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CARBON TRADING IN THE POLICY MIX

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The Kyoto Protocol is stimulating the development of emissions-trading schemes at the national and international levels. These are being introduced alongside existing policy instruments such as carbon taxes and negotiated agreements, leading to complex problems of policy interaction. But the topic of policy interaction remains under-researched. This paper aims to improve understanding of such interactions by examining the conditions under which a cap-and-trade scheme for carbon-dioxide emissions may usefully coexist with carbon/energy taxes, support mechanisms for renewable electricity, and policies to promote energy efficiency. The paper argues that each of these instrument combinations may be acceptable, provided they contribute to either improving the static or dynamic efficiency of the trading scheme, or delivering other valued policy objectives. But, since the coexisting instruments may raise overall abatement costs while contributing nothing further to emission reductions, the objectives and trade-offs within the policy mix must be explicit.

I. INTRODUCTION

Climate policy is relatively immature but growing rapidly in scale, scope, and complexity. In particular, the Kyoto mechanisms have created a unique international framework for market-based regulation which is stimulating the development of greenhouse-gas emissions-trading schemes at the national and international levels. The proposed EU Emissions Trading Scheme (ETS) is the most significant of these developments and is expected to cover some 45 per cent of EU carbon-dioxide (CO_2) emissions from 2005 onwards (CEC, 2003*a*).[2]

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[2] The proposed EU ETS is a harmonized, EU-wide 'cap-and-trade' scheme for CO_2 emissions. Participants include electricity generators, oil refineries, and energy-intensive manufacturing installations in sectors such as iron and steel, minerals, and paper. Phase 1 of the EU ETS runs from 2005 to 2008, and Phase 2 from 2008 to 2012. The size of the aggregate cap (total number of allowances) in Phase 1 has yet to be decided. Each participating member state will decide on the number of allowances to allocate to national participants—subject to approval by the EC—and the sum of these national caps will equal the overall EU cap.

As climate policy develops, so the number of policy instruments grows, together with the potential for interaction between these instruments. This interaction can be complementary and mutually reinforcing, but there is also the risk that different policy instruments might interfere with one another and undermine the objectives and credibility of each other. Furthermore, the unprecedented scope of the climate problem means that climate policy also interacts with a range of other instruments in the areas of energy, environment, transport, trade, fiscal, technology, agricultural, and social policy. These interactions are likely to have a determinate impact on the success of climate policy in general and on the development of emissions-trading schemes in particular. But despite this, the topic of policy interaction remains under-researched. With a few notable exceptions (Smith, 1999; Egenhofer, 2002; Johnstone, 2002), the majority of the economics literature confines itself to the study of individual instruments in isolation.

The political-science literature engages with the problem of policy interaction rather more successfully. Majone (1989) describes the problem of policy congestion where: 'solutions beget new problems in the form of policy overlaps, jurisdictional conflicts and unanticipated consequences'. Wildavsky (1979) considers this interaction to be a prime cause of much policy development, with the solution of these internally generated problems becoming as important as responding to external changes. A study by Glachant (2000) on the implementation of EU environmental directives found that the pervasiveness of interactions meant that a simple link between policy and outcome could not be found. Instead, the research question was recast as 'how can the implementation of a particular regulation cope efficiently with policy interactions?' Similarly, Gunningham and Gabrosky's (1998) detailed study of the operation of environmental regulation in the chemical and agricultural sectors concluded that approaches to environmental regulation based on single policy instruments were misguided. A better strategy was to harness the strengths of individual instruments while compensating for their weaknesses by the use of additional and complementary instruments within a broader policy mix. This includes the use of multiple instruments to achieve a single objective ('killing one bird with two stones') (Johnstone, 2002), as well as the use of multiple instruments to achieve multiple objectives (as exemplified by government regulation of the energy industry; Helm, 2003).

This paper aims to contribute to the literature on policy interaction by examining the conditions under which a cap-and-trade[3] ETS for CO_2 emissions may usefully coexist with other climate-policy instruments.[4] This involves two steps. First, an assessment of the extent to which, and mechanisms by which, other climate-policy instruments may contribute to or undermine the primary objective of a carbon ETS, which is to achieve a particular CO_2 emissions target at least cost. Second, an assessment of whether the coexistence of such instruments with an ETS can be justified, given both a particular set of policy objectives and a weighted set of policy evaluation criteria. Three types of coexisting instrument are explored, namely: carbon/energy taxes; support mechanisms for renewable electricity; and non-price instruments to overcome barriers to energy efficiency. Instruments in these categories form the focus of climate policy in most EU countries and each will interact with the proposed EU ETS in a variety of ways. The paper argues that such instrument combinations may be acceptable, provided they contribute either to improving the static or dynamic efficiency of the ETS, or to delivering other valued policy objectives. But successful combinations require both careful design and transparency in objectives. In practice, the incremental evolution and lock-in of individual instruments can militate against such successful combinations.

The remainder of the paper is in four parts. Section II explores some of the issues related to policy interaction with a cap-and-trade ETS, including the

[3] In a cap-and-trade ETS for CO_2 emissions, a fixed number of allowances is allocated each year to the participating sources. Each participant must surrender one allowance for every tonne of CO_2 emitted. Participants who face high abatement costs can continue to pollute by buying additional allowances, while participants who face low abatement costs can take abatement action and sell their surplus allowances for a profit. In this way, each participant can minimize its overall abatement costs. The scheme places an overall 'cap' on the annual quantity of CO_2 emissions equal to the number of allowances distributed, and the trading mechanism should allow this cap to be achieved at the lowest possible cost.

[4] The discussion in this paper is confined to the trading of CO_2 emissions. In practice, such trading schemes may also include other greenhouse gases.

distinction between directly and indirectly affected target groups, the cost incidence of an ETS, the different types of policy interaction, and the implications of an emissions cap. Section III applies these ideas to the three combinations of instruments identified above and seeks to identify the conditions under which such combinations may be justified. Section IV provides a brief overview of how these types of interaction may be triggered by the introduction of the EU ETS in the UK. Section V concludes.

