Energy Economics 31 (2009) S268–S273

Contents lists available at ScienceDirect

Energy Economics

journal homepage: www.elsevier.com/locate/eneco

THE EU 20/20/2020 targets: An overview of the EMF22 assessment

Christoph Böhringer [a], Thomas F. Rutherford [b], Richard S.J. Tol [c,d,e,*]

[a] Department of Economics, Carl on Ossietzky University, Oldenburg, Germany
[b] Centre for Energy Policy and Economics, ETH, Zurich, Switzerland
[c] Economic and Social Research Institute, Dublin, Ireland
[d] Institute for Environmental Studies, Vrije Universiteit, Amsterdam, The Netherlands
[e] Department of Spatial Economics, Vrije Universiteit, Amsterdam, The Netherlands

ARTICLE INFO

Article history:
Received 16 October 2009
Accepted 16 October 2009
Available online 24 October 2009

JEL classification:
Q54

Keywords:
Climate policy
European Union
Abatement costs
Renewables target
Emission reduction target

ABSTRACT

Three computable general equilibrium models are used to estimate the economic implications of a stylized version of EU climate policy. If implemented at the lowest possible cost, the 20% emissions reduction would lead to a welfare loss of 0.5–2.0% by 2020. Second-best policies increase costs. A policy with two carbon prices (one for the ETS, one for the non-ETS) could increase costs by up to 50%. A policy with 28 carbon prices (one for the ETS, one each for each Member State) could increase costs by another 40%. The renewables standard could raise the costs of emissions reduction by 90%. Overall, the inefficiencies in policy lead to a cost that is 100–125% too high. The models differ greatly in the detail of their results. The ETS/non-ETS split may have a negligible impact on welfare, while the renewables standard may even improve welfare. The models agree, however, that the distortions introduced by total EU package imply a substantial welfare loss over and above the costs needed to meet the climate target. The marginal, total and excess costs reported here are notably higher than those in the impact assessment of the European Commission.

1. Introduction

The European Union is committed to limiting the rise in global average temperature to 2 °C above pre-industrial levels (CEC 2008). It has set ambitious targets for greenhouse gas emissions reduction. At the same time, the EU has adopted equally ambitious targets for the portfolio for energy supply. The EU aims to meet these targets through a range of policy instruments at the Union, Member State and even subnational level. (Tinbergen 1952) cautioned policy makers over the welfare losses that are likely if the number of instruments and targets do not match. This paper provides some estimates of the size of such welfare losses and some insights into the mechanisms behind the inefficiencies.

Roughly, the EU has set the following targets: Greenhouse gas emissions should be reduced to 20% below their 1990 levels by 2020. About half of these emissions – essentially all energy-intensive industries[1] – are to be regulated under the European Trading Scheme (ETS). The target is −21% below 2005 levels.

The EU ETS is the first large-scale, international market for emissions permits (Convery 2009; Convery and Redmond 2007; Ellerman and Buchner 2007). It is a landmark environmental policy. The rationale of emissions trading is straightforward: The direct costs of meeting an exogenous emissions constraint (cap) would be minimized if all the emitters covered by the cap faced the same marginal abatement costs. In this case, there is no arbitrage in trading abatement efforts across emitters. Within a cap-and-trade system, cost-minimizing behaviour by individual emitters leads to a single price. Decentralized market interactions of economic agents assure the collective least-cost attainment of the system's emissions constraint.

The other half of greenhouse gas emissions are currently unregulated at the EU level, but subject to emissions control measures by individual Member States. The average target is −10%, but Member State targets range from a 20% decrease to a 20% increase relative to 2005; the average target is −18% relative to 1990. Achieving these targets is left to the Member States, but these are allowed to trade their non-ETS allocations among one another (Tol 2009b). Three percent emissions reduction may be achieved by investing in CDM-like projects in developing countries. The 3% limit is applied at the Member State level, but the access rights (CDM warrants) are again tradable among governments (Gorecki et al. 2009).

Although the four markets (ETS, non-ETS, CDM, CDM warrants) could jointly lead to a uniform price for all greenhouse gas emissions in the European Union, this is not guaranteed as it would be by a comprehensive market (Tol 2009a). Besides, the non-ETS and CDM warrant markets are untested, while the CDM market is less than perfect (Michaelowa and Jotzo 2005). Cost-effectiveness at the EU

* Corresponding author. ESRI, Whitaker Square, Sir John Rogerson's Quay, Dublin 2, Ireland.
E-mail address: richard.tol@esri.ie (R.S.J. Tol).

[1] The petrochemical, aluminium, and aviation industries will be included from 2012 onwards.

0140-9883/$ – see front matter © 2009 Elsevier B.V. All rights reserved.
doi:10.1016/j.eneco.2009.10.010

level would require cost-effective implementation of non-ETS emissions reduction at the Member State level (see below). The costs of meeting the EU emissions target raise a further concern.

