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Kyoto, Europe?—An Economic Evaluation of the European Emission Trading Directive

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Abstract

To meet its obligations accepted in the Kyoto Protocol cost effectively, the European Union introduces a scheme of Greenhouse Gas Allowance Trading for its member states. This paper evaluates the cost effectiveness, ecological accuracy and dynamic incentives of this approach.

The EU-emissions trading constitutes an important shift in the paradigm of environmental policy, from command and control to a market based approach. Still, the EU-system does not fully realize the economic potential of the transferable discharge permit policy. Especially, the limited scope of trading regarding geography, pollutants, sectors and activities reduces the quality of the system. Moreover, the EU-Directive is unspecific in many respects and it leaves many decisions defining the rules of the game to the individual member state. Uncertainty and heterogeneity increase transaction cost and thereby hamper the effectiveness of the system.

Keywords: EU emissions trading, transferable discharge permits, European environmental policy, global warming

JEL Classification: Q54, Q58, K32

1. Introduction

In the 1997 conference on climate change in Kyoto, Japan, a group of industrialized countries agreed to curb its greenhouse gas (GHG) emissions following a certain timetable.[1] The gases covered by the agreement are carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), hydrofluorocarbons (HFCs), perfluorocarbons (PFCs) and sulphur hexafluoride (SF_6). To assure comparability, the quantities of GHGs other than CO_2 are calculated in CO_2 equivalents. The goal is an average reduction of 5.2% compared to 1990 emission levels. The commitment period runs from 2008–2012.

The individual reduction loads of the signatory states vary considerably. The member states of the European Union (EU) are part of the most ambiguous group of countries, vowing to reduce GHGs by 8%. To achieve the goals in a cost effective way the protocol allows three "flexible mechanisms," *joint implementation* (JI, Article 6), *clean development mechanism* (CDM, Article 12) and *emission trading* (ET, Article 17). Even though different from each other in detail, these mechanisms share the idea that rights to pollute may be transferred between different sources of pollution. In that, the mechanisms are pragmatic variants of the *transferable discharge permit* approach, an idea proposed in the seminal papers of Crocker (1965) and Dales (1968) building upon Coase (1960). Subsequently, environmental economic mainstream has pleaded to grant a more prominent role for transferable discharge permits in the portfolio of practical environmental policy. Indeed, much of environmental economics can be interpreted to be a "25-years struggle to influence the pollution control debate, ... maintaining a steady drumbeat of support for the more flexible incentive based approach," as Goodstein (1995), pp. 267/8, 283 observes.

The EU-member states are allowed to meet their 8% reduction obligation as a group, using the *bubble concept* which is defined in Article 4 of the Kyoto Protocol: Compliance is defined to be achieved if the *aggregate* emission reduction of the EU-member states is 8%, no matter what the individual contribution of each member state is. Using this leeway, the governments of the EU-member states agreed on a differentiated burden sharing system (2002/358/EC). There is a wide variety between the individual contributions agreed upon in this system. The most ambiguous obligations are for Luxembourg (a 28% reduction), Denmark and Germany (each with an obligation to abate 21%). On the other end of the spectrum are Greece and Portugal who are allowed to increase their GHG-emissions by 25 and 27%, respectively.

To date, the Kyoto Protocol is not in force yet. The USA have withdrawn from the process altogether and Russia hesitates to ratify. In order to be effective the Kyoto Protocol, first, has to be ratified by 55 countries and, second, the aggregate 1990 emissions of the countries having ratified the agreement must add up to a minimum of 55% of the CO_2-emissions of the countries listed in annex I of the 1992 United Nations Framework Convention on Climate Change (UNFCC-Rio de Janeiro). It is this second requirement which is not yet met. Nevertheless, and surprising to some, the European Union is setting sails to reduce GHG emissions, taking unilateral action in global climate change policy.[2]

To enable the member states to carry their emission reduction load "in a cost-effective and economically efficient manner" (Article 1), the European Union introduces an emissions trading system specified in Directive 2003/87/EC.[3]

Starting from January 2005 the operator of an installation undertaking activities listed in Annex I of the Directive must hold a sufficient number of permits covering the emissions of GHGs by this installation. Permits may be traded among operators within the EU. The initial endowment of emission permits for each operator is defined in a *national allocation plan* (NAP) each member state has to design and to enforce. National plans must be submitted to the European Commission until March 31, 2004 for approval (Article 9).

Economically, the European GHG-emission trading system is a pragmatic variant of the transferable discharge permit idea, just like the flexible mechanisms of the Kyoto Protocol.

There are three fundamental economic approaches according to which a policy, like the EU-emissions trading system, may be evaluated:

- *The standard welfare economic procedure*
  Using this approach a social norm is introduced and the properties of a state of resource allocation meeting the requirements of this norm are specified. Then, the equilibrium allocation which is contingent upon the constraints defined by the policy under review is characterized. In a third step the norm and the equilibrium are compared. Divergences between the two motivate a discussion on whether it is possible to modify the constraints defined by the policy in order to move the equilibrium closer to the state meeting the norm.

The standard welfare economic procedure comes in two forms:

- The most ambitious approach is to take the social welfare maximizing resource allocation to be the norm. Applied to environmental policy the welfare maximizing allocation is characterized by the difference between the social benefits and the social cost of pollution reduction being maximized. In the target situation externalities are fully internalized.

- In a more modest approach the target level of emissions reduction is not optimized but exogenously given. The norm is reduced to the goal of achieving this predetermined level at minimum cost.[4] Of course, the norm mentioned above implies the one mentioned here because cost effectiveness is a necessary condition for welfare maximization.

- *Comparative institutional analysis*
  Using this approach, a given policy is compared to alternative policy designs serving the same goal as the first policy. In the context of this paper the EU-emissions trading policy would be compared to taxes or command and control policies aiming at the reduction of greenhouse gas emissions.
- *Public choice analysis*
  Using this approach the design of a given policy is explained as the result of the struggle of different interest groups. In environmental policy, the polluting industries, environmentalists and the bureaucracy are important actors in the game.

The paper at hand predominantly applies the *standard welfare economic procedure* to evaluate the EU-emissions trading policy. It does so using this procedure in the second of the two afore mentioned versions.

The main question dealt with below is whether the EU-Directive is tailored close enough to its theoretical "role model" to justify optimistic expectations regarding its cost effectiveness and other criteria.

To answer this question we first elaborate the main evaluation criteria for environmental policy instruments provided by environmental economic theory. We also briefly show how an "ideal" transferable discharge permit system fares when evaluated according to these criteria. Then, in the main part of this paper we compare the EU-Directive to the ideal permit system. Particularly, we assess the consequences which the divergencies between the two systems might have for the success of the coming European GHG-emission trading system.

Of course, no real policy is identical to its "textbook version". We use the comparison to make some suggestions for possible amendments of the policy under review.