II. POLICY INTERACTION AND CARBON ETS

(i) Direct and Indirect Impacts of a Policy Instrument

It is rare for policy innovations such as an ETS wholly to displace existing instruments. Instead, an ETS is likely to operate in parallel with existing instruments and to interact with them in a variety of ways. In exploring these interactions, a useful distinction is between *directly* and *indirectly* affected target groups—where a target group is defined as the collection of economic actors influenced in some way by a policy. A directly affected target group has obligations and incentives imposed upon it directly by a policy instrument, while an indirectly affected target group is influenced in some way by the behavioural changes that are made by the directly affected group. Of particular interest is the extent to which the additional costs imposed by a policy instrument on the business sector are indirectly borne by consumers, suppliers, and shareholders. So, for example, electricity generators participating in an ETS may either increase wholesale electricity prices (pass to consumers), reduce the consumption or unit price paid for supply inputs (pass to suppliers), or reduce dividends and capital gains (pass to shareholders) (Cramton and Kerr, 1998). In each case, the extent to which costs can be passed on will depend upon the market situation of the firm and the elasticities of demand and supply in each market. It will also depend upon the timeframe under consideration and the extent to which companies have the opportunity to change behaviour and invest.

Indirect effects permeate throughout the economy and require analysis within a general equilibrium framework. But if the electricity generators are participating in the ETS, the indirect impact on electricity consumers becomes of particular interest. This is because, first, the economic and environmental implications of carbon controls on electricity emissions are potentially very large; and, second, electricity consumers are typically subject to a wide range of other policies that will interact with the ETS.

The direct/indirect distinction is also relevant to the design of an ETS for CO_2 emissions. There is a basic choice between a *downstream* ETS, in which fossil-fuel users surrender allowances for their emissions, and an *upstream* scheme, in which fossil-fuel producers surrender allowances for the carbon content of the fuel. Within a downstream scheme, there is a second choice between the *direct* treatment of electricity emissions, where electricity generators surrender allowances, and an *indirect* treatment, where electricity consumers (or a subset of consumers) surrender allowances in proportion to the carbon content of delivered electricity. Each option has pros and cons and each has different implications for incentives and abatement options. For example, electricity generators have full and direct control over the carbon intensity of electricity generation, through investment and operational decisions, such as fuel switching, but they have only indirect and partial control over total electricity demand through electricity prices. In contrast, electricity consumers have full and direct control over their electricity demand, through investment and operational decisions, such as energy efficiency, but have no control over the carbon intensity of electricity generation unless some form of 'carbon labelling' of electricity is available. Similarly, while an upstream scheme ensures an economy-wide cap on fossil-fuel emissions and a single price for carbon throughout the economy, a downstream scheme will be confined to a subset of emission sources and may lead to carbon being priced differently between sectors and fuels.

(ii) The Indirect Impacts of an Emissions Trading Scheme

A fundamental choice in the design of an ETS is between the auctioning or free allocation of allowances (or some combination of the two). While this choice may lead to different costs for ETS participants, there should be no difference in the costs passed on in product prices. This important result rests on a number of assumptions, including that:

- the majority of firms in a particular product market are covered by the allowance programme;
- firms are profit maximizing and decisions about entry and exit are not affected by financing constraints;
- there is no market power in either the product or allowance market; and
- product prices are not subject to economic regulation.

With allowance auctioning, firms incur costs for abatement plus the allowances purchased in the auction which are used to cover residual emissions. Both are real accounting costs. With free allocation, firms only incur abatement costs, including the net cost of any acquisition of allowances. But the freely allocated allowances have an *opportunity cost* in that they could be sold on the allowance market. This opportunity cost should be treated identically to real accounting costs in a firm's pricing decisions (Harrison and Radov, 2002). Viewing the situation another way, the wealth provided by freely allocated allowances represents a lump-sum profit which should not influence product-pricing decisions since, in theory, these are based upon marginal costs.

The difference between auctioning and free allocation lies in the capture of the economic rent, rather than the cost increases for consumers. With free allocation, the rent is captured by the participating firms, thereby increasing their market value. With auctioning, the rent is captured by the government and may be used in a variety of ways throughout the economy, including compensating affected groups and reducing other forms of taxation. But the price impacts for consumers should be identical in both cases.

Whether this result holds in practice will depend upon the validity of the assumptions behind the economic model. For example, agency problems and other factors within firms may move them away from profit-maximizing behaviour or an individual firm may be able to exercise market power. In the US Acid Rain Program the participating electricity generators were subject to utility regulation, which distorted product pricing by valuing allowances at historic cost (zero) rather than opportunity cost (Bohi and Burtraw, 1992). In countries such as the UK, the electricity-generation market is liberalized with relatively limited market concentration, so neither of the last two problems should apply.

It is possible that there will be political objections to firms' pricing decisions. Free allocation is equivalent to a lump-sum subsidy to the participating firms which acts to increase shareholder wealth. Consumers face the same price impacts as in an auctioning scheme, but without the compensating use of auction revenues. It is possible that this will be seen as 'double charging', since consumers pay once as taxpayers, in creating the subsidy, and a second time as consumers in purchasing the sector's products. But the subsidy results from the social creation of scarcity (carbon emissions) and the forgoing of the corresponding economic rent (in the form of auction revenues), rather than through the explicit use of taxpayers' money.

It may be possible to compensate consumers for price increases by separating the *allocation* of allowances from the *compliance obligations* for emissions. For example, allowances could be allocated to electricity consumers, while electricity generators remained responsible for compliance. Here, the generators would need to purchase allowances from consumers, with the revenue transfers compensating for any increase in electricity prices. In practice, a combination of transaction costs, lobbying by ETS participants, and legal restrictions on who can receive allowances may limit the feasibility of this alternative. For example, the EU ETS restricts the allocation of allowances to participants in the scheme.[5]

(iii) A Typology of Policy Interaction

The distinction between directly and indirectly affected target groups leads naturally to a distinction between *direct* and *indirect* policy interaction. In addition, with an ETS there is the additional

[5] It would be possible to allocate allowances to energy-intensive industries participating in the EU ETS to compensate them for increases in electricity prices. However, the same compensation could not be extended to non-participants.

possibility of *trading* interaction. Each of these is introduced below.

