models are rooted in economic principles, bottom-up energy system modelling approaches have been mainly developed in engineering. These models depict the energy system "from below", i.e. the entire energy system from primary energy supply to energy services demand in the different end-use sectors, including all conversion steps, is described in a process-oriented manner. Thus, a large variety of technologies, both on the energy supply and demand side, are functionally modelled with their economic, technological and ecological parameters. In the model, the energy system is then represented as a network of processes (technologies) and commodities (energy carriers, materials, etc.), the so called reference energy system (RES). This makes it possible to base the substitution between different energy carriers and input factors on an explicit choice between different technologies. Given the high level of technological detail, energy system models are partial equilibrium models. The analysis focuses on the energy system, while macroeconomic repercussions are not taken into account in the traditional approach. A differentiation can be made between models where the demand for energy services and useful energy is exogenously given and fixed and those which assume own-price elasticities for the different demand categories (actual partial equilibrium approach). With respect to the mathematical formulation, two main approaches can be distinguished. Simulation models describe the development of the energy system based on exogenously defined scenario assumptions, while optimization models calculate the optimal configuration of the energy system given the objective function and a set of constraints that contain the technical limitations, demand assumptions, political objectives (e.g. emission reduction targets), etc. In most cases, the objective function comprises total energy system costs such that the optimization problem consists in either minimizing net total cost (with exogenously given demands) or maximizing net total surplus of suppliers and consumers (with own-price elastic demands) (cf. Remme 2006, pp. 79ff).

3.2. Ideal attributes of energy models for policy evaluation

The policy environment in which energy models are applied today has changed significantly since the 1970s. With the issue of climate change, energy policy has acquired a new focus. Hence, given the necessity of introducing new policy instruments, the role of energy modelling in designing and evaluating such instruments has increased considerably. Simultaneously, research priorities have undergone a substantial shift. In the 1970s and 1980s the main concern of energy policy was clearly energy security, such that the principal aim of energy system modelling was to assess the potentials of different technologies to derive cost-effective energy savings with financial costs being the main driver in these models. Today, in contrast, in addition to identifying technological pathways to arrive at certain climate goals, analyses concentrate also on the question of what could be achieved with different types of policy instruments taking into account the conditions under which such instruments would operate, e.g. behavioural and political constraints (cf. Worrell et al. 2004).

Against the background of this growing research need, the question arises what could be the contribution of energy system models in this process of evaluating policy instruments. Having the basic differentiation between bottom-up and top-down models in mind, one can also look at the question from another perspective examining what features an energy model would ideally need to possess to be appropriate for policy evaluation. Here, the approach initiated by Jaccard et al. (2003) provides a valuable starting point. It identifies three criteria that an ideal model would require to be useful for the assessment of different types of policy instruments: Technological explicitness, microeconomic realism and macroeconomic completeness (cf. Figure 3-1).

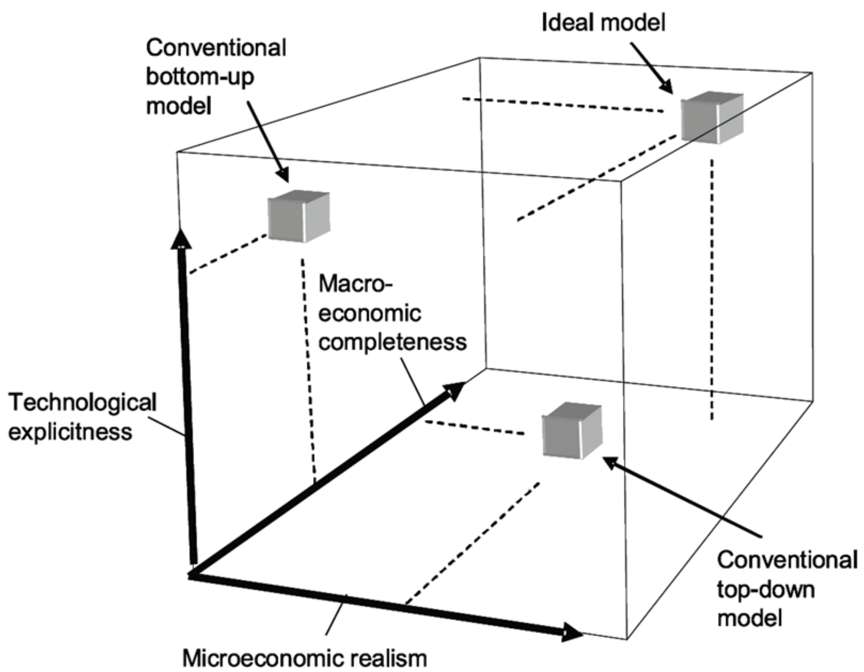

Figure 3-1: Dimensions of an ideal energy model for policy evaluation (Hourcade et al. 2006)

In this context, technological explicitness refers to the fact that energy models for policy evaluation should include information on a large variety of technologies as represented by their technical and economic characteristics. This is particularly important as many climate policy approaches tend to focus on a limited set of technologies, whose consequences for industrial production or household consumption is manageable and therefore enjoy greater political acceptability. The impact of such policy measures can only be evaluated when these technologies are described in an explicit way in the model. Moreover, a successful fulfilment of ambitious emission reduction goals depends greatly on the future breakthrough of different low-emission technologies, for which potentials, cost assumptions (including cost reductions through learning effects) and possible technological improvements also need to be incorporated in the model.