Moreover, to arrive at a more balanced exposition we include elements of comparative institutional analysis and of public choice analysis where appropriate. Also, at one point of the discussion, a policy relevant consideration of the standard welfare economic procedure using welfare maximization as a measuring rod is presented (see Section 3.2).

2. Transferable discharge permits and the environmental economist's measuring rod

The environmental economist's rod is a multidimensional tool. The evaluation criteria range from the compatibility of alternative instruments with a competitive market system to their political feasibility. In this paper we concentrate on three key criteria, the cost effectiveness, ecological accuracy and dynamic push of an instrument.

The *cost effectiveness* is the ability of an instrument to achieve a predetermined goal of environmental policy with minimal cost. In the most of the literature the goal is taken to be an aggregate emission standard. However, the definition can be applied to different goals like quality standards for environmental media. The costs are mostly taken to be emission abatement costs. However, it is possible to extend the analysis covering a wider understanding of cost. An important candidate for this broader view are transaction costs associated with the use of environmental policy instruments.[5]

The *dynamic push* of an instrument is its ability to induce environmentally friendly technical progress.

Ecological accuracy is the property of an instrument to achieve the predetermined goal of environmental policy with no time and resource consuming processes of trial and error.

In order to be able to apply these criteria in the evaluation of environmental policy instruments, they have to be operationalized. It is standard environmental economics wisdom that a predetermined aggregate level of emissions is achieved *cost effectively* in a situation where the marginal abatement costs of all contributing sources of the respective pollutant are equal to each other.[6] In the cost effective situation the relative contribution of each source to pollution abatement is the higher the lower the abatement cost of the respective source is, ceteris paribus. An ideal system of transferable discharge permits achieves just that: Given a certain equilibrium price in the permit market a cost minimizing operator of a source of pollution reduces emissions up to the point where the marginal abatement cost for this source is equal to the permit price. In a perfectly competitive permit market this price is identical for all emitters of the respective pollutant. It follows that in the permit market equilibrium marginal abatement costs are identical across sources. Thus, the equilibrium condition is identical to the cost effectiveness condition.

It is the key reservation of environmental economists against command and control policy that it does not achieve cost effectiveness, in general.

Better cost effectiveness is the most prominent reason of the EU-Commission for the choice of a trading system to meet the Kyoto targets. According to the Commission's own assessment the EU annual cost of climate policy with the intended trading system will be EUR 2.9 to 3.7 billions. Without trading the cost to achieve the same environmental goals would amount to EUR 6.8 billions.[7]

Regarding *ecological accuracy* economists check whether the object of regulation a certain environmental policy instrument chooses is closely connected to the goal of this policy. This requirement is met if the goal is a certain aggregate emission reduction and the instrument grants a certain quantity of emission rights to each polluter. Then, ecological accuracy requires individual emission rights to add up to the aggregate quantity which is compatible with this policy goal. This requirement can be met by design of the regulation. Ecological accuracy is much more difficult to achieve if the object of the regulation is only indirectly connected to the policy goal. This is so if effluent charges are used. These charges specify a "price for pollution". Each individual polluter is free to adjust to this price choosing the equilibrium quantity of pollution reduction. It is realistic to assume that the environmental policy agency does not know the individual marginal abatement cost functions of the polluters. Then, the agency does not know how polluters will react to a given tax rate in terms of emission reduction quantities either. Therefore it is much likelier that a predetermined aggregate emission standard will be achieved swiftly and accurately using transferable discharge permits compared to effluent charges.[8] According to Portney (2003), p. 18, higher ecological accuracy is the main reason why transferable discharge permit variants have flourished in the USA, compared to pollution taxes. Since the target emission level is achieved in a process of trial and error when emission taxes are used,[9] their ecological accuracy is lower than the one of TDPs.

To operationalize the criterion of *dynamic push* environmental economics uses the idea that a polluting firm, as a *homo oeconomicus*, introduces environmentally friendly technical progress to the extent to which this is compatible with the firm's profit maximisation goal. The decision to introduce a new technology is a matter of cost benefit analysis for the decision maker. It is crucial that this cost benefit analysis is contingent upon which environmental policy instrument is applied. Economic analysis of the dynamic push of transferable discharge permits may be summarized as follows[10]:

Permits produce a twofold incentive for polluters to apply new methods of pollution control. Whenever the new method achieves the firm's given ("pre-innovation") abatement level cheaper than the old one the firm benefits from introducing the new method by saving abatement cost. Moreover, when the new method abates more than the old equilibrium abatement level at a marginal cost lower than the permit price the firm saves permit expenditures (net of the added abatement cost) by pushing abatement beyond the old equilibrium level. The firm will balance these two gains from innovation against the cost of innovation.

Opposed to that, it has been shown that command and control fails to fully activate incentives to improve environmental technology applied by selfish decision makers. If command and control policy takes the form of an absolute emission limit, there is an incentive to develop and apply technologies achieving this limit cheaper, but not to go beyond this limit. On the other hand, if command and control takes the form of technology standards, even this selective incentive is attenuated. Command and control regulations have often been observed having "effectively 'frozen' pollution control techniques in place"[11].

All in all transferable discharge permits achieve favourable grading when evaluated according to the three criteria selected here. Cost effectiveness, dynamic push and ecological accuracy are perfect in an ideal system.

Below, we make an assessment of to what extent the EU-emissions trading system may live up to the great expectations which are possibly generated by the properties of the text book transferable discharge permit system sketched above.

3. An economic assessment of the EU-directive

3.1. Ecological accuracy

Obviously, the goal of international climate policy is to curb the increase of global temperature. In the light of this goal EU-emissions trading is certainly not accurate,—and neither is any other environmental policy system. This is so because the development of world climate is not exclusively anthropogenically determined. Moreover, there are many scientific uncertainties regarding the impact of economic activity on climate.

Nevertheless, there is some consensus among scientists that global climate is negatively affected by the emission of GHGs. Therefore, since the true goal of global policy, stabilizing the world climate, is not operational, a second best approach would be to use the reduction of world wide GHG-emissions as a proxy for this goal. However, when evaluated in the light of this "substitute goal " EU-emission trading cannot be expected to be accurate either. This is so because the goal is related to the world wide emissions of GHGs and the EU only accounts for a part of it. Moreover, EU-emissions trading (to date) does not cover all relevant GHGs but concentrates on CO_2.[12]

Even if we go one step further narrowing down the goal according to which ecological accuracy is assessed to the reduction of the CO_2 emissions of the EU the result is negative. EU-emissions trading is not accurate in this narrow setting since it only covers economic activities listed in Appendix I of the directive in the energy and industrial sector (Article 2). Important GHG producers like transport and the aluminium industry are not part of the system.[13]

Thus, the very limited scope of the EU-directive regarding geography, pollutants, sectors and economic activities hampers the ecological accuracy of the system. Moreover, there is another point of criticism regarding this criterion.