- *Direct interaction* is where the target groups directly affected by two policies overlap in some way. For example, some or all of the participants in a carbon ETS may already be subject to CO_2 emission limits or to a carbon tax on fuel use.

- *Indirect interaction* occurs when a target group is indirectly affected by one policy and either directly or indirectly affected by a second. So, for example, there is indirect interaction between a downstream ETS which includes the electricity generators and a tax on electricity at the point of consumption. Here, electricity consumers are indirectly affected by the ETS and directly affected by the tax. Similarly, there is indirect interaction between this type of ETS and obligations upon electricity suppliers to purchase renewable electricity, since both will lead to higher prices for electricity consumers and lower emissions from electricity generators.

- *Trading interaction* is where two policies influence one another by the exchange of an environmental trading commodity. For example, allowances from a trading scheme in one country may be exchangeable for allowances from a trading scheme in a second. Any such links would need to be governed by transfer and exchange rules, which in combination would define the *fungibility* of the different commodities. Trading interactions between the Kyoto mechanisms and national and international trading schemes are becoming a critical issue as the Kyoto regime develops and have important implications for abatement costs and environmental integrity (Haites and Mullins, 2001). In addition, the parallel development of schemes such as tradable green credits for renewable electricity opens up the additional possibility of linking schemes where the tradable commodities have different denominations (e.g. megawatt-hours (MWh) and tonnes of carbon dioxide (tCO_2), respectively; Morthorst, 2001; Sorrell, 2003*a*). This is possible because the tradable commodities in such schemes represent, in part, displaced carbon emissions and hence can be converted to tCO_2 by means of a suitable exchange rate.

Each type of interaction may have implications for abatement costs, administrative costs, environmental effectiveness, equity, and political feasibility. Furthermore, each type of interaction may lead to differential treatment, with some target groups being affected by both instruments and some by only one. Hence, the extent to which such interactions can be judged as beneficial, neutral, or counterproductive requires a careful examination of the nature and consequences of the interaction and an evaluation of those consequences within a multicriteria framework (Weimer and Vining, 1999). Sorrell *et al*. (2003) have developed a general methodology for analysing policy interaction, involving a systematic comparison of:

- the *scope* of each instrument, where scope means the sectors, sites, portions of sites, and individual emission sources that are directly or indirectly affected by each instrument;

- the *objectives* of each instrument and the extent to which these reinforce or conflict with one another;

- the *operation* of each instrument, including the aggregate effect of the different obligations and incentives when applied in combination;

- the *implementation* of each instrument, including the scope for rationalization and harmonization or regulatory responsibilities; and

- the *timing* of each instrument, including responses to 'triggers' and the scope for policy sequencing.

This methodology can be combined with a multicriteria evaluation against an agreed set of objectives, constraints, and evaluation criteria to provide a judgement as to whether the combination of instruments is useful, redundant, or positively harmful.

(iv) Double Regulation and Double Counting

In most OECD countries, the existing climate-policy mix contains a series of examples of the above categories of interaction. If an ETS is introduced into this mix, the number of interactions can be expected to multiply. If both the ETS and the existing instruments have significant economic impacts, the affected target groups may complain of *double regulation*, where they perceive themselves as paying twice for reducing carbon emissions (Sorrell, 2002). The validity of such perceptions will depend upon the objectives of each instrument, the clarity, transparency, and legitimacy of these objectives, and the degree of overlap between them. For example, two instruments may each have the objective of reducing carbon emissions, but if they are targeted at two different market failures (e.g. carbon externalities and asymmetric information), their coexistence may be considered acceptable.

In the fluid post-Kyoto policy mix, it is also possible that more than one type of carbon-trading scheme will coexist, or that a carbon ETS will coexist with trading schemes for other commodities, such as renewable electricity. In these circumstances, there may also be problems with *double counting* of carbon emissions, where the compliance obligations for the same emissions are either given to two separate parties, or given to the same party under two separate terms (Sorrell, 2002). These disputes may occur whether or not there is trading interaction between the two instruments and may have two consequences:

- *double coverage*, where two separate carbon allowances or carbon credits are surrendered for a 1-tonne increase in physical emissions; and
- *double crediting*, where two separate carbon allowances or carbon credits are generated from a 1-tonne decrease in physical emissions.

A cross-border example of double coverage would be the export of electricity from country A, which has an ETS with direct accountability (electricity generators hold allowances), to country B, which has an ETS with indirect accountability (electricity consumers hold allowances). Both the seller of the electricity (generators) in country A and the purchaser of the electricity (consumers) in country B would need to surrender allowances to cover the emissions associated with this electricity, which means the emissions would be covered twice, i.e. by two separate trading schemes. A primary motivation for introducing a harmonized ETS throughout the EU was to avoid such problems (Zapfel and Vainio, 2001).

In many cases the double coverage will be offset by double counting and there will be no threat to the environmental integrity of either scheme. But this may not always be the case. For example, the UK has proposed a scheme which allows for the generation of carbon credits from individual projects that improve electricity efficiency (Begg *et al.*, 2002). If, at the same time, the electricity generators were covered by a cap-and-trade scheme, such as the EU ETS, this would lead to double crediting without any compensating double coverage.

(v) Policy Interaction under a Cap

A defining feature of a cap-and-trade ETS is that, assuming adequate enforcement and full compliance, there is certainty that total emissions will be less than or equal to the aggregate cap. A second feature of a cap-and-trade scheme is that, under a standard set of assumptions regarding the competitive operation of the allowance market, the trading scheme will allow the target to be met at least cost. In the equilibrium, marginal abatement costs will be equalized across sources and equal to the allowance price.

As Sijm (2003) has argued, these idealized features of an ETS have important implications for policy interaction. Coupled with comparable assumptions regarding the idealized operation of the relevant product markets, they imply that the use of a second instrument that directly or indirectly interacts with the ETS will increase the overall costs of meeting the emissions cap, while at the same time having no influence on environmental effectiveness—where the latter is defined as assurance in meeting the ETS cap. The aggregate abatement costs of ETS participants may be either increased (e.g. by a carbon tax) or reduced (e.g. by a subsidy scheme) by the second instrument, but in all cases the aggregate social costs of meeting the cap will be increased and participant emissions will continue to be less than or equal to the cap. This result applies both to instruments which directly affect CO_2 emissions from ETS participants, such as a carbon tax on fuel use, and to instruments which indirectly affect those emissions, such as a tax on electricity consumption of both participants and non-participants (Sijm, 2003).

