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Global, Regional, and National Costs and Ancillary Benefits of Mitigation

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EXECUTIVE SUMMARY

The United Nations Framework Convention on Climate Change (UNFCCC) has as its ultimate goal the "stabilization of greenhouse gas concentrations in the atmosphere at a level that will prevent dangerous anthropogenic interference with the climate system." Whereas mitigation costs play only a secondary role in establishing the target, they play a more important role in determining how the target is to be achieved. UNFCCC states that "policies and measures to deal with climate change should be cost-effective so as to ensure global benefits at the lowest possible costs." This chapter examines the literature on the costs of greenhouse gas mitigation policies at the national, regional, and global levels. The net welfare gains or losses are reported, including (when available) the ancillary benefits of mitigation policies. These studies employ the full range of analytical tools described in Chapter 7, from the technologically rich bottom-up models to more aggregate top-down models, which link the energy sector to the rest of the economy.

Models can also be distinguished through their level of geographical disaggregation. Global models, which divide the world into a limited number of regions, can provide important insights with regard to international emissions trade, capital flows, trade patterns, and the implications of alternative international regimes regarding contributions to mitigation by various regions of the globe. National models are more appropriate for examining the effectiveness of alternative fiscal policies in offsetting mitigation costs, the short-term effects of macro shocks on employment and inflation, and the implications of domestic burden-sharing rules for various sectors of the economy.

To cope with their wide range of diversity, the studies are grouped into three categories. The first two focus on the near-to-medium term. In one of these, the focus is exclusively on domestic policies. In the other, the domestic/international interface is explored. The third category focuses on the long-term goals of climate policy and explores cost-effective implementation strategies. That is, what is the least-cost emission reduction pathway for accomplishing a prescribed goal? The major conclusions are summarized below.

For any class of models, the emissions baseline is critically important in determining mitigation costs. It defines the size of the reduction required for meeting a particular target. The growth rate in carbon dioxide (CO_2) emissions is determined by:

- growth rate in gross domestic product (GDP);
- decline rate of energy use per unit of output, which depends on structural change in the economy and on technological development; and
- decline rate of CO_2 emissions per unit of energy use.

Much of the difference in cost projections can be explained by differences in these key variables.

Economic studies vary widely in their estimate of mitigation costs (both across and within countries). These differences can be traced to assumptions about economic growth, the cost and availability of existing and new technologies (both on the supply and demand side of the energy sector), resource endowments, the extent of "no regrets" options and the choice of policy instruments.

Virtually all analysts agree on the existence of "no regrets" options. Such options are typically assumed to be included in the reference (no policy) scenario by economic modellers. Even so, the overwhelming majority of emission baselines show that emissions continue to rise well into the future. This suggests that zero cost options are insufficient to reduce emissions in the absence of policy intervention.

Mitigation costs to meet a prescribed target will be lower if the tax revenues (or revenues from auctioned permits) are used to reduce existing distortionary taxes (the so-called "double dividend"). The preferred policy depends on the existing tax structure. Most European studies find that cutting payroll taxes is more efficient than other types of recycling. A significant number of these studies conclude that, within some range of abatement targets, the net costs of mitigation policies can be close to zero and even slightly negative. Conversely, in the USA, studies suggest that reducing taxes on capital is more efficient, but few models report negative costs.

Policies aimed at mitigating greenhouse gases can have positive and negative side effects (or ancillary benefits and costs, not taking into account benefits of avoided climate change) on society. Although this report overall emphasizes co-benefits of climate policies with other policies (to reflect the reality in many regions that measures are taken with multiple objectives rather than climate mitigation alone), the literature that focuses on climate mitigation uses the term "ancillary benefits" of specific climate mitigation measures. In spite of recent progress in methods development, it remains very challenging to develop quantitative estimates of the ancillary effects, benefits and costs of GHG mitigation policies. Despite these difficulties, in the short term, ancillary benefits of GHG policies under some circumstances can be a significant fraction of private (direct) mitigation costs. In some cases the magnitude of ancillary benefits of mitigation may be comparable to the costs of the mitigating measures, adding to the no regrets potential. The exact magnitude, scale and scope of these ancillary benefits and costs will vary with local geographical and baseline conditions. In some circumstances, where baseline conditions involve relatively low carbon emissions and population density, benefits may be low. For the studies reviewed here, the biggest share of the ancillary benefits is related to public health.

Mitigation costs are highly dependent on assumptions about trade in emission permits. Cost estimates are lowest when there would be full global trading. That is, when reductions are made where it is least expensive to do so regardless of their geographical location. Costs increase as the size of the emissions market contracts. In the case of Annex B trading only, the availability of excess assigned amount units in Russia and Ukraine can be critical in lowering the overall mitigation costs. Carbon trade provides some means for hedging against uncertainties regarding emissions' baselines and abatement costs. It also reduces the consequences of an inequitable allocation of assigned amounts.

It has long been recognized that international trade in emission quota can reduce mitigation costs. This will occur when countries with high domestic marginal abatement costs purchase emission quotas from countries with low marginal abatement costs. This is often referred to as "where flexibility". That is, allowing reductions to take place where it is cheapest to do so regardless of geographical location. It is important to note that where the reductions take place is independent upon who pays for the reductions. The chapter discusses the cost reductions from emission trading for Annex B and full global trading compared to a no-trading case. All of the models show significant gains as the size of the trading market is expanded. The difference among models is due in part to differences in their baseline, the cost and availability of low-cost substitutes on both the supply and demand sides of the energy sector, and the treatment of short-term macro shocks. In general, all calculated gross costs for the non-trading case are below 2% of GDP (which is assumed to have increased significantly in the period considered) and in most cases below 1%. Annex-B trading would generally decrease these costs to well below 1 % of GDP for OECD regions. The extent to which domestic policies relying on revenue recycling instruments can lower these figures is conditional upon the articulation of these policies and the design of trading systems.

Emissions constraints in Annex I countries are likely to have so-called "spillover" effects on non-Annex B countries. For example, Annex I emissions reductions result in lower oil demand, which in turn leads to a decline in the international price of oil. As a response, non-Annex I countries may increase their oil imports and emit more than they would otherwise. Oil-importing non-Annex I countries may benefit, whereas oil exporters may experience a decline in revenue.

A second example of spillover effects involves the location of carbon-intensive industries. A constraint on Annex I emissions reduces their competitiveness in the international marketplace. Recent studies suggest that there will be some industrial relocation abroad, with non-Annex I countries benefitting at the expense of Annex I countries. However, non-Annex I countries may be adversely affected by the decline in exports likely to accompany a decrease in economic activity in Annex I countries.

The cost estimates of stabilizing atmospheric CO_2 concentrations depend upon the concentration stabilization target, the emissions pathway to stabilization and the baseline scenario assumed. Unfortunately, the target is likely to remain the subject of intense scientific and political debate for some time. What is needed is a decision-making approach that explicitly incorporates this type of uncertainty and its sequential resolution over time. The desirable amount of hedging in the near term depends upon one's assessment of the stakes, the odds, and the costs of policy intervention. The risk premium–the amount that society is willing to pay to reduce risk–is ultimately a political decision that differs among countries.

The concentration of CO_2 in the atmosphere is determined more by cumulative rather than year-by-year emissions. A number of studies suggest that the choice of emissions pathway can be as important as the target itself in determining overall mitigation costs. A gradual near-term transition from the world's present energy system minimizes premature retirement of existing capital stock, provides time for technology development, and avoids premature lock-in to early versions of rapidly developing low-emission technology. On the other hand, more aggressive near-term action would decrease environmental risks associated with rapid climatic changes, stimulate more rapid deployment of existing low-emission technologies, provide strong near-term incentives to future technological changes that may help to avoid lock-in to carbon intensive technologies, and allow for later tightening of targets should that be deemed desirable in light of evolving scientific understanding.

8.1 Introduction

8.1.1 Summary of Mitigation Cost Analysis in the Second Assessment Report

Chapters 8 and 9 of Second Assessment Report (SAR) (IPCC, 1996) reviewed the literature on costs of greenhouse gas (GHG) mitigation prior to 1995. At that period, the debate was dominated by the differences in results from "bottom-up" (B-U) models and "top-down" (T-D) models. The former contain more details of technology and physical flows of energy, and the latter give more consideration to linkages between a given sector and a set of measures and macroeconomic parameters like gross domestic product (GDP) and final household consumption.

B-U models showed that energy efficiency gains of 10%–30% above baseline trends could be realized at negative to zero net costs over the next two or three decades. However, the costs of stabilizing emissions at 1990 levels reported by T-D analysis for Organization for Economic Co-operation and Development (OECD) countries were less optimistic, in the range –0.5% to 2% of GDP. SAR devoted much effort to explain the reasons for these differences and their meaning for policymakers. B-U models identify negative-cost mitigation potentials because of the difference between the best available techniques and those currently in use; the key question is then the extent to which market imperfections that inhibit access to these potentials can be removed cost-effectively by policy initiatives. T-D analyses focus on the overall macroeconomic effect of new incentive structures, such as carbon taxes or subsidies for energy efficiency; their results reflect a judgement on the capacity of non-price policies (market reforms, information, capacity building) to enhance the effectiveness of such signals to decarbonize the economy. The lesson is that, for a given abatement target, the content of the policy mix (carbon tax, carbon-energy tax, or auctioned emissions trading system) is as important as the assumptions regarding technology.

A second lesson of SAR is that, for both B-U and T-D models, the differences in cost assessment usually result from differences in the definition of baseline scenarios and in the time frame within which a given abatement target has to be met. Less often, they result from divergences in the costs of achieving this target from the same baseline scenario. This, in turn, relates to:

- the structural features of the scenario (assumptions about population, the rate and structure of economic growth, consumption patterns, and technology development paths); and
- its level of suboptimality (higher efficiency in the baseline scenario results in higher mitigation-costs estimates for a given target, while the existence of market failure, which enhances GHG emissions, or of fiscal distortions provides a possibility for economic and environmental double dividends).

The third lesson is that some of the determinants of costs are beyond the field of energy and environmental policy *stricto sensu*. This is why SAR emphasizes the importance of developing multiple baseline scenarios to support policymaking. This issue of the multiplicity of baseline scenarios is specifically important for the developing countries and countries with economies in transition. These regions were underinvestigated compared with the number of studies available for OECD countries.

8.1.2 Progress since the Second Assessment Report

Since SAR, the most important advance is the treatment of new topics related to linkages between national policies and the international framework of these policies in the context of the pre-Kyoto and post-Kyoto negotiation process. Of specific interest is the articulation between international emissions trading systems and domestic policies (taxes, domestic emissions trading, and standards). This link has been made in national models and global models that provide a description of relationships among various regions of the world. Some models represent solely GHG emissions trading, while others also incorporate energy flows, trade of other goods, and capital flows. In this context, while SAR discussed only the carbon leakage between abating and non-abating economies, an increasing number of studies have captured spillover effects (see *Box 8.1*) such as those triggered by trade effects and the modification of the capital flows.

A second evolution is the emergence of studies on the local and regional ancillary benefits of climate policies.

The third evolution is the development of studies on various abatement pathways towards given long-run concentration targets and on rules for emissions quota allocation among countries. These approaches, more dynamic in nature, capture the consequences of various abatement timetables on the behaviour of carbon prices and on the sharing of the overall burden among countries. They provide basic information about the equity of various designs of climate policies.

8.1.3 Coverage

This chapter covers studies on global assessments of the net cost of GHG mitigation policies irrespective of the avoided costs of climate change: total mitigation expenditure, and welfare gains or losses resulting from the economic feedbacks of mitigation policies and from their environmental co-benefits.

A specific effort has been devoted to ensure a balanced representation of global models and national models. Global models incorporate linkages between regions and countries; they cannot, however, represent very precisely the specific characteristics of each country, such as differences in national fiscal policies, in regional arrangements, and in socioeconomic constraints. Results from these models are widely diffused within the scientific community through publications in international journals, but are less utilized by national policymakers. The second type of study uses models the scope of which is limited to within the national frame. Results of such studies are more frequently reported in local languages and are more reflective of national debates and the specifics of the country in question. They incorporate linkages with the rest of the world economy, although in a more simplistic manner than the global models.

The results of these studies group into three large clusters. Section 8.2 reviews the studies that entail near-to-mid term impacts of domestic mitigation policies on factors such as GDP, welfare, income distribution, and social and environmental co-benefits at the local and national levels. Section 8.3 contains the results of mitigation studies that examine the interface between these domestic policies and the international context: international trade regimes, and spillover effects of the implementation of mitigation measures by a country or a block of nations on other countries. Section 8.4 reviews studies that focus on social, environmental, and economic impacts of alternate pathways to meet a range of concentration stabilization pathways beyond the Kyoto Protocol. They encompass a longer time horizon and do not incorporate details of macro economic policies, but highlight the question of technological change over the long run and the consequences of various sets of national targets.

8.2 Impacts of Domestic Policies

Evaluation of the economic impacts of domestic mitigation policies can no longer be made independently of the linkages between these policies and the international framework. However, it is important to disentangle the mechanisms that are themselves independent of the international regimes from those specifically driven by the interplay between these regimes and domestic policies. In addition, the existence of an international framework does not rule out the importance of domestic policies for addressing the specific problems of each country.

This section basically relies on national studies, including integrated economic regions such as the European Union (EU), but it also reports the results of multiregional studies for the concerned countries or region.

8.2.1 Gross Aggregated Expenditures in Greenhouse Gas Abatements in Technology-rich Models

In technology-rich B-U models and approaches, the cost of mitigation is constructed from the aggregation of technological and fuel costs. These include investments, operation and maintenance costs, and fuel procurement, but also included (and this is a recent trend) are revenues and costs from imports and exports, and changes in consumer surplus that result from mitigation actions. In all the studies, it is customary to report the mitigation cost as the incremental cost of some policy scenario relative to that of a baseline scenario. The total cost of mitigation is usually presented as a total net present value (NPV) using a social discount rate selected exogenously (the NPV may be further transformed into an annualized equivalent). Many (but not all) report also the marginal cost of GHG abatement (in US$/tonne of CO_2-equivalent), which is the cost of the last tonne of GHG reduced. Chapter 7 discusses cost concepts and discount rates in more depth.

Current B-U analysis can be grouped in three categories:

- Engineering economics calculations performed technology-by-technology (Krause, 1995; LEAP (1995), Von Hippel and Granada (1993); UNEP, 1994a; Brown *et al*., 1998; Conniffe *et al*., 1997). The costs and reductions from the large number of actions are aggregated into whole-economy totals in these studies. Each technology (or other action on energy demand) is assessed independently via an accounting of its costs and savings (investment costs, operational and maintenance cost, fuel costs or savings, and emissions savings). Once these elements are estimated, a unit cost (per tonne of GHG reduction) is computed for each action. The unit costs are then sorted in ascending order, and thus the actions are ordered from least expensive to the most expensive, per tonne of abatement, to create a cost curve. This approach requires a very careful examination of the interactions between the various actions on the cost curve: it is often the case that the cost and GHG reduction attached to an action depends on those of other actions in the same economy. Although the simpler interactions are easily accounted for by careful analysis, there exist many other instances in which complex, multi-measure interactions are very difficult to evaluate without the help of a more complex model that captures the system's effects. As an example, consider simultaneously: (a) changing the mix of electricity generation, (b) increasing interprovincial trade of electricity, and (c) implementing actions to conserve electricity in several end-use sectors. As each of these three actions has an impact on the desirability and penetration of each other action, such a combination requires many iterations that assess the three types of action separately, before an accurate assessment of the full portfolio can be obtained.
- Integrated partial equilibrium models that facilitate the integration of multiple GHG reduction options and the aggregation of costs. To achieve this, the majority of B-U studies use the whole energy system (MARKAL, MARKAL-MACRO, MARKAL-MATTER, EFOM, MESSAGE, NEMS, PRIMES[1]). These models have the advantage of simultaneously computing the prices

[1] For references see *Table 8.1*.

Table 8.1: *List of the models referred to in this chapter*

Model	Region	Reference
ABARE-GTEM	USA/EU/Japan/CANZ	In Weyant, 1999
ADAM	Denmark	Andersen *et al*., 1998
AIM	USA/EU/Japan/CANZ	In: Weyant, 1999
	Japan	Kainuma *et al*., 1999; Kainuma *et al*., 2000
	China	Jiang *et al*., 1998
CETA	USA/EU/Japan/CANZ	In: Weyant, 1999
E3-ME	UK/EU/World	Barker 1997, 1998a, 1998b, 1998c, 1999
ELEPHANT	Denmark	Danish Economic Council, 1997; Hauch, 1999
ECOSMEC	Denmark	Gørtz *et al*., 1999
ERIS		Kypreos et al, 2000
G-Cubed	USA/EU/Japan/CANZ	In: Weyant, 1999
GEM-E3	EU	Capros *et al*., 1999c
GEM-E3	Sweden	Nilsson, 1999
GemWTrap	France/World	Bernard and Vielle, 1999a, 1999b, 1999c
GESMEC	Denmark	Frandsen *et al*., 1995
GRAPE	USA/EU/Japan/CANZ	In: Weyant, 1999
IMACLIM	France	Hourcade *et al*., 2000a
IPSEP	EU	Krause *et al*., 1999
ISTUM	Canada	Jaccard *et al*., 1996; Bailie *et al*., 1998
MARKAL	World	Kypreos and Barreto, 1999
	Canada	Loulou and Kanudia, 1998, 1999a and 1999b; Loulou *et al*., 2000
	Ontario (Canada)	Loulou and Lavigne, 1996
	Quebec, Ontario, Alberta	Kanudia and Loulou, 1998b; Kanudia and Loulou, 1998a; Loulou *et al*., 1998
	Canada, USA, India	Kanudia and Loulou, 1998b
	EU	Gielen, 1999; Seebregts *et al*., 1999a, 1999b; Ybema *et al*., 1999
	Italy	Contaldi and Tosato, 1999
	Japan	Sato *et al*., 1999
	India	Shukla, 1996
MARKAL-MACRO	World	Kypreos, 1998
	USA	Interagency Analytical Team, 1997
MARKAL-MATTER	EU	Gielen *et al*., 1999b, 1999c
MARKAL and EFOM	EU	Gielen *et al*., 1999a; Kram, 1999a. 1999b
	Belgium, Germany, Netherlands, Switzerland	Bahn *et al*., 1998
	Switzerland, Colombia	Bahn *et al*., 1999a
	Denmark, Norway, Sweden	Larsson *et al*., 1998
	Denmark, Norway, Sweden, Finland	Unger and Alm, 1999
MARKAL Stochastic	Quebec	Kanudia and Loulou, 1998a
	Netherlands	Ybema *et al*., 1998
	Switzerland	Bahn *et al*., 1996
MEGERES	France	Beaumais and Schubert, 1994
MERGE3	USA/EU/Japan/CANZ	In: Weyant, 1999
MESSAGE	World	Messner, 1995
MISO and IKARUS	Germany	Jochem, 1998
MIT-EPPA	USA/EU/Japan/CANZ	In: Weyant, 1999
MobiDK	Denmark	Jensen, 1998
MS-MRT	USA/EU/Japan/CANZ	In: Weyant, 1999
MSG	Norway	Brendemoen and Vennemo, 1994
MSG-EE	Norway	Glomsrød *et al*., 1992; Alfsen *et al*., 1995; Aasness *et al*., 1996; Johnsen *et al*., 1996
MSG-6	Norway	Bye, 2000
MSG and MODAG	Norway	Aaserud, 1996
NEMS + E-E	USA	Brown *et al*., 1998; Koomey *et al*., 1998; Kydes, 1999
Oxford	USA/EU/Japan/CANZ	In: Weyant, 1999
POLES	USA, Canada, FSU, Japan, EU, Australia, New Zealand	Criqui and Kouvaritakis, 1997; Criqui *et al*., 1999
PRIMES	Western Europe	Capros *et al*., 1999a
RICE	USA/EU/Japan/CANZ	In: Weyant, 1999
SGM	USA/EU/Japan/CANZ	In: Weyant, 1999
SPIT	UK	Symons *et al*., 1994
SPIT	Ireland	O'Donoghue, 1997
WorldScan	USA/EU/Japan/CANZ	In: Weyant, 1999

CANZ: Other OECD countries (Canada, Australia, and New Zealand); FSU: Former Soviet Union.

of energy and of end-use demand as an integral part of their routine. They are based on least-cost algorithms and/or equilibrium computation routines similar to those used in T-D approaches. They increasingly cover both the supply and demand sides, and include mechanisms to make economic demands responsive to the changing prices induced by carbon policies. Furthermore, many implementations of these models are multiregional, and represent explicitly the trading of energy forms and of some energy intensive materials.
- Simulations models (based on models such as ISTUM) that take into account the behaviour of economic agents when different from pure least cost. To accomplish this, economic agents (firms, consumers) are allowed to make investment decisions that are not guided solely by technical costs, but also by considerations of convenience, preference, and so on. Such models deviate from least-cost ones, and so they tend to produce larger abatement costs than least-cost models, all things being equal otherwise.

The boundaries between these three categories is somewhat blurred. For instance, NEMS and PRIMES do include behavioural treatment of some sectors, and MARKAL models use special penetration constraints to limit the penetration of new technologies in those sectors in which resistance to change has been empirically observed. Conversely, ISTUM has recently been enhanced to allow the iterative computation of a partial equilibrium (the new model is named CIMS).

Several studies go further: they are based on partial equilibrium models in which energy service demands are sensitive to prices. Therefore, even the quantities of energy services may increase or decrease in carbon scenarios, relative to the base case. For these models report not only the direct technical costs, but also the loss or gain in consumer surplus because of altered demands for energy services. The results of this new generation of partial equilibrium B-U models tend to be closer than those of other B-U models to the results of the general equilibrium T-D models, which are also discussed in this chapter. Loulou and Kanudia (1999) argue that, by making demands endogenous in B-U models, most of the side-effects of policy scenarios on the economy at large are captured. When a partial equilibrium model is used, the cost reported is the net loss of social surplus (NLSS), defined as the sum of losses of producers and consumers surpluses (see Chapter 7).

As is apparent from the results presented below, considerable variations exist in the reported costs of GHG abatement. Some of these differences result from the inclusion/exclusion of certain types of cost in the studies (*e.g.*, hidden costs and welfare losses), others from the methodologies used to aggregate the costs, others from the feedback between end-use demand and prices, and still others from genuine differences between the energy systems of the countries under study. However, the most significant cause of cost variations seems to lie not only (see also Chapter 9) in methodological differences, but in the differences in assumptions. Finally, although most recent B-U results consider the abatement of a fairly complete basket of GHG emissions from all energy-related sources, a few essentially focus on CO_2 abatement only and/or on selected sectors, such as power generation. In this chapter, only results are reported that have sufficient scope to qualify as GHG abatement costs in most or all sectors of an economy.

To facilitate the exposition of the various results, the rest of this subsection is divided into four parts, as follows:
- studies that assume a large potential for efficiency gains, even in the absence of a carbon price;
- other B-U studies for Annex I countries or regions;
- Annex I studies that account for trade effects; and
- studies devoted to non-Annex I countries.

8.2.1.1 National and Regional Cost Studies Assuming Large Potentials for Efficiency Gains (the Impact of No Regrets or Non-price Policies)

An important part of climate policy debates is underpinned by a lasting controversy between believers and non-believers in the existence of a large untapped efficiency potential in the economy. If there, this potential could be realized at such a small societal cost that it would be more than compensated by cost savings that accrue from the efficiency improvements. Options that have a negative net social cost add up to an overall negative cost potential that may be quite large. *Figure 8.1* is a sketch of the successive marginal costs of abatement, as a function of GHG reduction relative to some baseline point A. The total cost is simply the area between the curve and the horizontal axis. From A to B, marginal abatement costs are negative, and from B onwards, they are positive. The debate revolves around the size of the total (negative) cost from A to B. The studies discussed in this subsection argue that the negative cost area is potentially quite significant, and compensates

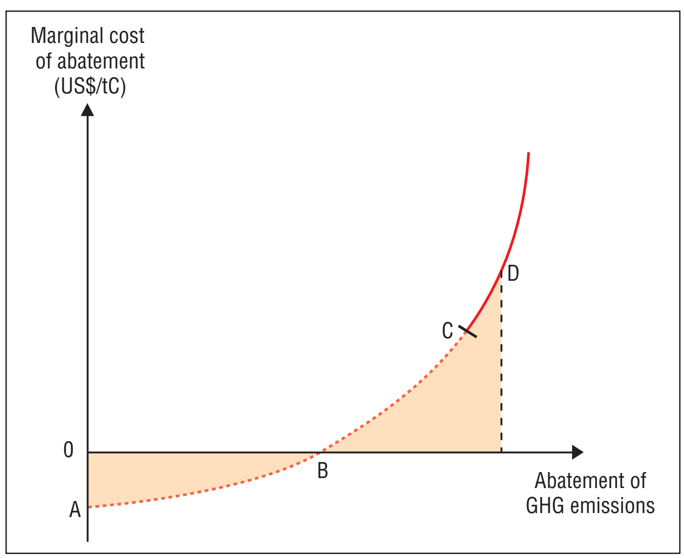

Figure 8.1: *A typical cost curve.*

to a large extent for the positive costs incurred after point B. Most other B-U studies analyzed in the next subsection do not even attempt to evaluate the relative positions of points A and B, since they optimize the system even in the absence of carbon constraint, and thus compute only the points beyond point B.