The second dimension, microeconomic realism, comprises all factors that influence the decision-making behaviour of firms and households. When choosing between different technologies that provide the same energy service, financial costs play of course the dominant role. Yet other aspects, like intangible cost terms representing differences in quality or other non-economic factors, perceived risk associated with the investment in new technologies or heterogeneous preferences, can also affect technology choice. Thus, not taking account of these factors in the model might result in over- or underestimating the impact of environmental policy instruments, especially in the household sector.

Finally, it has to be kept in mind when evaluating instruments of energy and climate policy that the energy system does not operate as a separate entity but is part of the economic system as a whole. Therefore, the third dimension highlights the necessity of integrating macroeconomic feedbacks into the model, since environmental policy instruments can have an effect on the structure of the economy and total output. It is useful to mention, however, that the importance of this aspect depends on the scope of the analysed instrument in terms of regional, sectoral and technological coverage, as the macroeconomic implications of a technology-specific measure clearly concentrated on one specific sector can be assumed to be rather limited.

It has to be noted that using energy system models for policy evaluation entails a change in perspective. Usually, such models assume the perspective of a social planner simultaneously minimizing total discounted costs (or maximizing welfare) of the entire system (cf. Keppo and Strubegger 2010). Accordingly, these analyses do not take into account any state influence on the market prices in terms of taxes or subsidies as these only constitute a redistribution of resources among different economic agents. Furthermore, a social discount rate is used reflecting the opportunity cost of capital for the economy as a whole. When, however, the aim consists in assessing the impact of environmental policy instruments, the perspective of the energy systems analysis needs to be modified, because the policy effects depend strongly on the individual decision-making of different economic agents facing market prices including all forms of state influence and applying private discount rates in their investment choices. Hence, the viewpoint can no longer be that of a social planner but the individual perspective of households, firms, etc. needs to be taken into consideration (cf. Ostertag et al. 2000, pp. 35ff). The change in the research focus also has an impact on the way scenarios are constructed: while most studies that estimate technical potentials utilize goal-oriented scenarios, i.e. scenarios that incorporate exogenously fixed target values, for example for emission reduction or the minimal use of renewable energies, analyses that look at the effect of a given policy instrument refrain from setting specific targets a priori and rather examine what the policy tool can contribute to a certain policy objective.

3.3. Strengths and weaknesses of energy system models in policy evaluation

At this point it has to be highlighted that the goal cannot consist in finding or creating the ideal modelling tool that will answer all questions related to policy evaluation. Each tool is developed for a specific purpose and will therefore have its strengths and weaknesses. Such being the case, it is more useful to look at the existing modelling approaches and assess them against the criteria established in the previous section.

The main advantage of bottom-up engineering models can be easily identified in their high level of technological detail. Hence, it is the only approach that can be applied to evaluate the effect of technology-specific measures and, even more importantly, to incorporate the impact of new technologies, for which no historical data is yet available (cf. Hoffman and Jorgenson 1977). Bottom-up energy system models also provide the possibility to model technology competition and to integrate the long-term trends in technology costs depending on their installed capacity with the help of learning curves. In estimating future developments, less reliance is put on historical data, whose main purpose usually is only to calibrate the model to the base year. In this way, it is feasible to model technological breakthroughs and other discontinuities, which can be surely expected in the face of ambitious emission mitigation targets (cf. Swan and Ugursal 2009).

At the same time, engineering energy models exhibit some drawbacks regarding the other two attributes of an "ideal" model outlined in the previous section. Bottom-up energy system models have often been criticized for ignoring critical aspects of the decision-making behaviour of different economic agents, especially private households. As Webler and Tuler (2010, p. 2690), put it: "*Getting the engineering right is not always enough*", there are other dimensions that influence decision-making. Engineering models usually rely on financial costs as the key decision variable for technology choice assuming that technologies that provide the same energy service can be regarded as perfect substitutes (cf. Jaccard 2009). This gives rise to a number of issues concerning consumer behaviour. First of all, limiting the analysis to pure financial costs implies that other significant cost elements, like transaction costs or intangible costs related to non-economic factors, are overlooked. Apart from that, bottom-up models generally have the underlying assumption of unbounded rationality, thereby disregarding important market imperfections (cf. Mundaca et al. 2010). This has often been reflected in the use of a social discount rate in the assessment of investment decisions of households and firms. Finally, the concept of the representative consumer adopted in energy system models ignores the influence that diverging preferences and market heterogeneity can have on technology diffusion.