To illustrate this, assume that the second instrument is a negotiated agreement (NA) which sets emission limits on a subset of ETS participants in terms of emissions per unit of output. It is straightforward to demonstrate that, where the NA limits are binding, they increase the marginal abatement costs of affected ETS participants by a factor λ, which is a measure of the change in costs for a marginal change in the NA emission limit (Sorrell, 2002, pp. 112–15). As a consequence of this double regulation the affected participants are likely to reduce emissions further than they would under the ETS alone, which means that they are likely either to sell more allowances or to purchase fewer allowances. The consequent reduction in allowance prices will make it easier for other ETS participants which are not affected by the NA to comply with their ETS obligations. Aggregate emissions will not have changed, since any 'freed-up' allowances will simply be used by other participants to cover increases in emissions. But aggregate abatement costs will have increased, since the distribution of abatement actions across participants will have departed from the cost-minimizing optimum (Sijm, 2003). Also, the differential treatment of participants may have introduced distortions to competition, with the participants subject to the 'double regulation' effectively subsidizing competitor participants which are not.[6] If *all* ETS participants are subject to the NA targets, the primary effect will be to increase overall abatement costs and lower the allowance price. If the emission limits from the NAs are sufficiently stringent, aggregate emissions will be reduced below the cap, the ETS will become redundant, and the price of allowances will fall to zero.

Very similar conclusions apply to instruments which indirectly affect ETS participants, such as an electricity tax. In this case, reductions in electricity demand will substitute for lower cost abatement by ETS participants (e.g. fuel switching), while overall emissions will be unchanged. Again, a very stringent electricity tax could reduce electricity consumption sufficiently that aggregate emissions are reduced below the cap and the ETS becomes redundant. However, if the electricity generators form only a subset of ETS participants, this is unlikely.

In practice, allowance and product markets may only approximate the theoretical ideal. Market failures will be pervasive in both markets and the political bargaining that led to the ETS cap is unlikely to provide an adequate reflection of the 'social optimum' for carbon externalities (to the extent that such a concept is meaningful for global climate change). In addition, governments have objectives which go beyond efficiency, such as the promotion of social equity. In these circumstances, there *may* be legitimate grounds for introducing or maintaining other climate-policy instruments that directly or indirectly interact with the ETS. These include:

- improving the static efficiency of the ETS by overcoming market failures other than CO_2 externalities;
- improving the dynamic efficiency of the ETS by overcoming market failures in the area of technology innovation and diffusion;
- delivering social objectives other than efficiency, such as equity and political feasibility; and
- compensating for deficiencies in the ETS design (Johnstone, 2002; Sijm, 2003).

However, the fact that positive combinations between an ETS and other instruments are theoretically possible does not mean that such combinations will result when an ETS is introduced into an existing policy mix. Furthermore, when an ETS is in place, aggregate emission reductions will be set solely by the ETS cap. Instruments which target emissions covered by the ETS cap will contribute nothing further to emission reductions—unless they are sufficiently stringent that they make the ETS redundant. This means that, once the ETS is in place, the justification for maintaining such instruments must rely upon one of the above rationales, rather than the contribution of the instrument to overall emission reductions (Sijm, 2003).

[6] This only applies to those competitors which are buyers of allowances. If they are sellers, the value of their sales will be reduced.

It is important to note that the same conclusion does not follow for instruments which do not directly or indirectly interact with the ETS. These will contribute emission reductions independently of and in addition to the ETS cap. So, for example, if the ETS is a downstream scheme that includes the electricity generators, emissions from household electricity consumption will be covered by the cap, while emissions from household fuel consumption will not. Hence, policies which affect the former will directly or indirectly interact with the ETS, while policies which affect the latter will not. Conversely, if the EU ETS is an upstream scheme, *all* fossil-fuel emissions will be covered by the cap and all policies that affect these emissions will interact with the ETS. Since none of these policies will contribute anything further to overall emission reductions, the justification for maintaining them must rely solely on one of the above categories.

The following sections use these insights to discuss possible justifications for combining a cap-and-trade ETS for CO_2 emissions with: (*a*) carbon/energy taxes; (*b*) policies to support renewable electricity; and (*c*) policies for promoting energy efficiency.

III. INTERACTIONS BETWEEN CARBON ETS AND SELECTED CLIMATE-POLICY INSTRUMENTS

(i) Carbon Trading and Carbon/Energy Taxation

Most OECD countries use some form of carbon/energy taxation. Unless these taxes are removed when a carbon ETS is introduced, or exemptions introduced for target groups directly and indirectly affected by the ETS, policy interaction becomes inevitable.

Carbon/energy taxation of the fuel used by a participant in a downstream ETS will distort the *substitution* objectives of each instrument. Marginal emissions will be priced twice for ETS participants, at different implicit or explicit rates, but only once for non-participants. Similarly, the coexistence of carbon/energy taxation of electricity with the participation of electricity generators in the ETS will distort the incentives to substitute between fuel and electricity consumption. In practice, the majority of existing tax regimes do not price carbon consistently between sectors and fuels (Newbery, 2001) and introducing a downstream ETS in this context is likely to distort the substitution incentives still further.[7]

Despite these distortions, there may be circumstances where the retention of the carbon/energy tax provides compensating benefits. This may particularly be the case when allowances are freely allocated to ETS participants rather than auctioned. Free allocation violates the polluter-pays principle in that participants only pay for the marginal damage of CO_2 emissions, while inframarginal emissions remain unpriced. This is in contrast to the use of allowance auctions which impose costs for all emissions. Free allocation may also undermine the incentives for technical innovation to reduce emissions. This topic is a subject of a growing literature, with most studies arguing that auctioning provides greater incentives for innovation than free allocation (Downing and White, 1986; Millman and Prince, 1989; Jung *et al.*, 1996).