Not every industrialized nation accepted to comply with the Kyoto Protocol and there are no obligations for developing countries to reduce emissions anyway. Therefore, even if the Kyoto Protocol will become effective only a part of the international community will reduce GHG emissions. On the first glance this might be welcomed to be a partial success. However, this success is reduced by the fact that GHG emission reductions in one part of the world may be the cause for GHG increases in other parts. This is so because climate change policy in the countries complying to the Kyoto Protocol induces changes in the economic structure to reduce the use of fossil fuels. In these countries there is a huge variety of programs under way to curb energy consumption and to substitute fossil by renewable energy. By these praiseworthy activities, demand for fossil energy is reduced in the world markets. Reducing demand, equilibrium prices go down ceteris paribus. Given decreasing prices the demanded quantities will increase in the countries not complying with the Kyoto Protocol. Increasing use of fossil fuels will lead to an increase of GHG emissions there. This kind of a *leakage effect* further attenuates the ability of the EU-directive to achieve ecological accuracy.[14] However, it must be emphasized that this problem is not generated by the EU-Directive itself but by the partial character of the Kyoto-Protocol, the international treaty the EU-Directive is building upon. Any unilateral climate policy of the EU, using TDPs or other instruments, would suffer from leakage. Nevertheless, this general problem has to be mentioned in the context of the ecological accuracy criterion. Moreover, in the light of the expectations which are directed towards alternative policy instruments, any TDP variant is particularly vulnerable regarding deficits in ecological accuracy. This is so because the textbook analysis of TDPs suggests that ecological accuracy is achieved. Opposed to that, policy instruments like effluent charges are not expected to be ecologically accurate anyway. For these, leakage is just an other reason why the are not.

3.2. Cost effectiveness

Worldwide GHG emissions are generated by many different activities. Billions of decision makers emit different *pollutants* by operating different *installations* as members of different economic *sectors* from different *locations*. The goal to reduce worldwide GHG emissions by a predetermined amount is reached cost effectively if all these polluting sources contribute to the goal to an extent that minimizes worldwide aggregate abatement costs. According to what has been said above the cost minimizing situation is characterized by the marginal abatement cost of all polluting sources being equal to each other. In order to achieve this unconstrained cost minimum in a transferable discharge permit system, generators of GHG emissions must be able to demand and supply permits irrespective of what kind of a greenhouse gas they emit, what kind of installation they operate, in which part of the world they are located and what sector they are a member of. A necessary property of an efficient transferable discharge permit system is that it is *comprehensive* in the sense that there are no constraints regarding the transferability of rights. Any attenuation on the transferability of pollution rights leads to the result that the equilibrium allocation of pollution reduction activities only qualifies for a *constrained cost minimum*. The difference between the aggregate abatement cost associated with a constrained cost minimum and the cost associated with the unconstrained one is the bigger the more binding constraints are introduced into the system.

It is obvious that the EU-Directive does not meet the requirements of an unconstrained permit system:

- According to this directive the framework for emissions trading is narrowed down to CO_2.[15] In addition to the detrimental effect this has on ecological accuracy, as mentioned above, it also hampers the cost effectiveness of the system. It is theoretically suggestive that aggregate abatement costs will be significantly reduced by integrating non CO_2 greenhouse gases. This is so, because these gases have a larger global warming potential than CO_2.[16] Moreover, most of them are less costly to abate. This *prima facie* assessment is confirmed by empirical analyses. Using a global trade and environment model (GTEM) Tulpulé et al. (1999) and Jakeman et al. (2001) assess the equilibrium marginal abatement cost in a certain emissions trading scenario using CO_2 only to be 36 USD (2000)/ton CO_2 versus 26 USD (2000)/ton CO_2-equivalent in a scheme which allows for additional gases to be traded. In a model evaluating the regional and global effects of greenhouse gas reduction policies (MERGE) used by Manne and Richels (1999, 2000, 2001) allowing for additional greenhouse gases to be traded makes equilibrium marginal abatement costs drop from 74 USD (2000)/ton CO_2 to 38 USD (2000)/ton CO_2- equivalent.[17]

The reason of the EU-Commission for not introducing non CO_2-GHGs into the system are difficulties to measure these substances. Even though measurement or reliable calculation is a prerequisite for ecological integrity of any emissions trading system the reason given may be disputed on the following grounds:

- *Even if* measurement problems are solved the directive does not make non CO_2-GHG trading an integral part of the EU system. Instead, the decision is left to the individual member state.

  In Article 27 the Commission restricts "opt out"-possibilities to the first trading period, for the sake of the system's comprehensiveness. However, the procedure regarding non CO_2-GHGs is a *de facto* opt out which is not mentioned in the article dealing with opt out and which is not subject to a time limit.
- The claim that these gases cannot be measured (or reliably calculated) is problematic for the following reasons: First, the EU-Commission's policy towards measurement of non CO_2-GHGs is not consistent. On the one hand it argues that these gases cannot be measured, on the other hand it requires its member states to report the emissions of non CO_2-GHGs.[18] Second, it is ignored that in the UK Emission Trading System, all GHGs are covered since 2002.[19] Third, the Norwegian Quota Commission argues that measurement is not only a question of the type of GHG but also of the source of emission.[20] According to this expertise, to date, four GHGs can be measured at reasonable cost as they are emitted from industrial sources.[21] The Commission recommends to include these in a GHG-emissions trading system. Doing so would make this system quite comprehensive, covering about 80% of the country's GHG-emissions (1990).

- The EU-Directive is on emissions trading among EU-member states. It is not completely clear yet to what extent it will be possible to integrate the flexible mechanisms of the Kyoto Protocol (CDM, JI and ET), as mentioned in the introduction, above, into the EU-system. The point of view taken by the Commission of the European Communities (2003b) is somewhat ambiguous.

  On the one hand it emphasizes that "the Community scheme creates an EU-wide market where allowances can be traded *without restriction*" (p. 7, emphasis added). Therefore, the member states are not entitled to decide to what extent they allow emission rights generated by applying the Kyoto mechanisms to be used in the EU trading system. Neither are they allowed to confine this use to certain *kinds* of rights generated by these mechanisms.