The second headline target is a 20% penetration of renewable energy by 2020. There are targets for every Member State, but these obligations are also tradable (Bertoldi and Huld 2006). Some of the Member States have adopted separate targets on the penetration of renewable energy in specific markets, such as transport and residential heating.

There is also an EU-wide *aspiration* to improve energy efficiency by at least 20% between 2005 and 2020, and perhaps there will be a market for this too (Oikonomou et al. 2008).

From the perspective of climate policy, these additional targets create excess cost. If targets for renewable energy and energy efficiency become binding, they produce an outcome different from the cost-effective solution generated by comprehensive emissions trading. This implies additional costs (Boehringer et al. 2008). The relative contribution of renewables and energy efficiency to emissions reduction should be determined by the markets and not by bureaucrats.[2]

Besides the various markets that operate at EU level, there are other instruments as well. Chief among these is the fuel efficiency target for passenger cars (European Parliament and Council of the European Union 2009), although one could also argue that this is a bilinear tax. Symbolically, incandescent light bulbs will be banned (Ecodesign Regulatory Committee 2008). The European Parliament has also considered other options, including banning patio heaters (European Parliament 2008) and the abolishment of daylight saving time (Doyle 2009). The EU has 27 Member States, and many of these have a wide variety of additional measures, including carbon taxes, appliance subsidies, tax breaks for bicycle owners, standards for tyre pressure, tests for efficient driving, and many others. At the same time, a number of Member States continue to support fossil fuels, car transport, agriculture, and other activities that emit disproportionate amounts of greenhouse gases.

Against this background, the primary objective of the EMF22 model analysis on EU climate policies is to provide quantitative estimates of the potential excess costs from restricted trading and overlapping regulation. The economic models used in this study cannot possibly reflect the true complexity of climate and energy policy in the European Union. Instead, we designed a set of stylized scenarios that (1) highlight the main inefficiencies and (2) attribute these to the various elements of the regulation.

The three models included in this analysis are multi-regional, multi-sector general equilibrium models. A key advantage of these models is that they provide a comprehensive representation of price-dependent market interactions based on microeconomic theory. The simultaneous explanation of the origin and spending of agents' incomes makes it possible to address both economy-wide efficiency as well as distributional impacts of policy interference. Policy measures in open economies can influence both domestic markets and international prices via changes in exports and imports. The changes in international prices, i.e., the terms of trade, imply secondary effects which can significantly affect the welfare impacts of the primary domestic policy.

The paper proceeds as follows. Section 2 details the study design. Section 3 discusses the shared results of the three models included in the study. Section 4 reviews the additional results from the individual papers. Section 5 concludes.

2. Study design

To a first approximation, the costs of emissions reduction are determined by two factors (Weyant 1993): the distance to the target and steepness of the abatement cost curve.[3] Costs increase as the cost curve steepens, or as the difference grows between emissions with policy and emissions without policy grows. The EU target for 2020 is set relative to 1990 emissions. The distance to target is thus given by the growth rate of emissions in the absence of additional policy. This is therefore an important variable in any model comparison.

In a textbook analysis, climate policy is least cost if all emitters face the same marginal costs. Some departures from this paradigm are possible when second-best effects are at work (Babiker et al. 2003; Baumol and Bradford 1970; Goulder et al. 1997; Parry et al. 1999; Parry 2000), but it makes sense to begin with a model in which excess costs of suboptimal policy are determined by the differentials in marginal costs. The gains from trade (in emissions permits) are larger if there are greater opportunities for arbitrage, that is, if the price differences before trade are larger (Montgomery 1972). Differences in marginal abatement costs between and within sectors, and between and within countries would occur if the targets set are incommensurate with cost curves and growth rates. This is likely in the European Union, as relatively uniform emissions reduction obligations were imposed on sectors and countries with very different dynamics. Again, a model comparison should consider differences in the baseline. Furthermore, it is important to consider the sectoral and regional resolution, as models typically assume homogeneity within the sectors and regions. A highly aggregate model thus has fewer options for arbitrage and hence lower gains from a uniform carbon price.

EU climate policy combines targets for emissions with targets for renewables. Compared with a policy on emissions only, this has two implications. First, the additional target is a supplementary constraint, and this can only increase total costs. Second, as the renewables target reduces greenhouse gas emissions, the incentive needed to meet the emissions target is lower. That is, the renewables target raises the total costs of policy, but reduces the price of carbon.

Using these priors, we designed the following scenarios:

1. No additional policy
2. 20% emissions reduction
   a. Uniform carbon price
   b. Uniform carbon price in ETS and non-ETS
   c. Uniform carbon price in ETS; non-ETS carbon prices vary by Member State[4]
3. 20% emissions reduction and a lower bound of 20% on renewables penetration
   a. Uniform carbon price
   b. Uniform carbon price in ETS and non-ETS
   c. Uniform carbon price in ETS; non-ETS carbon prices vary by Member State[5].