An additional point of criticism that has been voiced with respect to the traditional approach of bottom-up energy system models is their lack of taking repercussions on the rest of the economy into account. This holds especially true for models that use fixed demands for energy services or useful energy. In doing so, the flexibility of the energy system to respond to changes in prices or policy measures is clearly underestimated, possibly leading to an overestimation of the costs of emission abatement measures (cf. Worrell et al. 2004). Furthermore, it has to be kept in mind that important aspects of policy evaluation, especially the impact of environmental policy instruments on the structure and level of economic output, employment, income distribution, etc., cannot be carried out with the help of engineering models. Some other drawbacks of bottom-up energy system models that have been mentioned in a number of studies include the large data requirements, often at a level of detail not easily available, and the fact that in policy evaluation additional cost parameters, like administrative or programme costs, have in some cases not been considered (cf. Swan and Ugursal 2009).

When looking at top-down energy modelling tools, it can be observed that areas where bottom-up models exhibit weaknesses are usually those where top-down models have their greatest strengths. An intrinsic characteristic of the top-down approach consists in its inclusion of all macroeconomic feedbacks. Moreover, as all decisive model parameters are generally estimated from time-series data, behavioural aspects are also, at least roughly, taken into account (cf. Swan and Ugursal 2009). At the same time, however, this strong dependence on historical data also constitutes a major drawback of these models in policy evaluation. It is doubtful that, especially for crucial parameters like the elasticities of substitution and the autonomous energy efficiency index, the historical development correctly reflects future trends given the substantially different political challenges, energy price levels and technological options (cf. Bataille et al. 2006). Thus, there is a risk that models tend to only project current trends into the future and thereby reinforce the status quo (cf. Laitner et al. 2003). Most climate policy instruments rely heavily on the realization of significant technological breakthroughs, which cannot be modelled if technological change is treated exogenously. In general, the most critical weakness of top-down models is their high level of aggregation making it impossible to evaluate technology-specific policy instruments. On the other hand, this is also why data requirements for top-down approaches are much more limited than for bottom-up tools (cf. Swan and Ugursal 2009).

In the past, both modelling approaches have been applied to evaluate policy scenarios with different emission reduction levels. Considerable differences can be observed when looking at the estimated carbon abatement costs, with cost results from bottom-up models usually being substantially lower than those from top-down approaches. Several reasons can be brought forward for this trend (cf. Schäfer 2012). Firstly, by disregarding behavioural factors, like barriers to energy-efficient investment, and using social discount rates, engineering models tend to indicate large potentials for emission mitigation at low costs. Secondly, important economic factors, like rebound effects, that dampen the expected energy savings from improvements in energy efficiency, are not taken into account. Finally, the assumption of perfect foresight underlying most bottom-up energy system models makes sure that the most cost optimal transition path is found in the long-term. Factors that lead to an overestimation of abatement costs in top-down models comprise the low level of technological flexibility and the dependence on parameters that are estimated from historical data.

Given the limitations and strengths of both model approaches, efforts have been undertaken, especially with the growing focus of energy policy on greenhouse gas abatement, to combine them and create so-called "hybrid" models (cf. Hourcade et al. 2006) (cf. Figure 3-2). Here, the strategy is either to increase the technological detail in existing top-down approaches (cf. for example the models WITCH (Bosetti et al. 2009), ReMIND (Schmid et al. 2012) and IMACLIM-R (Sassi et al. 2010)) or to include macroeconomic feedbacks and behavioural parameters in bottom-up tools (cf. for example the models CIMS (Jaccard et al. 2003), GCAM (Calvin 2011) and MARKAL-MACRO (Loulou et al. 2004)). In the following, the two main weak points of traditional bottom-up energy system models when applied for policy evaluation, concerning the behavioural and macroeconomic dimension, will be described in more detail with the major aim to illustrate approaches that have been developed so far to address these issues.