  On the other hand the commission emphasizes that the Marrakesh Accords require "that the use of the mechanisms shall be supplemental to domestic action"[22]. In the light of this provision the Commission of the European Community proclaims that it will induce a review process in case it turns out that more than 6% of the total quantity of allowances allocated by the member state are generated by using the flexible Kyoto mechanisms. In this case "the Commission *may* consider" to introduce a quantitative constraint of a maximal level of 8%.[23] These documents seem to breathe the air of political compromise. In their ambiguity they are detrimental to the success of emissions trading because the involved decision makers need clear information about the framework within which they are supposed to operate.[24]

Still it seems to be save to say that there will be some cap on the possibility to integrate the flexible Kyoto mechanisms into the EU-emissions trading system.
Moreover, the EU-Commission does not object to NAPs of member states taking a very detached position towards the integration of Kyoto mechanisms into the trading system. An example is the NAP of the UK stating that "it is not intended that any use be made of the flexible mechanisms by the UK government to meet the burden sharing agreement target" (Section, 1.32 on p. 15).[25] In the NAP of Finland the consequences of lacking specification, repeatedly picked upon in this paper, are obvious: ". . . there is not enough certainty as to . . . what extent Finland will use the so-called flexible mechanisms of the Kyoto Protocol . . . " (Section 6.2). This assessment has not been challenged by the EU-Commission.
It follows from what has been said above that this kind of a constraint prevents a permit market from operating efficiently. It cannot be ruled out that greenhouse gases may be reduced less costly using flexible mechanisms than by transferring emissions rights within the EU. This is particularly suggestive for the Clean Development Mechanism which allows to generate pollution rights by reducing emissions in developing countries under certain conditions. Marginal abatement cost in these countries are often much lower than they are in the industrialized world. This *prima facie* impression is supported by empirical analysis. In an empirical study by Hillebrand et al. (2002), pp 71–79, the marginal abatement cost in the equilibrium of a "narrow" permit system is estimated to be 12.70 USD per ton CO_2 by the year 2012. In a wider system allowing JI and CDM projects to be integrated the 2012 equilibrium marginal abatement costs are estimated to be 2.60 USD per ton CO_2.
It must be emphasized that the unrestricted use of (environmentally sound) flexible Kyoto mechanisms is a must for any cost effective climate policy. It is not a *specific* requirement for a permit trading system. But the use of flexible mechanisms is closer connected to the permit system than it would be to any system using alternative instruments of climate policy: Given a policy using taxes (or command and control), the tax rates (or the performance standards, respectively) would not automatically vary with a change in the policies to use flexible Kyoto mechanisms. However, in a TDP system, the equilibrium permit price would change. Therefore a TDP system, like the EU trading scheme, is more susceptible to distortions caused by uncertainties about the use of the Kyoto mechanisms than alternative forms of climate policy.

- The criticism raised above regarding the constraints on tradability with respect to only one of several greenhouse gases and with respect to the integration of the flexible Kyoto mechanisms analogously holds for the fact that emissions trading is constrained to the activities and installations listed in Annex I of the directive. Thereby, emissions trading is limited to about 98% of the CO_2 emissions from energy production and 60% of the CO_2 emissions from industry[26] (and no emissions from transport at all). Given that, the equalizing of marginal abatement costs is attained only for a subset of GHG emitting firms. Marginal abatement cost between two decision makers of which one is part of the trading sector and the other one is not, are not equalized within this system. Therefore the goal of overall cost minimization is missed. This problem does not vanish if the non trading sector is regulated by a different instrument of climate change, e.g. some *eco tax*. In the equilibrium of the tax regulated sectors marginal abatement costs are equal to the tax rate. Therefore, cost minimization between activities regulated by these two different instruments is only attained in the special (and very unlikely) case of the tax rate being equal to the equilibrium permit price.

The problems of the cost divergencies between the trade and the non-trade sector are aggrevated by the following feature of the system:
Each individual member state is free to decide on how to allocate the burden it accepted within the EU-burden sharing agreement between its own non-trading and its own trading sector. The problem of diverging marginal abatement cost would be pragmatically minimized if the member states would strive to allocate the reduction loads between the two sectors according to abatement cost criteria, in their respective National Allocation Plans. However, there is no guarantee for this. To the contrary, there is empirical evidence that in the case of Germany the distribution of reduction loads between the two sectors has been the result of a competition among interested industry groups for lower reduction loads.[27] Initially, according to the plan of the German Federal Ministry of the Environment the trading sector was scheduled to receive emission rights of 488 million tons of CO_2. This proposal was designed without any reference to abatement cost comparisons between the trade and the non-trade sectors. Instead, it used the figures of a voluntary commitment of German industry to reduce its greenhouse gas emissions as a base line.[28] As a result of heavy pressure from the German energy sector the endowment of the trading sector was increased to 503 million tons in the final German national allocation plan. Of course, the reduction level of the German non-trading sector has to be increased accordingly in order not to jeopardise the German commitment within the EU-burden sharing.[29]

- According to Article 24 (1), "(f)rom 2008, Member States *may* apply emission allowance trading . . . to activities, installations and greenhouse gases which are not listed in Annex I . . . ". Moreover, "(f)rom 2005 Member States *may* . . . apply emissions allowance trading to installations carrying out activities listed in Annex I below the capacity limits refered to in this Annex."[30]

  Opening up the possibility to expand the trading sector is certainly a step into the direction of improving the cost effectiveness of the system. It is therefore to be welcomed.[31] The beneficial effect would be augmented if the scope of the trading sector defined by the EU-Commission would be generally widened instead of leaving the decision on extensions to the individual member state: Given the decisions of the individual member states will turn out to be different from each other the rules of transferability of emission rights will be vary within the EU. Accordingly, marginal abatement costs will vary in the equilibria of national permit markets and EU-wide cost minimization will be missed.[32]

  Moreover, by the opening clause quoted above, the decision on the final design of the trading system is considerably delayed. This creates high uncertainty for the involved firms regarding the properties of the framework within which they are supposed to operate. It cannot be expected that firms commit themselves by making costly GHG saving investments when they are told that they will learn about the game they are going to play in a few years. This uncertainty, detrimental to the success of the program, is intensified by Article 30 on "Review and Further Development". In this article the Commission "threatens" to make a proposal to the European Parliament and Council by December 31, 2004 to change the rules of the game regarding many important issues. Among these issues are

  - the integration of additional sectors and activities,

- the relationship between Community emission allowance trading and the international emissions trading which is supposed to start in 2008 according to the Kyoto Protocol
- the use of credits from projects mechanisms of this protocol,[33]
- the degree of harmonization between the criteria according to which individual states allocate emission rights,
- the relationship between emissions trading and other policies implemented at member states and between community level,
- the level of penalties,
- other issues.

Above, we have criticised the EU-directive on the grounds that it is not *comprehensive*.[34] On the other hand, it must be acknowledged that the EU scheme is much more comprehensive than a system in which each member state would "autistically" strive to meet its own obligations as specified in the EU burden sharing agreement. According to a recent study by Hidalgo et al. (2004), the cost of meeting the EU Kyoto target is reduced by 50% using the EU emissions trading scheme compared to using uncoordinated national schemes.

In addition to the principle of comprehensiveness, the principle of *smoothness* has to be observed.[35] Frictions can take on many forms. Especially, the coherence of environmental policy is damaged if market based instruments, like the emission trading system, are conflicting with other regulations. This is particularly so if command and control policy reduces the transferability of emission rights in a permit market system.[36] On first glance the EU system does not seem to be threatened by these kinds of problems since there is (to date) no end of pipe technology to reduce CO_2 emissions. Therefore, no technology based standards related to that kind of a technology exist. Obviously, this is different in case the system is expanded to other greenhouse gases. Moreover, CO_2 is often jointly emitted with other pollutants and for the latter command and control regulations might exist. Also, it is not only technological standards which might hamper the trading system. It is also possible that Clean Energy Regulations requiring firms to meet a certain percentage of their energy demands by using energy from regenerative sources put a constraint on the transferability of CO_2 emissions rights.[37]

In addition to comprehensiveness and smoothness of the policy, a third concern has to be raised in the context of cost effectiveness, *the initial allocation* of emission rights.