Krause (1995, 1996) identifies two main reasons why the negative cost area may be quite large: untapped potential for efficiency gains mainly in end-use technologies, both on the demand and on supply sides. Several major studies concretize this view in Europe as well as in North America (USA and Canada). For Europe, the monumental IPSEP reports (summarized in Krause *et al*. (1999)) conclude that emissions could be reduced by up to 50% below the 1990 level by 2030, at a negative overall cost. This involves the judicious implementation of technologies and practices in all sectors of the economy, and the application of a large number of government policies (incentives, efficiency standards, and educational). In the US, some of the 5-Lab studies (Brown *et al*., 1997a, 2000, particularly the HE/LC scenario) indicate that the Kyoto reduction target could be reached at negative overall cost ranging from –US$7 billion to –US$34 billion. Another study based on the NEMS model (Koomey *et al*., 1998) indicates that 60% of the Kyoto gap could be bridged with an overall increase in the US GDP. The latter study contrasts with another NEMS study (Energy Information Administration, 1998) that indicates GDP losses from 1.7% to 4.2% (depending on the extent of permit trading and sink options) for the USA to reach the Kyoto targets. Laitner (1997, 1999) further stresses the impact of efficient technologies on the aggregate cost of mitigation in the USA. In Canada, the MARKAL model was used with and without certain efficiency measures in various sectors (Loulou and Kanudia, 1998; Loulou *et al*., 2000): the results show costs of Kyoto equal to US$20 billion without the additional efficiency measures, versus –US$26 billion when efficiency measures are included in the database. Again in Canada, the ISTUM model was used (Jaccard *et al*., 1996, Bailie *et al*., 1998) considering a set of pro-active options. For example, in the residential sector large emissions reductions of 17% to 25% relative to 1990 could be achieved as early as 2008 with many negative costs options, and beyond that level of reduction, the marginal costs is ranging from US$25 to US$89/tC.

As extensively discussed in SAR, many economists argue that the real magnitude of negative cost options is not so large if account is taken of:

- Transaction costs of removing market imperfections that inhibit the adoption of the best technologies and practices;
- Hidden costs, such as the risks of using a new technology (maintenance costs, quality of services);
- "Rebound effect" because, for example, an improvement in motor efficiency lowers the cost per kilometre driven and has the perverse effect of encouraging more trips; and
- Real preferences of consumers: options such as driving habits and modal switches towards rail and mass transit are considered to entail negative costs. This does not consider enough the reality of consumers' behaviour preferences for flexibility and non-promiscuity in transportation modes, or even "symbolic" consumption (such as the preference for high-power cars even in countries with speed limits).

These arguments should not be used to refute the very existence of negative cost potentials. They indicate that the applicability of non-price policy measures apt to overcome barriers to the exploitation of these potentials must be given serious attention. Some empirical observations do confirm that active sectoral policies can result in significant efficiency gains, in demand-side management for electricity end-uses for example. However, the many sources of gaps between technical costs and economic costs cannot be ignored (see the taxonomy of Jaffe and Stavins, 1994). The few existing observations (Ostertag, 1999) suggest that the transaction costs may represent, in many cases, a large fraction of the costs of new technology, and there is always an uncertainty about the efficiency and the political acceptability of the policies suggested in the above studies. This issue is clearly exemplified by the set of studies carried out in the USA and collected under the name "5-LAB studies". In these, some scenarios produce positive incremental costs and others negative costs, depending on the aggressiveness with which efficiency measures are implemented (Interlaboratory Working Group, 1997; Brown *et al*., 1998).

8.2.1.2 Bottom-up Costs Resulting from Carbon Pricing (Developed Countries)

Contrary to the studies discussed above, the partial equilibrium studies reviewed in this section do not report negative costs. This is because the least-cost algorithms employed, which are powerful to compute the incremental cost of the system with and without a carbon constraint (*i.e*., point B in *Figure 8.1*), demand a set of somewhat arbitrary parameters to be calibrated in such a way that they calculate a suboptimal baseline; but such an operation demands resorting to a set of somewhat arbitary parameters and the results are less easy to interpret. This is why the B-U studies reported hereafter explore only the section of the cost curve with positive carbon prices (section CD in *Figure 8.1*).

It is very hard to encapsulate in a short presentation the many studies carried out with a B-U approach using a crosscutting, carbon-pricing instrument. *Figure 8.1* summarizes a number of these results, obtained with a variety of B-U models applied to a single Annex I country or region, ignoring the trade effects. Included are those studies that contain enough information to present the marginal abatement cost along with the level of GHG emission variation from 1990 (other studies that reported only the total abatement cost are discussed separately). In *Figure 8.2*, each point represents one particular reduction level (relative to 1990) and the corresponding marginal cost of reduction. Points that are linked together by a line correspond

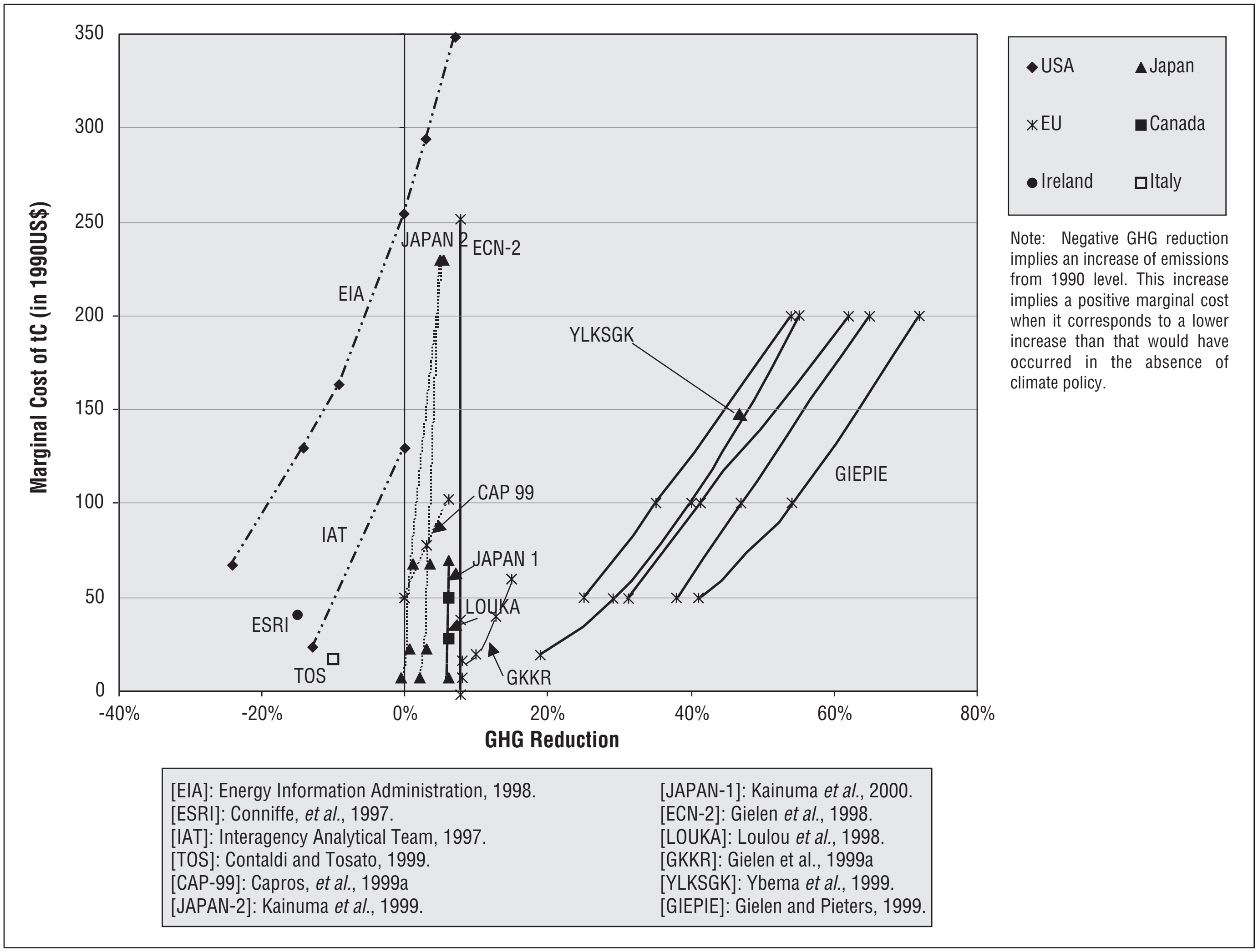

Figure 8.2: *Country results with bottom-up studies using a crosscutting instrument.*

to a multi-run study effected with the same model, but in which the amount of reduction was varied.

Evidently, *Figure 8.2* shows considerably discrepancies from study to study. These large variations are explained by a number of factors, some of which reflect the widely differing conditions that prevail in the countries studied, while others result from the modelling and scenario assumptions. These variations are discussed next, illustrated by examples from *Figure 8.2*.

8.2.1.2.1 Cost Discrepancies that Result from Specific Country Conditions

- *Energy endowment*. Countries that are richly endowed with fossil fuels find it generally less expensive to replace coal with gas, and thus have a greater potential than other countries to reduce emissions with readily available means. (This assumes that the change is not done very rapidly, so as to affect as little as possible the turnover rate of the existing investments.) For instance, this is the case for the USA (coal and oil products).[2] At the other extreme of the spectrum some countries have fuelled their economy almost exclusively on hydropower, nuclear power, and some gas, and will thus find fewer opportunities to switch to less CO_2-intensive fuels. This occurs for Norway (hydro), France (nuclear and hydro), Japan (nuclear and some fossil fuels), Switzerland (hydro and nuclear), and to some extent Canada (hydro and nuclear).
- *Economic growth*. An economy with high growth rate faces the following dilemma. On the one hand, the growth allows for a rapid capital turnover, and thus many opportunities to install efficient or low-carbon technologies. On the other hand, the same economy requires more energy precisely because of its fast growth. The net result is that such countries have a tendency to decrease markedly their energy intensity

[2] However, although the direct cost of switching away from coal may be relatively low, the indirect costs (including the political cost) of disrupting the coal sector may be high.

(energy per GDP), but to increase significantly their total emissions. For such countries, a net reduction of GHG emissions below a base year's emissions is usually costly. Typically, many fast-developing countries in East and South East Asia are in this category. In the studies cited above, their emissions "reductions" are often computed relative to the baseline rather than to a fixed base year.

- *Energy intensity*. The degree of energy intensity of an economy acts in opposing directions when the economy wishes to achieve net emissions reductions: on the one hand, a high degree of energy intensity may occur because that the country has not yet implemented some efficiency measures implemented elsewhere. On the other hand, such an economy may have been built on energy, and may thus find it hard to veer to a different, less energy-intensive mode, in a short time. Its development path is somewhat frozen, at least in the short term. The higher the carbon intensity, the more important the time frame of abatement. Such a pattern is observable in North America; despite its rapid capital turnover in the industry, the large inertia in sectors such as transportation is a determining factor of high abatement cost when the required abatement implies short term actions on these sectors.
- *Other specific conditions*. For example, Germany faces a very special situation because of the reunification in 1990 to 1991. The East German part of the country emits much less now than in 1990, and the country as a whole is able to effect significant reductions at essentially zero or very low cost, up to a certain point, beyond which its marginal cost may well accelerate considerably.

8.2.1.2.2 Discrepancies in Results Due to Modelling and Scenario Assumptions

- *Policy assumptions*. The results summarized in *Figure 8.2* are based mostly on partial equilibrium models, which tend to approach general equilibrium computations, such as AIM, NEMS, MARKAL, MARKAL-MACRO, PRIMES, CIMS, etc. Some of these models allow evaluation of the impact on mitigation cost of the redistribution of the proceeds of a carbon tax (the results obtained with the AIM model for Japan (Kainuma *et al*., 1999, 2000) show very clearly that suitable redistribution reduces the marginal cost of abatement).
- *Modelling differences*. Some models include partial economic feedbacks in the form of demand elasticities, as for example MARKAL (Loulou and Kanudia, 1999a), and for these models the abatement marginal costs are generally lower than when demands are fixed, because it becomes unnecessary to tap the most costly technological options. MARKAL-MACRO and NEMS include macroeconomic components in the computation of the equilibrium, and therefore qualify as general equilibrium models, albeit simplified ones. In addition, these two models include behavioural considerations in the calculation of the equilibrium, which tend to raise the cost of abatement, compared to least-cost models such as MARKAL.
- *Scenario variation*. The variety of scenarios used is quite large, as a result of varying some or all of the relevant elements. These include whether the technologies are allowed to penetrate freely or in a limited fashion (typically renewables, nuclear, and some new end-use technologies), the basket of GHG gases considered (CO_2 alone versus multigas studies), assumed economic growth, and sectoral scope (energy only *versus* whole economy).
- *Example*. To illustrate the above comments, *Figure 8.2* indicates that at a marginal cost of less than US\$100/tCO_{2eq}, the US emissions would still be larger in 2010 than in 1990, according to the NEMS (Energy Information Administration, 1998) and MARKAL (Interagency Analytical Team, 1997) studies. Note that NEMS predicts higher marginal costs than MARKAL for the same emission level, as expected, since NEMS includes many behavioural considerations, whereas MARKAL is a least-cost model. Japan's emissions would be reduced by 1% to 8% (AIM studies (Kainuma *et al*., 1999, 2000); Ireland's (Conniffe *et al*., 1997) and Italy's (Contaldi and Tosato, 1999) emissions would also increase, whereas Canada's emissions would decrease by 6% (MARKAL study (Loulou *et al*., 1998)). The several European studies show a wide range of reductions, from relatively small reductions (PRIMES study (Capros *et al*., 1999a)) to medium or large reductions with the various MARKAL studies (Gielen and Pieters, 1999; Gielen *et al*., 1999a; Ybema *et al*., 1999). These large variations are mainly explained by the modelling and scenario assumptions: PRIMES marginal costs are expected to be larger than MARKAL's (just as NEMS costs were larger than MARKAL's in the US case). In addition, scenario assumptions vary across studies: the number of gases modelled, degree of efficiency of the instrument used across the EU countries, and availability of international permits trading.

Several studies are not represented in *Figure 8.2*, since only incremental or average costs were reported. For instance, a German study (Jochem, 1998) indicates reductions of 30% to 40% in 2010 at average costs ranging from US\$12 to US\$ 68/tCO_{2eq}. In Canada (Loulou and Lavigne, 1996), a measure of the impact of demand reduction is obtained by running MARKAL with and without elastic demands for energy services: the total cost is US\$52 billion with fixed demands, andUS\$42 billion with elastic demands. Chung *et al*. (1997) arrive at much higher total costs for Canada, using a North American equilibrium level (the higher cost apparently results from fewer technological options than in MARKAL) A Swedish MARKAL study (Nystrom and Wene, 1999) find total cost of 210 billion Swedish krona for a stabilization scenario, against 640 billion Swedish krona for a 50% emissions reduction in 2010. This same study investigates the opportunity cost of a nuclear phase out, and evaluates a rebound effect on the demands of a 9% emissions reduction for Sweden.

8.2.1.3 Country Studies for Developing Countries

Several recent studies have been carried out as part of internationally co-ordinated country study programs conducted by the United Nation Environment Programme (UNEP) Collaborating Centre of Energy and Environment (UNEP, 1999a–1999g), and by the Asian Development Bank, United Nations Development Programme (UNDP), and the Global Environment Facility (ALGAS, 1999c–h). Summaries and analyses appear in Halsnaes and Markandia (1999). These recent studies supplement a number of earlier ALGAS studies of Egypt, Senegal, Thailand, Venezuela, Brazil, and Zimbabwe. The relevant results on aggregate cost are presented as individual country reports and summarized in ALGAS (1999) and in Sathaye *et al*. (1998). National study teams undertook the UNEP and ALGAS studies, using a variety of modelling approaches. The study results reported in *Table 8.2* are based primarily on energy sector options, which are supplemented with a number of options in the transportation sector, waste management, and from the land-use sectors. The GHG emissions reductions are defined as percentage reductions below baseline emissions in 2020 or 2030, or as accumulated GHG emission reductions over the timeframe of the analysis. These analyses are very useful to indicate the extent and cost of clean development mechanism (CDM) potentials in all countries studied.

The ALGAS cost curves show a total accumulated CO_2 emission reduction potential of between 10% and 25% of total emissions in the period 2000 to 2020. The marginal reduction cost is below US\$25/tCO_2 (see *Table* 8.2) for a major part of this potential, and a large part of the potentials in many of the country studies are associated with very low costs which even in some cases are assessed to be negative. The magnitude of the potential for low cost options in the individual country cost curves depends on the number of options that have been included in the studies. Countries like Pakistan and Myanmar have included relatively many options and have also assessed a relatively large potential for low-cost emission reductions.

Most of the country studies have concluded that options like end-use energy efficiency improvements, electricity saving options in the residential and service sectors, and introduction of more efficient motors and boilers are among the most cost-effective GHG emission reduction options. The studies have included relatively few GHG emission reduction options related to conventional power supply.

The UNEP cost curves exhibit a number of interesting similarities across countries. All country cost curves have a large potential for low cost emission reductions in 2030, where 25% (and in some cases up to 30%) of the emission reduction can be achieved at a cost below US\$ 25/tCO_2 (See *Table 8.2*). The magnitude of this "low cost potential" is like in the ALGAS studies, influenced by the number of climate change mitigation options included in the study. Individual studies indicate that some of the countries like Ecuador and Botswana experience a very steep increase in GHG emission reduction costs when the reduction target approaches 25%. It must be noted that these country studies primarily have assessed end use energy efficiency options and a few renewable options and have not included major reduction options related to power supply which probably could have extended the low cost emission reduction area. The studies for Hungary and Vietnam estimate a relatively small emission reduction potential, which primarily can be explained by the specific focus in the studies on end use efficiency improvements and electricity savings that do not include all potential reduction areas in the countries.

The options in the low-cost part of the UNEP cost curves typically include energy efficiency improvements in household and industry, and a number of efficiency or fuel switching options for the transportation sector. The household options include electricity savings such as compact fluorescent light-

Table 8.2: *Emission reduction potentials achievable at or less than US\$25/tCO_2 for developing countries and two economies in transition*

Annual reduction relative to reference case		
Country	**MtCO_2/yr**	**%**
Argentina (UNEP, 1999a)	–	11.5
Botswana (UNEP, 1999c)	2.87	15.4
China (ALGAS, 1999c)	606	12.7
Ecuador (UNEP, 1999b)	12.7	21.3
Estonia (UNEP, 1999g)	9.6	58.3
Hungary (UNEP, 1999f)	7.3	7.6
Philippines (ALGAS, 1999h)	15	6.2
South Korea(ALGAS, 1999d)	5.3	5.7
Zambia (UNEP, 1999d)	6.09	17.5
Brazil (UNEP, 1994)	–	29
Egypt (UNEP, 1994)	–	52
Senegal (UNEP, 1994)	–	50
Thailand (UNEP, 1994)	–	29
Venezuela (UNEP, 1994)	–	24
Zimbabwe (UNEP, 1994)	–	34
Cumulative reduction relative to reference case		
Country	**MtCO_2/yr**	**%**
Myanmar (ALGAS, 1999e)	44	23
Pakistan (ALGAS, 1999f)	1120	23.7
Thailand (ALGAS, 1999g)	431	4.2
Vietnam (UNEP, 1999e)	1016	13.4

bulbs (CFLs) and efficient electric appliances and, for Zambia, improved cooking stoves. A large number of end-use efficiency options have been assessed for electricity savings, transport efficiency improvements, and household cooking devices, but very few large scale power production facilities.

There are a number of similarities in the low cost GHG emission reduction options identified in the ALGAS and UNEP studies. Almost all studies have assessed efficient industrial boilers and motors to be attractive climate change mitigation options and this conclusion is in line with the conclusions of earlier UNEP studies (UNEP 1994b). A number of transportation options, in particular vehicle maintenance programmes and other efficiency improvement options, are also included in the low-cost options. Most of the studies have included a number of renewable energy technologies such as wind turbines, solar water thermal systems, photovoltaics, and bioelectricity. The more advanced of these technologies tend to have medium to high costs in relation to the above mentioned low-cost options. A detailed overview of the country study results is given in the individual country study reports (UNEP 1999a-g; ALGAS a-h, 1999).

Apart from the UNEP and ALGAS studies presented above, several additional independent studies were carried out for large countries with the help of equilibrium models. Examples are the ETO optimization model (for India, China, and Brazil), the MARKAL model for India, Nigeria, and Indonesia, and the AIM model for China. *Table 8.3* reports the marginal costs (or other cost in some cases) for the abatement levels considered in the studies (relative to baseline). Marginal costs vary from moderate to negative, depending on the country and model used, for emission reductions that are quite large in absolute terms compared to the baseline emissions.

These studies point out the interest of the same set of technologies for most of the countries, such as efficient lighting, efficient heating or air-conditioning (depending upon the region), transmission and distribution losses, and industrial boilers.

Importantly, it should be emphasized that in the way these studies are conducted, the potential for cheap abatement increases in proportion the baselines. In reality, this may not be the case because, in cases of rapid growth, an acceleration of the diffusion of efficient technologies is expected, which

Table 8.3: *Abatement costs for five large less-developed countries*

Country	**China**	**India**	**Brazil**	**China**	**India**	**Nigeria**	**Indonesia**
Reference	Wu *et al*. (1994)	Mongia *et al*. (1994)	La Rovere *et al*., (1994)	Jiang *et al*. (1998)	Shukla (1996)	Adegbulugbe *et al*. (1997)	Adi *et al*. (1997)
Span of study	1990–2020	1990–2025	1990–2025	1990–2010	1990–2020	1990–2030	1990–2020
Emissions in 1990 ($MtCO_2$)	2411	422	264				
Emissions in final year, baseline ($MtCO_2$)	6133	3523	1446				
% change	154%	735%	447%	130%	650%		
Emissions in final year, mitigation ($MtCO_2$)	4632	2393	495				
% change	92%	467%	88%	53%	520%		
% change: mitigation versus baseline, final year	**–40%**	**–36%**	**–80%**	**–59%**	**–20%**	**–20%**	**–20%**
Marginal cost in final year (US$/$tCO_2$)	**32**	**–16**	**–7**	**28**	**28**	**<30**	
Average cost in final year (US$/$tCO_2$)						**<5**	
Annual cost in final year (billion US$/yr)							**47**

would lower the magnitude of the negative cost potentials. A second caveat to be placed is that an increase of the GDP per capita is consistent with the increase of wages and purchasing power parities which would increase the cost of carbon imported from these countries through CDM projects.

8.2.1.4 Common Messages from Bottom-up Results

Clearly, the impact of policy scenarios has a large influence on abatement costs. Certain studies propose a series of public measures (regulatory and economic) that tap deep into the technical potential of low carbon and/or energy-efficient technologies. In many cases, such policies show low or negative costs. A comparison with least-cost approaches is difficult because these evaluate systematically both the baseline and the policy scenario as optimized systems and do not incorporate market or institutional imperfections in the current world. It would be of great interest to conduct a more systematic comparison of the results obtained via the various B-U approaches, so as to establish the true cause of the discrepancies in reported costs. A timid step in this direction is illustrated in Loulou and Kanudia (1999a).

This leads to a general discussion about the extent to which all these results suffer from a lack of representation of transaction costs, which are usually incurred in the process of switching technologies or fuels. This category of transaction cost encompasses many implementation difficulties that are very hard to capture numerically. The general conclusion from SAR (that costs computed using the B-U approach are usually on the low side compared to costs computed via econometric models, which assume a history-based behaviour of the economic agents) is no longer generally applicable, since some B-U models take a more behavioural approach. Models such as ISTUM, NEMS, PRIMES, or AIM implicitly acknowledge at least some transaction costs via various mechanisms, with the result that market share is not determined by visible (market-based) least-cost alone. Least-cost modellers (using MARKAL, EFOM, MESSAGE, ETO) also attempt to impose penetration bounds, or industry-specific discount rates, which approximately represent the unknown transaction costs and other manifestations of resistance to change exhibited by economic agents. In both cases these improvements result in partially eschewing the "sin" of optimism and blur the division between B-U and T-D models. While the former, indeed, tend to be less optimistic when they account for real behaviours, it is symmetrically arguable that the latter underestimate the possibility of altering these behaviours through judicious policies or better information. All this area still remains underworked.

A common message is the attention that must to be paid to the marginal cost curve. Despite the limitations and differences in results discussed above, B-U analyses convey important information that lies beyond the scope of T-D models, by computing both the total cost of policies and their marginal cost. Very often, indeed, the marginal abatement cost of a given target is high, although the average abatement cost is reasonably low, or even negative. This is because the initial reductions of GHG emissions may have a very low (or negative) cost, whereas additional reductions have, in general, a much higher marginal cost. This fact is captured in the curve representing marginal abatement cost versus reduction quantity, which starts with negative marginal costs, as illustrated in *Figure 8.1*. The initial portion of the curve (section A–B) exhibits negative cost options, which may add up to a significant portion of the reductions targetted by a given GHG scenario. As the reduction target increases (section B–C–D of the curve), the marginal cost becomes positive, and also eventually the total mitigation cost if the reduction target is large enough. But there is systematically a wedge between the marginal and total costs of abatement, and this wedge is all the more important as the macroeconomic impacts of climate policies are driven in large part by the marginal costs (because the latter dictate the change in relative commodity prices). They are driven only modestly by the total amount of abatement expenditures.

A crucial, albeit indirect, message, is the importance of innovation: indeed, B-U models depend on a reasonable representation of emerging or future technologies. When this representation is deficient, the models present a pessimistic view of the costs of more drastic abatements in the long term. This issue is not one of the modelling paradigm, but rather of feeding the models with good estimates of technical progress. Some works are currently underway to make explicit the drivers of technical change, such as learning-by-doing (LBD) or uncertainty. These studies are discussed further in Section 8.4.