Free allocation also means that the ETS can contribute nothing to the government's fiscal objectives. No revenue is being raised and the economic rent from allowance distribution is distributed wholly to shareholders. This is in contrast to an auction scheme where allowance revenue can be used to compensate affected groups or to reduce other forms of taxation. Numerous studies have demonstrated how the recycling of auction or tax revenue to reduce other forms of taxation can provide a net welfare benefit (de Mooij, 1999). While the extent of this 'double dividend' is contested, the efficiency benefits of auctioning compared to free allocation are not.

Despite the theoretical benefits of auctioning, free allocation is the norm and is likely to remain so for the foreseeable future. Political opposition to large-

[7] For example, the UK Climate Change Levy (CCL), which applies to the business and public sectors, corresponds approximately to a £8.15/tCO_2 tax for natural gas, a £4.55/tCO_2 tax for coal, and a £9.35/tCO_2 tax for the primary fuel input to electricity generation. Oil products are excluded on the grounds that these are already subject to excise duties, which in turn correspond to an equivalent carbon tax of £8.9/tCO_2 for heavy fuel oil and £11.6/tCO_2 for gasoil.

scale rent transfers undermines the feasibility of auctioning in the same manner as it constrains carbon/energy taxes to sub-optimal levels and/or necessitates the use of extensive tax exemptions. In these circumstances, the retention of existing carbon/energy taxes after an ETS is introduced may be seen as a pragmatic, second-best alternative to the use of allowance auctions. If applied directly to ETS participants, it will: (*a*) ensure that a portion of inframarginal emissions are priced—albeit at a different rate to marginal emissions; (*b*) increase the incentives for technical innovation—although this will be balanced by the reduction in the value of allowance holdings that a tax creates; and (*c*) provide a means to recover some of the windfall rent from allowance allocation. If applied either directly to participants or indirectly to consumers, it will ensure that the revenues from the carbon/energy tax are maintained (Johnstone, 2002).

In practice, the last benefit is likely to be particularly important. Existing carbon/energy taxes have generally been established in the face of vigorous opposition, sometimes as part of a broader programme of environmental tax reform. Having established such a tax, a government will be reluctant to relinquish the income benefits simply to rationalize the substitution incentives of the policy mix. Carbon/energy tax revenues are commonly used to offset other forms of taxation, fund R&D programmes or provide subsidies and tax allowances for the adoption of low-carbon technologies. If the tax is removed, this revenue must either be recovered from other sources or the relevant programmes abandoned.

(ii) Carbon/Energy Taxation as a 'Back-up'

The relative merits of retaining or abandoning the coexisting tax will depend upon its marginal rate relative to the anticipated allowance price in the ETS, and the consequent expectations regarding the economic impacts of the double regulation on affected groups. Tax exemptions or reductions for ETS participants may provide one mechanism to reduce these impacts, with the full rate of the tax continuing to be applied to non-participants. Alternatively, if allowance prices are anticipated to be low, there may be an argument for retaining the carbon/energy tax as a 'back-up' to ensure a minimum level of abatement by ETS participants.

In the first instance, allowance prices will be determined by the size of the cap relative to the aggregate marginal abatement cost curve. But an important complication with a carbon ETS is the possibility of trading interaction with other schemes or with the Kyoto mechanisms. A combination of the refusal of the USA to ratify the Kyoto Protocol, the surplus 'hot air' in the assigned amounts to Russia and the Ukraine, and the generous provisions for crediting carbon sequestration by 'sinks' has created the possibility of very low carbon prices after 2008 (Den Elzen and de Moor, 2003). In this context, any interface between a carbon ETS and the international carbon market could have the effect of reducing the ETS allowance price and substituting the purchase of fungible carbon commodities from outside the ETS for abatement by ETS participants. The debate over 'supplementarity' demonstrates that this is an ongoing concern for many EU countries, several of whom may prefer to prioritize domestic abatement over aggregate emission reductions. While one method of incentivizing such abatement would be to restrict trading links, this option may not always be available. For example, the proposed EU ETS incorporates links with both the Kyoto mechanisms and other national trading schemes, despite the reservations of environmental groups and some participating member states (CEC, 2003*b*; Climate Action Network Europe, 2003).

If existing carbon/energy taxes are maintained as an alternative means of ensuring 'supplementarity', the net result will be to increase abatement costs within the domestic ETS, reduce emissions from ETS participants, and either increase allowance sales to or reduce allowance purchases from the linked trading scheme. If both the domestic and linked ETS are of the cap-and-trade form, the aggregate emissions within the two schemes will remain unchanged. Conversely, if the linked ETS uses relative rather than absolute targets, or if credits from project mechanisms are used, the environmental integrity of an overall cap will be lost.

A possible rationale for giving priority to domestic abatement would be to put a country 'on course' for achieving much greater reductions in CO_2 over the next half century. This objective relies on a set of explicit or implicit assumptions regarding: the uncertainty and potential severity of climate threats; the appropriate global targets for CO_2 emissions over the medium to long term; the appropriate contribution of different countries towards those targets; the importance of domestic action by developed countries to encourage the subsequent participation of developing countries in the Kyoto Protocol; the need to ensure a transition away from long-lived, CO_2-intensive capital stocks and infrastructure; and the consequent threat of high adjustment costs in the future should that transition be delayed (Grubb, 1997; RCEP, 2000). While each of these assumptions can be questioned, the 'pathways' objective has become explicit in the climate policy of some OECD countries. For example, the UK has a 'goal' of reducing CO_2 emissions to 20 per cent below 1990 levels by 2010 (DETR, 2000), a target which goes beyond requirements under the EU burden sharing agreement, and has recently expressed a commitment to 'put the UK on a path' to reducing CO_2 emissions by some 60 per cent below current levels by 2050 (DTI, 2003).