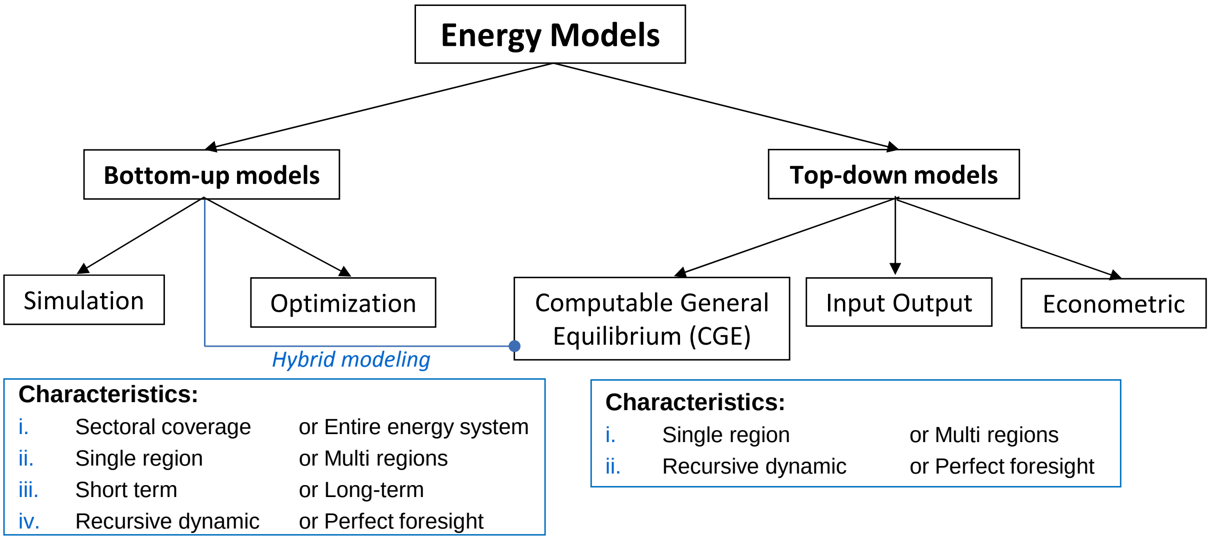

Figure 3-2: Classification of energy models

3.4. Main challenges (1): Consumer behaviour

In the previous sections it has already been established that investment decisions by households or firms do not only depend on financial costs but on a number of additional drivers. Hence, for a realistic assessment of the impact of policy instruments, these factors need to be taken into account in energy system models. Before looking at different modelling strategies, it is, however, crucial to have some background information on the issue of decision-making behaviour.

The debate on the energy paradox

The discussion on the varying representation of consumer behaviour in energy modelling evolved mainly around the debate on the energy paradox in the 1980s and 90s (cf. Hourcade et al. 2006). This phenomenon, also referred to as the energy-efficiency gap, describes the fact that there seems to exist a substantial gap between the actual investment in more energy-efficient technologies and the level perceived as socially optimal (cf. Jaffe and Stavins 1994). In economic terms, the decision for or against an energy efficient device should be based on a net present value calculation weighing the higher up-front investment costs against the long-term (discounted) savings in operating costs. It has been observed, however, that in reality investments which prove to be cost efficient based on their net present value are often not realized. Accordingly, bottom-up energy system models identify significant potentials for cost efficient investments in energy efficiency, while in the economic perspective of top-down models the existence of an energy-efficiency gap cannot be acknowledged based on the assumption of rational behaviour and perfect markets (cf. Hourcade et al. 2006).

It is essential to understand the reasons behind the energy paradox in order to be able to evaluate the effectiveness of different types of policy instruments in reducing the energy-efficiency gap and to arrive at a realistic modelling approach. Here, it needs to be pointed out that increasing energy efficiency cannot be considered as a goal in itself, but only as a means to achieve a more efficient resource allocation, i.e. higher economic efficiency. Jaffe and Stavins (1994) summarize all factors that may explain the low investment activity in energy efficiency under the term *market barriers*. A differentiation is then made between *market failures* and *non-market failures*.

Market failures describe all incidents in which the market mechanism does not lead to the optimal allocation of resources. Thus, a government intervention might be justified, if it leads to an improvement in overall welfare. With respect to investments in energy efficiency, a main source for market failures is often identified in the limited availability of information (cf. Howarth and Sanstad 1995). Information is generally viewed as a public good resulting in an underprovision through private markets. Thus, consumers might not possess all the necessary information about possible future energy savings in order to make an appropriate investment choice. Additional market failures that can be ascribed to information problems consist in the positive knowledge externalities that early adopters create through learning-by-using for potential other adopters. For this, they receive, however, no compensation thereby lowering the incentive to adopt new technologies (cf. Gillingham et al. 2009). Especially in the building sector another type of market failure can be observed in terms of the split-incentive problem, where the party that decides on and pays for an investment (e.g. the builder or landlord) is not the one who profits from the reduced energy costs (e.g. the purchaser or tenant) (cf. Murtishaw and Sathaye 2006). In some studies, liquidity constraints are also named as a cause of less than optimal investments in energy efficiency (cf. Blumstein et al. 1980). There exists, though, less empirical prove for this incident and if it is the case, it is no problem specific to investments in energy efficiency.

Yet, it is not certain that the energy-efficiency gap can be explained entirely on the basis of market failures. Jaffe and Stavins (1994) highlight another set of factors that cannot be attributed to a malfunction of the market mechanism. These non-market failures mainly comprise costs factors that consumers incur when investing in energy efficient devices, but are not included in simple net present value calculations. Hence, their behaviour is still rational and the optimal investment choice is made. First of all, transaction costs fall into this category which are often higher for new technologies where less user experience has been gained so far. Closely related to that is the higher perceived risk that consumers might attach to a new technology (cf. Groves 2009). Apart from that, one must not forget that investments in energy efficient products usually have long payback periods such that uncertainties regarding future energy prices and the market trends for new technologies need to be taken into account in the investment decision (cf. Sorrell 2004). Furthermore, consumers might perceive a new, more energy efficient technology not as a perfect substitute to an old one due to differences in quality (as for example fluorescent compared to incandescent lighting). In this context, market heterogeneity also plays a crucial role. For example, the potential energy savings will have more weight in the investment decision of a consumer who will use the product very frequently compared to another consumer who rarely makes use of it (cf. Jaffe et al. 1999). Here, a number of non-economic factors also come into play, like differences in comfort, design, etc. (cf. Mundaca et al. 2010).