8.2.2 Domestic Policy Instruments and Net Mitigation Costs

Tapping the technical abatement potentials requires setting up new incentive structures (taxes, emissions trading, technical standards, voluntary agreements, subsidies) for production and consumption, i.e. climate policies. In the following, empirical models that measure net mitigation costs of climate policies are reviewed in order to disentangle the reasons why certain policy packages have similar or different outcomes in various countries. As a first step, the results are presented at an aggregated level; then the impact of measures meant to mitigate the sectoral and distributional consequences of climate policies is examined. Finally, in a third step, the ancillary benefits from the joint reduction of carbon emissions and other pollutants are considered to complete the picture.

8.2.2.1 Aggregate Assessment of Revenue-raising Instruments

Introducing a carbon tax (or auctioned tradable permits) provides an incentive to change the technology over the short and long term. Such policies generate tax revenues and the way these revenues are used has major impacts on the social costs of the climate policy. The reason is that these revenues are, in principle, available to offset some or all of the costs of the mitigation policy. When emission targets go beyond the negative cost potentials, there is a general agreement among economists (see Chapters 6 and 7) that if standards are used (or if emissions permits are allocated for free) the resultant social cost is higher than the total abatement expenditures. Producers pass part of the marginal abatement cost on to consumers through higher selling prices, which implies a loss of consumer surplus. If the elasticity of supply is quite high, this might lead to a net loss of producer surplus. However, if the elasticity of supply is fairly low, overall (or net) producer surplus can rise when policies cause a restriction in output, because the policy-generated rents per unit of production enjoyed by producers more than compensate for the net decrease in sales.

In the 1990s there was considerable interest in how revenue-neutral carbon taxes may mitigate this effect on the economy by enabling the government to cut the marginal rates of pre-existing taxes, such as income, payroll, and sales taxes. The possibility is a double dividend policy (Pearce, 1992), by both (1) improving the environment and (2) offsetting at least part of the welfare losses of climate policies by reducing the costs of the tax system (see the discussion in Chapter 7). The same mechanism occurs when nationally auctioned permits are used; for simplicity, the term carbon tax is used in the rest of this chapter, except when the distinction between these two instruments is necessary.

The starting point in a discussion of a double dividend is how expensive it is to raise government income, that is, how big is the marginal cost of funds (MCF). A high MCF gives more scope for a double dividend than a small MCF in the economy. This arises because the parameters that determine the magnitude of the double-dividend (see Chapter 7) are:

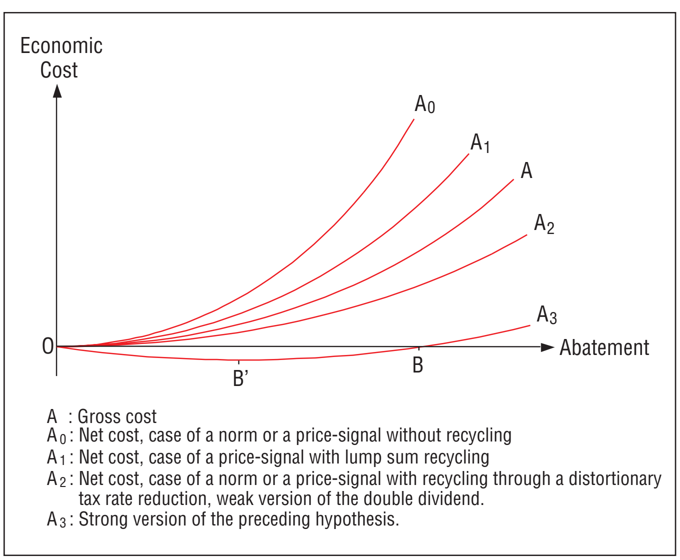

Figure 8.3: *Carbon taxes and the costs of environmental policies.*

- direct cost to the regulated sector (sector's changes in production methods or installation of pollution-abatement equipment);
- tax-interaction effect (prices are increasing);
- revenue-recycling effect associated with using revenues to finance cuts in marginal tax rates.

When the revenues of carbon taxes are returned in a lump-sum fashion to households and firms, the tax-interaction effect is systematically higher than the revenue-recycling effect. Also the net cost of climate policy is higher than its gross cost (while lower than that with a no-tax policy, see A_1 and A_2 in *Figure 8.3*). However, it is possible to improve this result by targetting tax revenues to cuts in the most distortionary taxes; this can yield either a weak or a strong form of double dividend (Goulder, 1995a). The weak double dividend occurs as long as there is a revenue-recycling effect due to the swap between carbon taxes and the most distortionary taxes. Mitigation costs are systematically lower when revenues are recycled this way than when they are returned lump sum. The strong double dividend is more difficult to obtain. It requires that the (beneficial) revenue-recycling effect more than offset the combination of the primary cost and the tax-interaction effect. In this case, the net cost of abatement is negative (at least within some range). As discussed in Chapter 7, this is possible if, prior to the introduction of the mitigation policy, the tax system is already highly inefficient along non-environmental dimensions. In terms of *Figure 8.3*, the revenue-recycling effect is represented by the downward shift from curve A_1 to curve A_2 or A_3. If the shift is from A_1 to A_2, the weak double dividend occurs, but not the strong double dividend. If the shift is from A_1 to A_3, not only does the weak double dividend occur, but the strong double dividend is realized as well, since the net costs are negative within a range.

While the weak form of double dividend enjoys broad support from theoretical and numerical studies, the strong double dividend hypothesis is less broadly supported and more controversial. Indeed, reaching an economical dividend is impossible when the economy is at full employment and if all other taxation is optimal (abstracting for the environmental externality). Therefore, it may be argued that the double dividend accrues from the tax reform, independently of the climate policy. However, empirical models capture the fact that, in the real world, a carbon tax or auctioned emissions permits will not be implemented after the enforcement of an optimal fiscal reform. To the contrary, introducing a new tax may be a *sine qua non* condition to the fiscal reform. For a given carbon tax revenue, models help interpret the best way to recycle this revenue.

Specific features of the tax systems and markets of the production factors (labour, capital, and energy) ultimately determine the presence or absence of a strong double dividend. For example, a double dividend is likely if production factors are very distorted by prior taxation or specific market conditions, if there is a problem of trade-balance because of the import of fossil energy, or if consumer choice is highly distorted because of tax-deducible spending provisions (Parry and Bento, 2000).

Table 8.4: *Energy Modelling Forum Results: carbon tax and GDP losses in 2010 with lump-sum recycling (in 1990 US$)*

	Carbon tax in 2010				GDP losses in 2010 (%)			
Model	**USA**	**OECD-E**	**Japan**	**CANZ**	**USA**	**OECD-E**	**Japan**	**CANZ**
ABARE-GTEM	322	665	645	425	1.96	0.94	0.72	1.96
AIM	153	198	234	147	0.45	0.31	0.25	0.59
CETA	168				1.93			
G-Cubed	76	227	97	157	0.42	1.50	0.57	1.83
GRAPE		204	304			0.81	0.19	
MERGE3	264	218	500	250	1.06	0.99	0.80	2.02
MIT-EPPA	193	276	501	247				
MS-MRT	236	179	402	213	1.88	0.63	1.20	1.83
Oxford	410	966	1074		1.78	2.08	1.88	
RICE	132	159	251	145	0.94	0.55	0.78	0.96
SGM	188	407	357	201				
WorldScan	85	20	122	46				

Source: Weyant (1999). The carbon tax required (either explicitly or implicitly) and the resultant GDP losses are calculated to comply with the prescribed limits under the Kyoto Protocol for four regions under a no trading case: the USA, OECD Europe (OECD-E), Japan, and Canada, Australia, and New Zealand (CANZ).

Empirical studies try to gauge the impact of these many determinants and to understand why the effects of a given recycling strategy (reducing payroll, personal income, corporate income, investment income, or expenditure taxes) differ from one country to another.

8.2.2.1.1 Net Economic Costs under Lump-sum Recycling

The simplest way to simulate the recycling of a carbon tax or of auctioned permits is through a lump-sum transfer. Such recycling does not correspond to any likely policy in the real world. However, these modelling experiments provide a useful benchmark to which other forms of recycling can be compared. In addition, they allow an easy intercountry comparison of the impacts of emissions constraint before the impacts of the many types of possible recycling policies are considered.

The comparative study carried out by the Energy Modeling Forum (EMF, Stanford University) is very useful in this respect: EMF-16 (1999) examined the costs of compliance with the Kyoto Protocol as calculated by more than a dozen modelling teams in the USA, Australia, Japan, and Europe (*Table 8.4*). Most of the models used are general equilibrium models. While not strictly comparable to the marginal technical abatement costs reported in Section 8.2.1, the magnitude of the carbon tax used in these models is determined again by the difference between the costs of marginal source of supply (including conservation) with and without the target. As in the B-U models, this parameter depends in turn on such factors as the size of the necessary emissions reductions, assumptions about the cost and availability of carbon-based and carbon-free technologies, the fossil fuel resource base, and short- and long-term price elasticity. Also important is the choice of base year: a model that provides 3 years to adapt to a constraint beginning in 2010 shows higher marginal abatement costs than one that provides 8 years.

Figures 8.4-a to *8.4-d* show the incremental cost of reducing a ton of carbon for alternative levels of CO_2 reductions in the USA, OECD Europe, Japan, and Other OECD countries (CANZ) when all reductions are made domestically. Note there are two differences with the B-U studies:

- these numerical experiments do not consider negative cost abatement potentials and presume that if an action is economically justifiable in its own right, it will be undertaken independent of climate-related concerns; and
- because they incorporate demand elasticity and multiple macroeconomic feedback, these marginal cost curves do not behave as those found in the B-U studies.

A first conclusion that could be drawn from *Table 8.4* is that no strict correlations occur between the necessary carbon tax to reach a certain emission target and the GDP loss faced by a country. While the carbon tax in Japan is systematically higher than that for the USA, most studies conclude lower GDP losses in Japan than in the USA. In general, the carbon taxes are highest in OECD Europe and Japan, while the GDP losses are highest in the USA and Other OECD countries. This absence of strict correlation between marginal taxes and GDP losses is explained by the pre-existing energy supply, the structural economic features, and the pre-existing fiscal system. For instance, if a country relies more on renewable energy, and is specialized in low carbon-intensive industry, the impact of a given level of carbon tax will be lower. However, as the burden of emission reductions falls only on a few sectors, the carbon tax for a given target will be higher.

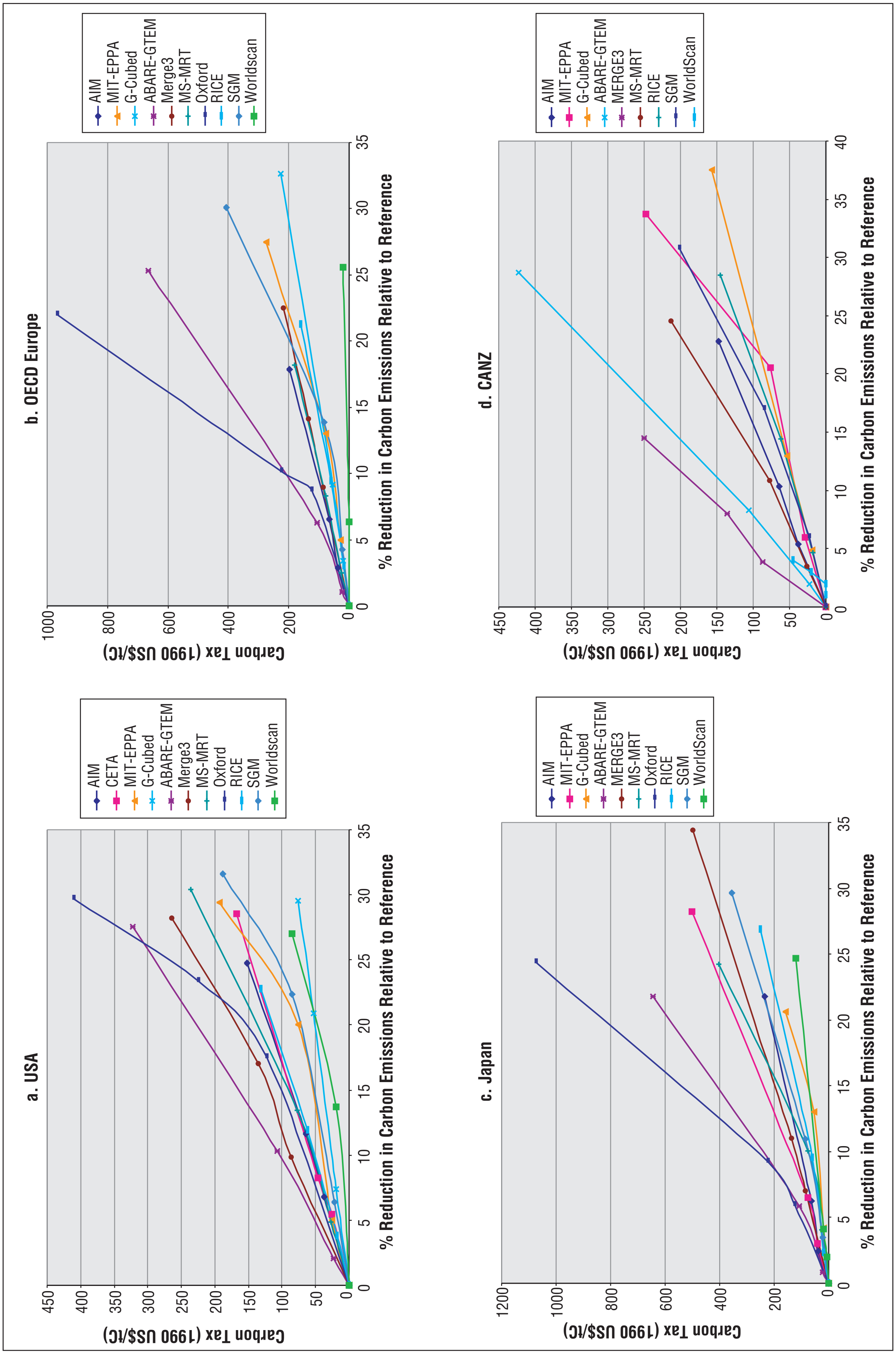

***Figure 8.4:** Incremental cost (US$/tC) by regions.*

A second observation from this comparison is that the lump-sum recycling of tax revenue never gives a strong double dividend, which is in accordance with theory and is confirmed by country studies. This is the case in particular for Denmark (Frandsen *et al*., 1995; Jensen, 1998; Gørtz *et al*., 1999), France (Bernard and Vielle, 1999a), Finland (Jerkkola *et al*., 1993; Nilsson, 1999), and Norway (Brendemoen and Vennemo, 1994; Johnsen *et al*., 1996; Håkonsen and Mathiesen, 1997). These studies demonstrate welfare losses of the same order of magnitude as those of global models, ranging from 0.14% to 1.2% for various levels of emissions abatement ranging from 15% to 25% over a 10-year time period. Only a very few studies conclude to some strong form of double-dividend but do not explain the contradiction between this result and lessons from analytical works.

8.2.2.1.2 Carbon Taxes and Reducing Payroll Taxes

Figure 8.5 plots the range of the numerical findings for a wide set of countries. In comparison with the previous results, these findings are far more optimistic. This confirms the theoretical results that the gross costs of meeting given abatement targets can be significantly reduced by using the revenue of carbon taxes to finance cuts in the existing distortionary taxes, instead of returning the revenues to the economy in a lump-sum fashion. Only a few studies provide results that allow for a systematic assessment of the attractiveness of payroll recycling through comparing its welfare implication with that of lump-sum recycling. For Norway, Håkonsen and Mathiesen (1997) use a static computable general equilibrium (CGE) model to compare lump-sum recycling to private households and a reduction in employers' social security contributions. Welfare is measured by a combined index of commodity and leisure consumption. When CO_2 emissions are reduced by 20% (*i.e.*, stabilizing emissions in 2000 at the 1990 level), welfare is reduced by 1% with lump-sum recycling, but only by 0.3% when tax income is used to reduce social security contributions. These authors also found ancillary benefits that decrease welfare losses even further (see Section 8.2.4 below).

In this report, it is impossible to identify all the sources of discrepancies in results across models. Only the differences between results concerning the USA and European economies are considered. These discrepancies arise because labour taxes represent one of the most important sources of distortion in European countries as a result of the pre-existing tax structure and of the type of labour-market regulation that prevails in these countries. Note that a systematic outcome of these studies is that an increase in employment is easier to obtain than an increase in total consumption or social welfare, which leads some authors to discuss the employment double dividend as distinct from the efficiency dividend.

While studies conclude that the swap between carbon and payroll taxes reduces the net burden of climate policies but does not avoid net welfare losses in the USA (Goulder 1995b; Jorgenson and Wilcoxen, 1995; Shackleton *et al*., 1996), a strong double dividend often occurs in Europe. As suggested by theoretical analyses (Carraro and Soubeyran, 1996), these differences can be explained by the differences both in taxation

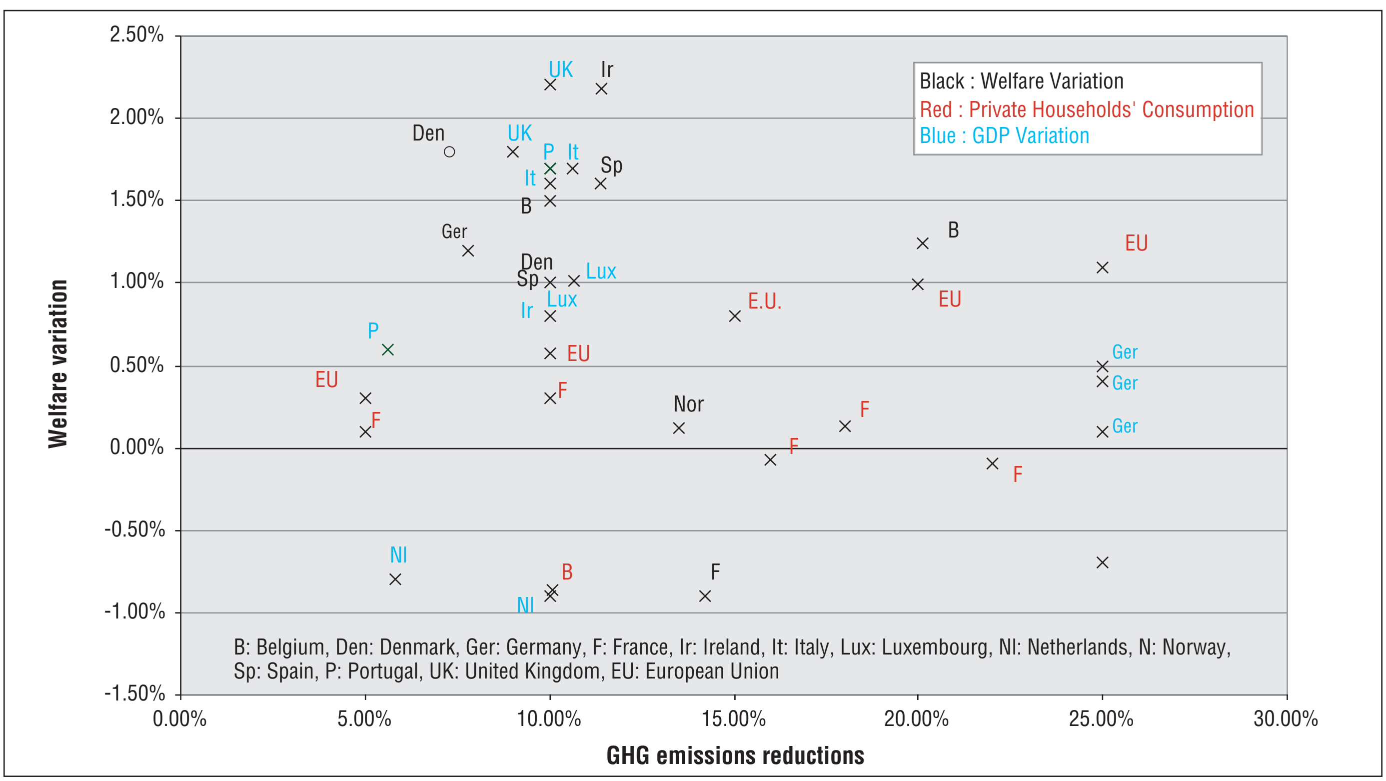

Figure 8.5: *Welfare variation with recycling through payroll taxes reduction.*

systems and in the rigidities of the labour markets. Capros *et al.* (1999b) demonstrate (*Figure 8.6*) that the increase of employment in the EU countries due to payroll tax reduction is far higher under the assumption of wage rigidities than under the assumption of a classic flexible labour market. In the same way, Bernard and Vielle (1999c) do not conclude to a strong double dividend in France, while Hourcade *et al.* (2000a) find a modest increase in total consumption of households (up to 0.2% for carbon taxes up to US$100/tC) because they incorporate structural unemployment. This is also why the E3ME model (Barker, 1999), econometrically driven and neo-Keynesian in nature, provides the most optimistic results; they indeed incorporate the rigidities of the real labour markets. It systematically finds a net increase in GDP in Europe (from 0.8% to 2.2%), except for the Netherlands, with a maximum in the UK. The DRI and LINK models, similar in nature to E3M3, do not find such a gain for the US economy, but a loss of 0.39%.

The magnitude of the double dividend for the European countries is lower in general equilibrium models than in Keynesian models: the welfare effects in different studies are between –1.35% and 0.57%. Even if these estimates cover different emission reduction levels for different time periods, they confirm the attractiveness of payroll recycling. In addition, it is remarkable that negative figures are found for small economies such as Belgium (Proost and van Regemorter, 1995) and Denmark (Andersen *et al*., 1998) in the situation of a unilateral policy, which confirms the specific interest of these countries in international coordination.

The magnitude of the second dividend (the net economic benefit of tax recycling) is not independent of the abatement target. For a given fiscal system, it is determined by parameters for which sizes vary with the taxation levels (*e.g.,* the elasticity of decarbonization in the production sector and in household consumption, the crowding-out effect between carbon-saving technological change and non-biased technological change). Unfortunately, only a few studies report the range of taxes in which the double-dividend hypothesis holds. Hourcade *et al.* (2000a) found a curve similar to A_3 in *Figure 8.3*; after an optimum around US$100/tC, the double dividend tends to vanish in the same way. Håkonsen and Mathiesen (1997) found that tax recycling is actually welfare improving in the range of a 5% to 15% reduction in CO_2 emissions. Capros *et al*. (1999c) are more optimistic in this respect. They found that the final consumption of households in the EU is increases (about 1%) when the abatement target increases from 20% to 25%. The marginal increase is, however, lower than when the abatement target increases from 5% to 10%.

8.2.2.1.3 Other Forms of Taxes Reduction

Other forms of tax reductions, such as value-added tax (VAT), capital taxes, and other indirect taxes have also been studied in addition to recycling via the national debt and public deficit reductions.

Studies for the USA confirm that the nature of the existing fiscal system matters. While no study found a strong double div-

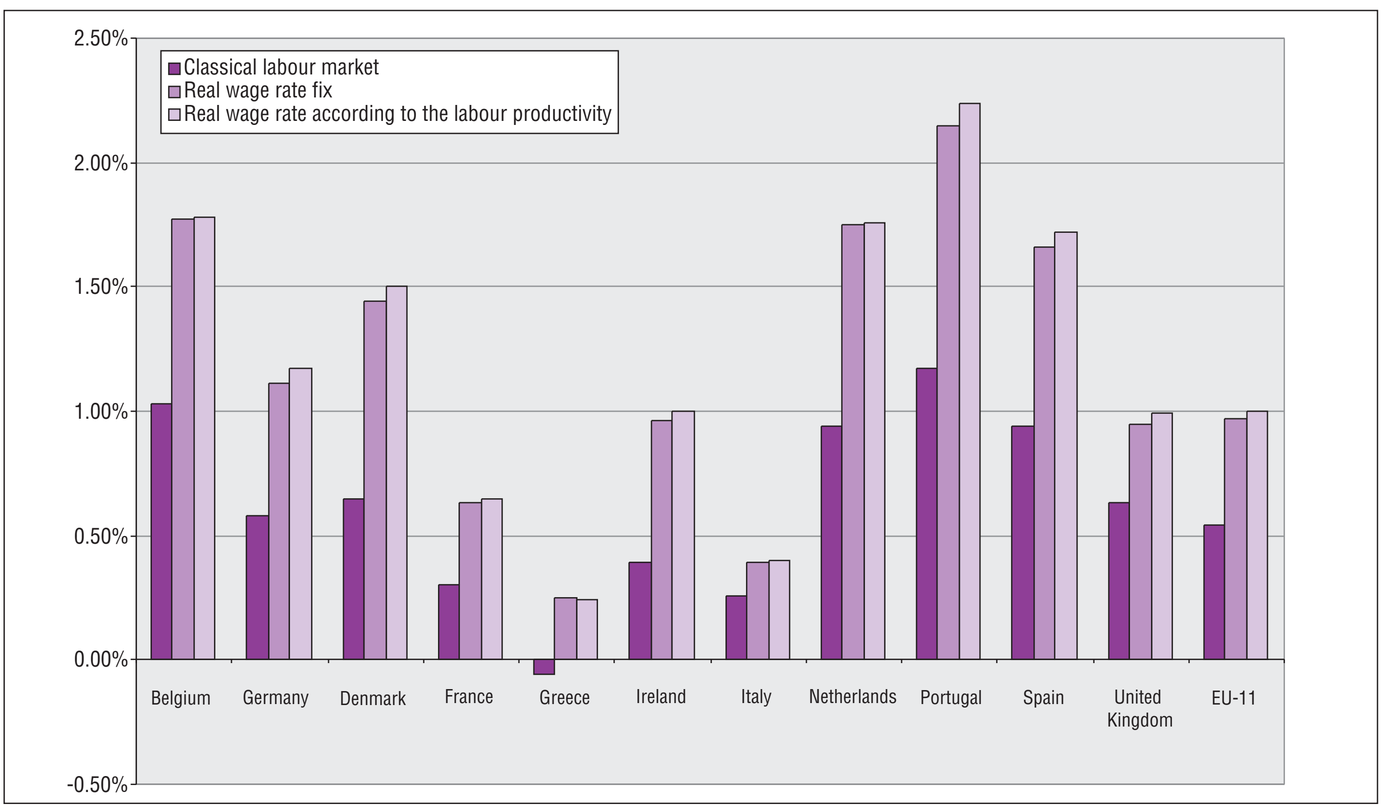

Figure 8.6: *Variation in employment.*

idend for the USA in the case of labour-tax recycling, the Jorgenson–Wilcoxen model supports this notion when recycling takes the form of a reduction in capital taxes (Shackleton, 1998). The pre-existing marginal distortions from taxes on capital are considerably larger than those from labour taxes. Consequently, according to Jorgenson (1997), if the revenues were rebated to consumers in the form of reduced taxes on wage and salary incomes, the cost would be reduced to 0.6%, or by a factor of three compared to the lump-sum recycling case. But if the taxes were rebated on capital income instead, the loss would turn into a gain (0.19%). This higher attractiveness of capital taxation recycling is not found in European countries, with the exception of the Newage model for Germany (Boehringer, 1997).