3. Common results

We first consider the first best policy, with a uniform price of carbon for all sources and countries. Fig. 1 shows the EU-wide emissions reduction target of ETS and non-ETS emissions according to the three models. The nominal targets (relative to the base year) are identical, but because the different models use different growth rates,

[2] Admittedly, there could be other objectives behind renewable energy quotas and energy efficiency targets. But here too these are imperfect proxies for energy security or strategic technological innovation.

[3] The costs of emissions reduction are moderated by three complications. First, climate policy may interact with other policies. Second, the revenues of climate policy may be used to alter prior distortions. Third, implementation of climate policy would change the competitive position of the economy.

[4] Note that the PACE model cannot run this scenario because of its crude regional aggregation.

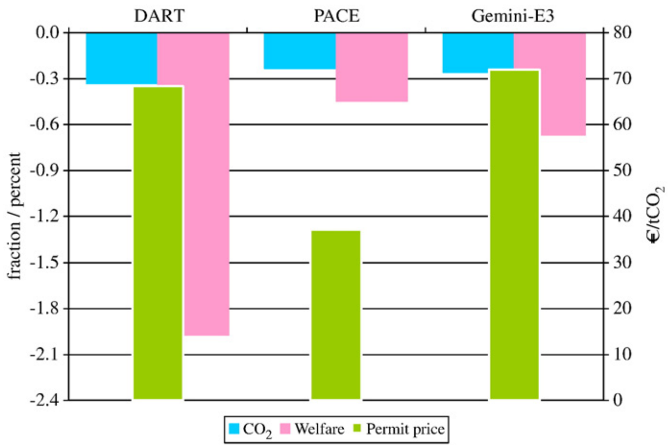

Fig. 1. The 2020 emissions reduction target (from the baseline scenario) for ETS plus non-ETS emissions, the price of carbon dioxide emissions permits (right axis), and the loss of welfare (from the baseline scenario) according to the three different models.

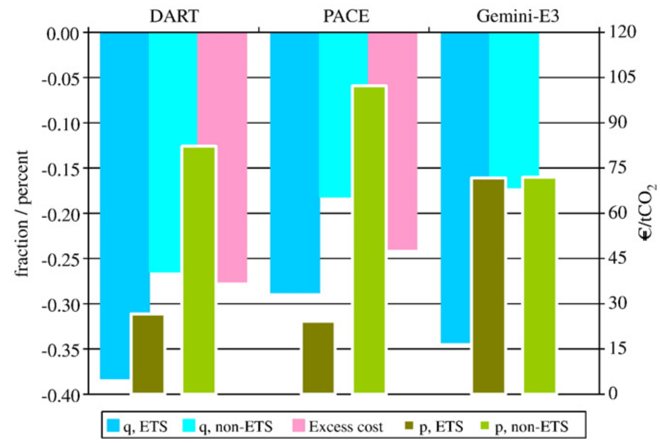

Fig. 2. The 2020 emissions reduction target (from the baseline scenario) for ETS and non-ETS emissions, the price of ETS and non-ETS carbon dioxide emissions permits (right axis), and the excess loss of consumption (from the first best scenario) according to the three different models.

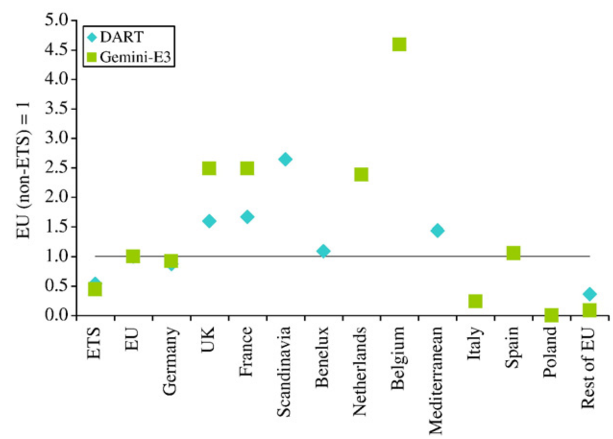

Fig. 3. The 2020 price of non-ETS emissions permits in the countries and regions of the European Union according to two different models; for reference, the ETS price is shown too.

the actual targets differ substantially. Emissions grow slowest in PACE, so that emissions have to be cut by 23%. Emissions grow fastest in DART, so that emissions reduction is 33%. Gemini-E3 is in between, with a target of 26%. This compares to a 19% cut according to the impact assessment of the European Commission (Capros et al. 2008).