In the environmental economics literature, two methods to distribute emissions rights among the polluters at the beginning of a TDP policy have been discussed: Auctions and free distribution. According to Article 10 of the EU-Directive, in the first trading phase (2005–2007) at least 95% of emission rights must be distributed free of charge. In the second phase (2008–2012) the rate is decreased to 90%. Criticism against free distribution of rights on cost effectiveness grounds is threefold:

- In a Coasean world the initial distribution of rights is allocatively irrelevant. There is an unique cost effective equilibrium of emission reductions by various sources of pollution. This is achieved in the permit market, no matter whether permits are auctioned of or distributed free of charge in the initial situation.[38]

However, things change in a world with transaction costs. It has been shown by Stavins (1995) that the cost effectiveness of the auctioning system is superior to free distribution if the notion of cost does not only include abatement cost (as in the main part of this paper) but is extended to include transaction cost.[39]

- In the paper at hand we use cost effectiveness as a measuring rod for the equilibrium which is produced under the constraints defined by environmental policy. (This is the "modest" version of the standard welfare economic procedure, mentioned in Section 1., above.) If we open the perspective to use social welfare maximization as the measuring rod (the "ambitious" version), additional light is shed on allocative divergencies between actions and free distribution. In this alternative setting, the benefits and cost of the environmental policy have to be compared and the quest is for a program design which maximizes net benefits. In a realistic setting, which is characterized by the acknowledgement of the existence of distortionary taxes in the economy, auctions fare better than free distribution. The reason for "a strong case on efficiency grounds for using carbon taxes or auctioned permits over grandfathered carbon permits" (Parry, 2003:p. 385) is that the revenue of auctions can be used to reduce distortionary taxes, creating the potential of a "double dividend".[40] In their study on the US electricity market, Burtraw et al. (2001) assess the societal cost of the auction to be about one half of the costs of two alternative forms of free distribution considered (grandfathering and generation performance standards). This result is generated even though the assumptions made on how the auction revenues are used do not exploit the full potential of a double dividend.[41]
- In the two points raised above, we did not refer to the specific manner in which emission rights are distributed free of charge. The procedure of free distribution which is currently followed in the EU-member states may be called "consecutive grandfathering": The emission rights in the first trading phase are calculated using historic emissions of output data of the polluting firms. The rights in the second phase will (as far as it is known to date) be allocated on the basis of emissions or output data at the end of the first phase.

  If the amount of emission rights received free of charge by a polluter at the beginning of a certain trading phase is conditioned on emissions or output (used as a proxy for emissions) data in the previous period the incentives for cost effectiveness of the system are distorted: Different from a situation under auctions, the benefit of an additional (marginal) emission reduction is not equal to the price of a permit (net of marginal abatement cost). This benefit of reduction for the reducing firm is diminished by the value of the permits the polluter does *not* receive at the beginning of the following trading period due to the additional reduction in the first period. It is known from the literature that allocation procedures of the kind used in the EU system create distortions like an output subsidy. [42]

3.3. Dynamic push

Regarding the incentives generated by the EU emissions trading system to introduce environmentally friendly technical progress the criticism elaborated for the efficiency criterion above, analogously holds.

Given that the playing field for emissions trading is constrained to certain gases, installations, activities, sectors and regions, it cannot be expected that innovations are induced beyond these limits. This is particularly regrettable because the market mechanism is not only a means to realize static efficiency but also a mechanism to generate information. It is not known where, in terms of the gases, activities and the other dimensions mentioned above, there is the biggest potential for technical progress. Therefore, the incentive generated by environmental policy for individual decision makers to search for innovations must be *broad* in the sense that it covers all of these dimensions. However, a permit system surrounded by multidimensional constraints as the EU system, can only provide *narrow* incentives to discover new technical solutions. Therefore, the system will display the dynamic push generally expected from market forces only to a very limited extent. The dynamic incentive will be much more like the effect of targeted governmental subsidy programs designed to achieve prespecified forms of technological change.

Regarding the procedure used to initially allocate permits, the picture regarding dynamic push is analogous to the one described above regarding cost effectiveness: Auctions are better at inducing technical change than grandfathering.[43]

3.4. Climate policy as stock management

Above, we have implicitly dealt with greenhouse gas emissions as *flow* problems. Climate policy has been imagined to strive for the reduction of periodical (say: annual) GHG emissions. This is convenient as a first step. According to present scientific knowledge, however, changes in the average world wide temperature seem to be much closer related to the *stock* of greenhouse gases in the atmosphere than to the flow of emissions. Therefore, issues of accumulation that have been ignored above must be dealt with.

The first consequence is that the criterion of *ecological accuracy*, unlike with most other pollutants, is not to be applied to annual emissions in the context of greenhouse gases. Hence, if some goal of emission reduction is missed in one period of time, in principle, it is possible to make amends in the next period. Given it is the stock that matters, underachievement and overachievement in individual periods may be tolerated as long as they even out over time.[44] However, this qualification of the criterion of ecological accuracy in the context of climate policy does not soften the criticism regarding the ecological accuracy of the EU system raised above. According to what has been said above the EU system will fall short of the goals of climate stabilization *in any period* of time. Therefore, it is as ineffective in terms of GHG stocks as it is in terms of GHG flows.

In terms of the *efficiency criterion* dealing with stocks instead of flows opens up a new dimension to reduce abatement costs. Not only can abatement costs be reduced by shifting emission reductions from one source to another at each point of time but also by shifting abatement activities among different periods.[45] Particularly, if we expect marginal abatement cost in a certain period to be lower than in another period it is efficient to shift emission rights from the former to the latter period.

In the EU system intertemporal transferability of emission rights is a one way street. Discharge permit which are not used in the present may be transferred to the future (*emission banking*).[46] On the other hand there is no provision to allow for future emissions rights to be used in the present (*emission borrowing*). If we expect abatement technology to be superior in the future compared to today's technology, then abatement costs will decrease over time. Therefore, borrowing would be an instrument to improve the efficiency of the system.[47]

The plead to allow borrowing is ambitious for two reasons:

- There is no emissions trading program which allows borrowing and thereby might be used as model.[48]
- Even though the EU-Directive does not use the term "borrowing" there is an element of this in Article 16 of the Directive: A firm polluting in excess of its emission rights must buy these rights in the following period and is subject to a sanction.[49] Effectively, under these circumstances, a firm polluting in excess of its permit endowment in the present uses its future endowment and pays a fee for this intertemporal transaction. So this might be interpreted as "*de facto* borrowing". However, it is not only the monetary penalty/fee which distinguishes borrowing which would be a possibility explicitly allowed in an emission trading program from the rules of Article 16 of the EU-Directive. The reason for this assessment is that a firm using Article 16 is in non-compliance.[50] It is therefore subject to the "naming and shaming"[51]—strategy with which the EU-Commission intends to further firms' compliance with the directive. If this strategy is effective and firms indeed suffer from image damages, the cost of a firm to use "borrowing according to Article 16" is much higher than the fee.[52] It is plausible that the transaction costs of "borrowing according to Article 16" will considerably reduce the extent to which borrowing will be used compared to the equilibrium amount of borrowing in a system where it is an integral and accepted part.