The other recycling modes have been scrutinized less systematically, but yield in general less favourable results than labour- and capital-taxation recycling. *Figure 8.7* synthetises these results. For Australia, McDougall and Dixon (1996) found that for all the scenarios in which energy taxes were used to offset reductions in payroll taxes, rises in GDP and employment were achieved. A decrease in GDP and employment resulted in the only scenario in which energy taxes were used to reduce the budget deficit. Fitz Gerald and McCoy (1992) found the same type of result for repayment of national debt in Ireland (1% GDP loss). These results are also confirmed in the German case, which is particularly interesting, because several models (Almon, 1991; Welsch and Hoster, 1995; Conrad and Schmidt, 1997; Boehringer *et al.,* 1997) simulate the same emission target (–25%) for the same year (2010) with different types of recycling. They generally conclude to a strong double dividend, and they find a significantly more pessimistic variation in welfare (–4.2% against –0.7% in Almon (1991), –0.03% against +0.1% in Conrad and Schmidt (1996)) when the revenues of the carbon tax are used to lower public deficit rather than reduce social contribution. The results are less clear concerning the relevance of recycling via a capital tax reduction in this country.

For France, Schubert and Beaumais (1998) found, for a carbon tax of US$140/tC, that these tax recycling schemes are less efficient in terms of welfare than recycling through payroll tax, because they trigger no mechanism that enhances employment and general activity. Bernard and Vielle (1999c) confirm this result for the same country. In a short-run analysis for Sweden, Brännlund and Gren (1999) found that private income remains almost unchanged if a reduction in VAT is implemented, because it compensates the regressive income effect of carbon taxation. Nevertheless, as the income increase in this study is relatively important compared to the changes in prices, taxes can be raised without altering consumer behaviour in any considerable way. But this balance may not be preserved in the case of higher carbon taxes.

There are few studies on mitigation costs and recycling for developing countries, but China is one exception. Zhang (1997, 1998) analyzed the implications of two scenarios under which China's CO_2 emissions in 2010 will be cut by 20% and 30% relative to the baseline. Gross National Product drops by 1.5% and

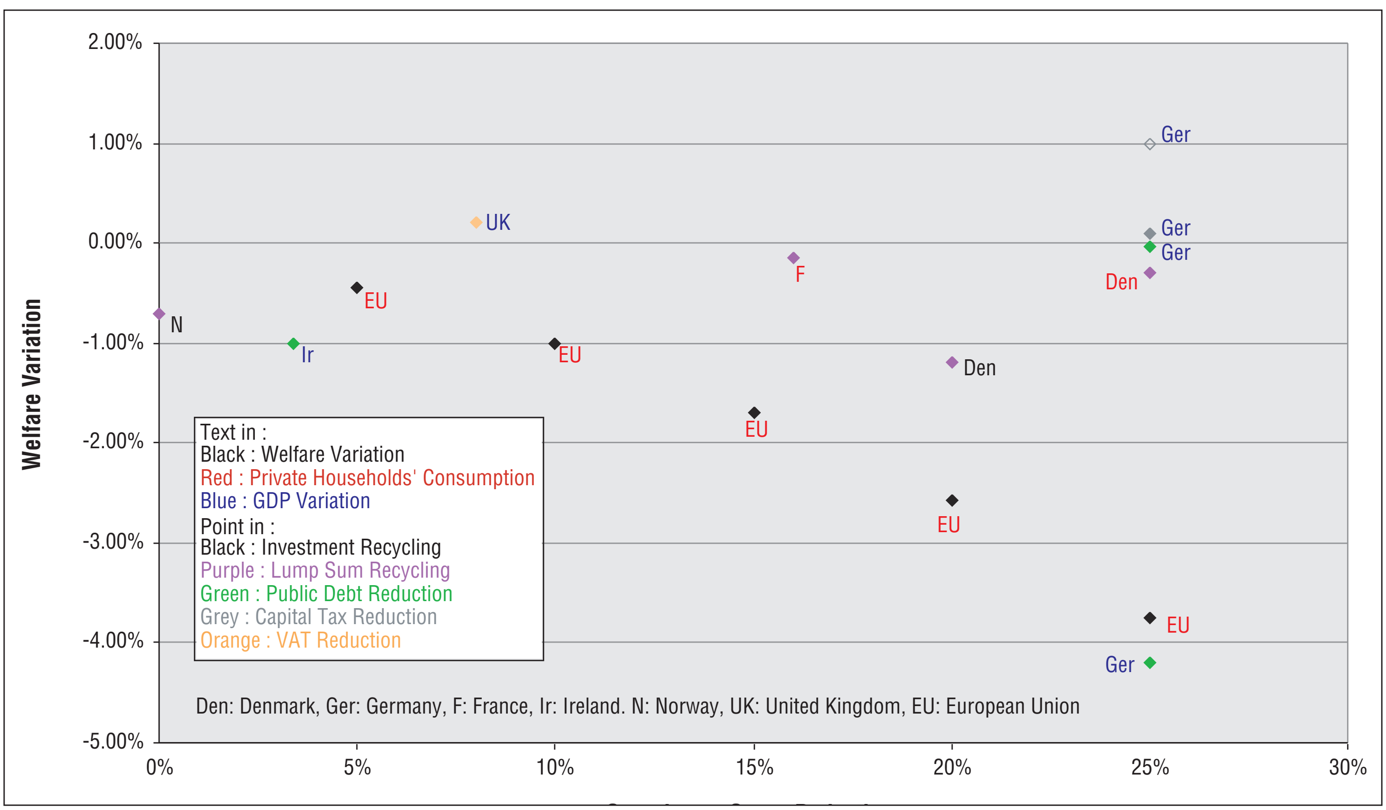

Figure 8.7: *Welfare variation with different recycling policies.*

2.8%, respectively, in 2010 relative to the baseline, and welfare, measured in Hicksian equivalent variation (defined in Chapter 7), drops by 1.1% and 1.8%. If part of the revenues raised by carbon taxes is recycled by equally reducing indirect taxes by 5% and 10%, respectively, for all sectors the welfare effect is markedly improved, and there may even be a gain. Garbaccio *et al*. (1998) report an even more optimistic view from their simulations on a dynamic CGE model for China. Uniform emissions reductions of 5%, 10%, and 15% from baseline were studied, and carbon tax revenues recycled by reducing all other taxes proportionally. In all of the alternative scenarios, a very small decline in GDP occurs in the first year of the simulation. However, in each case, GDP is increased in every year thereafter. The result arises through a shift from consumption to investment brought about indirectly through the imposition of the carbon tax. Thus, a double dividend may be achieved in China.

8.2.2.1.4 Conclusions: Interest and Limits of Aggregate Analysis

A lesson from this section is that, despite their great diversity, the findings of empirical models confirm the theoretical diagnosis. Revenue-raising instruments such as carbon taxes or auctioned emissions permits are, if properly utilized, the most efficient instrument for minimizing the aggregate welfare losses (or maximizing the welfare gains) of climate policies.

It should be noted however, that, even if the only one available study for China suggests that opportunities for revenue recycling exist in developing countries, no swapping generalization can be made at this stage. While theoretical modelling and empirical evidence suggest that such opportunities are available in many OECD countries, developing countries in many cases start from a different fiscal baseline (e.g., fewer entrenched distortionary payroll taxes). They also have other potentially underused tax bases that may become more developed as their economies grow at rates that typically exceed growth rates in OECD countries. In developing countries, direct welfare losses associated with a carbon tax may, therefore, reduce opportunities for mitigation within the fiscal reform policy envelope. At this stage, however, insufficient evidence exists either to confirm or to substantiate these hypotheses; studies to date have mainly concentrated on developed countries and their conclusions may not be directly transferable.

Beyond controversies about the capacity of government to warrant fiscal neutrality, that is the fact that the total fiscal burden remains unchanged, the adoption of carbon taxes or auctioned permits confronts the fact that their enforcement must be done in the heterogeneity of the real world, and can have very significant distributive implications:

- *Across economic sectors*. The carbon content of the steel, aluminium, cement, basic chemical, and transport industries are, indeed, four to five times higher per unit of value added than for the rest of industry. For unilateral initiatives, carbon taxes drastically impact the competitiveness of these sectors (with potential economic shocks at the regional level); even with an internationally co-ordinated policy, their equity value will be lowered compared with the rest of industry.
- *Across households income groups*. Carbon taxation increases the relative prices of energy services such as heating, lighting, and transport. The resultant impact on welfare is then more negative for low income levels and people living in cold areas and in low density areas. It is also higher for high income groups and more beneficial for medium income groups in case of swap with other taxation.

Economic analysis can define the compensation necessary to offset these negative distributional effects but, in the real world, winners cannot (or are not willing to) compensate losers. This is especially relevant when the losers suffer heavy impacts and the winners enjoy only marginal gains, which leads to the so-called political mobilization bias (Olson, 1965; Keohane and Nye, 1998) when the losers are more ready to organize a lobbying and incur mobilization costs than the winners (Williamson, 1996). Under such circumstances, policies yielding the largest aggregate net benefits may prove very difficult to enforce. Economic models provide no answer to this issue, but can try to frame the debate by providing the stakeholders with appropriate information. This is the objective of Sections 8.2.2.2 and 8.2.2.3.

8.2.2.2 Mitigating Sectoral Implications: Tax Exemptions, Grandfathered Emission Permits, and Voluntary Agreements

In all countries in which CO_2 taxes have been introduced, some sectors are exempt, or the tax is differentiated across sectors (see, e.g., ECON, 1997). Typically, households pay the full tax rate, whereas export-oriented industries pay either nothing or a symbolic rate.[3] Very few countries have actually implemented a CO_2 tax, and (unsurprisingly) tax exemptions are more systematically analyzed in these countries, such as the Scandinavian countries. Concerns about the sectoral implications of revenue-raising policies have led to four types of responses being studied:

- exemption of the most carbon-intensive activities;
- differentiating the carbon tax across sectors;
- compensation subsidies; and
- government's free provision of emissions permits to firms on a grandfathering basis or on the basis of voluntary agreements on sectoral objectives.

8.2.2.2.1 Tax Exemption

Lessons from the few modelling exercises suggest that the efficiency cost for the whole economy of offsetting the sectoral

[3] Some exceptions occur. For instance, in Norway emissions of CO_2 from oil and gas production have traditionally been charged the maximum rate.

impacts of carbon taxes through tax exemptions are very high. Böhringer and Rutherford (1997) show for Germany that exemptions to energy- and export-intensive industries increase the costs of meeting a 30% CO_2 reduction target by more than 20%. Jensen (1998) has similar findings for Denmark with respect to a unilateral reduction of CO_2 emissions by 20% (Jensen, 1998). To exempt six production sectors that emit 15% of Denmark's total emissions implies significantly greater welfare costs (equivalent variation) than full taxation to meet the same abatement target. Namely, welfare loss of 1.9% and a carbon tax on the non-exempted sectors of US\$70/tCO_2, against a welfare loss of 1.2% and a carbon price of US\$40/tCO_2 in the no-exemption case (uniform taxes). A similar result is found in Hill (1999) for Sweden: the welfare costs of using exemptions are more than 2.5 times higher than in the uniform carbon tax case for a 10% emission decrease. The high costs of tax exemption are also confirmed by a US study (Babiker *et al*., 2000).

8.2.2.2.2 Tax Differentiation

Tax differentiation is studied in a CGE model for Sweden in Bergman (1995), who compares its effect with a uniform tax for given emission targets. The tax rate applicable to the industrial firms is set to one-quarter of the tax rate for non-industrial firms and households. The GDP loss increases slightly compared to the uniform tax, but it is still quite small. However, the purchasing power of the aggregated incomes of labour and capital is significantly reduced. Consequently, tax differentiation does not seem to have as much of an adverse effect as full tax exemption. The reason is that all sectors pay a carbon tax when taxes are differentiated, while this is not the case for tax exemptions. Thus, the burden on sectors that pay the highest carbon tax is not that large, and hence results in lower welfare losses.

8.2.2.2.3 Compensating or Subsidizing Mitigation Measures

Böhringer and Rutherford (1997) as well as Hill (1999) envisage labour subsidies used to keep a given employment target. They conclude that – compared to tax exemptions for energy- and export-intensive industries – a uniform carbon tax cum wage subsidy achieves an identical level of national emission reduction and employment at a fraction of the costs.

A second option is a special case of voluntary agreements. In most of the literature, voluntary agreements result from negotiations on emission levels between public authorities and firms adversely impacted by environment policies. Carraro and Galeotti (1995) examined another form of voluntary agreement for European countries: firms receive financial benefits if they have engaged in environmental research and development (R&D) spending. This option is justified because economic tools may be inefficient in reaching the optimal R&D level, even in a pure and perfect market competition (Laffont and Tirole, 1993). According to this study, a strong double dividend could occur in all European countries except Belgium and the UK, even if the impact on employment is weak. One of the reasons for this double dividend is the technical progress induced by this policy.

8.2.2.2.4 Free Allocation of Emissions Permits

Annex B countries are currently considering the creation of a market for GHG emissions on the basis of grandfathered quotas or of quotas delivered in function of voluntary agreements of sectors to given emissions targets (see Chapter 6). This option does not generate revenue, but (contrary to tax exemptions) implies participation of the carbon-intensive industry to climate policy and does not transfer the full burden to households and the rest of industry. However, the welfare impacts are systematically found to be less favourable than under a full revenue-neutral taxation. Jensen (1998) found a welfare loss of 1.4% in Denmark and a permit price of US\$110/tCO_2, while a uniform tax to meet the same –20% target is only US\$40 and the resultant welfare loss is 1.2%. Bye and Nyborg (1999) investigated the effects on welfare (total discounted utility) of both uniform taxes and tradable permits issued freely compared to the current system of tax exemptions. To keep total tax revenues unchanged for the government, the payroll tax is adjusted accordingly. They found that a permit system gives a welfare loss of 0.03% compared to the current system, while with uniform taxes there is a gain of similar size. The main reason is that payroll taxes must increase to maintain the budget balance when carbon taxes are not used. There are similar findings for the USA. Parry *et al.* (1999) show that the net economic impact (after accounting for environmental benefits but not without climate benefits) of carbon abatement is positive when permits are auctioned, but switches to negative when permits are grandfathered.

Other allocation rules have been tested, but do not improve the result compared with grandfathered permits. For Denmark, Jensen and Rasmussen (1998) examined the aggregate welfare loss (equivalent variation) of an emission target of 80% of 1990 levels from 1999 to 2040; they found 0.1% with a permit auction, 2.0% for grandfathered permits, and 2.1% when the permits are given to firms in the proportion of market shares and sectoral emissions, similar to an output subsidy.

Such results are obtained because, in the case of free delivered permits, the interactions with the tax system occur without the compensating effect of tax-revenue recycled, as in the cases of environmental taxes and auctioned permits. Studies by Parry (1997), Goulder *et al*. (1997), Parry *et al*. (1999), and Goulder *et al*. (1999) show that the costs of quotas or marketable permits are higher if there are prior taxes on the production factors concerned than in if there are no such taxes. Quotas or permits tend indeed to raise the costs of production and the prices of output. This reduces the real return to labour and capital, and thereby exacerbates prior distortions in relevant markets and decrease the overall efficiency of the economy.

Bovenberg and Goulder (2001) found that avoiding adverse impacts on the profits and equity values in fossil fuel industries involves a relatively small efficiency cost for the economy. This arises because CO_2 abatement policies have the potential to generate revenues that are very large relative to the potential loss of profit for these industries. By enabling firms to retain only a very small fraction of these potential revenues, the government can protect the firms' profit and equity values. Thus, the government needs to grandfather only a small percentage of CO_2 emissions permits or, similarly, must exempt only a small fraction of emissions from the base of a carbon tax. This policy involves a small sacrifice of potential government revenue. Such revenue has an efficiency value because it can finance cuts in pre-existing distortionary taxes. These authors also found a very large difference between preserving firms' profits and preserving their tax payments. Offsetting producers' carbon tax payments on a dollar-for-dollar basis (through cuts in corporate tax rates, for example) substantially overcompensates firms, raising their profit and equity values significantly relative to the situation prior to the environmental regulation. This reflects that producers shift onto consumers most of the burden from a carbon tax. The efficiency costs of such policies are far greater than the costs of policies that do not overcompensate firms.

8.2.2.2.5 Conclusions

The costs of meeting the Kyoto targets are very sensitive to the type of recycling used for the revenue of carbon taxes or auctioned permits. In general, however, modelling results show that the sum of the positive revenue-recycling effect and the negative tax-interaction effect of a carbon tax or auctioned emission permits is roughly zero. Thus, in some analyses the sum is positive, while in others it is negative. In economies with an especially distortive tax system (as in several European analyses), the sum may be positive and hence confirm the strong double-dividend hypothesis. In economies with fewer distortions, such as in various models of the US economy, the sum is negative. Another conclusion is that even with no strong double-dividend effect, a country fares considerably better with a revenue-recycling policy than with one that is not revenue recycling, like grandfathered quotas. Analyses of the US economy found that revenue recycling reduces the cost of regulation by about 30%–50% for a certain range of targets, while European analyses report cost savings that are even higher than 100%.

However, at this stage insufficient evidence exists either to confirm or to substantiate these results in the context of developing countries. Studies to date have concentrated on developed countries and, while these studies are comprehensive and rigorous, their conclusions may not be directly transferable. It can be argued that, in developing countries, direct welfare losses typically associated with specific factor taxes (such as a carbon tax) may have fewer opportunities for mitigation within the fiscal-reform policy envelope. Nevertheless, the complex linkages between formal and informal sectors of the economy may show this intuition to be incorrect; the only existing study for China reviewed here suggests that this may be the case but further research is needed to confirm this more generally.

8.2.2.3 The Distributional Effects of Mitigation

A policy that leads to an efficiency gain may not improve overall welfare if some people are in a worse position than before, and vice versa. Notably, if there is a wish to reduce the income differences in a society, the effect on the income distribution should be taken into account in the assessment.

An evaluation of the distributional incidence of higher energy prices is significantly conditional upon the indicator used. Distributional impacts appear to be higher when additional costs are measured in terms of percentage of total household expenditures rather than income, and higher if current income is considered instead of lifetime income. Lifetime income is relevant in the sense that households can borrow or save, and also move between different income classes. According to Poterba (1991), a person had only a 41% chance of being in the same quintile of income distribution in 1971 and in 1978. This percentage rises to 54% if the person initially belongs to the poorest quintile. However, current income is relevant to studies on the short-term intergeneration impact of a new tax. For example, an elderly person is more adversely affected by new taxes on expenditures than are those on an income, even if subsequent generations pay the same lifetime tax bill under each factor influencing the macroeconomy.

International competition limits the ability of firms to pass the tax onto prices, thus reducing the size of the indirect distributional effect. In the same way, the degree of production factor substitution determines the extent to which the tax changes prices. Moreover, as the substitution is generally supposed to be limited in the short term, but increasing as existing plants are replaced, the distributional effect of an environmental tax changes over time. Last, but not least, the distributional effect depends basically on the utilization of the tax revenue.

Two British studies looked at distributional effects of climate policies. Barker and Johnstone (1993) investigated the distributional effects of a carbon–energy tax. Revenues are recycled through an energy efficiency programme and compared to lump-sum transfers. The results show that the burden of a carbon–energy tax falls most heavily on low-income groups. At the same time, for these low-income groups the potential gains to be realized by increasing energy efficiency are higher to offset this regressive outcome. Symons *et al.* (1994) investigated other various assumptions of revenue recycling for the UK, and found that to introduce a carbon tax without recycling or with recycling through VAT or petrol excise-duty reductions is significantly regressive. Conversely, recycling the carbon tax by a combination of VAT rate reductions and benefits reforms directed towards poorer households results in favourable distributive effects.

The conclusion is similar for other countries. For Ireland, O'Donoghue (1997) found that carbon taxation is generally regressive, but that recycling the carbon tax through a fixed basic income for all individuals allows the distributional effects to become almost neutral. For Norway, Brendemoen and Vennemo (1994) concluded that a global carbon tax of US$325/tC in 2000 and US$700/tC in 2025 (1987 prices) has no significant impact on the regional distribution of welfare. For Australia, Cornwell and Creedy (1996) found that a carbon tax only affecting households (the input–output matrix is constant in their model) is clearly regressive, but can become neutral if adequate recycling is implemented. In addition, the distributional differences across income are not affected much. On the other hand, Aasness *et al*. (1996) conclude for the same country that poor households are less favourably affected than rich households, because of smaller budget shares on consumer goods (which imply relatively more CO_2 emissions) in the rich households. Harrison and Kriström (1999) studied the general equilibrium effects of a scenario in Sweden in which the existing carbon taxes increase by 100% and labour taxes are reduced to maintain constant governmental revenue, but without removing the existing exemptions from carbon taxes. All households lose from this carbon tax (with tax exemptions) increase. They point out that the distributional effects are very dependent on the size of the household (the more affected being those with children). In a study for 11 EU member states, Barker and Kohler (1998) examined emission reductions of 10% below baseline by 2010. They concluded that the changes would be weakly regressive for nearly all the member states if revenues are used to reduce employers' taxes, and strongly progressive if they are returned lump-sum to households.

In summary, most studies show that the distributional effects of a carbon tax are regressive unless the tax revenues are used either directly or indirectly in favour of the low-income groups (see also Poterba, 1991; Barker, 1993; Hamilton and Cameron, 1994; OECD, 1995; Cornwell and Creedy, 1996; Oliveira Martins and Sturn, 1998; Smith, 1998; Fortin, 1999). This undesirable effect can be totally or partially compensated by a revenue-recycling policy if the climate policy is implemented through carbon taxes or auctioned permits.

Three other issues of distributional effects, not dealt with here, are industry sector impacts, regional effects, and how people are affected by environmental damage. For instance, a tax on CO_2 emissions obviously leads to very different effects in energy-intensive industries than in sectors producing labour-intensive services (see Chapter 9). In addition, the poor household generally lives in the most polluted area and then benefits first from the amelioration of air quality induced by GHG reduction policy (see Section 8.2.4).

8.2.3 The Impact of Considering Multiple Gases and Carbon Sinks

The overwhelming majority of T-D mitigation studies concentrate upon CO_2 abatement from fossil fuel consumption, while an increasing number of B-U studies tend to incorporate all the GHG emissions from the energy sector, but still not include emissions from the agricultural sector and sequestration. However, the Kyoto Protocol also includes methane (CH_4), nitrous oxide (N_2O), perfluorocarbons, hydrofluorocarbons, and sulphur hexafluoride (SF_6) as gases subject to control. The Protocol also allows credit for carbon sinks that result from direct, human-induced afforestation and reforestation measures taken after 1990. This may have significant impacts on abatement costs.

A recent study (Reilly *et al*., 1999) estimated the mitigation costs for the USA and included consideration of all of these gases and forest sinks. The study assumes that the Kyoto Protocol is ratified in the USA and implemented with a cap and trade policy. The analysis considers the effects of including the other gases in the Kyoto Protocol in terms of the effect on allowable emissions, reference emissions, the required reduction, and the cost of control.

For the USA, the authors estimate that base year (1990) emissions were 1,654MtC_{eq}, converting non-CO_2 gases to carbon equivalent units using 100-year global warming potential indices (GWPs) as prescribed in the Kyoto Protocol. This compares with 1,362MtC_{eq} for carbon emissions alone. The result is that allowable emissions are 1,539MtC_{eq} in the multigas case compared with 1,267MtC_{eq} if other gases had not been included in the agreement.