Fig. 1 also shows the EU-wide loss of welfare.[5] This is two orders of magnitude smaller than the emissions abatement. The costs follow the same pattern as the targets. Costs are lowest in PACE (0.45%) and highest in DART (1.98%), with Gemini-E3 in between (0.67%). The European Commission reports a cost of 0.50% (Capros et al. 2008).[6] When implemented at the lowest possible cost, climate policy is not expensive.

Finally, Fig. 1 shows the price of carbon dioxide emissions permits. Here the pattern is broken. PACE has the lowest price at €36/tCO_2, but the price in Gemini-E3 (€72/tCO_2) is slightly higher than in DART (€68/tCO_2). This suggests that the implicit emissions reduction cost curve in Gemini-E3 is steeper than in the other two models. According to the European Commission, the price of carbon would be €49/tCO_2 in 2020 (Capros et al. 2008).

We next turn to a single distortion in policy: The price of carbon differs between the ETS and the non-ETS emissions, but does not differ between Member States.

Fig. 2 shows ETS and non-ETS emissions separately. In PACE, ETS emissions have to be cut by 29% in the ETS and 18% in the non-ETS. In DART, ETS emissions have to be cut by 38% and non-ETS emissions by 26%. In Gemini-E3, the ETS reduction is 34% and non-ETS abatement 17%. This compares with a 26% cut in the ETS and a 12% cut outside the ETS according to the European Commission (Capros et al. 2008). In all three models as well as in the EU impact assessment, ETS emissions reduction is more stringent than non-ETS emissions reduction. All models assume that ETS emissions grow faster (in the absence of policy) than do non-ETS emissions.

Fig. 2 also shows the permit prices in the ETS and non-ETS. In Gemini-E3, the price is almost equal[7] (and thus equal to the price in the first-best policy; cf. Fig. 1). That is, the initial allocation of ETS and non-ETS abatement obligations is almost cost-effective; there are few opportunities for arbitrage. The other two models disagree. DART and PACE show a non-ETS price that is substantially higher than the ETS price even though the non-ETS target is less stringent. That is, DART and PACE have an implicit abatement cost function that is much steeper for non-ETS than for ETS emissions.

Finally, Fig. 2 shows the excess costs, that is, the welfare loss on top of the costs of the first-best policy as shown in Fig. 1. This is 0.28% in DART and 0.24% in PACE. In DART, the policy with two carbon prices is 14% more expensive than the policy with one price, while in PACE the cost increase is 53%. In Gemini-E3, the initial allocation between ETS and non-ETS is almost cost-effective, so the effect of separating the two markets is very small.[8]

We next turn to the policy scenario with differentiated prices for the non-ETS. Only two of the three models could run this scenario. Fig. 3 shows the 2020 permit price relative to the respective EU average. The models agree that prices would vary widely across the EU, the price in Germany would be close to the EU average, the price in the UK

[5] Hicksian Equivalent Variation in DART and PACE; Hicksian Compensating Variation in Gemini-E3.

[6] Note that this is given as "total energy system cost" as a fraction of Gross Domestic Product, which is an incomplete welfare measure; GDP is otherwise unspecified, but if scenario-specific GDP is used, the welfare measurement is inconsistent as well as incomplete.

[7] The price difference is 0.15 €/tCO_2.

[8] In fact, the welfare impact is positive in the EU. The two scenarios cannot be compared. The first scenario was simulated with a uniform carbon tax on all emissions. The second scenario was simulated with a permit market for non-ETS emissions. At the global level, welfare falls if ETS and non-ETS are regulated with different prices (as one would expect). This result does not carry over to the EU because carbon taxes and permits have a different impact on the terms of trade. As a result, the two scenarios are incomparable, and the welfare gain found is meaningless.