  Within the framework of the Kyoto Protocol integrating borrowing into the EU system would require an amendment of the Marrakesh accords. The Marrakesh accords accept banking and even borrowing but the latter only at a penalty. A full integration of borrowing into the EU-system does not seem to be feasible without conflicting with these provisions. The situation is complicated by the fact that, to date, it is unclear whether the Kyoto Protocol will become effective. The EU has committed to introduce the emissions trading scheme from 2005 irrespective of the status of the Kyoto Protocol at that time. If Kyoto fails it is easier to integrate borrowing into the EU system.

3.5. Compliance

In the main part of this paper we have analyzed the economic incentives created by the EU-emissions trading directive under the tacit assumption that compliance will be perfect. Of course, this assumption is heroic.

Compliance issues are complicated warranting a paper of their own. Moreover, to date, no comprehensive empirical assessment is possible, since trading will not start before 2005. Therefore, we will confine the exposition to a few remarks;

Regarding EU-policy compliance must be analysed at two stages, first, compliance of individual member states to the EU-Directive, second, compliance of polluters to the national trading policy as enacted by the individual member state in which the polluter is located.

- On the first level of compliance certain pre-policy problems have been visible. At the time while this paper is revised (July 2004), two of the “established” member states of the EU and five of its new members have missed their deadlines for submission of their national allocation plans.[53] The Commission of the EU has started infringement procedures against the two EU 15 states and announced the same step against the new members in case of continuing non-compliance. In addition the Commision is sending “final written warnings“ to eleven of the EU 15 states for not fully transposing the Directive into national law.[54] These problems may be understood to indicate diverging national compliance attitudes between member states regarding climate policy.

  In the history of the EU conflicts between the EU-Commission and individual member states regarding compliance did not always result in full compliance. If resistance of the individual state(s) is strong, often a political compromise is reached. A recent example is the case of Germany and France not complying with the Maastricht Agreements on stability within the European Monetary Union.

  Even if we assume that the problems indicated above are transitional, compliance at the level of individual member states may constitute a point of concern. This is so because it may be misleading to take a situation where compliance is complete to be ideal: the approach of the EU-Directive, leaving many important decisions to the individual member state will certainly increase compliance of these states compared to a more general approach. It is in this context, where the possibly diverging national compliance attitudes of different member states may lead to a very differentiated system. As has been argued in this paper having different trading rules and other properties of the policy in different member states may be a method to achieve compliance which is costly: During the time the individual states need to define their rules uncertainty prevails. After the rules have been established divergencies constitute attenuations in the ability of the system to equalize marginal abatement costs. Moreover, transaction costs increase since experiences gathered by executing trades within an individual member state cannot necessarily be transferred to an intended trade between firms which are located in different states.
- Regarding compliance of firms to national rules, monitoring and sanctions have been identified to be the key issues.[55] In the EU trading system a high degree of transparency is achieved by making the national allocation plans accessible to the general public and by a elaborated system of reporting and documentation of emissions and trades. Monitoring of emissions is also achieved at a satisfactory level since emissions have to be measured or inferred by an established calculation procedure using activity data, emission factors and oxidation factors.[56]

  A point of criticism is the penalty for excess pollution. The decision of a polluter to generate emissions in excess to its permit endowment depends on the probability of “being caught,” the size of the penalty and on the permit expenditure saved by not buying permits. (In addition, image concerns of the firm play a possible role, as mentioned above.) In an earlier version of the directive the nature of this “cost-benefit- analysis of non compliance” had been taken into account saying that the penalty would be Euro 50 per ton of CO_2 in the first trading period and Euro 100 afterwards or twice the average market price, which ever is higher.[57] In the final Directive, not only has the penalty in the first phase been decreased by Euro 10 but the connexion to the permit market price has been deleted from the directive altogether.

Moreover, the possibility that a firm may pollute in excess of its permit endowment is not the only issue in firm non-compliance. Another form of non-compliance is not to meet reporting obligations. Thereby, a polluter might try to reduce the probability that excess pollution is detected. The EU-Directive does not contain specifications of the penalty to be applied in this case. It is likely that different member states will impose different penalties. Thereby, incentives to comply will vary within the EU.

4. Conclusions

Cost effectiveness is a must in environmental policy. This is so on pure economic grounds because the burden of environmental policy on industry and consumers is to be minimized. Moreover, cost effectiveness is an ecological requirement because more ambitious environmental policy goals are politically feasible if the cost to attain these targets are reduced.

With the GHG emissions trading directive the EU Commission attempts to design a framework within which the EU obligations of the Kyoto Protocol can be met with minimal cost. The design of the EU emissions trading system is a remarkable shift of paradigm in environmental policy. This policy has been predominantly relying on a command and control approach using technology and other standards to achieve ecological goals. The EU directive now introduces a significant element of a market based instrument, a move that has traditionally been advocated by environmental economists. Choosing climate change as the area of application, issues regarding the dispersion characteristics of pollutants (including the *hot spot*-problem) that have complicated the design of permit markets for many pollutants do not play any role here: GHGs are pure public bad.

Unfortunately, the EU system will not be able to meet the high expectations that may have been generated by the traditional environmental economics evaluation of "textbook" transferable discharge permit systems. Opposed to these ideal systems transferability of emission rights is constrained in many ways, following the design of the EU directive. Of course, no practical system can be the twin of the original idea. Still, the cost effectiveness , ecological accuracy and dynamic push of the coming EU system would be higher if it had been tailored closer to its theoretical model.

To summarize the proposed amendments:

- The EU intends to fulfil its Kyoto commitments using a hybrid system: The economy of each member state is divided into a trading and a non-trading sector. Currently, each member state is free how to divide the emission reduction load defined in the EU burden sharing arrangement between the two sectors in its National Allocation Plan.

  To improve cost effectiveness across these two sectors the EU should issue a guideline requiring its member states to (even if pragmatically) pursue the goal of cost minimization in the decision on how to distribute reduction burdens between trade and non-trade sector.
- It has been shown in some detail, above, that the current system leaves many important decisions on how to proceed within the trading sector to the individual member state. The EU should strive for greater homogeneity of the system. In addition to improving the chances

to equalize marginal abatement costs of sources located in different member states, harmonization would also contribute to reduce the transaction costs for transnational trades.[58]

- The key to effectiveness of any market based instrument is its ability to exploit divergencies in the marginal abatement costs of different polluting sources.