These domestic targets have led to a debate over whether the allocation to UK participants in the EU ETS should be consistent with the 20 per cent goal or with the UK's burden-sharing target. But the former would not necessarily succeed in reducing *domestic* UK emissions because participants could simply purchase allowances from other member states. Although the overall EU cap would be tightened by the UK's actions, the overall size of that cap will depend upon the allocation decisions of other member states. In contrast, the coexistence of the EU ETS with a carbon/energy tax would achieve greater emissions reductions in the UK (but not in the EU) at the expense of higher abatement costs and potential damage to the competitiveness of UK industry.

In sum, the direct or indirect interaction of a carbon/energy tax with an ETS may potentially be justified through the polluter-pays principle, the increased incentives for innovation, the capture of windfall rent from allowance allocation, the maintenance of fiscal revenue, or the perceived need for a 'back-up' regulation to ensure a minimum level of abatement by groups directly or indirectly affected by the ETS. It is a matter of judgement as to whether these benefits offset the distortions in substitution incentives that such double regulation creates.

(iii) Carbon Trading and Policies to Promote Renewable Electricity

Many OECD countries use one or more instruments to support the innovation and diffusion of renewable electricity sources. These instruments may be price based, such as a guaranteed purchase price for renewable electricity; quantity based, such as an obligation upon electricity suppliers to purchase a certain percentage of electricity from renewable sources; or a hybrid, in which the price risk of a quantity scheme is mitigated by placing a ceiling on the total costs of achieving the renewables target (Roberts and Spence, 1976; Menantau *et al.*, 2003). Each effectively combines a financial subsidy to renewable generators (generally paid for by electricity consumers rather than taxpayers) with a long-term take-or-pay contract to reduce investment risk in an increasingly liberalized electricity market (Helm, 2002). The equivalent abatement costs of such mechanisms are typically high compared to competing options. For example, the price ceiling in the UK Renewables Obligation (a hybrid scheme) corresponds to maximum abatement cost of approximately €100/tCO_2, which is least ten times higher than first-year allowance prices in the UK ETS. While a comparable quantity of renewables deployment could be achieved through an ETS, the cap would need to be much more stringent than is required to meet existing targets under the Kyoto Protocol.

As with a tax, the coexistence of a support scheme for renewable electricity with a carbon ETS will raise overall abatement costs while contributing nothing to the aggregate cap. However, there may be legitimate grounds for supporting renewables, independent of their contribution to carbon abatement over the short term. For example, it is well established that private markets will under-invest in R&D as a consequence of both the uncertainty and intangibility of R&D outcomes and the inability of innovators fully to appropriate the social returns of such investment (Stoneman and Vickers, 1988). Also, the development of new technologies is characterized by learning-by-doing, where performance improves and costs fall as production experience is accumulated (Ibenholt, 2002). Learning creates an additional source of positive externality, as the act of investment benefits future investors, but the benefit is not paid for by the market (Arrow, 1962). These arguments point to a general role for government in both funding R&D and in steering innovation in desired directions (Grubb, 1997; Gross and Foxon, 2002). This can be achieved in a variety of ways, but it is generally recognized that policies need to avoid 'picking winners' (Hall, 2002). Quantity-based support schemes are designed to provide aggregate targets for renewables deployment while not specifying the contribution from individual technologies, but they still require the government to define those technologies which qualify as 'renewable'.

Two other factors may legitimate government support for renewables deployment.

- The increasing returns to adoption as a result of learning-by-doing may combine with other factors both to *lock in* dominant technologies and to *lock out* viable alternatives (Arthur, 1989). These other factors include scale economies in production, the inertia of long-lived capital stock, and the network economies associated with the relations between technologies, infrastructures, interdependent industries, suppliers, users, public and private institutions (e.g. trade associations, universities, etc.), and public expectations (Unruh, 2000). As with the 'pathways' objective, this evolutionary perspective on technical change implies that a failure to invest in the development of low-carbon technologies, such as renewables, could lead to lock-in to a high-emissions path. In turn, this creates the risk that the cost of switching to alternative technologies could become prohibitive, particularly if climate impacts turn out to be more severe than anticipated. Again, the policy recommendation is for targeted support in areas that offer significant opportunities for low-carbon innovation (Grubb and Ulph, 2002).

- The expectation that global targets on carbon emissions will tighten creates the possibility that early support for renewable technologies may drive down unit prices sufficiently to form the basis for viable industries with significant export potential. An example of this 'early mover advantage' is German support for wind power, which saw investment costs decreasing from €4,500 per kilowatt-hour (kWh) in 1992 to below €1,000/kWh in 2002, and enabled German firms to capture much of the world market (Haas, 2002).

A further rationale for supporting renewables relates to the possibility of improving security of supply through the promotion of diversity in generation sources. This argument has a long history within energy policy, but has rarely been subject to serious scrutiny. A notable exception is Stirling (1994), who distinguished between variety (number of options), balance (extent of reliance on individual options), and disparity (difference between options), and developed a systematic approach to optimizing portfolio diversity taking into account multicriteria appraisals of the performance of different options and the willingness to pay for diversity implicit in previous government support for nuclear power. A robust conclusion from this analysis is that a 'diversity optimal' generation mix would have a higher contribution from renewables than at present (e.g. up to 30 per cent) and that under a range of scenarios, renewables investment at the margin was a more effective route to increasing diversity than investment in nuclear power. Security of supply has recently re-emerged as a concern within energy policy, but the extent to which it justifies policy intervention is contested (NERA, 2003). Furthermore, diversity represents only one dimension of security of supply, and existing support mechanisms may lead to a high reliance on a single renewable technology (wind) with intermittent output.

In sum, both technology market failures and supply security objectives could potentially provide a rationale for supporting the diffusion of renewable technologies. These rationales are less well established than environmental externalities as a basis for government intervention and there is a lack of consensus over either the extent of intervention that is appropriate or the particular instruments that should be used. But if such instruments are to coexist with a cap-and-trade ETS, these rationales become more important. This is because aggregate emission reductions will be set solely by the ETS cap, and the renewables policy will simply shift abatement to technologies which impose significantly higher costs in the short term. The question then becomes whether the perceived, long-term benefits of renewables are considered sufficient to offset these short-term costs.