The authors also projected emissions of other gases to grow substantially through 2010 in the absence of GHG control policies, so that total emissions in the reference case reach 2,188MtC_{eq} compared with 1,838MtC_{eq} of carbon only. The combination of these factors means that the required reduction is 650MtC_{eq} in the multigas case compared with 571MtC_{eq} if only carbon is subject to control. To analyze the impact of including the other gases in the Kyoto Protocol the authors consider three policy cases:

- *Case 1*, *fossil CO_2 target and control*. This case includes only CO_2 in determining the allowable emissions under the Kyoto Protocol and includes only emissions reductions of CO_2, unlike the requirements in the Kyoto Protocol that require consideration of multiple gases.
- *Case 2, multigas target with control on CO_2 emissions only*. This case is constructed with the multigas target (expressed as carbon equivalents using GWPs) as described in the Kyoto Protocol, but only carbon emissions from fossil fuels are controlled.
- *Case 3, multigas target and controls*. The multigas Kyoto target applies and the Parties seek the least-cost control across all gases and carbon sinks.

Case 1 is thus comparable to many other studies that only consider CO_2 and provides an approximate ability to normalize results with other studies. For Case 1 the resultant carbon price is US$187 in 1985 price (US$269 in 1997 price). Case 2 illustrates that, if the USA does not adopt measures that take advantage of abatement options in other gases and sinks, the cost could be significantly higher (US$229 in 1985 price or US$330 in 1997 price). In 1997 US$, the total cost in terms of reduced output is estimated to be US$54 billion for Case 1, US$66 billion in Case 2, and US$40 billion in Case 3.

By comparison with Case 1, the introduction of all gases and the forest sink results in a 20% decline in the carbon price to US$150 (1985 price, US$216 in 1997 price).

Cases 2 and 3 are comparable in the sense that they nominally achieve the same reduction in GHGs (when weighted using 100-year GWPs). Thus, for a comparable control level, the multigas control strategy is estimated to reduce US total costs by nearly 40%.

The Reilly *et al*. (1999) study did not conduct sensitivity analyses of the control costs, but noted the wide range of uncertainties in any costs estimates. Both base year inventories and future emissions of other GHGs are uncertain, more so than for CO_2 emissions from fossil fuels. Moreover, some thought will be required to include other GHGs and sinks within a flexible market mechanism such as a cap and trade system. Measuring and monitoring emissions of other GHGs and sinks could add to the cost of controlling them and so reduce the abatement potential.

8.2.4 Ancillary Benefits

"Co-benefits"[4] are the benefits from policy options implemented for various reasons at the same time, acknowledging that most policies resulting in GHG mitigation also have other, often at least equally important, rationales. "Ancillary benefits" are the monetized secondary, or side benefits of mitigation policies on problems such as reductions in local air pollution associated with the reduction of fossil fuels, and possibly indirect effects on congestion, land quality, employment, and fuel security. These are sometimes referred to as "ancillary impacts" to reflect that these impacts may be either positive or negative. *Figure 8.8* shows the conceptual framework for analyzing ancillary and co-benefits and costs. The figure shows that climate and social/environmental benefits can be direct benefits, ancillary benefits, or co-benefits, depending on the objectives of the policies.

The term co-benefits is used in this report despite its limited literature because it shows the case for an integrated approach, linking climate change mitigation to the achievement of sustainable development. However, there appear to be three classes of literature regarding the impacts of climate change mitigation: (1) literature that primarily looks at climate change mitigation, but that recognizes there may be benefits in other areas (illustrated in the top panel of *Figure 8.8*); (2) literature that primarily focuses on other areas, such as air pollution mitigation, and recognizes there may be "ancillary benefits" in the area of climate mitigation (illustrated in the centre panel of *Figure 8.8*), (3) literature that looks at the combination of policy objectives and examines the costs and benefits from an integrated perspective (illustrated in the bottom panel of *Figure 8.8*). In this report, the term "co-benefits" is used when speaking generically about this latter perspective and when reviewing class (3) literature. The term "ancillary benefits" is used when addressing class (1) and (2) literature. This section covers primarily class (1) literature, which is the most extensive.

Very few economic modelling studies that examine the impacts on economic welfare of various GHG abatement policies explicitly consider their ancillary consequences, *i.e*. effects which would not have occurred in the absence of specific GHG policies. These range from public health benefits through reduced air pollution to reduced CH_4 from animal farms, and through impacts on biodiversity, materials, or land use (see Rothman, 2000).

Existing studies provide evaluations of net ancillary benefits ranging from a small fraction of GHG mitigation costs to more than offsetting them (see Burtraw *et al*., 1999, and reviews by Pearce, 2000; Burtraw and Toman, 1997; and Ekins, 1996). Such variation in estimates is not surprising because the underlying features differ by sectors considered and the geographic area being studied; but this variation also reflects the lack of agreement on the definition, reach, and size of these impacts and on the methodologies to estimate them. This literature is growing, particularly with respect to the impacts on public health[5,6], so a critical review of it is given in this section. Most of the studies reviewed focus on public health, which is the largest quantifiable impact; therefore this assessment also focuses on it. Ancillary impacts to specific economic sectors are reviewed in Chapter 9.

Most of the key ancillary benefits quantified to date are relatively short term and 'local', that is affecting the communities relatively close to the sources of the emissions changes. In both these respects ancillary benefits can be thought of as offsetting all or part of the welfare losses associated with the costs of reducing GHGs. In this regard the best measure of ancillary

[4] See Chapter 7 for a more formal definition of ancillary and co-benefits and costs of GHG mitigation.

[5] A number of possibly important side benefits are not amenable as yet to either quantitative or economic analyses (e.g., ecosystem damages, biodiversity loss).

[6] In SAR, IPCC estimated that, for European countries and the USA, benefits such as reduced air pollution could offset between 30% and 100% of the abatement costs (IPCC, 1996, p. 218). These estimates were controversial and not supported by a standardized methodology. After SAR, extensive debates arose regarding suitable costing methods to quantify the relative economic impacts of various policies in distinct regions, with as yet no consensus on the most suitable methods to be employed (Grubb et al., 1999). However, a consensus is now beginning to emerge on how to quantify some ancillary benefits. See OECD, Proceedings from Workshop on the Ancillary Benefits and Costs of Climate Change Mitigation (OECD, 2000).

impacts may be the percentage (or absolute) variation in welfare loss from considering a carbon tax (or other instrument) that does not include direct climate-mitigation benefits. Few studies provide such estimates (Dessus and O'Connor (1999), is an exception).

Other metrics in the literature help to shed light on the size and uncertainty associated with ancillary impacts estimates. The first normalizes ancillary benefits with carbon reductions, that is, ancillary benefits per tonne of carbon reduced (e.g., Burtraw *et al*., 1999). The second is the average ancillary benefits per tonne as a fraction of the carbon tax. This latter measure is useful because it has some linkage to the net benefits question. Private marginal carbon mitigation costs are equalized to the tax in the models in the literature. Given that average mitigation costs are less than (or equal to) marginal costs, if the met-

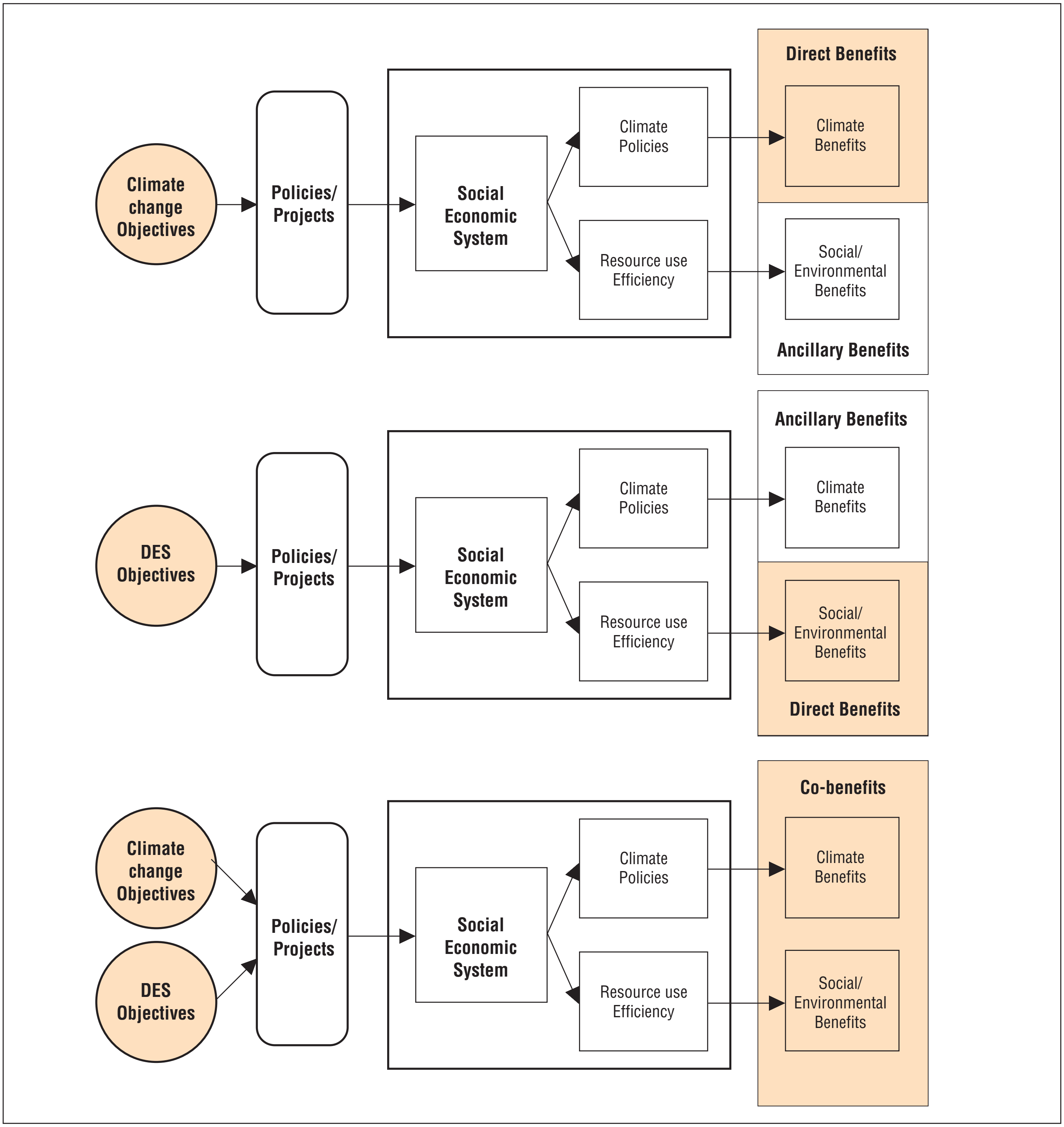

Figure 8.8: *Conceptual framework for analyzing ancillary and co-benefits and costs.*

ric equals more than one then the carbon policy modelled has net private benefits even without counting the direct climate benefits. If (average) ancillary benefits are lower than private marginal costs no claim for net benefits can be made. However, the lower the fraction, the less likely the policy will have net benefits.

A few important caveats are, however, in order. The most important is that the relevant cost measure in the above fraction is *social* not *private* cost. In this case, marginal *social* costs are likely to exceed marginal *private* costs because of tax interaction effects. Thus, ancillary benefits may not exceed social marginal cost even if the former exceeds private marginal cost (equal to the carbon tax). Second, ancillary benefit (cost) measures need to measure social welfare gains (or losses) if they are to be comparable to losses on the mitigation side. But, many measures of ancillary benefits understate social welfare gains and other benefits remain unmonetized or even unquantified, while in other cases, the ancillary benefits overstate welfare gains (say by counting all traffic fatality reductions as external benefits). Thus, *reported* ancillary benefits can under- or overstate actual ancillary benefits. If this indicator is greater than one, then the carbon policy has net private benefits even without counting the direct climate benefits.

The section reviews some of the recent studies estimating ancillary benefits of GHG mitigation policies. The studies are briefly described and examined for the credibility of their methods and estimates.

8.2.4.1 The Evaluation of the Ancillary Public Health Impacts

Studies estimating ancillary public health impacts from climate policies were examined, relying on three surveys of this literature (Ekins, 1996; Burtraw *et al*., 1999; Kverndokk and Rosendahl, 2000) and on summaries of the older literature, supplemented by some of the newer studies. *Table 8.5* provides a description of each study, as well as the estimates of ancillary benefits per tonne of carbon. *Table 8.6* summarizes the modelling choices of the studies reviewed.

The Burtraw *et al*. (1999) review of US ancillary benefit studies of public health impacts linked to mitigation policies applied to the electricity sector came to several important conclusions:

- Estimates from early studies of ancillary benefits tended to exceed later ones because of the former's use of more crude and less disaggregate modelling.
- Studies that did not factor into the baseline the reductions in conventional pollutants required under the 1990 Clean Air Act estimated benefits an order of magnitude larger than the studies that did include the 1990 Clean Air Act in the baseline. Analyzing Ekins (1996), Burtraw *et al*. (1999) found that whether the Second Sulphur Protocol is added to the baseline or not can alter the estimate of ancillary benefit by over 30%.
- Some studies did not consider the "bounceback" effect (*i.e*., the offsetting increase in conventional pollutants) when a less carbon-intensive technology is substituted for a more intensive one in response to a carbon mitigation policy.
- Ancillary benefit estimates are very sensitive to assumptions about the mortality risk coefficient and the value of statistical life (VSL). Routine values used in the literature can lead to a difference of 300% in ancillary benefit estimates.
- Burtraw *et al*. (1999) and earlier studies to reconcile US and European estimates for the social costs of fuel cycles found that population density differences between Europe and the USA account for 2 to 3 times larger benefit estimates in Europe. Also, the fact that much East Coast US pollution is blown out to sea while European pollution is blown inland can account for large ancillary benefit differences.
- With a cap on SO_2 emissions, abatement cost savings are considered ancillary benefits of a carbon policy unless the reductions are so large that the cap becomes non-binding. When this happens, with SO_2 effects on mortality being as large as they appear to be, ancillary benefits increase in a discontinuous and rapid fashion, as the health benefits begin to be counted.

Kverndokk and Rosendahl (2000) review much of the recent ancillary benefit literature in the Nordic countries, UK, and Ireland, concluding that benefits are of the same order of magnitude as gross (*i.e*., private) mitigation costs. They also conclude that the benefits should be viewed as highly uncertain, because of the use of simplistic tools and transfers of dose–response and valuation functions from studies done in other countries. They point out that most of the Norwegian studies use expert judgement instead of established dose–response functions and estimates of national damages per tonne rather than distinguishing where emissions changes occur and exposures are reduced. Also, they point out that large differences in ancillary benefits per tonne across several Norwegian studies can be attributed to differences in energy demand and energy substitution elasticities. If energy production is reduced rather than switched to less carbon-intensive fuels, ancillary benefits will be far larger. Kverndokk and Rosendahl (2000) point out also that studies that feed environmental benefits back into the economic model add significantly to ancillary benefits.

8.2.4.2 Summarizing the Ancillary Benefit Estimates

8.2.4.2.1 Presentation of the Studies

Figure 8.9 summarizes the ancillary benefits per tonne of carbon from 15 studies, along with available confidence intervals around the mid estimate. Multiple entries for a study on the Figure result from modelling of multiple policy scenarios. Most of the studies focus solely on public health impacts.

Table 8.5: *Scenarios and results of studies on ancillary benefit reviewed*

Study	Area and sectors	Scenarios (1996 US$)	Average ancillary benefits (US$/tC; 1996 US$)	Key pollutants	Major endpoints
Dessus and O'Connor, 1999	Chile (benefits in Santiago only)	Tax of US$67 (10% carbon reduction) Tax of US$157 (20% carbon reduction) Tax of US$284 (30% carbon reduction)	251 254 267	Seven air pollutants	Health—morbidity and mortality, IQ (from lead reduction)
Cifuentes *et al*., 2000	Santiago, Chile	Energy efficiency	62	SO_2, NO_x, CO, NMHC Indirect estimations for PM_{10} and resuspended dust	Health
Garbaccio *et al*., 2000	China – 29 sectors (4 energy)	Tax of US$1/tC Tax of US$2/tC	52 52	PM_{10}, SO_2	Health
Wang and Smith, 1999	China – power and household sectors	Supply-side energy efficiency improvement Least-cost per unit global-warming-reduction fuel substitution Least-cost per unit human-air-pollution-exposure-reduction fuel substitution		PM, SO_2	Health
Aunan *et al*. 2000, Kanudia and Loulou , 1998a	Hungary	Energy Conservation Programme	508	TSP, SO_2, NO_x, CO, VOC, CO_2, CH_4, N_2O	Health effects; materials damage; vegetation damage
Brendemoen and Vennemo, 1994	Norway	Tax US$840/tC	246	SO_2, NO_x, CO, VOC, CO_2, CH_4, N_2O, Particulates	Indirect: health costs; lost recreational value from lakes and forests, ; corrosion Direct: traffic noise, road maintenance, congestion, accidents
Barker and Rosendahl, 2000	Western Europe (19 regions)	Tax US$161/tC	153	SO_2, NO_x, PM_{10}	Human and animal health and welfare, materials, buildings and other physical capital, vegetation

(continued)

Table 8.5: *continued*

Study	Area and sectors	Scenarios (1996 US$)	Average ancillary benefits (US$/tC; 1996 US$)	Key pollutants	Major endpoints
Scheraga and Leary, 1993	USA	US$144/tC	41	TSP, PM_{10}, SO_x, NO_x, CO, VOC, CO_2, Pb	Health – morbidity and mortality
Boyd *et al.*, 1995	USA	US$9/tC	40	Pb, PM, SO_x, SO_4, O_3	Health, visibility
Abt Associates and Pechan-Avanti Group, 1999	USA	Tax US$30/tC Tax US$67/tC	8 68	Criteria pollutants	Health – mortality and illness; Visibility and household soiling (materials damage)
Burtraw *et al.*, 1999	USA	Tax US$10/tC Tax US$25/tC Tax US$50/tC	3 2 2	SO_2, NO_x	Health

NMHC, non-methane hydrocarbons; PM, particulate matter; PM_{10}, particulate matter less than 10 microns; TSP, total suspended particulate; VOC, volatile organic compounds; IQ, intelligence quotient

Table 8.6: *Modelling choices of studies on valuation of ancillary benefits reviewed*[21]

Study	Baseline (as of 2010)	Economic modelling	Air pollution modelling	Valuation	Uncertainty treatment
Dessus and O'Connor, 1999	4.5%/yr economic growth; AEEI: 1% Energy consumption: 3.6% PM: 1% Pb: 4.1% CO: 4.8%	Dynamic CGE	Assumed proportionality between emissions and ambient concentrations	Benefits transfer used: PPP of 80% US VSL: $2.1 mill. VCB: $0.2 mill. IQ loss: $2500/point	Sensitivity tests on WTP and energy substitution elasticities
Cifuentes *et al*., 2000	For AP control, considers implementation of Santiago Decontamination Plan (1998 to 2011)	No economic modelling Only measures with private, non-positive costs considered	Two models for changes in $PM_{2.5}$ concentrations: (1) Box model, which relates SO_2 and CO_2 to $PM_{2.5}$ (2) Simple model assumes proportionality between $PM_{2.5}$ concentrations apportioned to dust, SO_2, NO_x, and primary PM emissions. Models derived with Santiago-specific data and applied to nation	Benefits transfer from US values, using ratio of income/capita Uses original value for mortality decreased by 1 standard deviation VSL = US$407,000 in 2000	Parameter uncertainty through Monte Carlo simulation. Reports centre value and 95% CI
Garbaccio *et al*., 2000	1995 to 2040 5.9% annual GDP growth rate; carbon doubles in 15 years; PM grows at a bit more than 1%/yr	Dynamic CGE model; 29 sectors; Trend to US energy/consumption patterns; Labour perfectly mobile; Reduce other taxes; Two-tier economy explicit.	Emissions/concentration coefficients from Lvovsky and Hughs (1998); three stack heights	Valuation coefficients from Lvovsky and Hughs (1998); VSL: US$3.6 million (1995) to RMB 82,700 Yuan (RMB 8.3 yuan = $1) in 2010 (income elasticity = 1). 5%/yr increase in VCB to US$72,000	Sensitivity analysis
Wang and Smith, 1999		No economic modelling	Gaussian plume	Benefit transfer using PPP. VSL = US$123,700, 1/24 of US value	

[21] AEEI, Autonomous Energy Efficiency Improvement; $PM_{10,\ 2.5}$, particulate matter less than 10 or 2.5 microns, respectively; CGE, Computable General Equilibrium Model; PPP, Purchasing Power Parity; W TP, Willingness To Pay; AP: air pollution; CAA: Clean Air Act; NAAQS: National Ambient Air Qaulity Standards; SIP: State Implementation Plan; CRF: concenrtration-response function; CL: confidence level; VSLY and VSL: Value of Statistical Life Year, Value of Statistical Life; RIA: Regulatory Impact Analysis; VCB, value of a case of bronchitis.

(continued)

Table 8.6: *continued*

Study	Baseline (as of 2010)	Economic modelling	Air pollution modelling	Valuation	Uncertainty treatment
Aunan *et al.*, 2000	Assumes status quo emissions scenario	Two analyses: bottom-up approach and macroeconomic modelling	Assumes proportionality between emissions and concentrations	Benefit transfer of US and European values using "relative income" = wage ratios of 0.16	Explicit consideration through Monte Carlo simulation Reports centre value and low, high
Brendemoen and Vennemo, 1994	2025 rather than 2010 2%/yr economic growth 1% increase in energy prices 1%–1.5% increase in electricity and fuel demand CO_2 grows 1.2% until 2000, and 2% thereafter.	Dynamic CGE		Health costs of studies reviewed based on expert panel recommendations Contingent valuation used for recreational values	Assume independent and uniform distributions
Barker and Rosendahl, 2000	SO_2, NO_x, PM expected to fall by about 71%, 46%, 11% from 1994 to 2010	E3ME Econometric Model for Europe		US$/emissions coefficients by country from EXTERNE: €1,500/t NO_x for ozone (€1= $1); NO_x and SO_2 coefficients are about equivalent, ranging from about €2,000/t to €16,000/t; PM effects are larger (2,000–25,000) Uses VSLY rather than VSL: €100,000 (1990)	
Scheraga and Leary, 1993	1990 to 2010 7% growth rate carbon emissions Range for criteria Pollutants 1%–7%/yr	Dynamic CGE			
Boyd *et al.*, 1995	Static CGE			US$/emissions coefficients	

(continued)

Table 8.6: *continued*

Study	Baseline (as of 2010)	Economic modelling	Air pollution modelling	Valuation	Uncertainty treatment
Abt Associates and Pechan-Avanti Group, 1999	2010 baseline scenarios – 2010 CAA baseline emission database for all sectors, plus at least partial attainment of the new NAAQS assumed. Benefits include coming closer to attainment of these standards for areas that would not reach them otherwise. Includes NO_x SIP call	Static CGE		From Criteria Air Pollutant Modelling System (used in USEPA Regulatory Impact Analysis and elsewhere)	SO_2 sensitivity – SO_2 emissions may not go beyond Title IV requirements – NO_x sensitivity – NO_x SIP call reductions not included in final SIP call rule
Burtraw *et al*., 1999	Incorporates SO_2 trading and NO_x SIP call in baseline	Dynamic regionally specific electricity sector simulation model with transmission constraints. The model calculates market equilibrium by season and time of day for three customer classes at the regional level, with power trading between regions.	NO_x and SO_2. Account for conversion of NO_x to nitrate particulates	Tracking and Analysis Framework: the numbers used to value these effects are similar to those used in recent Regulatory Impact Analysis by the USEPA.	Monte Carlo simulation for CRF and valuation stages.

Box. 8.1. Global Public Health Effects of Greenhouse Gas Mitigation Policies

It is useful to estimate ancillary benefits through quantitative indicators, even if they are not monetized (Pearce, 2000). One such global scale effort was produced by the WHO/WRI/EPA Working Group on Public Health and Fossil Fuel Combustion on the range of avoidable deaths that could arise between 2000 and 2020 under current policies, and under the scenario proposed by the EU in 1995. This EU Scenario assumed that by 2010 GHG emissions would be 15% below 1990 levels for Annex I countries, and 10% below projected emissions for 2010 for non-Annex I countries (Davis, 1997; Working Group on Public Health and Fossil Fuel Combustion, 1997). The total change in carbon emissions was estimated globally, based on a source–receptor matrix for four specific sectors (industry, transport, household, and energy) that was adjusted for local temperature and humidity. Applied to nine regions and adjusted for temperature and humidity, this matrix yielded changes in projected fuel types and formed the basis for calculating total emissions of particulates. Mortality tied with particulates was calculated based on best estimates (Borja-Aburto *et al*., 1997, 1998; Pereira *et al*., 1998; Gold *et al*., 1999; Braga *et al*., 1999; Linn *et al*., 2000).

The report included a sensitivity analysis of the range of deaths, predicting that by 2020, 700,000 avoidable deaths (90% Confidence Interval, 385,000–1,034,000) will occur annually as a result of additional particulate matter (PM) exposure under the baseline forecasts when compared with the climate policy scenario. From 2000 to 2020, the cumulative impact on public health related to the difference in PM exposure could reach 8 million deaths globally (90% CI, 4.4–11.9 million). In the USA alone, the number of annual deaths from PM exposure in 2020 (without control policy) would equal in magnitude deaths associated with human immunodeficiency diseases or all liver diseases in 1995. "The mortality estimates are indicative of the magnitude of the potential health benefits of the climate-policy scenario examined and are not precise predictions of avoidable deaths. While characterized by considerable uncertainty, the short-term public-health impacts of reduced PM exposure associated with greenhouse-gas reductions are likely to be substantial even under the most conservative set of assumptions."