The theory on market and non-market failures is still based on the assumption of rational behaviour. Studies from the behavioural economics literature have, however, observed that this assumption cannot always be applied to consumer choices and have therefore introduced the concept of bounded rationality or other heuristic decision-making methods (cf. McFadden 1999). This implies that consumers do not always possess the ability and resources to process all the information required to arrive at the optimal solution (cf. Shogren and Taylor 2008). That is why Gillingham et al. (2009) have introduced *behavioural failures* as a third category to explain the energy-efficiency gap. Here, social and cultural norms as well as the influence of family and acquaintances (cf. Dawnay and Shah 2011) also can have a decisive impact on investment choices. It might be justified to implemented specific policy instruments, like educational or information programmes, to address these behavioural failures given that this leads to an increase in welfare.

Jaffe et al. (1999) have shown that the different concepts of the energy-efficiency gap can result in alternative notions on the optimal level of energy efficiency. In Figure 3-3 different levels of economic efficiency are contrasted with different levels of energy efficiency. As has already been stated, the ultimate aim consists in maximising economic efficiency. Starting from the zero point which represents the reference case where no policy instruments are in place, there exists the possibility for specific policy interventions dealing with market (and behavioural) failures. As a result, both economic and energy efficiency are increased, creating a “win-win” or “no regrets” situation. This is referred to as the *Economists’ Narrow Optimum*. In contrast, bottom-up engineering models tend to arrive at a different optimum when simply minimizing investment and operating costs, the *Technologists’ Optimum*. Here, a higher level of energy efficiency is reached at the expense of the overall welfare level, as both market and non-market failures are eliminated. Jaffe et al. (1999) then consider an additional optimum, the *True Social Optimum*, where all other market failures, most importantly environmental externalities, are removed up to the point where benefits (in terms of economic efficiency) exceed the costs.

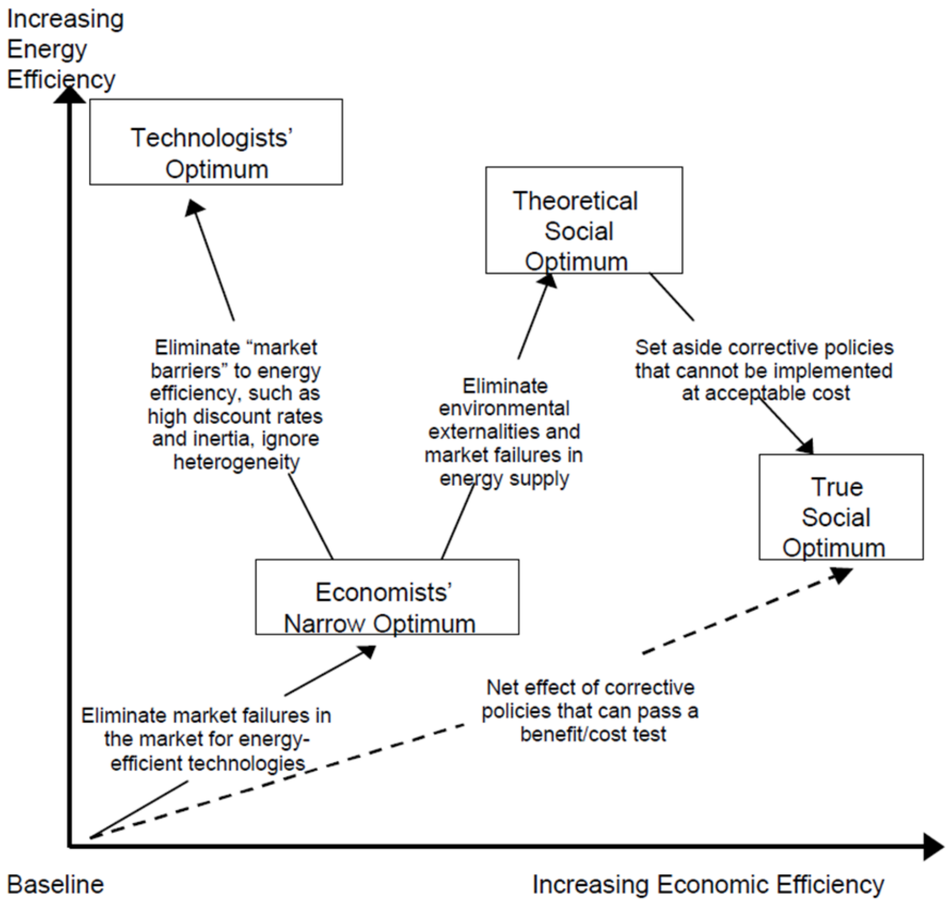

Figure 3-3: Alternative concepts on the optimal level of energy efficiency (Jaffe et al. 1999)

To conclude, it can be observed that by ignoring market barriers to energy efficient investments, bottom-up energy system models tend to overestimate the potentials for cost efficient energy savings. When modelling these barriers, the differentiation between market and non-market failures needs to be kept in mind.