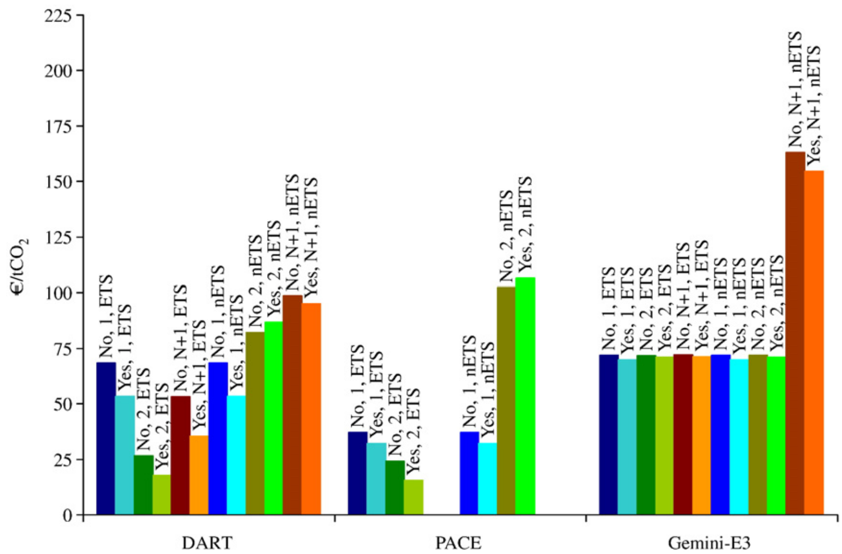

Fig. 4. The 2020 price emissions permits in the European Union according to three different models; no = no target for the share of renewables in energy supply; yes = target for the share of renewables in energy supply; 1 = uniform price for ETS and non-ETS emissions; 2 = separate prices for ETS and non-ETS emissions; N+1 one Europe-wide price for ETS emissions, different prices for non-ETS emissions in different Member States; ETS = ETS price; nETS = non-ETS price.

and France would be above average, and the price in Eastern Europe would be below average. For the Benelux, DART finds that the non-ETS prices are close to the EU average, but Gemini-E3 finds large differences; in Belgium, the price would be 359% above average. As the two models have different regionalizations, one cannot read too much from Fig. 3, but it is clear that the distribution of costs between the Member States is particularly uncertain.

In Gemini-E3, the excess cost of differentiated non-ETS prices is 0.25% of welfare in 2020, 39% higher than in the case of a single non-ETS price; in DART, the excess cost is 0.34%, 15% higher than for a single non-ETS price. At the same time, in Gemini-E3, the coefficient of variation of the non-ETS carbon prices is 0.96, while in DART this is 0.52. Based on first principles, one would expect that greater price variation implies greater excess cost. Again, the explanation lies in the curvature of the implicit abatement cost curve. As noted above, the cost curve in Gemini-E3 is rather steep, so that marginal costs rapidly escalate while total costs rise much more slowly.

Finally, we consider the effect of the renewable penetration standard in the scenario with a uniform price for all emissions, and in the scenario with ETS and non-ETS split into two markets. The renewables target in itself reduces carbon dioxide emissions. Climate policy would therefore need to be less stringent, and the price of carbon duly falls in all three models. In the scenario with a single permit market for all emissions, the carbon price falls by 3% in Gemini-E3 and by 22% in DART. PACE is in between with a price drop of 13% (see Fig. 4). That is, renewables play a greater role in climate policy in Gemini-E3, while DART instead seeks to reduce emissions by energy efficiency and switching from coal to gas. The European Commission finds a price drop of 19% (Capros et al. 2008).

If the market is split between ETS and non-ETS, different results emerge. The renewables standard reduces the ETS price to a greater extent: by 33% in DART and by 37% in PACE, but by only 1% in Gemini-E3. In DART and PACE, the non-ETS price rises by 5% and 4%, respectively; but it falls by 1% in Gemini-E3. In Gemini-E3, renewables are deployed in the non-ETS sector. The same result is found by the European Commission (Capros et al. 2008). In DART and PACE, the renewables standard hardly affects the non-ETS sectors. Therefore, in these models, the non-ETS permit price changes are the result of second-order effects. Particularly, the renewables standard drives down the price of fossil fuels; this needs to be compensated by a higher carbon price in the non-ETS sector.

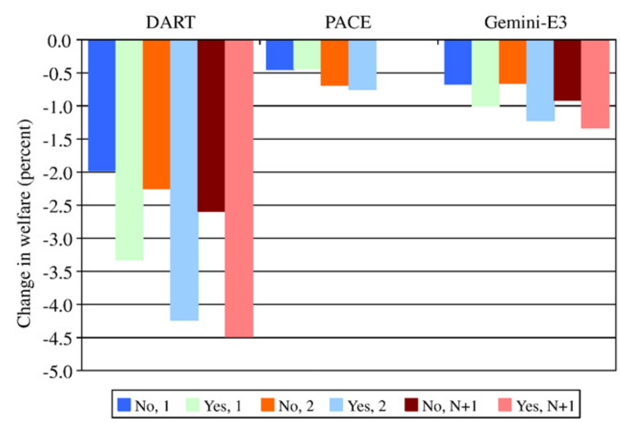

Fig. 5. The change in welfare in the European Union in 2020 according to three different models; no = no target for the share of renewables in energy supply; yes = target for the share of renewables in energy supply; 1 = uniform price for ETS and non-ETS emissions; 2 = separate prices for ETS and non-ETS emissions; $N+1$ = one Europe-wide price for ETS emissions, different prices for non-ETS emissions in different Member States.

If the non-ETS market is also split by Member State, the ETS price rises by 33% in DART and by 1% in Gemini-E3.[9] The non-ETS price rises by 3% and 5% in DART and Gemini-E3, respectively.