The framework for this assessment is described in more detail in Abt Associates (1997); Pechan and Associates (1997).

From *Figure 8.9*, it can be observed observed that:

- midpoint estimates are mostly less than US$100/tC, but range from less than US$2 up to almost US$500/tC;
- US estimates are the lowest while estimates from one study for Chile and several for Norway are the highest (the latter includes a broader range of benefits);
- significant divergence in estimates occurs across studies for the same country; and
- uncertainty bounds are quite large for most of the studies that report them.

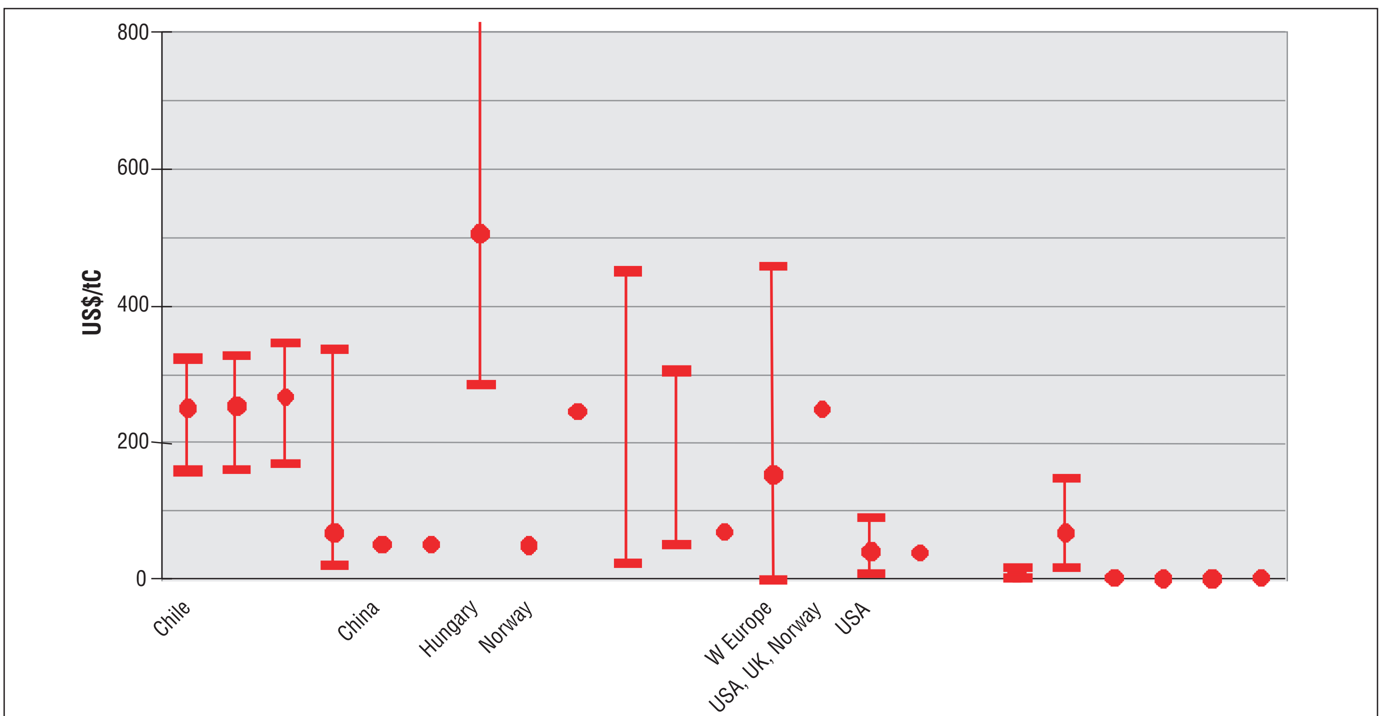

Figure 8.9: *Summary of ancillary benefits estimates in 1996 US$/tC.*

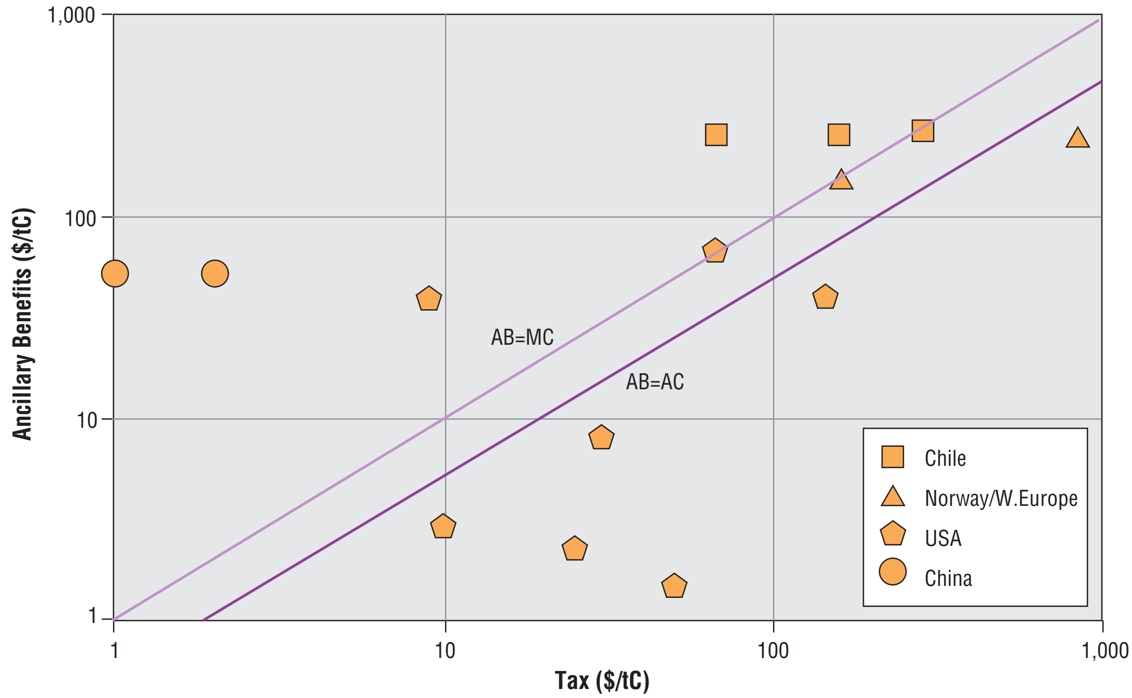

Figure 8.10: *Ancillary benefits in 1996 US$/tC versus levels of carbon tax.*

Figure 8.10 provides ancillary benefits per tonne estimates related to the size of the carbon tax (in 1996 US$/tC). Points on the diagonal line AB = MC indicate that marginal private mitigation costs (MC) equate to the tax. Some points fall on this line; more appear above it than below, with the Norwegian/Western Europe and the US studies split. If the damage (benefit) function is linear, then average benefits equate marginal benefits. Thus, points on the diagonal imply that the carbon tax is "quasi-optimal" (Dessus and O'Connor, 1999), in that the tax is optimal without considering either the direct climate mitigation benefits or any social costs over private costs (such as deadweight losses from the tax interaction effect). Alternatively, it can be assumed that the private mitigation cost function is quadratic (Total Cost=b(X^2)), where X is carbon reduction. In this case, the tax rate equals marginal private mitigation cost and average private mitigation cost is half marginal private mitigation cost. The heavy diagonal line equates ancillary benefits to average private mitigation cost. Points above this line imply there are net benefits to carbon policy, with the same important caveats as above. More points appear above the corresponding line (AB=AC) on the graph than above the AB=MC line.

In the general case, a larger carbon tax should lead to progressively smaller carbon reductions (if the marginal abatement cost curve is upward sloping). For all but one study (Abt Associates and Pechan-Avanti Group, 1999), the ratio of ancillary benefits to the tax rate does fall. As for the change in ancillary benefits per tonne of carbon, Burtraw *et al*. (1999) show this ratio falling dramatically in percentage terms with higher carbon taxes. In contrast, Dessus and O'Connor (1999) show it rising slightly, and in the Abt study the ratio of benefits to the tax rate rises dramatically (Abt Associates and Pechan-Avanti Group, 1999). This last result reflects that this analysis treated the SO_2 cap as non-binding considerably below the higher tax-rate modelled. In addition, this study treated the National Ambient Air Quality Standards as a cap, with abatement below these "caps" treated as benefits, but reductions above these caps treated as saving abatement costs.

8.2.4.2.2 Evaluation of the Studies

Almost all the studies analyze the effects of a GHG reduction policy through a tax on carbon. The ranges of the tax are from modest levels (RMB Y9/tC[7]) in 2010 for Garbaccio *et al.* (2000), US$10/tC for Burtraw *et al*. (1999); up to high levels (US$254/tC for Dessus and O'Connor (1999), and US$840/tC for Brendemoen and Vennemo (1994). The US studies employ relatively modest taxes, between US$10/tC and US$67/tC. Only two studies consider alternative programmes: Aunan *et al*. (2000) considers a National Efficiency Programme, and Cifuentes *et al*. (2000) considers energy efficiency improvements. The level of abatement considered by these two studies is relatively modest, however.

Baseline

An analysis of ancillary benefits requires a time line and a clear definition of the key constituents of the baseline against which the prospective scenario can be measured, including the eco-

[7] The exchange rate is US$1 vs Remminbi yuan (RMB Y) 8.3.

nomic, demographic, regulatory[8], environmental[9], and technological conditions, and their implications for emissions or other inputs to an ancillary benefit calculation.

The importance of the baseline is evident in a review of previous studies for the USA in Burtraw *et al*. (1999). Assessments varied in their estimates of ancillary benefits, chiefly because they employed different assumptions regarding the regulatory baselines, that is the 1990 US Clean Air Act Amendments and, especially, the tradable permit programme for SO_2. Among these baseline parameters, the most critical are the spatial location of emissions relevant to potentially exposed populations, regulatory conditions, and available technologies (Morgenstern, 2000). The importance of the location of emission reductions and exposed populations means that highly disaggregated models are the preferred tools of analysis. This may conflict with other goals for the analysis of GHG mitigation strategies. For example, large CGE models, which are used for cost estimation, operate at a different scale than the more localized models relevant to estimating ancillary benefits.

Economic Modelling

Most of the studies in *Table 8.5* use static or dynamic CGE models (one uses an econometric model) that provide T-D and sectorally aggregate estimates of ancillary benefits and/or costs. The Burtraw *et al*. (1999) model stands out for the location specificity of its economic model (although only for the electricity sector), which permits more credible modelling of population exposure reductions than that from spatially aggregate models. Another specific feature is its detailed representation of investment choices and their dependence on other factors covered in the model. Finally, several studies do not use an economic model. Instead, they follow a B-U approach, positing some increase in energy efficiency or reduction in carbon and estimating the ancillary benefits that would result, at a reasonably detailed spatial level. Such studies suffer from not accounting for behavioural adjustments, such as energy substitutions, which could alter their estimates of ancillary benefits considerably. The high ratio of ancillary benefits to the carbon tax for Garbaccio *et al*. (1999) appears to arise from very optimistic assumptions about energy substitution elasticities.

[8] For example, if they are implemented, the recent proposed tightening of the US standards for ozone and particulates and associated improvements over time imply that benefits from reductions in the criteria air pollutants that result from climate policies will be smaller in the future than if carried out now.

[9] Some environmental effects exhibit thresholds or non-linearities that imply benefits do not move directly with reductions in local and regional pollutants. Acidification is an interesting example because damage may result only after critical load thresholds are violated. On the other hand, recovery may not occur with a reduction in conventional pollutants until some new threshold is achieved or after a significant time lag.

Emissions and Environmental Media Modelling

All the studies in *Table 8.5* account for the most important pollutant affecting public health – particulates. Most, however, do not consider secondary particulate formation from SO_2 and NO_x, or do so in a very simplistic manner. In a developing country, direct particulate emissions are likely to be a large fraction of particulate mass, making the lack of attention to secondary products less important. In developed countries, however, secondary products are likely to be far more important than primary particulates. Omitting these products could bias ancillary benefit estimates downwards; using proportionality assumptions or other simple approaches raises uncertainties and may carry biases. Only one study considered lead emissions (Dessus and O'Connoer, 1999); few address ozone.

The Abt study (Abt Associates and Pechan-Avanti Group, 1999) is the most comprehensive in its modelling of secondary particulate formation and dispersion. It found that 12 urban areas in the USA would come into compliance with the recently promulgated standard for particulate matter less than 2.5 microns ($PM_{2.5}$)[10] for a carbon tax of US$67 (US$1996). Without this tax, these areas would not be able to meet the new standard. With there being at best sparse information on the actual $PM_{2.5}$ concentrations in US urban areas, these estimates should be viewed as highly speculative.

Health Effects Modelling

Three recent studies (Hagler-Bailly 1995; Lee *et al*., 1995; European Commission, 1999) developed methods that set the stage for much of the recent estimates of ancillary benefits. However, studies that draw on this literature, but reduce its information to coefficients that link emissions directly to health effects (or values) ignore spatial and demographic heterogeneity. This is particularly so when such coefficients are generated for one country or region and then directly applied to another, without taking into account local conditions. In the absence of country-specific information, transfer of risk information may be made between countries, with appropriate caveats to take into account underlying differences in health status, access to care, and other important factors (see *Box 8.2*).

Most of the studies rely on concentration–response functions from the literature on health, and apply them using a standard methodology (Ostro, 1996; EPA, 1999). The most important health effects are premature mortality and chronic respiratory effects.

Aside from differences in the base rates of the effects[11], due to local characteristics such as the age distribution of the population and health care services, other factors help explain the different outcomes of the studies. First, some use PM_{10}, while

[10] The new US $PM_{2.5}$ standard and the tighter ozone standard have been remanded to the EPA by the D.C. Court of Appeals and aspects of the case are currently being heard by the US Supreme Court. Thus, these standards are not yet in effect (November 2000).

Box 8.2. The Impact of Air Pollution on Health Differs by Country

For any society, deaths at earlier ages result in more productive years of life lost than for those that occur at later ages. One study in Delhi, India, found that children under 5 and adults over 65 years of age are not at risk from air pollution, because other causes of death (notably infectious diseases) predominated in those who survive to reach these age groups (Cropper *et al.,* 1997). However, people between 15 and 45 years of age are at increased risk of death from air pollution relative to those in developed countries. Since the population distribution in India includes many more people in these middle age groups, the net impact on the country from air pollution measured in terms of years of life lost is similar to that of a developed country.

others use fine particles ($PM_{2.5}$), or serveral components of them (sulphates and nitrates). When the individual components of $PM_{2.5}$ are used, the implicit assumption is that their risk is similar to that of $PM_{2.5}$. To date, this has not been verified (especially for nitrates, the secondary particulate product from NO_x emissions). Second, studies that look at age groups separately generally report higher impacts (Aunan *et al*. (2000), for example, used a steeper dose–response coefficient for people older than 65 years of age than that used by other studies). Very few consider the chronic effects on mortality, derived from cohort studies (*e.g*., Pope *et al*., 1995) (Abt Associates and Pechan-Avanti Group, 1999 is one, while others consider it for their "high estimate" only). Use of the latter results in estimates of death three times larger than use of the time series studies. Also, few studies consider effects on child mortality. Finally, different studies consider different health endpoints, which is important for reconciling morbidity estimates.

Valuation of Effects

The most important monetary benefit is related to mortality risk reductions, which can be expressed in terms of the VSL (see Chapter 7). The VSL should ideally be indigenously estimated (Krupnick *et al*., 2000)[12] but almost of the studies build on a consensus on the appropriate values to use (Davis *et al*., 2000), given the state of research on valuation (albeit concentrated in the UK and USA).

A major difference in the treatment of values across the studies is whether these values are adjusted for different income levels and increased for future income growth. Adjustments that assume an income elasticity of willingness to pay (WTP) of 1.0 are inconsistent with the admittedly thin literature. A number of studies found elasticities in the 0.2-0.6 range based on income differentials *within* a country. Such elasticities, when applied to transfers *among* countries, yield quite high values. Most of the developing country ancillary benefit studies reported in *Table 8.6* use an income elasticity of 1.0. The US Science Advisory Board has endorsed the idea of making adjustments for future income growth within a country.

The state of the art of the valuation of air pollution-related mortality effects is currently in ferment, with serious questions being raised about the inappropriateness of basing such valuation on labour market studies. Ad hoc adjustments for the shorter life span of those thought to be most affected by air pollution (the elderly and ill) have been made but more credible estimates of willingness to pay await new research. Such efforts are more likely to lower such estimates relative to current estimates than raise them (see Davis *et al*., 2000 and Krupnick *et al*., 2000).

Environmental Externalities

All the studies, except those in the USA, assume that improvements in public health count as externalities and, hence, as ancillary benefits. As noted in Krupnick *et al*. (2000), this assumption may not always hold. Burtraw *et al*. (1999) and Abt Associates and Pechan-Avanti Group (1999) count the abatement cost savings from reducing SO_2 emissions in response to a carbon tax because SO_2 emissions are capped in the USA. Similar adjustments are not made for SO_2 and other pollutant taxation in Europe. Moreover, not all ancillary benefits are necessarily externalities. In some cases, these effects may be already "internalized" in the price of goods and services: for example, where accident insurance against road fatalities exists, much of this effect is already accounted for through purchasing insurance and the penalties for failure to obtain it.

Treatment of Uncertainty

The uncertainty that surrounds the estimates of benefits is no less than that associated with mitigation costs, extending from physical modelling, through valuation, to modelling choices. Several of the studies use Monte Carlo simulation, but others use less sophisticated sensitivity analyses to characterize uncertainties.

Allowance for Ancillary Costs

None of the studies reviewed in this assessment reported estimates of ancillary costs. Some studies, such as Burtraw *et al.*

[11] Most of the concentration-response functions for health effects of air pollution are based on relative risks models, which give the percentage increase in the number of health effects due to a change in air pollution concentration. This percentage change needs to be applied to the base rate of the effects (i.e. the number of effects observed without change in air pollution). For example, for the non-accidental mortality in the USA, this base rate is about 800/100,000.

[12] Where there is a lack of local information on willingness to pay, one option is to use studies from developed countries and "adjust" the estimates for local conditions. This procedure is called benefit transfer: "an application of monetary values from a particular valuation study to an alternative or secondary policy-decision setting, often in another geographic area than the one where the original study was performed" (Navrud, 1994). The problems of such transfers are discussed in greater detail in Davis *et al.* (2000).

(1999), discuss the bounce-back effect associated with energy substitution to natural gas and other less carbon-intensive fuels. However, even these studies, not surprisingly, estimate positive net ancillary benefits from GHG mitigation policies. The issue is whether the models were designed to capture ancillary costs. In general, our conclusion is no, except for fossil fuel substitution in the power and transport sectors. From an energy substitution perspective, substitution to nuclear power or hydropower does not generate reported ancillary costs because these ancillary effects are not present in the studies. Other sources of ancillary costs were also left out of the modelling exercise, either because of model boundaries or through making some standard modelling choices. All the studies examined effects on one country or region, and therefore do not consider the leakage effect. None of the studies considered health linkages that might result from slower income and employment growth following the implementation of a GHG mitigation policy.

8.2.4.3 *Why Do Studies for the Same Country Differ?*

It is enlightening to consider why estimates of ancillary benefits (or costs) for two different studies of the same country differ.

In the case of Chile, Dessus and O'Connor (1999) estimate benefits of about US$250/tC, as compared to US$62/tC in Cifuentes *et al*. (2000). Half of the Dessus and O'Connor (1999) benefits are attributable to effects on intelligence quotient (IQ) associated with reduced lead exposure, an endpoint not considered by Cifuentes *et al*. (2000) and by most studies. The large lead–IQ effect seems to be at variance with US and European studies that consider this and more conventional endpoints. Also, the VSL used by Dessus and O'Connor (1999) is more than twice as large as that used by Cifuentes *et al*. (2000; US$2.1 million versus US$0.78 million by the year 2020). These choices were driven by alternative benefit transfer approaches: Dessus and O'Connor (1999) used 1992 purchasing power parity to transfer a mid estimate of US VSL, while Cifuentes *et al*. (2000) used 1995 per capita income differences and the exchange rate to transfer a lower bound US VSL. This comparison illustrates that the choice of benefit transfer approach in estimating ancillary benefits dominates by far the modelling choices (Dessus and O'Connor (1999) used a T-D model while Cifuentes *et al*. (2000) used a B-U approach).

For the USA, Burtraw *et al*. (1999) found that for a US$25 carbon tax, the ancillary benefits per tonne are US$2.30, while Abt Associates and Pechan-Avanti Group (1999) found that for a slightly larger tax (US$30), the ancillary benefits per tonne are US$8. For a US$50/tC tax, Burtraw *et al.* (1999) found ancillary benefits of only US$1.50/tC, while for an even larger tax (US$67), Abt Associates and Pechan-Avanti Group (1999) estimated the ancillary benefits to be US$68/tC. These differences are explained by:

- The effect of a unit change in particulate nitrates (derived from NO_x emissions) on the mortality rate which in Burtraw *et al*. (1999) are about one-third of those used by Abt Associates and Pechan-Avanti Group (1999).
- The value of statistical life used to value mortality risk reductions (about 35% lower in Burtraw *et al*. (1999) who adjust the VSL for the effects of pollution on older people rather than on those of average age).
- Sectors included (Burtraw *et al*., 1999) are restricted to the electricity sector by 2010, and NO_x emissions per unit carbon are projected to be lower for this sector than in the general US economy.
- Effect of carbon tax on SO_2 emissions (Abt Associates and Pechan-Avanti Group, 1999) finds that the US$67 carbon tax is large enough to bring SO_2 emissions significantly under an SO_2 cap 60% lower than the current cap. It also cuts NO_x emissions enough to bring significant numbers of non-attainment areas under the national ambient standards. Burtraw *et al*. (1999) does not find such a large effect.
- Baseline emissions (Burtraw *et al*., 1999) do not account for new, tighter ozone and PM standards, but Abt Associates and Pechan-Avanti Group (1999) do (while assuming only partial attainment of the standards). This baseline assumption leaves lower emissions of conventional pollutants to be controlled in the Abt Associates and Pechan-Avanti Group (1999) study than in the Burtraw *et al*. (1999) study.

8.2.4.4 *Conclusions*

The diffusion of methods and key studies to estimate health effects and their monetization has contributed to a reasonable degree of standardization in the literature. However, some of the differences in estimates result from different assumptions and/or methodologies used to estimate them:

- Selection of concentration–response functions, such as use of time series rather than the cohort mortality studies.
- Consideration of more and/or different endpoints, such as considering the lead effects on IQ.
- Use of different assumptions to perform benefit transfers across countries and across time. For example, considering per capita income as opposed to purchasing power parity to perform the unit value transfer; choice of the income elasticity value.
- Defining the baseline differently: most of the literature on ancillary benefits systematically treats only government regulations with respect to environmental policies. In contrast, other regulatory policy baseline issues, such as those relating to energy, transportation, and health, are generally ignored, as have baseline issues that are associated to technology, demography, and the natural resource base.

Therefore, although the standard methodology is generally accepted and applied, a number of assumptions or judgements can lead to estimates of ancillary benefits in terms of US$/tC for a given country that differ by more than an order of magnitude. The least standardized, least transparent and most uncertain component for modelling ancillary benefits is the link from emissions to atmospheric concentrations, particularly in light of the importance of secondary particulates to public health.

Also, the above review reveals implicitly the lack of studies estimating non-health effects from GHG mitigation policies (damages from traffic crashes, the effects of air pollution on materials, and air pollution effects on crops losses, which have been shown to be quite high in some regions). Depending upon the GHG mitigation policies selected, some of this damage could well be reduced, but the nature of this relationship remains a speculative matter. More information can be found in sectoral studies reviewed in Chapter 9, but no comprehensive evaluation can be derived from them.

For all these reasons, it remains very challenging to arrive at quantitative estimates of the ancillary benefits of GHG mitigation policies. Despite the difficulties, it can be said that the ancillary benefits related to public health accrue over the short term, and under some circumstances can be a significant fraction of private (direct) mitigation costs. With respect to this category of impacts alone mortality tends to dominate. The exact magnitude, scale, and scope of these ancillary benefits varies with local geographical and baseline conditions; if the baseline scenario assumes a rapid decrease in non-GHG pollutant emissions, benefits may be low, especially in low density areas. Net ancillary costs (i.e., where the ancillary benefits are less than ancillary costs) may occur under certain conditions, but the models reviewed here are generally not designed to capture these effects. While most of the studies assessed above address ancillary benefits of explicit climate mitigation measures, it should be noted that in many cases, these ancillary benefits can be expected to be as least as important as climate mitigation for decision making. Hence, the terms co-benefits is also used in this report. Therefore, there is a strong need for more research in the area of integrated policies addressing climate mitigation alongside other environmental, social or economic objectives.

8.3 Interface between Domestic Policies and International Regimes

For every country, the costs of achieving a given level of abatement will be dramatically affected by the interface between its domestic policy and international regimes. Since a co-ordination on the basis of simple reporting mechanisms has not be adopted from the outset because it would not have been stringent enough for UNFCCC objectives, some studies were devoted to clarifying the differences between the two main tools for co-ordinating climate policies: country emissions quotas or agreed carbon taxes.