(iv) Carbon Trading and Policies to Promote Energy Efficiency

Most OECD countries use a range of policies to promote energy efficiency, including information programmes, labelling schemes, tax allowances, subsidies, regulatory standards, and market-transformation programmes. These are typically justified as a means to overcome non-price 'barriers' to energy efficiency which prevent individuals and organizations from investing in highly cost-effective efficiency improvements. Economists have been more sceptical of such approaches and consider that only a subset of these barriers can be considered as market failures which justify public intervention (Jaffe and Stavins, 1994; Sutherland, 2000). For example, if efficiency investments are associated with real but 'hidden' costs, such as disruptions to production, this creates a barrier but does not justify intervention. Nevertheless, a wide range of failures in energy service markets have been identified, including imperfect information on efficiency opportunities, and asymmetric information between contracting parties leading to problems of split incentives, adverse selection, and moral hazard (Sorrell *et al.*, 2000). A standard example is landlord–tenant relationships in the housing market, where neither party has the incentive to invest in energy efficiency. Similarly, asymmetric information between buyers and sellers prevents house prices reflecting the discounted value of efficiency investments.

Following Arrow (1969), it is perhaps more useful to consider these market failures in a relative rather than absolute sense using 'a broader category, that of transaction costs, which in general impede and in particular cases completely block the formation of markets'. Transaction costs are a feature of all contractual relationships and are shaped by both the nature of the transaction and the associated legal, organizational, and institutional arrangements (Williamson, 1985). Combined with the behavioural assumptions of bounded rationality and opportunism, transaction costs can provide a valuable insight into the nature of barriers to energy efficiency (Sorrell *et al.*, 2000). For example, transaction costs help explain why the landlords and tenants do not enter into shared-savings contracts to share the benefits of efficiency investments. Similarly, the bounded rationality of economic agents, coupled with the invisibility of energy-efficiency performance, helps explain why purchase decisions are biased towards equipment with low capital cost, but high running costs. In manufacturing, the asymmetric information between senior management and individual departments helps explain the use of capital budgeting procedures that prevent all but the most cost-effective energy-efficiency projects from going ahead (Ross, 1986). And in the construction industry, opportunism and asymmetric information help explain why subcontractors substitute cheaper and less-efficient building services equipment, leading to higher running costs for the client (Sorrell, 2003*b*).

Proponents of energy efficiency argue that public intervention can overcome barriers such as these through reducing transaction costs, economizing on bounded rationality, aligning the incentives of different groups in the direction of improved efficiency, and safeguarding against opportunism (Golove and Eto, 1996). Furthermore, they argue that such measures can be more effective than relatively small increases in energy prices and can deliver net social benefits (Krause, 1996). Economists are more sceptical of such claims and question whether many existing energy-efficiency policies are in fact cost effective (Joskow and Marron, 1992; Sutherland, 2000).

This is a long-standing debate within energy policy and there is an extensive literature on both the economic potential for energy efficiency (IPPC, 2001) and the win–win opportunities within individual organizations (Lovins and Lovins, 1997). But, owing in part to measurement difficulties, evaluations of the costs and benefits of individual policy instruments are much harder to find. Nevertheless, some evidence suggests that common types of energy-efficiency programme can be cost effective. For example, Levine *et al.*'s (1994) study of US appliance standards found a total benefit-to-cost ratio of more than 2.5 (at a discount rate of 6 per cent), excluding environmental externalities. Eto *et al.*'s (1994) study of US utility lighting programmes found the weighted average cost of electricity saved was \$0.039/kWh. And an assessment of a UK scheme to promote household energy efficiency found a benefit-to-cost ratio of 2.3, considering energy savings alone, or 3.0 if the benefits of improved comfort to low-income households were included (EST, 2001).

If the proponents of energy efficiency are correct, the coexistence of such instruments with an ETS will lead to an improvement in static efficiency and a reduction in overall abatement costs. If they are not correct, the coexistence of such instruments will increase overall costs. Ultimately, this is an empirical question, the answer to which will depend upon the specific market, technology, and policy under examination. It may be expected that such policies will be more effective for sectors such as households and small business, which have both a low energy price elasticity and a substantial economic potential for efficiency improvement. While these sectors are unlikely to participate directly in a carbon ETS, they may be indirectly affected by increases in fuel or electricity prices. But allowance prices would need to be very high to have a significant impact on energy efficiency in these sectors, and the resulting impacts are likely to be regressive without explicit compensation (Johnson *et al.*, 1990).

In addition to overcoming a variety of market failures, many energy-efficiency policies have social objectives. This is particularly true in the UK, where a combination of income inequality and a poor-quality housing stock leads to 4.5m households living in 'fuel poverty'. Efficiency investments in these households are commonly taken up in improved levels of energy service, such as warmer homes, rather than reduced consumption. While the CO_2 benefits of such investments are close to zero, quality of life is improved and savings may be made in areas such as health care. In view of this, the energy-efficiency obligations imposed on UK electricity and gas suppliers require that at least 50 per cent of investment be in low-income households.

Policies to promote energy efficiency may also be justified in terms of employment benefits and the mitigation of non-CO_2 externalities of energy supply such as acid rain, nuclear waste, and the visual impacts of wind farms. But these rationales appear less convincing. While jobs are frequently created in priority sectors, locations, and skill groups, research suggests that the cost effectiveness of this employment creation is relatively low (UKACE, 2000). Similarly, non-CO_2 externalities are increasingly reflected in energy prices through regulations such as the EU Large Combustion Plant Directive.

In sum, the use of additional policies to promote energy efficiency among target groups directly or indirectly affected by a carbon ETS may be justified through their contribution to overcoming a range of market and organizational barriers to energy efficiency, or to delivering non-efficiency objectives such as improved social equity. Taken together, these appear to provide a good case for the continuation of such policies. However, since the costs and benefits of such instruments are contested, each case will need to be judged on its merits.