Modelling approaches to incorporate consumer behaviour

As it can be understood from the previous section, the large variety of factors influencing consumer behaviour makes it all the more difficult to arrive at a realistic representation of decision making in energy system models. What is more, after integrating parameters on consumer behaviour, an understanding needs to be developed on how policy instruments might influence these parameters.

A basic approach to model investment barriers in energy system models has consisted for a long time in using high implicit discount rates (cf. Mundaca et al. 2010). These can be described as the subjective rates that can be empirically observed in the investment decisions of private households or other economic agents. This has the advantage that extensive empirical research has been conducted on these discount rates, starting with Hausman (1979), Gately (1980) and Ruderman et al. (1987) who arrive at high estimates of in some cases over 100 % for implicit discount rates of energy-efficient equipment in households. Accordingly, in many energy system models sector-specific discount rates are applied, with higher rates for private households and private transport and lower ones for energy conversion and industry (cf. for example E3M-Lab (2011) for the PRIMES model and IEA (2005) for the ETP model). Discount rates can also be differentiated between technologies to reflect social acceptance. It becomes more problematic, however, when the aim shifts from describing the actual status quo to modelling the impact of policy interventions as many of these instruments target exactly those market failures that the high discount rate are supposed to reflect. Some analyses have taken the approach to mimic the impact of policy measures by lowering the discount rates (cf. Mundaca 2008; Božić 2007). Yet, it has to be noted that there exists no good empirical foundation on the effect of policy instruments on implicit discount rates such that this approach appears rather intransparent and dependent on expert judgement. Moreover, it involves the risk of mixing up the two components that are behind the market barriers (market and non-market failures) or even of ignoring the non-market failures by reducing the discount rates to the level of social discount rates (cf. Mundaca et al. 2010).

A more sophisticated method to include consumer behaviour in energy system modelling has been developed in the hybrid model CIMS (cf. Jaccard 2009). CIMS is a bottom-up technology-rich simulation model of the energy system that also includes macroeconomic feedbacks. To account for behavioural aspects, the calculation of the market shares of competing technologies is not only based on financial costs, but extended by the following parameters. A weighted average time preference rate is used for discounting which is the same for all technologies providing the same energy service, but can vary between different energy uses. Apart from that, a cost term is added to capital and operational costs that reflects all intangible costs and benefits of a certain technology. This might be, for example, the perceived drawbacks of using public transport instead of one's own car to fulfil the demand for personal transport. Finally, an additional parameter is incorporated to capture market heterogeneity, which prevents that the technology with the lowest life-cycle cost covers the entire market.

Murphy and Jaccard (2011) have illustrated the impact that integrating consumer behaviour into the framework of an energy systems analysis can have on the results, especially regarding greenhouse gas abatement costs. They have contrasted the marginal abatement cost curve for the US calculated (1) by McKinsey (2007) based on a conventional bottom-up approach and (2) with the CIMS model using, as far as possible, the same scenario assumptions. It shows that generally marginal abatement costs are higher when consumer behaviour is taken into consideration. Furthermore, in the calculations with CIMS, the contribution of energy efficiency to the entire reduction potential is less pronounced compared to the other options of greenhouse gas abatement.

As Jaccard (2009) highlights himself, the value of such a more complex modelling technique depends greatly on a good empirical foundation. In CIMS, discrete choice models based on stated preference surveys are mainly used to estimate the three behavioural parameters. Other modelling teams have taken up and further developed the approach from the CIMS model, as for example the Res-IRF model focusing on the residential sector in France (cf. Giraudet et al. 2011) and the BLUE model depicting the energy system of London (cf. Strachan and Warren 2011). In general, it has to be noted that this method has greatly increased the transparency of representing decision-making behaviour in energy system models. However, substantial uncertainties with respect to the estimation of the behavioural parameters persist. And just as it was the case with using implicit discount rates, it gets even more complicated when trying to evaluate the impact of different policy instruments on these parameters.

Another approach to deal with the sociological dimension of investments in energy efficient equipment has been introduced through the SOCIO-MARKAL model (cf. Nguene et al. 2011). Within the scope of the conventional bottom-up energy system optimization model MARKAL, the effect of awareness campaigns is modelled by introducing "virtual technologies". These include the cost of the awareness campaign, which, if used, directly has an effect on the investment decision (e.g. using more efficient light bulbs instead of the conventional ones). This technique has the advantage of being easily integrable into an existing linear optimization model. Yet, comprehensive sociological surveys need to be conducted in order to get realistic values for all the required parameters.

3.5. Main challenges (2): Economic flexibility

Problems arising from the partial equilibrium approach

Energy system models are constructed by definition as partial equilibrium models such that repercussions on the rest of the economy are not taken into account. Consequently, the impact evaluation of environmental policy instruments is also restricted to the energy system when using this modelling technique. To measure economy-wide effects, for example on GDP, employment and trade, other approaches, like CGE models, can be applied. Nonetheless, some problematic issues arise from using a partial equilibrium approach for policy evaluation, even when restricting the analysis to the energy system.