Fig. 5 shows the implications of the renewables standard. The renewables standard exerts a downward pressure on the carbon cap, thereby lowering carbon prices (cf. Fig. 4) and distorting the cost-effective contribution of emissions reduction from different channels (fuel switching, energy efficiency improvements, energy savings) from a first-best perspective. For uniform emissions pricing, Gemini-E3 finds that the renewables constraint imposes a small cost (0.33%), whereas DART find that the lower bound on renewables induces a welfare loss of 1.35%. PACE reports a slight welfare increase (0.01%) because prior distortions on the energy markets call for a more favourable treatment of renewable energy. If the market is split between ETS and non-ETS, the welfare loss is greater — that is, the inefficiencies interact with one another. The excess costs go up by 0.64% in DART and by 0.23% in Gemini-E3, while PACE swings from a 0.01% benefit to a 0.06% loss. If the non-ETS market is split between

[9] This scenario was not run by PACE.

Member States, DART shows an excess cost of 1.90% of consumption and Gemini-E3 a smaller 0.42%.

Overall, PACE finds that the inefficiencies in policy imply that welfare losses are 67% above those in the cost-effective implementation. DART finds excess costs of 127%. Gemini-E3 is in between with 98%. In contrast, the European Commission reports excess costs of only 3% (Capros et al. 2008).

4. Other findings

Using the Gemini-E3 model Bernard and Vielle (2009-this issue) focus on the implications of climate policy for international trade. They decompose the welfare impact (Hicksian Equivalent Variation) into the terms of trade effect and a residual, which they refer to as the deadweight loss of taxation. The terms of trade effect is positive for Western European countries, alleviating the costs of emissions abatement. For Belgium and the Netherlands, the terms of trade gains are so large that the overall impact on welfare is positive. The USA and Japan also gain in terms of trade, but the eastern part of the European Union, the countries of the former Soviet Union, and the rest of the world (Latin America, Africa, and Asia) lose.

Bernard and Vielle introduce a new way to define carbon leakage. Previously, carbon leakage was defined as the increase in greenhouse gas emissions in non-abating countries relative to the case without climate policy. Bernard and Vielle call this “gross carbon leakage”. They define “net carbon leakage” as the increase in greenhouse gas emissions in non-abating countries relative to the case with climate policy but without carbon leakage. Both gross and net leakage are defined against a counterfactual. Gross leakage confounds all general equilibrium effects of abatement in one group of countries on the rest of the world. Net leakage, on the other hand, is limited to substitution effects only, that is, the relocation of production from abating to non-abating countries. Net leakage thus seems to be the more appropriate definition. Bernard and Vielle show that gross leakage is more than twice as large as net leakage.

Finally, Bernard and Vielle analyze a policy that is close to the proposed EU directive. This includes limited CDM, but also restricted trade in non-ETS allowances. The trade restrictions bring a welfare loss, but this is more than offset by cheap (€14/tCO_2) emissions reductions outside the EU. The net welfare gain is 0.06%. The CDM offsets would be primarily used in the non-ETS sectors, and the average price falls from €155/tCO_2 to €74/tCO_2, very close to the ETS price of €71/tCO_2.

Using the DART model, (Kretschmer et al., 2009-this issue) focus on biofuels. Besides its targets for greenhouse gas emissions and the share of renewables in overall energy supply, the European Union also aims for a 10% market share of biofuels in transport by 2020. The analysis indicates that the biofuels target is binding. Neither the emissions target nor the overall renewables target would induce the required uptake of biofuels in transport. There are cheaper options to reduce emissions, and there are cheaper ways to use renewable energy. As a result, one would expect that imposing the biofuels target implies a loss in welfare. This is true in the scenario in which the biofuels target is imposed on each Member State, but not if the biofuels target is EU-wide. This counterintuitive result is explained by the fact that DART is a *general* equilibrium model with many distortions in the base case. Specifically, biofuels crowd out food production. Indeed, DART reports a substantial increase in food prices. Because EU food production is both subsidized and shielded from the world market, a forced reduction of agricultural production is a benefit. This benefit is larger, according to DART, than the welfare loss induced by the biofuels target.

Using the PACE model, (Böhringer et al., 2009-this issue) focus on the impact of alternative baseline scenarios on the macroeconomic costs of EU climate policy. The baseline projections not only determine the magnitude of the effective abatement requirement but also the ease of emissions abatement. The critical importance of baseline projections for the magnitude of the costs of complying with emissions reduction targets is widely ignored in the policy debate. Böhringer et al. show that alternative baseline variants explain drastic differences in compliance cost – in their case by a factor 4 to 6. They also show that the excess cost of deviating from uniform pricing depends on the baseline scenario. Böhringer et al. argue that uniform emissions pricing may not be the preferred policy in the presence of initial tax distortions and international market power. They show that deviation from uniform emissions pricing across ETS and non-ETS sectors can be welfare-improving as long as the increase in direct abatement costs due to differential emissions pricing is more than offset through potential terms-of-trade gains or the amelioration of initial tax distortions.