Theoretically, both solutions are equivalent in a world with complete information (the optimal quota leads to the same marginal abatement cost as the optimal level). However, Pizer (1997), building on a seminal work by Weitzman (1974), demonstrated that this is not the case if uncertainties about climate damages and GHG abatement costs are considered. Indeed, welfare losses due to an error of anticipation are not the same in these two approaches, depending upon whether the steepness of marginal abatement cost curve is higher or lower than the steepness of the damage curve. If the marginal abatement cost curve is steeper, then it is preferable to agree on a pre-determined level of taxation because if this level is either too low or too high, the resulting welfare losses trough climate impacts will not be dramatic. This is the case in most modelling efforts as long as there is no large probability of dramatic non-linearity in climate systems over the middle term. This policy conclusion can be reverted if one considers a high level of risk-aversion to catastrophic events (which makes the damage curve steeper), or a large proportion of "no regret" policies (which make the mitigation cost curve flatter). The main message, however, is that in a tax harmonization approach, the costs of complying with commitments on climate policies are known in advance (but the outcome is not predictable), while in a quota approach the outcome is observable but there is an uncertainty about the resultant costs. In this respect, emissions trading is logically a companion tool to a system of emissions quotas, to hedge against the distributional implications of surprises regarding abatement costs and emissions baselines.

After the Berlin Mandate (1995), a quota co-ordination approach was implicitly adopted and the focus of analysis was placed on linkages between emissions trading regimes and national policies. Contrary to the preceding period, very few works were devoted to the case of co-ordinated carbon taxes. Hourcade *et al*., (2000a) confirmed that, because of the existing uneven distribution of income, discrepancies in pre-existing taxation levels, and differences in national energy and carbon intensities, a uniform carbon tax would result in very differentiated losses in welfare across countries, unless appropriate compensation transfers operated. However, a differentiated taxation does not minimize total abatement expenditures (rich countries would have to tap more expensive abatement potentials) and creates distortions in international competition. The suggested solution, a uniform tax for carbon-intensive industry exposed to international competition and a differentiated taxation for households, has to be at least adapted to the Kyoto framework which does not preclude the use of carbon taxes but changes the condition of their applicability. However, the underlying issue of how to minimize abatement expenditures while guaranteeing a fair distribution of welfare costs still remains.

Under the Kyoto framework, the interface between domestic policies and the international regime passes through three main channels: the impact of international emissions permit trading (under Article 17), international trading in project-related credits (under Articles 6 and 12 (Read, 1999)) on abatement costs, and spillover effects across economies through commercial and capital flows.

Table 8.7: *Energy Modelling Forum main results; marginal abatement costs (in 1990 US$/tC; 2010 Kyoto target)*

Model	No trading				Annex I trading	Global trading
	USA	OECD-E	Japan	CANZ		
ABARE-GTEM	322	665	645	425	106	23
AIM	153	198	234	147	65	38
CETA	168				46	26
Fund					14	10
G-Cubed	76	227	97	157	53	20
GRAPE		204	304		70	44
MERGE3	264	218	500	250	135	86
MIT-EPPA	193	276	501	247	76	
MS-MRT	236	179	402	213	77	27
Oxford	410	966	1074		224	123
RICE	132	159	251	145	62	18
SGM	188	407	357	201	84	22
WorldScan	85	20	122	46	20	5
Administration	154				43	18
EIA	251				110	57
POLES	135.8	135.3	194.6	131.4	52.9	18.4

Source: cited in Weyant, 1999; Council of Economic Advisors, 1998; EIA (Energy Information Administration), 1998; Criqui *et al*., 1999.

8.3.1 International Emissions Quota Trading Regimes

8.3.1.1 "Where Flexibility"

Table 8.7 synthesizes marginal abatement costs for the USA, Japan, OECD-Europe, and the rest of the OECD (CANZ) calculated by 13 world T-D models co-ordinated by the Energy Modeling Forum. It also includes the results obtained with the POLES model, which provides a multiregional partial equilibrium analysis of the energy sector, and two other studies of the economic impacts of Kyoto conducted by the US Government, the Administration's Economic Analysis (Council of Economic Advisors, 1998), and a study by the Energy Information Administration (1998). These results cannot be directly compared with those of the B-U analysis reported in Section 8.2.1.1, because they incorporate feedback on energy demand, oil prices, and macroeconomic equilibrium. They give, however, an idea of the assumptions on technical abatement potentials retained for each region in these exercises, the main difference with B-U analysis being that these exercises do not explicitly consider negative cost potentials (they are implicit in most optimistic baselines).

Despite the wide discrepancies in results across models, the robust information is that, in most models, marginal abatement costs appear to be higher in Japan than in the OECD-Europe. CANZ and the USA have comparable results, approximately two-thirds the European one, and much lower than in Japan.

This means that Kyoto targets are likely to be unequitable. This risk is confirmed by uncertainty analyses based on existing

Table 8.8: *Energy Modelling Forum main results; GDP loss in 2010 (in % of GDP; 2010 Kyoto target)*

	No trading				Annex I trading				Global trading			
Model	USA	OECD-E	Japan	CANZ	USA	OECD-E	Japan	CANZ	USA	OECD-E	Japan	CANZ
ABARE-GTEM	1.96	0.94	0.72	1.96	0.47	0.13	0.05	0.23	0.09	0.03	0.01	0.04
AIM	0.45	0.31	0.25	0.59	0.31	0.17	0.13	0.36	0.20	0.08	0.01	0.35
CETA	1.93				0.67				0.43			
G-CUBED	0.42	1.50	0.57	1.83	0.24	0.61	0.45	0.72	0.06	0.26	0.14	0.32
GRAPE		0.81	0.19			0.81	0.10			0.54	0.05	
MERGE3	1.06	0.99	0.80	2.02	0.51	0.47	0.19	1.14	0.20	0.20	0.01	0.67
MS-MRT	1.88	0.63	1.20	1.83	0.91	0.13	0.22	0.88	0.29	0.03	0.02	0.32
Oxford	1.78	2.08	1.88		1.03	0.73	0.52		0.66	0.47	0.33	
RICE	0.94	0.55	0.78	0.96	0.56	0.28	0.30	0.54	0.19	0.09	0.09	0.19

models which provide a pretty wide range of outcomes that can be interpreted as covering the uncertainties prevailing in the real world. This can be shown in the results of domestic cost of carbon: from US$85 to US$410 in the USA, US$20 to US$966 for the OECD-Europe, US$122 to US$1074 for Japan, US$46 to US$423 for CANZ. The variance remains significant if the extreme values:

- from US$76 to 236/tC for the USA if one excludes GTEM, Merge 3, and Oxford;
- from US$159 to US$276/tC for the OECD-Europe and from US$145 to US$250 for CANZ if one excludes Worldscan, GTEM, and Oxford; and
- a continuum from US$122 to US$645/tC for Japan if Oxford is excluded.

In terms of GDP losses, the ranking of impacts differs because of the various pre-existing structures of the economy and of the energy supply and demand in various countries and because these studies do not consider the domestic policies targeted to tackle these pre-exisiting conditions; the GDP losses are from 0.45% to 1.96% for the USA, from 0.31 to 2.08 for the EU, from 0.25 to 1.88 for Japan. This variation is reduced under emissions trading; 0.31 to 1.03 for the USA, 0.13 to 0.73 for the OECD-Europe, from 0.05 to 0. 52 for Japan.

This discrepancy in results reflects differences in judgements about parameters such as technical potentials, emissions baselines, how the revenues of permits are recycled, and how near-term shocks are represented. Another important source of uncertainty is the feedback of the carbon constraint on the demand for oil; a drop in oil prices requires indeed higher prices of carbon to meet a given target since the signals not conveyed by oil prices as to be passed through price of carbon which leads to a totally different incremental cost of the carbon constraint.

These uncertainties about mitigation costs are reflected in the net welfare losses. The preceding discussion in Section 8.2 demonstrated the many sources of a wedge between total abatement costs and welfare losses, including the double dividend from fiscal reforms and the very structures of the economy (share of carbon intensive activities) and of the energy system.

The wide range of cost assessments, far from resulting from purely modelling artefacts, help to capture the range of possible responses of real economies to emissions constraints and to appreciate the magnitude of uncertainties that governments have to face.[13] They demonstrate that without emissions trading, the Kyoto targets lead to a misallocation of resources, a non-equitable burden-sharing (notwithstanding its mitigation through double-dividend domestic policies analyzed in Section 8.2.2.1) and distortions in international competition. Even in the most optimistic models regarding abatement costs such as Worldscan, trading offers the potential for countries with high domestic marginal abatement costs to purchase emissions permits in countries with low marginal abatement costs and hence a way of minimizing total abatement costs and of hedging against risks of a too high and unequitable burden.

The full global trading scenarios presented in *Tables 8.7* and *8.8* assume non-restricted trade within Annex I and ideal CDM implementation that can exploit all cost effective options in developing countries with unlimited trading. Beyond the fact that the price of carbon is drastically reduced, it is remarkable that the variance of results is far lower than in the no-trade scenarios (between US$15/tC and US$86/tC). Uncertainty about costs persists, but this lesser variance arises because uncertainty is higher on each regional cost curve than on the aggregation of the same regional cost curves, which is exploited in the case of full trading.

In the case of Annex I trading (without considering the CDM) the price of permits ranges from US$20 to US$224/tC instead of US$15 to US$86/tC in the full trade case, which represents a far greater variance. This is mainly from the amount of so called "hot air"[14] retained in simulations. Some countries in Eastern Europe and the former Soviet Union have had a decline in emissions in the 1990s, resulting from the economic dislocations associated with restructuring. As a result, their emissions during the first commitment period are projected to be lower than their negotiated target. If trading is allowed within Annex I, these excess emissions quota may be sold to countries in need of such credits. Hence, the assumption regarding the availability of "hot air" is important. This, of course, will be governed in part by the rate of economic recovery, but also by the role of energy efficiency improvements and fuel switching during the restructuring process.

The main lessons from the above studies using T-D approaches (namely that trade has a marked, beneficial effect on costs of meeting mitigation targets), are confirmed by a series of recent studies using B-U approaches. These provide a more detailed information on the potentials for CDM projects. The MARKAL, MARKAL-MACRO, and MESSAGE models have been adapted and expanded to facilitate such multicountry studies. In North America, Kanudia and Loulou (1998) report MARKAL results for a three-country Kyoto study (Canada, USA, India). The total cost of Kyoto for Canada and the USA amounts to some US$720 billion with no trade, versus US$670

[13] This is exemplified by two others studies of the economic impacts of Kyoto conducted by the US Government. It is remarkable that GDP losses span from virtually zero to 3.5% and are correlated with the level of marginal abatement cost. The EIA assessment is the highest because it accounts for near-term shocks, such as inflationary impacts of higher energy prices (requiring higher interest rates, which dampen the investment), and for a 5-year delay in the response of agents). The EIA estimates rise to 4.2% when the non-CO_2 gases and carbon sinks are excluded from the analysis.

[14] Hot air: a few countries, notably those with economies in transition, have assigned amount units that appear to be well in excess of their anticipated emissions (as a result of economic downturn). This excess is referred to as "hot air".

billion when North American emissions and electricity trading is unimpeded, and only US$340 billion when India is added to the permit trading. MARKAL studies in the Nordic states (see Larsson *et al*. (1998) for Denmark, Sweden, and Norway, and Unger and Alm (1999) for the same plus Finland) show the considerable value of trading electricity and GHG permits within the region when severe GHG reductions are sought. Another MARKAL study computes the net savings of trading GHG permits between Belgium, Switzerland, Germany, and the Netherlands (Bahn *et al*., 1998) at about 15% of the total Kyoto cost without trading. Another study (Bahn *et al*., 1999a) shows that Switzerland's Kyoto cost may be reduced drastically if it engages in CDM projects with Columbia, in which case the marginal cost of CO_2 drops to US$12/tC. This type of B-U analysis has also been extended to the computation of a global equilibrium between Switzerland, Sweden, and the Netherlands, using MARKAL-MACRO (Bahn *et al*., 1999b), with the conclusion that GDP losses resulting from a Kyoto target are 0.2% to 0.3% smaller with trade than without. More ambitious current research aims at building worldwide B-U models based on MARKAL (Loulou and Kanudia, 1999a) or on MARKAL-MACRO (Kypreos, 1998).

8.3.1.2 Impacts of Caps on the Use of Trading

From the above results, it is seen that all OECD countries have an interest in making the market as large as possible. Some Parties to the UNFCCC, however, have suggested that the supplementarity conditions of Articles 6.1.d, 12 and 17 of the Kyoto Protocol be translated into quantitative limits placed on the extent that Annex I countries can satisfy their obligations through the purchase of emission quotas. The rationale for the supplementarity condition is that, if the price of permits were too low, this would discourage domestic action on structural variables (infrastructure, transportation) or on innovation apt to modify the emissions trends over the long run. These measures are very often liable to high transaction costs and governments may prefer to import additional emissions permits instead of adopting such measures. In other words, minimization of the costs of achieving Kyoto targets may not guarantee minimization of the costs of climate policies over the long run; this is the case when the inertia of technical systems is considered (Ha-Duong *et al*., 1999) and when one accounts for the long term benefits of inducing technical change through abatements in the first period (Glueck and Schleicher, 1995).

Some works have studied the consequences of enforcing the supplementarity condition through quantitative limits: one of the EMF scenarios imposed a constraint on the extent to which a region could satisfy its obligations through the purchase of emission quota (the limit was one-third).

However, the models cannot deliver any response without an assumption about *ex ante* limits on carbon trading, resulting into a stable duopoly between Russia and Ukraine or into a monopsony (Ellerman and Sue Wing, 2000). In the first case, the price of carbon will be higher than in a non-restricted market, and most of the additional burden will fall on countries in which the marginal cost curve is high because they have a lesser potential for cheap abatement. This is typically the case for Japan and most of the European countries (Hourcade *et al*., 2000b). The other possibility is for the market power to be controlled by the carbon-importing countries; in this case, the risk is that all or most of the trading will be of "hot air" at a very low price. Which of these alternatives will be realized cannot be predicted but, in both cases, quantitative limits to trade lead to outcomes that contradict the very objective of the supplementarity condition. Criqui *et al*. (1999) assessed the order of magnitude at stake with the POLES model, and examined a scenario in which the carbon tax is US$60/tC with unrestricted trade. They found that the carbon prices under the concrete ceiling conditions proposed by the EU fall to zero (with no market left for the developing countries) if the market power is held by the buyers. Alternatively, the carbon prices increase up to US$150/tC if the market power is held by the sellers, this risk being increased in the case of caps on hot air trading which increases the monopolistic power of Russia and Ukraine. Böhringer (2000) assesses the economic implications of the EU cap proposal within competitive permit markets. He concludes that part of the efficiency gains from unrestricted permit trade could be used to pay for higher abatement targets of Annex-B countries which assure the same environmental effectiveness as compared to restricted permit trade but still leaves countries better off in welfare terms.

8.3.1.3 The Double Bubble

Here the case of the "double bubble" is examined, in which countries belonging to the EU have a collective target, making use of the flexibilty to shift emission quota within the group and the remaining Annex I countries trade among themselves to reach their individual targets.

Figure 8.11 shows the incremental value of carbon emission for the two groups and compares them with that of full Annex I trading. Notice that for the USA, the tax is lower in the case of the "double bubble" than with Annex I trading. The reason is that without the EU bidding for the Russian "hot air", the demand for emission quotas falls as does its price. The EU on the other hand is disadvantaged under such a scenario. With their access to low cost emission quotas limited, the incremental value rises.

8.3.2 Spillover Effects: Economic Effects of Measures in Countries on Other Countries

In a world in which economies are linked by international trade and capital flows, abatement by one economy induces spillover effects and has welfare impacts on other economies. It matters to understand the conditions under which both abating and non-abating economies will experience positive or negative impacts from the policy adopted in other groups of countries; it matters also to understand the results of these spillover

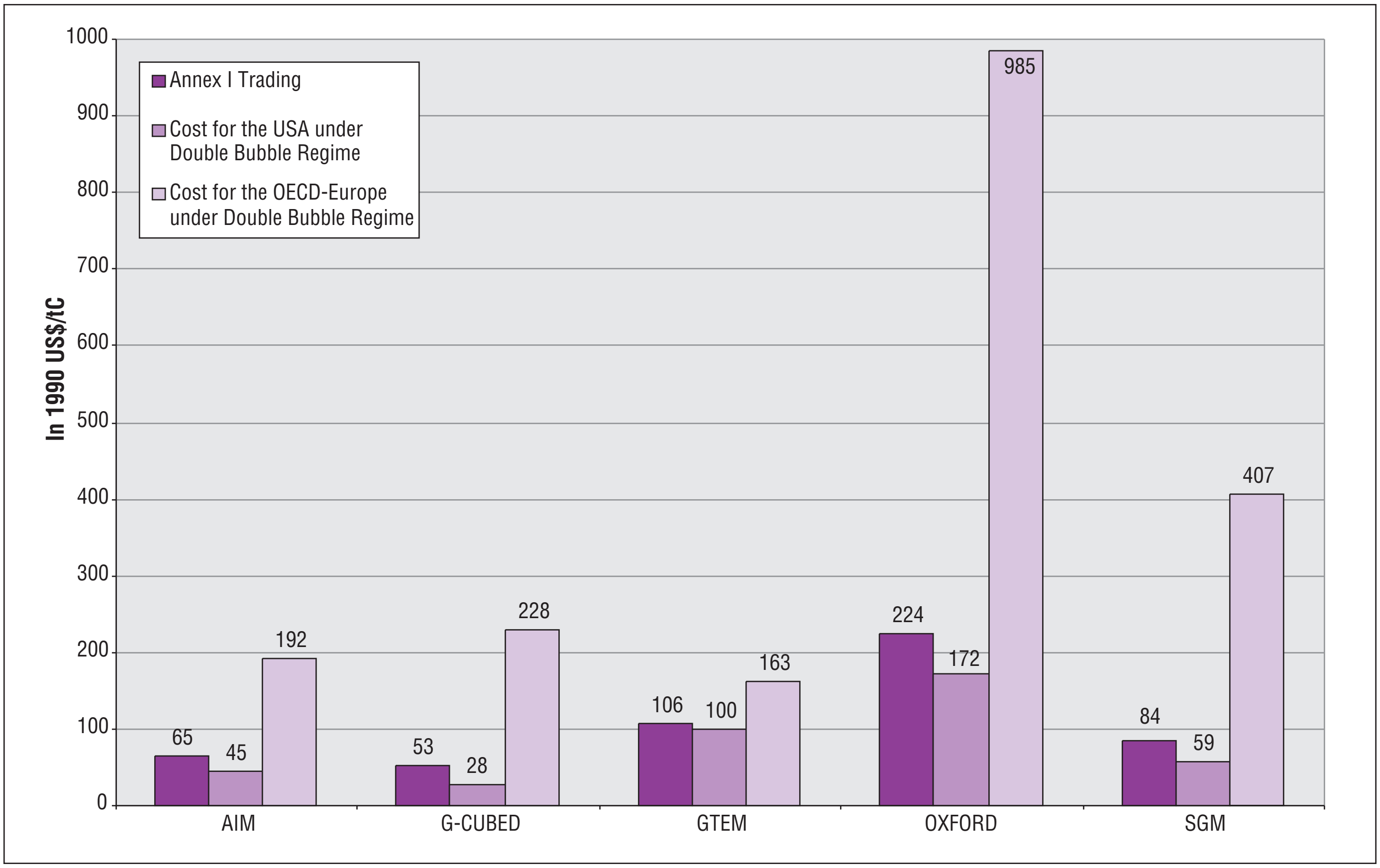

Figure 8.11: *The double bubble.*

effects in terms of carbon leakage. Chapter 7 provides the basic concepts of such an analysis and here some brief comments are added to explain the strengths and weaknesses on the results found by modelling exercises.

In static terms, without international capital mobility, the welfare costs of abatement for an open economy can be decomposed into two components (Dixit and Norman, 1984):

- costs that would be incurred if the economy were closed; and
- changes in the terms of trade, which are the first transmission mechanism for spillover effects (see Chapter 7).

If they require to go beyond "no regrets" potentials, binding emissions constraint comes to increasing the cost of carbon-intensive products and, if emissions arise from the production of its export goods, the abating economy benefits from better terms of trade. If, indeed, the importing economy cannot produce a perfect substitute easily, it will sell the same product at a higher price and increase its purchasing power of imported goods. The non-abating economy will symmetrically suffer a welfare loss because of more expensive imports, while the net result for the abating economy depends on the size of improvement in the terms of trade relative to the production costs of abatement. The welfare impacts are more important in the economies that are very dependent on foreign trade.

In the real world, an emission constraint simultaneously affects both export and import goods, but this does change the nature of the mechanism. Increased production of emission-intensive goods in non-Annex I regions is stimulated by both increased non-Annex I consumption and increased exports to Annex I regions. The net relative balance between these parameters is influenced by the extent to which Annex I emission constraints fall on export competing industries (when the country is specialized in such industries) as opposed to import-competing industries (when it imports carbon intensive goods). If a constraint predominantly affects export industries, it encourages increased non-Annex I production for internal consumption. If the constraint predominantly affects import-competing industries, increased non-Annex I production is mainly exported to Annex I regions. Emissions leakage is beneficial to non-Annex I economies only in the second case, since it is associated with an improvement in their terms of trade, whereas their terms of trade deteriorate in the first case.

Another factor that affects the increase of emission-intensive goods in non-Annex I regions is the effect of Annex I abatement on the intermediate demand for fossil fuels. As discussed above, Annex I abatement will reduce fossil fuel prices. Lower prices for fossil fuels will encourage the production of more emission-intensive goods and the use of more emission-intensive production techniques in non-Annex I regions.[15]

So far, it was assumed that changes in the production structures in both Annex I and Non-Annex I countries result only in changes in final demand and in price structures. The introduction of international capital mobility complicates the analysis since, in addition to production costs and changes in the terms of trade, carbon constraints alter the relative rates of return in the abating and non-abating countries. If capital flows from the first country to the second in response to these changes, there will be a further restriction of the production frontier (the set of possible productive combinations) for the abating economy and an outward shift for the other economy. Factor rewards in both countries are also affected. Part of the income from foreign investment accrues to the home economy and subtracts from income in the foreign economy; abating economies are affected by changes in income and factor prices that result from changes in international capital flows, with symmetric gains for non-abating economies.

No theoretical results for complex and empirically relevant cases can be obtained as to the extent that international capital mobility modifies the conclusions of the static analysis of the role of the trade effects. However, modelling results are seldom reported on the welfare impact of changes in international capital flows, although McKibbin *et al.* (1999) emphasize the macroeconomic repercussions. It is still, indeed, impossible to derive clear conclusions about the role of these changes, because of the methodological difficulties in interpreting the results from complex CGE models. It is usually conceded that modelling international capital flows is one of the more contentious issues; technically indeed, such a modelling relies on equalizing rates of return on capital across countries, but, because this makes capital flows too reactive, various "ad hoc" devices are used to obtain less irrealistic outcomes. Differences in the riskiness of rates of returns are clearly relevant to explain most of the real behaviours, but how this can be "best" dealt with in a deterministic model is an open question. Progress depends on the further development of techniques.[16] It depends also on progress in theoretical and empirical analyses to capture more effectively how the exchange rate of currencies reacts to external payment deficits. This depends on the level of confidence on the future economic expansion of each country and how monetary policies (including the determination of the public discount rate) employed to mitigate adverse impacts can change the return to capital in a country relative to other countries.

Models reviewed in this section have in common features that must be clearly borne in mind when interpreting the results:

- They assume perfect competition in all industries.
- Most of them use the so-called Armington specification that identical goods produced in different countries are imperfect substitutes: it is known that the results may then be sensitive to the particular commodity and chosen regional aggregation models (Lloyd, 1994).
- All of the models, apart from the G-cubed model of McKibbin and Wilcoxen (1995), are long-term growth models with international trade, without explicitly modelled financial markets that affect the macroeconomic adjustment.[17]
- Emission reductions involve only carbon dioxide.[18]
- The bias in technological change is unaffected by the emissions constraints and the production possibilities frontier always lies below the unconstrained frontier. Under such a hypothesis, the aggregate impact is unlikely to be positive, but some economies may benefit from favourable changes in their terms of trade and from changes in international capital flows.

Simulation studies covered in this report were conducted prior to and after the negotiation of the Kyoto Protocol. Pre-Kyoto studies consider more stringent emissions reduction targets for Annex I regions than the average 5.2% actually adopted under the Protocol. The major findings are that Annex B abatement would result in welfare losses for most non-Annex I regions under the more stringent targets. The magnitude of these losses is reduced under the less stringent Kyoto targets. Some non-Annex I regions that would experience a welfare loss under the more stringent targets experience a mild welfare gain under the less stringent targets.

Studies using a variety of more stringent pre-Kyoto targets include Coppel and Lee (1995; the GREEN model), Jacoby *et al*. (1997; the EPPA model), Brown *et al*.1997b) and Donovan *et al* (1997; the GTEM model), and Harrison and Rutherford (1999; the IIAM model). The last two models are based on the Global Trade Analysis Project (GTAP) database (Hertel, 1997).

In these studies, most non-Annex I countries suffer deterioration in their terms of trade and also welfare losses. Since the analysis at the region or country level depends on the type of

[15] To the extent that increased non-Annex I emissions result from more emission-intensive production techniques and increased production of emission-intensive goods for internal consumption, policies to control emission leakage by curbing the imports of emission-intensive goods into Annex I regions are likely to be counterproductive. Curbing imports may restrict substitution options in Annex I economies, requiring further cuts in output and exports that would stimulate greater non-Annex I emission-intensive production.

[16] These include techniques such as decomposition analysis (Huff and Hertel, 1996) and multiple simulations, in which some variables are held constant to isolate their influence on the final results. Verikios and Hanslow (1999) employed such a framework to asses the welfare impacts of international capital mobility.