First of all, by fixing the demand for energy services or useful energy, as it is the case in many conventional bottom-up energy system models, the flexibility of the energy system in reacting to rising energy prices or emission reduction targets is considerably restricted (cf. Worrell et al. 2004), because one important abatement option is disregarded: while the abatement options of increased energy efficiency, fuel substitution as well as, in most cases, of carbon capture and storage are taken into consideration in such models the possibility of reducing demand for energy services is neglected.

Secondly, it has to be kept in mind that changes in energy prices or greenhouse gas reduction targets can also have an impact on the structure of the economy, as for example the production in energy-intensive industry branches, like iron and steel, needs to be diminished (cf. Bataille et al. 2006). These adjustments cannot be estimated with the help of partial equilibrium models, yet they can have a significant effect on the energy demand. Closely related to this issue is the aspect of carbon leakage, i.e. the shift of (energy/emission-intensive) production from countries with stringent mitigation objectives to countries with no or weak target values (cf. Barker et al. 2007). This mechanism can only be evaluated with the help of supranational/global CGE models.

Finally, an intense debate has developed in recent years regarding the integration of rebound effects in energy system models. The results from bottom-up approaches tend to indicate large energy saving potentials from improvements in energy efficiency. However, the assessment changes when taking into account the energy cost savings that follow from the higher level of energy efficiency. Then it is likely that energy efficiency improvements fall short in delivering the expected energy saving potential, since the associated cost savings may actually encourage greater demand for energy services.

A differentiation can be made between different types of rebound effects (cf. Figure 3-4). A *direct rebound effect* arises when the demand for the energy service where the efficiency improvement was realized is increased due to the decrease in energy costs (cf. Berkhout et al. 2000). For example, a consumer who has bought a more fuel-saving car might choose to drive more given the lower fuel costs. According to microeconomic theory, this effect can be divided into a substitution and income (for households) or output (for producers) effect. The substitution effect defines how the now cheaper energy service substitutes for the consumption of other goods and services (or input factors) at a constant level of utility/output. The income (or output) effect describes the movement to a higher level of utility (or output) through an increase in consumption of all goods and services (or inputs) (cf. Sorrell 2007). The *indirect rebound effect* also comprises several components. First of all, the energy that is used to produce the equipment that is needed to improve energy efficiency has to be taken into account (denoted as embodied energy). Apart from that, the cost savings associated to one energy service can also be used to raise the consumption of other goods and services that require energy as well. Lower heating costs might, for example, allow for an additional holiday trip. In addition, the possibility of more economy-wide effects stemming from a higher level of productivity and lower energy prices need to be considered.

All these effects can significantly influence the effectiveness of policy instruments promoting energy efficiency. According to the Khazzoom-Brookes postulate, it might even occur that improvements in energy efficiency entail an increase rather than a decline in energy consumption (cf. Saunders 1992). Even though this hypothesis has not been verified empirically, taking rebound effects into consideration still can have strong implications on the results of energy systems analyses.

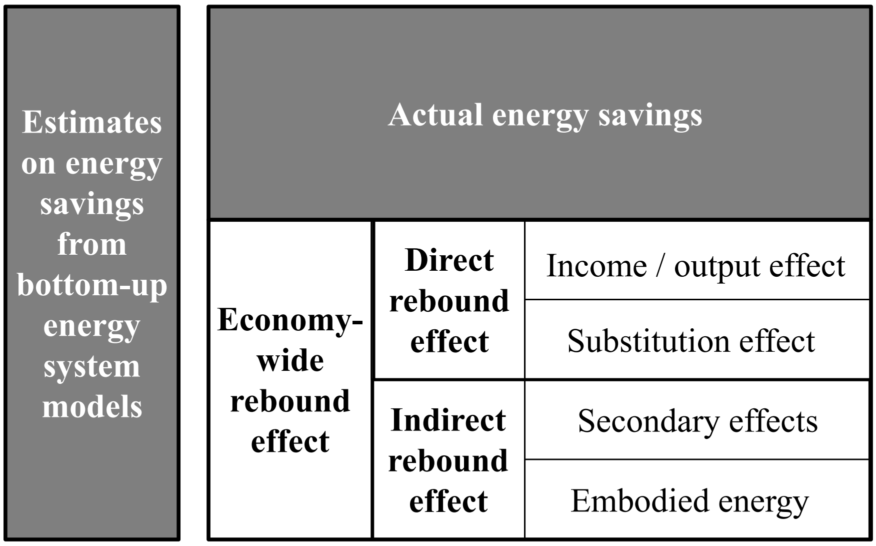

Figure 3-4: Classification of rebound effects from energy efficiency improvements and impacts on energy savings (own illustration based on Sorrell 2007, p. 4)

Increasing economic flexibility in bottom-up energy system models

In general, it can be stated that approaches for the inclusion of macroeconomic feedbacks in bottom-up energy system models have been much more widely explored than the integration of behavioural parameters.