5. Discussion and conclusion

In this paper, we present simulation results from three computable general equilibrium models on the economic implications of EU climate policies. Obviously, these models and our stylized policy scenarios cannot possibly reflect the true complexity of climate and energy policy in the European Union. However, they do provide important insights into key determinants of climate policy costs, and serve as a post-hoc check on policy choices and the impact assessment of the European Commission.

The following results emerge. If implemented at the lowest possible cost, the 20% emissions reduction for 2020 would lead to a welfare loss of 0.5–2.0%. The policy period is 2013–2020, so the pessimistic estimate suggests a loss of 1 year of growth in eight. It should be noted that these are results from comparative-static computable general equilibrium models. That is, transitional frictions are disregarded, and the effect of emissions reduction policy on economic growth is not treated thoroughly if at all.

The second result is that second-best policies increase costs. A policy with two carbon prices (one for the ETS, one for the non-ETS) rather than one could increase costs by 50%. A policy with 28 carbon prices (one for the ETS, one each for each Member State), could increase costs by another 40%. The renewables standard could raise the costs of emissions reduction by 90%. Overall, the inefficiencies in policy lead to a cost that is 100–125% too high.

The third result is that the models differ greatly in the detail of their results. The previous paragraph cites the most pessimistic findings. However, we also find that, according to some model results, the ETS/non-ETS split may have a negligible impact on welfare, and that the renewables standard may even improve welfare. Splitting the non-ETS target of the EU into targets for each Member State is costly in both models that considered this. However, while the models disagree whether it is the renewables target or the ETS/non-ETS split that causes excess costs, the models agree that the two together imply a substantial and unnecessary welfare loss.

Comparing the above results to those of the impact assessment of the European Commission (Capros et al. 2008), we find that the marginal, total and excess costs reported here are higher. In some cases, the numbers of the European Commission are in the lower end of the range found here; in other cases, the European Commission's are below the lowest numbers found here. This suggests that there would be scope for closer scrutiny of the European Commission's impact assessments.

The above results should be treated with caution. The numbers are neither accurate nor precise. They are ballpark estimates. What really matters are the insights: Climate policy need not cost a lot, but imperfect implementation implies excess costs. The excess costs are substantial relative to the costs of the first-best policy, but modest in absolute terms.

There is scope for further research. We review the results of three computable general equilibrium models, which are good for certain types of analyses. One aspect that deserves further attention is the interactions between climate policy and pre-existing tax and trade distortions, and the interactions between climate policy and market power. First-best policy such as a uniform carbon tax may well be suboptimal, but the size of the welfare loss is not clear at present, nor is it known how policies should deviate from the first-best prescription. Furthermore, policy makers occasionally use second-best arguments to deviate from first-best prescriptions, but in a haphazard way. The welfare loss of using the wrong second-best policy has yet to be estimated. Another open question concerns the split between ETS and non-ETS emissions. The models used here and elsewhere make this distinction on a sectoral basis but the actual distinction is based on the size of the installation. This means that, in reality, small and large companies *in the same sector* are regulated differently. Typically, tax differentiation within a sector (that is, between companies that compete on input and output markets) is worse than tax differentiation between sectors (that is, between companies that compete on input markets only).

Other types of economics models are needed to analyze other policy questions. The year 2020 is sufficiently close that the business cycle matters. The same is true for vintages of capital and durable consumption goods. In a short policy period such as 2013–2020, the construction period of infrastructure would have an impact too. In the long run, the economic impact of climate policy is driven by its effects on the growth rate of the economy, while the environmental impact is driven by its effects on the rate and direction of technological progress (Baker et al. 2006; Clarke et al. 2008; Gillingham et al. 2008; Pizer and Popp 2008). None of these issues is well understood. The current papers do not shed much light on them either, so we defer them to future research.

Acknowledgements

David Goldblatt edited the text. Alain Bernard, Bettina Kretschmer, Andreas Loeschel, Ulf Moslener, Daiju Narita, Sonja Peterson and Marc Vielle did the modelling work that supports this paper. Many EMF participants had constructive comments, but Stefan Boeters and Tom van Ierland stand out. Leon Clarke and John Weyant masterly guided the EMF22 project.

References

Babiker, M.H., Metcalf, G.E., Reilly, J.M., 2003. Tax distortions and global climate policy. Journal of Environmental Economics and Management 46, 269–287.

Baker, E., Clarke, L.E., Weyant, J.P., 2006. Optimal technology R&D in the face of climate uncertainty. Climatic Change 78, 157–179.

Baumol, W.J., Bradford, D.F., 1970. Optimal departures from marginal cost pricing. American Economic Review 60 (3), 265–283.