  Therefore, the EU should critically review its directive identifying all the instances where the transferability of emission rights is restricted. This holds for the transferability across GHGs, installations, locations, trading subsectors and along the time axis. An amendment should strive to eliminate these constraints as far as possible.
- In the process of distributing emission rights at the beginning of a trading period auctions should play a much more significant role than allowed in the EU-Directive. Of course, this request might sound naïve since free initial distribution is a result of interested parties exercising pressure in the process of political decision making. Still, the acceptability of a switch from free distribution to auctions may be greater than it seems to be at first glance. The reason for this assessment is that the welfare gains from this switch are probably very high making the policy maker able to (more than) compensate the losers of this switch. According to a recent study by Burtraw et al. (2002), it only takes 7.5 per cent of the revenue raised in an auction to compensate US electricity generating industry for the losses due to auctions compared to free distribution.

  It is a very interesting research question whether these results can be transferred to EU climate policy.
- Each policy is subject to the possibility of revision. Beyond this general fact, the EU Commission has written a very detailed "threat" to change most of the rules constituting the game into Article 30 of the Directive. This is a signal of instability which does not create a sound basis for the firms to make investment decisions. The EU commission should reduce uncertainty (and thereby transaction cost) signalling more trust in the persistence of its own policy.

The issues raised in this paper do not deny the value of trading (and other market based instruments in environmental policy) compared to command and control. To the contrary, it is acknowledged that the theoretical case for the superiority of market based instruments to command and control, often made in the literature and briefly reviewed in section II, above, has been confirmed by numerous empirical studies. Most recently, *Resources for the Future* has conducted a major study comparing economic incentive policies to command and control as analysed in twelve case studies referring to six environmental problems.[59] The result is that cost effectiveness of market based instruments is dramatically higher than of command and control. E.g., estimates regarding US SO_2-policy show that the policy goal has been achieved by tradable emission permits at about one quarter of the cost of various command and control policies. Market based instruments are also superior to command and control in terms of their dynamic push. The theoretical hypothesis sketched in section II, above, that "in a CAC regime only cost-reducing innovations are encouraged, while under EI both cost-reducing and emissions-reducing innovations are encouraged"[60] is empirically confirmed.

Still, the degree of superiority of market based instruments depends upon the factual design of these policies in each individual case under review.[61]

The rationale for the better cost effectiveness of market based instruments is their ability to exploit abatement cost heterogeneities among different sources of pollution. It is this key issue (and some other points of concern) with respect to which this paper argues that EU emissions trading should be amended in order to better enable the system to actually realize its theoretical potential.

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Notes

1. Below, the contents of the Kyoto Protocol are summarized very briefly. For an extensive exposition see Oberthür/Ott (1999). The *law and economics* of the convention are analysed in Faure/Gupta/Nentjes (2003).
2. A possible motivation is to set a good example, inducing others to follow. On the other hand, unilateral action might constitute an "invitation" to be exploited. This critical view is prevalent in the game theoretical literature since Hoel (1991). See Finus (2001) for an updated discussion.
3. See the Commission of the European Communities (2003a, 2003b). In the paper at hand we describe the contents of the system only very briefly and concentrate on the economic assessment.
4. The classical reference for this approach is Baumol and Oates (1971).
5. See Eckermann et al. (2003) and Michaelowa and Stronzik (2002).
6. See, e.g., Barde (2000), Endres (2000), Faure and Skogh (2003) and Hanley, Shogren and White (2001).
7. See document MEMO/04/44 on the web page of the European Commission.
8. In this line of reasoning it is assumed that compliance to the rules of environmental policy is perfect. This (unrealistic) assumption does not distort the comparison between alternative environmental policy instruments unless there are reasons to presume that alternative instruments create different incentives for non compliance.
9. Trial and error analysis in the economics of environmental policy goes back to the seminal paper of Baumol and Oates (1971).
10. See the literature given in fn. 6.
11. Portney (2003), p. 16.
12. From 2005–2008 the concentration on CO_2 is binding. From 2008 member states *may* include other GHGs into the trading system (Article 24). In how far they will actually do that and how the trading system will be designed if they do is unclear.
13. In Section 15 of the preamble, it is emphasized that the Directive does not prevent individual Member States from operating national emission trading schemes with a wider scope. If the states use this leeway in different ways, trading rules will be heterogeneous within the EU.
14. We have explained the leakage effect using comparative static analysis of the world markets for fossil fuels above. Game theoretic analysis shows that there are additional reasons for this effect: A welfare maximizing country produces higher equilibrium emissions given other countries reduce pollution in a Cournot-Nash framework. (Emission reaction functions are negatively sloped.) For this result it is sufficient that marginal damage increases with increasing emissions and marginal abatement cost increases with the quantity of abatement. These are quite conventional assumptions which carry high plausibility. See Finus (2001) for a detailed analysis. Of course leakage also occurs if firms escape strict environmental regulations by relocation.
15. From 2008, member states *may* extend trading to other GHGs. (see Article 24 (1).)
16. See Manne and Richels (2002).