A first attempt to enhance the economic flexibility of bottom-up energy system models can consist in introducing price-elastic demands (cf. Loulou et al. 2005). By assigning own-price elasticities to the different categories of demand for energy services and useful energy, the different economic agents can react more flexible to changes in energy prices or more stringent emission mitigation targets. This also implies a modification in the optimization approach: instead of minimizing net total energy system cost, as it is the case with fixed demands, now the net total surplus of producers and consumers is maximized. In addition, price elastic demands have been used to study the direct rebound effect in the framework of an engineering energy model (cf. Wang 2011).

In the hybrid model CIMS, a macroeconomic module is added where the different demand categories are modified according to the changes in energy prices (cf. Bataille et al. 2006). For the residential and commercial sector, elasticities of substitution to adjust home energy consumption versus other goods, consumption versus savings and goods versus leisure are applied. The demand for personal transportation depends on its own-price elasticity, while the the demand for freight transportation is connected to the value added of the industrial sector. The industrial output is adjusted with the help of Armington elasticities of substitution, describing the degree of substitutability between domestic and foreign goods.

On the whole, it has to be pointed out, however, that to account for all macroeconomic feedbacks in an energy systems analysis some sort of coupling of the bottom-up model with a top-down CGE model is required. Combining the two approaches is challenging due to the fact that they are based on different mathematical programming approaches – linear programming versus mixed complementarity programming. According to Böhringer and Rutherford (2005), three different coupling strategies can be distinguished. Firstly, attempts have been made to “soft-link” existing large-scale bottom-up and top-down models (cf. for example Hoffman and Jorgenson 1977; Hogan and Weyant 1982; Schäfer and Jacoby 2006). In this case, the different models are still run separately but important model drivers are exchanged between the two models in an iterative process. For example, the top-down model delivers data on GDP, industrial output, etc. to the bottom-up model, whereas data on energy prices, the rate of technological progress, etc. is taken from the bottom-up model and implemented in the top-down approach. This method is relatively easy to handle, but, as Böhringer and Rutherford (2005) point out, issues of inconsistency might arise due to different theoretical assumptions.

Secondly, some modellers have followed the approach of “hard-linking” an existing bottom-up with a reduced form representation of a top-down model. A notable example is the link of the energy system model MARKAL with the single sector general equilibrium model MACRO into a single, self-contained model (cf. Loulou et al. 2004). While this strategy ensures a high level of consistency, the representation of the economic system remains relatively superficial.

The third approach makes use of the possibility to specify market equilibrium models as mixed complementarily problems to create entirely integrated models containing both the detailed representation of technologies and all macroeconomic feedbacks (cf. Böhringer 1998; Böhringer and Rutherford 2005). The strong appeal of this methodology lies in its overall consistency and flexibility, but at the same time issues of dimensionality and algebraic complexity clearly reduce its practicability.

4. Conclusion

The growing concern about climate change has brought on a new focus in energy policy. A large variety of policy instruments has been implemented to deal with this unprecedented challenge – ranging from command-and-control policies to market-based measures and technology-oriented instruments. To evaluate those policy tools, environmental economic theory delivers a number of criteria, including cost efficiency, dynamic efficiency, ecological precision as well as political feasibility. Given the fact that usually multiple instruments are introduced, special attention needs to be put on the issue of policy interaction both within climate policy and with other policy areas. Moreover, any assessment of climate policy instruments needs to take into consideration that energy policy follows additional goals like energy security or affordability of energy services.

The changing focus of energy policy has also affected the role of quantitative energy modelling with a rising need for studies that evaluate the impact of different emission mitigation measures. This report has shown that bottom-up energy system models can make a significant contribution to this new task. The greatest strength of these engineering models can be found in their high level of technological detail allowing to assess the effects of technology-specific measures and technological breakthroughs in ambitious emission reduction scenarios. Furthermore, these comprehensive modelling tools provide the appropriate framework to analyse all interactions within the energy system.

Limitations of conventional bottom-up models have been identified with respect to a realistic representation of the decision-making behaviour of different economic agents and the inclusion of macroeconomic feedbacks. A rising number of hybrid model approaches, with the aim of combining the top-down and bottom-up perspectives in energy modelling, have been developed to address these challenges. However, these new techniques only offer an added value if the additional parameters are based on a good empirical foundation. Here, an interdisciplinary research approach with inputs from disciplines like behavioural economics, social psychology and other social sciences is required.

At the same time, it must not be forgotten that the goal cannot consist in creating one large model that can answer all questions related to policy evaluation. There is an important trade-off between flexibility as well as transparency on the one hand and complexity on the other hand. With respect to bottom-up energy system models, it seems to be essential that even if new aspects are added to the modelling approach, basic characteristics, like the linearity and the process-oriented representation, should be maintained, as these features constitute some of the main strengths of these models. In the end, a heuristic strategy with a diversity of methodological approaches will be needed for the assessment of climate policy instruments. In this context, special emphasis should be put on transparency, openly stating both the advantages and drawbacks of the respective modelling techniques.

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