Bernard, A., Vielle, M., 2009. Assessment of European Union transition scenarios with a special focus on the issue of carbon leakage. Energy Economics 31, S274–S284 (this issue).

Bertoldi, P., Huld, T., 2006. Tradable certificates for renewable electricity and energy savings. Energy Policy 34 (2), 212–222.

Boehringer, C., Koschel, H., Moslener, U., 2008. Efficiency losses from overlapping regulation of EU carbon emissions. Journal of Regulatory Economics 33 (3), 299–317.

Boehringer, C., Löschel, L., Moslener, U., Rutherford, T.F., 2009. EU Climate policy up to 2020: an economic impact assessment. Energy Economics 31, S295–S305 (this issue).

Capros, P., Mantzos, L., Papandreou, V., Tasios, N., 2008. Model-based analysis of the 2008 EU policy package on climate change and renewables, E3M Lab. National Technical University, Athens.

CEC, 2008. Proposal for a decision on the European Parliament and of the Council on the effort of member states to reduce their greenhouse gas emissions to meet the community's greenhouse gas emission reduction commitments up to 2020 COM (2008) 17 final. Commission of the Euopean Communities, Brussels.

Clarke, L.E., Weyant, J.P., Edmonds, J., 2008. On the sources of technological change: what do the models assume. Energy Economics 30 (2), 409–424.

Convery, F.J., 2009. Origins and development of the EU ETS. Environmental and Resource Economics 43 (3), 391–412.

Convery, F.J., Redmond, L., 2007. Market and price developments in the European Union emissions trading scheme. Review of Environmental Economics and Policy 1 (1), 88–111.

Doyle, A. (2009), *Extending Summer Time*, Parliamentary Questions E-2223/09.

Ecodesign Regulatory Committee, 2008. Member states approve the phasing-out of incandescent bulbs by 2020 IP/08/1909. Council of the European Union, Brussels.

Ellerman, A.D., Buchner, B.K., 2007. The European Union Emissions trading scheme: origins, allocation, and early results. Review of Environmental Economics and Policy 1 (1), 66–87.

European Parliament and Council of the European Union, 2009. Regulation (EC) No 443/2009 of the European Parliament and the of the Council of 23 April 2009 setting emission performance standards for new passenger cars as part of the communities' integrated approach to reduce CO_2 emissions from light-duty vehicles. Official Journal of the European Union L140, 1–15.

Gillingham, K., Newell, R.G., Pizer, W.A., 2008. Modeling endogenous technological change for climate policy analysis. Energy Economics 30 (6), 2734–2753.

Gorecki, P.K., Lyons, S., Tol, R.S.J., 2009. EU Climate Change Policy 2013–2020: Using the Clean Development Mechanism more effectively, Working Paper 299. Economic and Social Research Institute, Dublin.

Goulder, L.H., Parry, I.W.H., Burtraw, D., 1997. Revenue-raising versus other approaches to environmental protection: the critical significance of preexisting tax distortions. RAND Journal of Economics 28 (4), 708–731.

Kretschmer, B., Narita, D., Peterson, S., 2009. The economic effects of the EU biofuel target. Energy Economics 31, S285–S294 (this issue).

Michaelowa, A., Jotzo, F., 2005. Transaction costs, institutional rigidities and the size of the clean development mechanism. Energy Policy 33, 511–523.

Montgomery, W.D., 1972. Markets in licences and efficient pollution control programs. Journal of Economic Theory 5, 395–418.

Oikonomou, V., Jepma, C., Becchis, F., Russolillo, D., 2008. White Certificates for energy efficiency improvement with energy taxes: a theoretical economic model. Energy Economics 30 (6), 3044–3062.

European Parliament, 2008. *Action Plan for Energy Efficiency: Realising the Potential* INI/2007/2106. European Parliament, Brussels.

Parry, I.W.H., 2000. Tax deductions, environmental policy, and the “double dividend” hypothesis. Journal of Environmental Economics and Management 39, 67–96.

Parry, I.W.H., Williams III, R.C., Goulder, L.H., 1999. When can carbon abatement policies increase welfare? The fundamental role of distorted factor markets. Journal of Environmental Economics and Management 37, 52–84.

Pizer, W.A., Popp, D., 2008. Endogenizing technological change: Matching empirical evidence to modeling needs. Energy Economics 30 (6), 2754–2770.

Tinbergen, J., 1952. *On the Theory of Economic Policy*. North Holland, Amsterdam.

Tol, R.S.J., 2009a. Intra- and extra-union flexibility in meeting the European Union's emission reduction targets. Energy Policy 37, 4329–4336.

Tol, R.S.J., 2009b. Intra-union flexibility of non-ETS emission reduction obligations in the European Union. Energy Policy 37 (5), 1745–1752.

Weyant, J.P., 1993. Costs of reducing global carbon emissions. Journal of Economic Perspectives 7 (4), 27–46.