IV. POLICY INTERACTION BETWEEN THE EU ETS AND UK CLIMATE POLICY

The introduction of the EU ETS into a crowded 'policy space' within each member state is likely to bring these issues of policy interaction to the fore. The issues discussed in the previous three sections are likely to be faced by the majority of member states, together with related issues such as the future of negotiated agreements with energy-intensive industry. The complex, elaborate, and interdependent mix of climate policies developed in the UK provides a particularly rich example of the challenges to be faced. This mix includes:

- a downstream, revenue-neutral energy tax for business and the public sector;
- negotiated agreements for energy-intensive industry which give participants exemption from 80 per cent of the energy tax;
- an elaborate greenhouse-gas trading scheme, developed in collaboration with industry;
- the UK implementation of the EU Integrated Pollution Prevention and Control (IPPC) Directive, which includes provisions on energy efficiency;
- a tradable green credit scheme, in which obligations to purchase renewable electricity are imposed upon electricity suppliers; and
- an obligation upon electricity suppliers to invest in energy efficiency in the household sector, with priority given to low-income households.

Table 1
The Nature of the Potential Interaction between Selected UK Policy Instruments and the EU ETS

Category	Name	Direct	Indirect	Trading
Carbon/energy taxes	Climate Change Levy	✓	✓	
Negotiated agreements	Climate Change Agreements	✓	✓	✓
Emissions trading	UK ETS—cap-and-trade scheme	✓	✓	✓
Emissions trading	UK ETS—project scheme	✓	✓	✓
Industrial pollution control	IPPC Directive	✓	✓	
Support for renewables	Renewables Obligation		✓	✓
Promotion of energy efficiency	Energy Efficiency Commitment		✓	✓

These instruments are closely inter-linked. For example, the negotiated agreements provide exemption from the energy tax as well as forming part of the trading scheme. Similarly, there is trading interaction between the carbon-trading scheme and the instruments for promoting renewable electricity and household energy efficiency. The carbon trading scheme itself includes: a voluntary cap-and-trade scheme, with participation incentivized by direct subsidy; a 'baseline and credit' trading scheme for the negotiated agreement participants; and a scheme for emission-reduction projects. Table 1 lists all these instruments and indicates the nature of their interaction with the EU ETS.

The interactions summarized in Table 1 lead to a series of examples of double regulation and double counting (Sorrell, 2003*c*). Unless resolved, these interactions could lead to substantial economic impacts for the affected groups, and/or threaten the overall environmental integrity of the policy mix. For example:

- organizations eligible for the energy tax will also face electricity price increases as a consequence of the generators participating in the EU ETS. Under a number of simplifying assumptions,[8] an EU ETS allowance price of €7/tCO_2 could increase average electricity prices by some 0.7c€/kWh, which is approximately equivalent to the current level of the tax.

- the UK trading scheme allows a project to be awarded carbon credits for improving downstream electricity efficiency. But this action also 'frees up' allowances held by the electricity generators participating in the EU ETS. If the project credits are subsequently traded into the EU ETS, the cap will be breached and the environmental integrity of the scheme will be undermined.

To avoid these problems, the existing UK policy mix will need to be rationalized. But such changes are likely to create administrative costs for both government and industry. They are also likely to encounter resistance from a range of sources—particularly since none of the above instruments is more than 3 years old.

In the UK, as elsewhere, policy instruments resist replacement even when a more viable alternative is available. This inertia may derive from a number of sources. For example: a legislative framework will have been established which may be difficult to change; regulatory institutions will have been established, or responsibilities assigned to existing institutions; procedures and standards will have been established for functions such as monitoring, reporting, and verification; a network of private organizations will have become involved in implementation; and the target groups themselves will have invested substantial time and money in gaining familiarity with the policy instruments and putting the appropriate procedures in place. All these activities are separate from investment in abatement, but each will cultivate vested interests and encourage

[8] Namely: (*a*) the trading scheme is introduced overnight without companies having the opportunity to change behaviour; (*b*) allowance prices are passed on in full to electricity consumers; and (*c*) the marginal plant on the UK electricity system price is coal fired, leading to an average carbon intensity of delivered electricity of ~1.0 kgCO_2/kWh. In practice, these assumptions are likely to overstate the impact on electricity consumers.

resistance to change. As a result, there is a strong possibility that many of these instruments will continue after the ETS has been introduced, whether or not this is helpful to overall government objectives.

In the case of the UK, the government is reviewing the policy mix in the light of the EU ETS and is anticipated to make some small changes—such as exempting EU ETS participants from a portion of the energy tax. But a major overhaul of the policy mix does not appear likely, at least not before 2008. This means that policy interaction could have a determinate impact on the success of the EU ETS in the UK.

V. SUMMARY AND CONCLUSIONS

Policy interaction has been neglected within the economics literature, but is of central importance in determining the success of individual instruments and of the overall policy mix. This is particularly true within climate policy, where the introduction of a carbon ETS into an already overcrowded 'policy space' poses a particularly difficult challenge.

In theory, cap-and-trade schemes should provide assurance of meeting an overall emissions target at least cost. It follows that, if we assume a perfect economy with no market failures, any instruments which directly or indirectly interact with a carbon ETS will raise overall abatement costs while providing no additional contribution to emission reductions. Hence, once a cap is in place, the rationale for introducing or retaining such instruments must rely upon either their contribution to overcoming market failures other than carbon externalities, or in delivering social objectives other than efficiency. Their contribution to emission reductions can no longer form part of their rationale.

This paper has explored the rationales for the coexistence of a carbon ETS with carbon/energy taxation, support mechanisms for renewable electricity, and non-price instruments to overcome barriers to energy efficiency. In each case, there are possible justifications which fall into one of the above categories and which may have validity. But these are contested and raise both theoretical issues regarding the legitimacy of government intervention and empirical issues regarding the design of individual instruments. In many cases there will be trade-offs between long-term and/or non-efficiency objectives and short-term increases in abatement costs. If the policy mix is to gain legitimacy, these objectives and trade-offs need to be explicit.

The fact that positive combinations between an ETS and other instruments are theoretically possible does not mean that such combinations will result in practice. While the introduction of a carbon ETS should provide an opportunity to rationalize the policy mix, this may not always be taken. Governments may be reluctant to abandon tried and tested instruments in favour of an unfamiliar alternative, and the inertia of existing instruments may make them difficult to displace. The net result may be a mix of overlapping, interacting, and conflicting instruments which lack any overall coherence. In short, a policy mix may easily become a policy mess.

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