17. The empirical studies presented in the literature show considerable differences regarding equilibrium reduction quantities, marginal cost etc. This is due to diverging projections of emission growth in the "business as usual"-scenario and to different modelling approaches. A comparative study of the literature mentioned above and many other empirical analyses is in Springer (2003). Even though different in many respects these studies share the view regarding the point under consideration here: Expanding the scope of a transferable permit system from CO_2 to a broader basis will considerably improve the cost effectiveness of the system.
18. An example of precise reporting is in the Danish NAP.
19. See National Audit Office (2004). Moreover, in appendix 4 of this report three other trading schemes are listed which integrate non CO2-GHGs.
20. The quota commission has been established by the Norwegian Government to make recommendations for a domestic GHG-emission trading system. The recommendations of the Commission are to be found on the web page of the Norwegian Ministry of the Environment.
21. These gases are carbon dioxide, nitrous oxide, perfluorocarbons and sulphur hexafluoride.
22. Decision 15/CP.7: "Principles, nature and scope of the mechanisms pursuant to Article 6, 12 and 17 of the Kyoto Protocol".
23. See the Commission of the European Communities (2003b), p. 7, emphasis added.
24. The ambiguity lamented here is also obvious in Article 30 (2d), (3).
25. The UK NAP has been approved by the EU-Commission "on the condition that technical changes are made." Making these changes will make the NAP automatically acceptable. These changes are not related to the issue discussed above. They concern provisions for new entrants and installations in Gibraltar. See document IP/04/862 on the web page of the European Commission.
26. Surprisingly, the chemical industry is exempt from the Directive. Boemare/Quirion (2002) attribute this to heavy lobbying activities during the genesis of the Directive.
27. Industries' *rent seeking* activities targeting an individually favourable initial endowment of permits may not only weaken the cost effectiveness of the system but also its social acceptability and credibility.
28. See Jochem and Eichhammer (1999) for the details of this commitment and Carraro and Siniscalco (1996), Carraro and Lévêque (1999) for a critical view of the (lacking) cost effectiveness of voluntary industry commitments.
29. S. Kallbekken (2004) uses a CGE model for a quantitative assessment of the point raised in qualitative terms, above. The result is that the cost of using sectoral differentiation increases the EU cost for meeting its Kyoto commitments by a factor of 2.5.
30. Commission of the European Union (2003a), emphasis added.
31. A critical view is taken by Montero (1999) showing that opt-in might lead to adverse selection.
32. This criticism of heterogeneous rules within the EU holds for all the design issues for which the Commission leaves the decision to the individual member state. There are many of those issues picked upon in this paper.
33. It should be noted that these issues are related to but not identical with the issue referred to in Note 23, above. In the present context we refer to article 30 of the EU-emission directive. Here, we find a *general* provisio of the Commission that it may change the rules of the game fundamentally. As the criteria for revision this article quotes the experience with the application of the Directive and international developments. Neither of the two criteria is specified. Opposed to that, the earlier remark does not refer to the Directive itself, but to an additional document, as explained in Note 23. Here, we find a *specific* provisio that a constraint will be imposed if a certain quantitative limit will be violated.
34. See Endres (1999).
35. See Endres (1999).
36. There are many negative experiences with this kind of regulatory interactions in market based environmental policy like the bubble and offset concepts used in the United States as elements of a Clean Air Act as well as the US Acid Rain Program (see e.g. Cronshaw and Kruse 1996, Harrison, 2002 and the papers reprinted in part II of Tietenberg 2002). The same analogously holds for the efficiency of the German waste water charges act (Abwasserabgabengesetz) which is considerably hampered by interaction with the command and control regulations of the German Clean Water Act (Wasserhaushaltsgesetz).
37. According to Article 26, "Member States *may* choose not to impose requirements relating to energy efficiency in respect of . . . units emitting carbon dioxide . . . " (emphasis added).
38. See Montgomery (1972) for rigorous proof.

39. The reason is that using auctions, the initial allocation of pollution rights can be expected to be close to the equilibrium allocation. This is completely different under a system of free initial distribution. Given transaction costs vary with the volume of trades these costs are minimized when the rights are initially allocated in a way that minimizes trade.
40. In their study of US SO2-allowance trading Goulder/Parry/Burtraw (1997) show that the cost to achieve the same policy goal are significantly lower using auctions compared to free distribution. See also Bovenberg and Goulder (1996), Parry and Williams (1999) and Parry and Williams et al. (1999).
41. This approach seems to be particularly interesting since a public choice view suggests that deviations from welfare maximizing behaviour of the government (deciding how revenues are used) should be expected.
42. See Boemare and Quirion (2002), Burtraw et al. (2001, 2002). According to an empirical assessment based on a CGE-model by Böhringer and Lange (2003) the detrimental effect of "consecutive grandfathering" appears to be very large. The authors conclude that for "dynamic allocation rules where the assignment of emission allowances depends on endogenous output decisions of firms, the implied distortion will nearly offset all efficiency gains" (p. 2).
43. See Fisher and Parry et al. (2003) and Goulder and Schneider (1999).
44. Therefore, with respect to ecological accuracy the advantage of transferable discharge permits compared to effluent charges, argued in section II, above, is not that important in the case of stock pollutants. See Endres (1999) and Pizer (2002).
45. It has been shown by Rubin (1996) that temporal flexibility is a prerequisite for full cost effectiveness.
46. However, banking is among the many features of the system which are optional to EU-member states.
47. The argument carried above implies that emissions trading is an instrument to meet *predetermined* goals to reduce the stock of greenhouse gases. There is no interdependency between instrument and goal in this line of reasoning. Taking a *public choice* perspective reveals that there are reasons to be cautious with regard to borrowing: Assume that polluters have intensively used borrowing and due to that, prices for emission permits reach a dramatic level at some future period. Then GHG polluting industry might argue that its existence is threatened by high permit cost and might pressure policy makers to allow emissions beyond the amount of permits held. Policy makers might give in and environmental policy goals are softened.
48. The comparison of a GHG-program with other programs regarding the possibilities for intertemporal trades has to be handled with caution. Borrowing is much more problematic regarding flow pollutants than it is regarding stock pollutants (like GHGs). In the latter case, the danger that borrowing might create "intertemporal hot spots" is much lower than it is in the former.
49. The sanction is €40 per excess ton of CO_2-equivalent in the first trading period and €100 in the second.
50. The argument presented above is analogous to the one given by Boemare and Quirion (2002), p. 224 in the context of the Bonn agreement regarding the Kyoto Protocol.
51. EU-Commission (2003c).
52. There is an ongoing discussion in the law and economics literature on the effectiveness of using stigma as an instrument to further compliance. See, e.g. Funk (2004) and the literature given therein. It would be an interesting research question whether the results of this literature can be transferred to the case of "naming and shaming" within the EU-emissions trading system.
53. The deadline for EU 15 member states was March 31, 2004, for new member states May 1, 2004.
54. For details of these procedures see documents IP/04/861, IP/04/862 and MEMO/04/44 on the web page of the EU-Commission.
55. See, e.g., Boemare and Quirion (2002), Braun and Jacobson (1992), Keohane (1995), Reeve (2002), Sand (1992) and Victor et al. (1998).
56. The principles for calculation, measurement and reporting are specified in annex IV of the Directive. A possible criticism of this procedure is that to date, the formula required for emission calculation from the data mentioned above does not allow to take new technologies to abate CO_2 into account, like carbon sequestration.
57. Boemare and Quirion (2002), p. 225.
58. A possible counter point to this idea would be to say that allowing different procedure allows competition of different national ideas and thereby facilitates learning. However, competition is able to further the achievement of the common good (here: cost effective GHG reduction within the EU) only if competition takes place within a set of rules designed to create the correct incentives for the competitors. To date, the EU commission did not design a framework aiming at this goal.

59. See Harrington and Morgenstern (2004).
60. Harrington and Morgenstern (2004), p. 16. In the quotation "EI" stands for "economic incentive" instrument.
61. A sceptical view regarding this aspect applied to the EU emissions trading directive is taken by Böhringer et al. (2004). The authors estimate the total cost for the EU to comply to its Kyoto obligations to be EUR_{1997} 1,913.9 in a system with no trading at all, 1,234.0 in an ideal trading system and 13,470.6 in the trading system which is currently defined by the EU-Directive (p. 15). Even if the authors caution their readers to interpret these figures to be "illustrative" (p. 21), they maintain that the given system causes "efficiency (to) detoriate in a drastic way" (p. 18) and call for the current system to be reconsidered "to avoid substantial excess cost of regulation" (p. 21).

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