[17] In the G-cubed model such a mechanism is superimposed on the structure of a long-term growth model.

[18] It is evident from simulations with the GTEM model (Brown *et al.,* 1999) that somewhat different results may be obtained if emission reductions involve a least-cost mix of the different GHGs identified under the Kyoto Protocol.

aggregation, it is difficult to give a comprehensive list of exceptions. The reasons for these exceptions are, however, easy to explain. Brazil and South Korea are, in many models, found to enjoy welfare gains from Annex I abatement policies because, unlike other non-Annex I regions, they are net importers of fossil fuels and have a high relative dependence on exports of iron and steel and non-ferrous metal products. In addition, in Brazil these products are far less intensive in fossil energy than in many other economies.[19] Brazil gains from lower prices for fossil fuel imports and higher prices for exports of iron and steel and non-ferrous metal products. Conversely, non-Annex I regions with the greatest dependence on fossil fuel exports, such as the Middle East and Indonesia, suffer the greatest deterioration. Non-Annex I regions that are net importers of manufacture goods that are fossil-fuel intensive also suffer a deterioration even if they benefit from lower oil prices.

One of the most important conclusions is that a number of those among non-Annex I regions that experienced a welfare loss under the pre-Kyoto targets experience a welfare gain under the Kyoto targets. For example, in the GREEN model, India and the Dynamic Asian Economies experienced a loss in real income in the pre-Kyoto simulation (Coppel and Lee, 1995). They experience a mild gain in real income under simulations of the Kyoto Protocol that involve varying degrees of policy co-ordination among the non-Annex I regions (van der Mensbrugghe, 1998). In pre-Kyoto simulations of the GTEM model (Brown *et al.*, 1997b; Donovan *et al.*, 1997), Chinese Taipei, India, Brazil, and the Rest of America were all found to experience welfare losses; with Kyoto targets (Tulpulé *et al.*, 1999) these regions experience mild welfare gains.

There is one key reason why some regions that experienced welfare losses under the more stringent targets experience mild gains in welfare under the Kyoto targets: the changing balance between substitution and output reduction with the level of abatement. GDP losses or the required level of a carbon tax for Annex I regions are, indeed, an increasing function of the level of abatement and the milder Kyoto targets are expected to be achieved with a greater reliance on substitution relative to output reduction than the more stringent targets.

A fairly similar regional pattern of non-Annex I welfare changes is found in simulations of Kyoto targets in a number of studies in which comparable pre-Kyoto target simulations are not available. These studies include Kainuma *et al.* (1999; the AIM model drawing on the GTAP database), McKibbin *et al.* (1999; the G-Cubed model), Bernstein *et al.* 1999; MS-MRT, drawing on the GTAP database), and Brown *et al.*, (1999; the multigas (CO_2, CH_4, and NO_x) version of GTEM) and Böhringer and Rutherford (2001).

[19] These calculations are based on version 3 of the GTAP database after reconciliation with energy data, mainly from the International Energy Agency.

8.3.2.1 *Impact of Emissions Trading*

All of the above studies considered various forms of emissions trading for Annex I economies. It was universally found that most non-Annex I economies that suffered welfare losses under uniform independent abatement also suffer smaller welfare losses under emissions trading. This is also the case in all of the studies for which results on movements in the terms of trade are published (Coppel and Lee 1995; Harrison and Rutherford, 1999).

Why are overall welfare losses to non-Annex I regions reduced by emissions trading? A key point is that because the marginal and average cost of abatement for the aggregate Annex I is lower under emissions trading than under uniform abatement, a higher GDP is achieved for a given reduction in emissions. This means that the reduction in emissions is achieved through a heavier reliance on substitution relative to output reduction (substitution involves the substitution of less emission-intensive for more emission-intensive Annex I produced inputs). The heavier reliance on substitution means that there is a less severe decline in fossil fuel prices and a lower increase in the price of manufactured goods that are fossil-fuel intensive. There is also less increase in non-Annex I exports of fossil-fuel intensive manufactured goods to Annex I regions under emissions trading than independent abatement. However, these increased exports divert resources from activities in which the original non-Annex I comparative advantage was higher and the overall result is less beneficial to most non-Annex I economies.

Some non-Annex I economies that experience welfare gains under independent abatement also experience smaller gains under emissions trading; however, the aggregate effect of emissions trading is found to be positive for non-Annex I economies: those that suffer welfare losses under independent abatement suffer smaller losses under emissions trading.

To summarize, despite a number of identifiable numerical discrepancies, there is agreement that the mixed pattern of gains and losses under the Kyoto targets results in a more positive aggregate outcome than under the assumed and more stringent pre-Kyoto targets. Similarities in the regions that are identified as gainers and losers are also quite marked. Oil-importing economies that rely on energy-intensive exports are gainers (and more so if the exports' carbon intensity is low), economies that rely on oil exports experience losses, and the results are more unstable for economies between these two extremes.

8.3.2.2 *Effects of Emission Leakage on Global Emissions Pathways*

As discussed above, a reduction in Annex I emissions tends to increase non-Annex I emissions, reducing the environmental effectiveness of Annex I abatement. Emissions leakage is measured as the increase in non-Annex I emissions divided by the reduction in Annex I emissions.

A number of multiregional models have been used to estimate carbon leakage rates (Martin *et al*. 1992; Pezzey 1992; Oliveira-Martins *et al*. 1992; Manne and Oliveira-Martins, 1994; Edmonds *et al*., 1995; Golombek *et al*., 1995; Jacoby *et al*. 1997; Brown *et al*. 1999). In SAR (IPCC, 1996, p. 425) a high variance in estimates of emission leakage rates was noted; they ranged from close to zero (Martin *et al*. (1992) using the GREEN model) to 70% (Pezzey (1992) using the Whalley–Wigle model). In subsequent years, some reduction in this variance has occurred, in the range 5%–20%. This may in part arise from the development of a number of new models based on reasonably similar assumptions and data sources, and does not necessarily reflect more widespread agreement about appropriate behavioural assumptions. However, because emission leakage is an increasing function of the stringency of the abatement strategy, this may also be because carbon leakage is a less serious problem under the Kyoto targets than under the targets considered previously.

Technically, there is a clear correlation between the sign and magnitude of spillover effects analyzed above and the magnitude of carbon leakage. It is important, however, to recognize those parameters that have a critical influence on results:

- The assumed degree of substitutability between imports and domestic production. This is why models based on the Armington assumption that imports and domestic production are imperfect substitutes produces lower estimates of emission leakage than models based on the assumption of perfect substitutability.
- The ease of substitution among technologies with different emissions intensities in the electricity and the iron and steel industries in Annex I regions.
- The assumed degree of competitiveness in the world oil market; this issue is considered in Section 8.3.2.3.
- The existence of an international carbon-trading system: for a given abatement strategy, emission leakage is lower under emissions trading than under independent abatement. This conclusion flows logically from the discussion above on movements in terms of trade. Greater Annex I output reduction under independent abatement stimulates greater emission-intensive production in non-Annex I regions, through both higher prices for emission intensive products and lower prices for fossil fuels. Support for the above conclusions on the impact of emissions trading is found in ABARE-DFAT (1995), Brown *et al. (*1997b), Hinchy *et al*. (1998), Brown *et al*. (1999), McKibbin *et al*. (1999), Kainuma *et al*. (1999), and Bernstein *et al*. (1999).

8.3.2.3 Effects of Possible Organization of Petroleum Exporting Countries (OPEC) Response

In the preceding discussion, a competitive equilibrium in the world economy was assumed. However, OPEC may be able to exercise a degree of monopoly power over the supply of oil. The issue has been raised in the literature as to the possible nature of an OPEC response to reduced demand for oil as a result of Annex I abatement. If in the short term OPEC were to reduce production to maintain prices in the face of lower demand, the time path for Annex I carbon taxes may need to be modified. See also Chapter 9.

A number of theoretical papers examined how a carbon tax might alter the optimal timing of extraction of given reserves of oil and, symmetrically, how significantly the potential supply response could alter the optimal time path of the price of carbon tax (Sinclair, 1992; Ulph and Ulph, 1994; Farzin and Tahovonen, 1996; Hoel and Kverndokk, 1996; Tahvonen, 1997). However, the severity of the potential problem depends on a number of key parameter values and implementation issues. Although it has been assumed that OPEC can "Granger cause" the world price of oil (Güllen, 1996), there is some question about the degree of cartel discipline that could be maintained in the face of falling demand (Berg *et al*., 1997a). Any breakdown in the cartel would tend to increase the supply of oil on the market, which in the short term may require a higher carbon tax to meet a given abatement target. On the other hand, Bråten and Golombek (1998) suggest that implementing an Annex I climate change agreement might be seen by OPEC members as a hostile act and could strengthen the resolve to maintain cartel discipline. The OPEC response is likely to be related to the size of its potential loss in revenue to OPEC and these potential losses would be smaller under Annex I emissions trading than under independent abatement.

A number of empirical studies have tried to assess the significance of the potential OPEC response within a game theoretic framework. To do so, Berg *et al*. (1997b) resorted to a Cournot–Nash dynamic game in which parameter values are based on empirical estimates. They also identify (non-OPEC) "fringe" oil producers and other fossil fuel sources. A scenario is examined in which a carbon tax is maintained at a level of US$10 per barrel of oil. Initially, OPEC cuts back on production to try to maintain price, but this is partly offset by increased production by the fringe. Bråten and Golombek (1998) derive a similar pattern of OPEC response in a static model. Berg *et al*. (1997b) found that the optimal OPEC policy is not heavily influenced by intertemporal optimization in shifting supplies from one time period to another to maximize discounted net revenue.

If OPEC acts as a cartel, the extent of emissions leakage in response to Annex I abatement may be reduced (Berg *et al.,* 1997b), because the resultant higher price for oil reduces the incentives for increased emission-intensive activity in non-Annex I regions. Lindholt (1999) examined the Kyoto Protocol in an enhanced version of the same model and assumed that an efficient tradable permit scheme is established between Annex B countries. Whether or not OPEC acts as a cartel does not affect the shape of the time path of permit prices, only their level according to Lindholt (1999). A permit price of US$14/t$CO_2$ would be required in 2010 if OPEC acts as a cartel, whereas it would be US$24/t$CO_2$ in a competitive oil market. The lower permit price when OPEC acts as a cartel stems from OPEC cutting back production to maintain a higher oil price, which slows the growth in emissions in Annex B countries.

These studies mentioned demonstrate that whether or not OPEC acts a cartel will have a modest effect on the loss of wealth to OPEC and other oil producers and the level of permit prices in Annex B regions. A natural extension of this research would be to trace through all the ramifications of cartel behaviour by OPEC in the more complex CGE models discussed in this section.

8.3.2.4 Technological Transfers and Positive Spillovers

In a dynamic context, a progressive outward shift in the production possibilities frontier occurs over time as a result of technical change. A strand of literature (Goulder and Schneider, 1999) argues that climate policies will bias technical change towards emissions savings. In that case, there will be an outwards shift in the production possibilities frontier at some points, and an inwards shift at other points relative to the baseline.

One potentially important related issue not captured in the above models is that cleaner technologies, developed in response to abatement measures in industrialized countries, tend to diffuse internationally. The question is to what extent this will offset the negative aspects of leakage noted above and to amplify positive spillover. Grubb (2000) presents a simplified model, which represents this spillover effect in terms of its impact on emissions per unit GDP (intensities). The results suggest that, because the impact of cleaner technologies is cumulative and global, this effect tends to dominate over time, provided the connection between industrialized and developing country emission intensities is significant (higher than 0.1 on a scale where 0 represents an absence of connection and 1 a complete convergence of intensities by 2100). At this stage, empirical analysis is still lacking to derive a robust conclusion from this result. A recent work by Mielnik and Goldemberg (2000) suggests that the potential for technological leap-frogging in developing countries is important, but to what extent climate mitigation in Annex B accelerates this leap-frogging is still unclear. However, this demonstrates that the trickling down of technical change across countries deserves more attention in modelling exercises, all the more so since theoretically it (see Chapter 10 of this report) demonstrates that technological spillovers may be a major stabilizing force of any climate coalition.

8.4 Social, Environmental, and Economic Impacts of Alternative Pathways for Meeting a Range of Concentration Stabilization Pathways

The appropriate timing of mitigation pathways depends upon many factors including the economic characteristics of different pathways, the uncertainties about the ultimate objective, and the risks and damages implied by different rates and levels of atmospheric change. This section focuses upon the mitigation costs of different pathways towards a predetermined concentration ceiling. No policy conclusion should be derived from it before reading Chapter 10, which discusses mitigation timing in the wider context of uncertainties, risks and impacts.

8.4.1 Alternative Pathways for Stabilization Concentrations

A given concentration ceiling can be achieved through a variety of emission pathways. This is illustrated in *Figure 8.12*. The top panel shows alternative concentration profiles for stabilization at 350-750ppmv. The bottom panel shows the corresponding emission trajectories. In each case, two different routes to stabilization are shown: the IPCC Working Group I profiles (from IPCC, 1995) and Wigley, Richels and Edmonds (WRE) profiles (from Wigley *et al*., 1996).

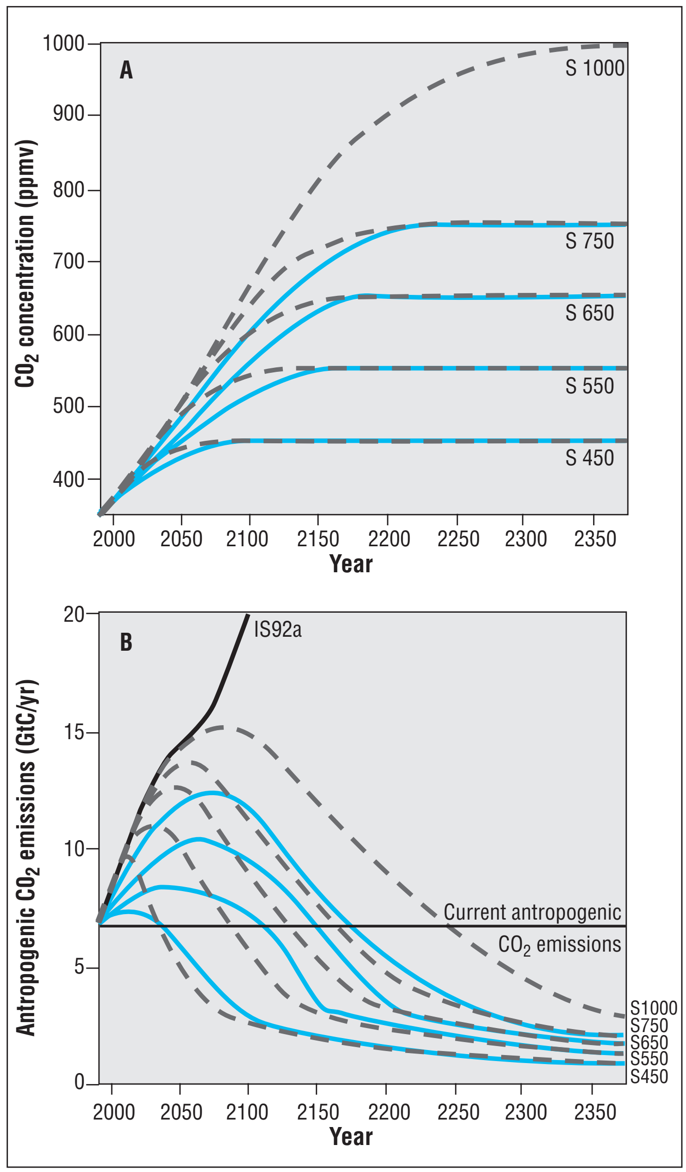

Figure 8.12: *Alternative pathways to stabilization.*

The choice of emission pathways can be thought of as a carbon budget allocation problem. To a first approximation, a concentration target defines an allowable amount of carbon to be emitted into the atmosphere between now and some date in the future. The issue is how best to allocate this budget over time. A number of modellers have attempted to address this issue. Unfortunately, to model stabilization costs is a daunting task. It is difficult enough to forecast the evolution of the energy and economic system to 2010. Projections over a century or more are necessary, but must be treated with considerable caution. They provide useful information, but their value lies not in the specific numbers but in the insights.

This section examines how mitigation costs might vary both with the stabilization level and with the pathway to stabilization. Also discussed are key assumptions that influence mitigation cost projections. Important, this discussion begins with the assumption that the stabilization ceiling is known with certainty and neglects the costs of different damages associated with different pathways (discussed in Chapter 10). Here, the challenge is to identify the least-cost mitigation pathway to stay within the prescribed ceiling. In Chapter 10, the issue of decision-making under uncertainty is discussed regarding the ultimate target and impacts of different pathways. Decision making under uncertainty requires indeed examining symmetrically the costs of accelerating the abatement in case of negative surprises about damages of climate change and adopting a prudent near-term hedging strategy. That is, one that balances the risks of acting too slowly to reduce emissions with the risks of acting too aggressively.

8.4.2 *Studies of the Costs of Alternative Pathways for Stabilizing Concentrations at a Given Level*

Some insight into the characteristics of the least-cost mitigation pathway can be obtained from two EMF studies (EMF-14, 1997; EMF-16, 1999) and from Chapter 2 in the SRES mitigation scenarios (IPCC, 2000). In the first EMF study, modellers compared mitigation costs associated with stabilizing concentrations at 550ppmv using the WGI and WRE profiles (see *Figure 8.12*), Note that the WGI pathway entails lower emissions in the early years, with less rapid reductions later on. The WRE pathway allows for a more gradual near-term transition away from carbon-venting fuels. *Figure 8.13* shows that in these models the more gradual near-term transition of the two examined results in lower mitigation costs.

The above experiment compares mitigation costs for two emission pathways for stabilizing concentrations at 550ppmv. It does not identify the least-cost mitigation pathway, however. This was done in the subsequent EMF (1997) study. The results are presented in *Figure 8.14*. In these studies the least-cost mitigation pathway tends to follow the models reference case in the early years with sharper reductions later on.

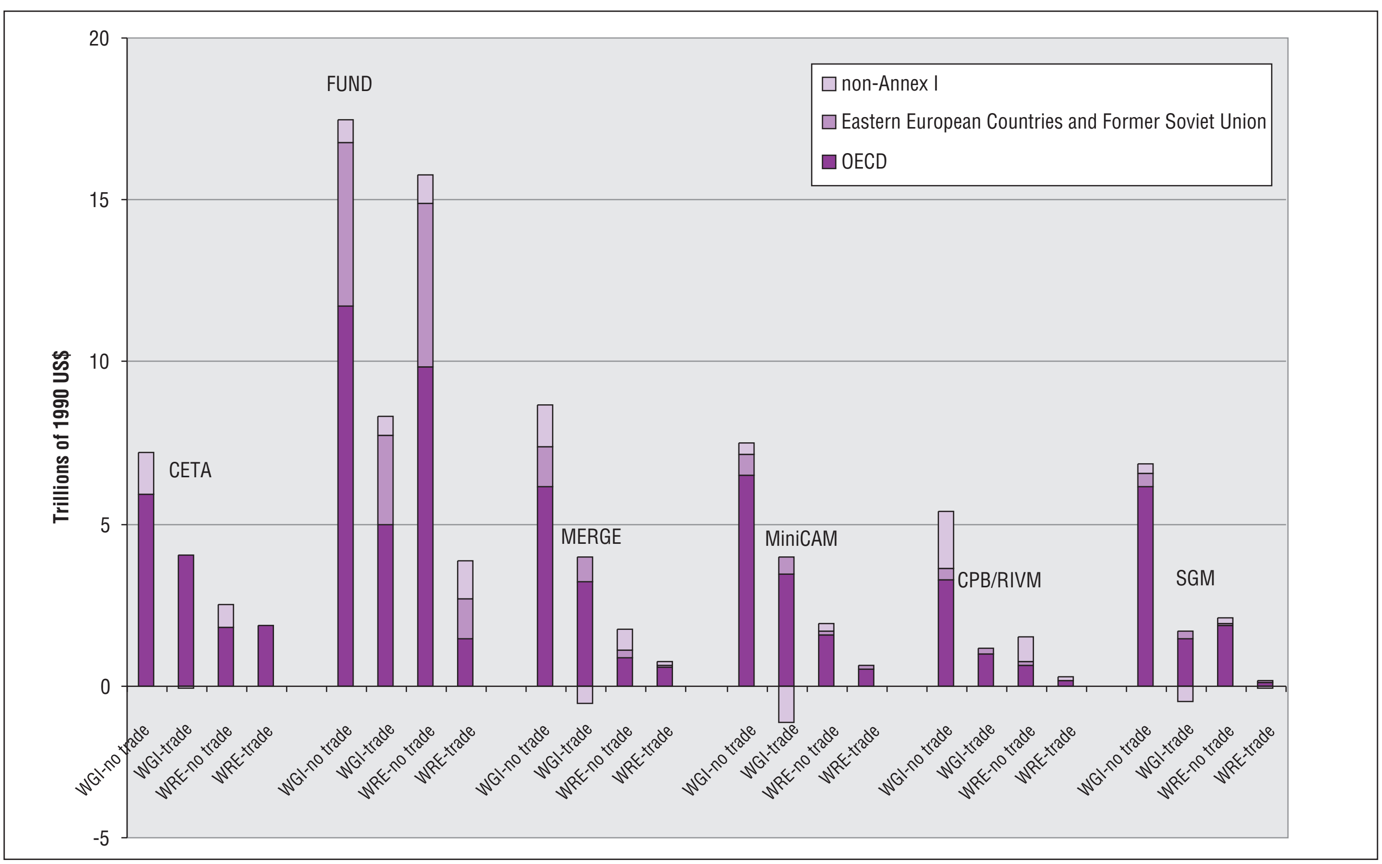

Figure 8.13: *Costs of stabilizing concentrations at 550ppmv; discounted to 1990 at 5%*

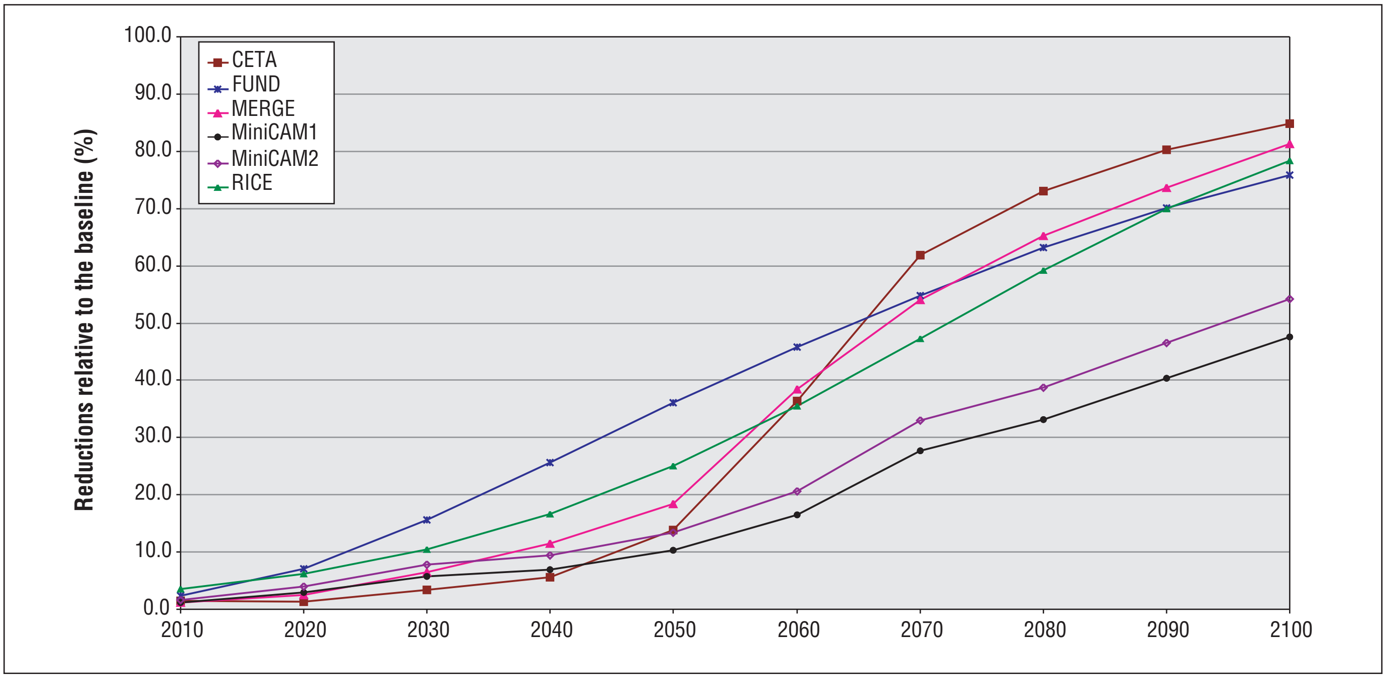

Figure 8.14: *Rate of departure from the baseline corresponding to least-cost mitigation pathway for a 550ppmv stabilization target.*

The selection of a 550ppmv target was purely arbitrary and not meant to imply an optimal concentrations target. Given the present lack of consensus on what constitutes "dangerous" interference with the climate system, three models in the EMF-16 study examined how mitigation costs are projected to vary under alternative targets. The results are summarized in *Figure 8.15*. As would be expected, mitigation costs increase with more stringent stabilization targets.

In Chapter 2, nine modelling groups reported scenario scenario results using different baseline scenarios. An analysis focused on the results of stabilizing the SRES A1B scenario at 550 and

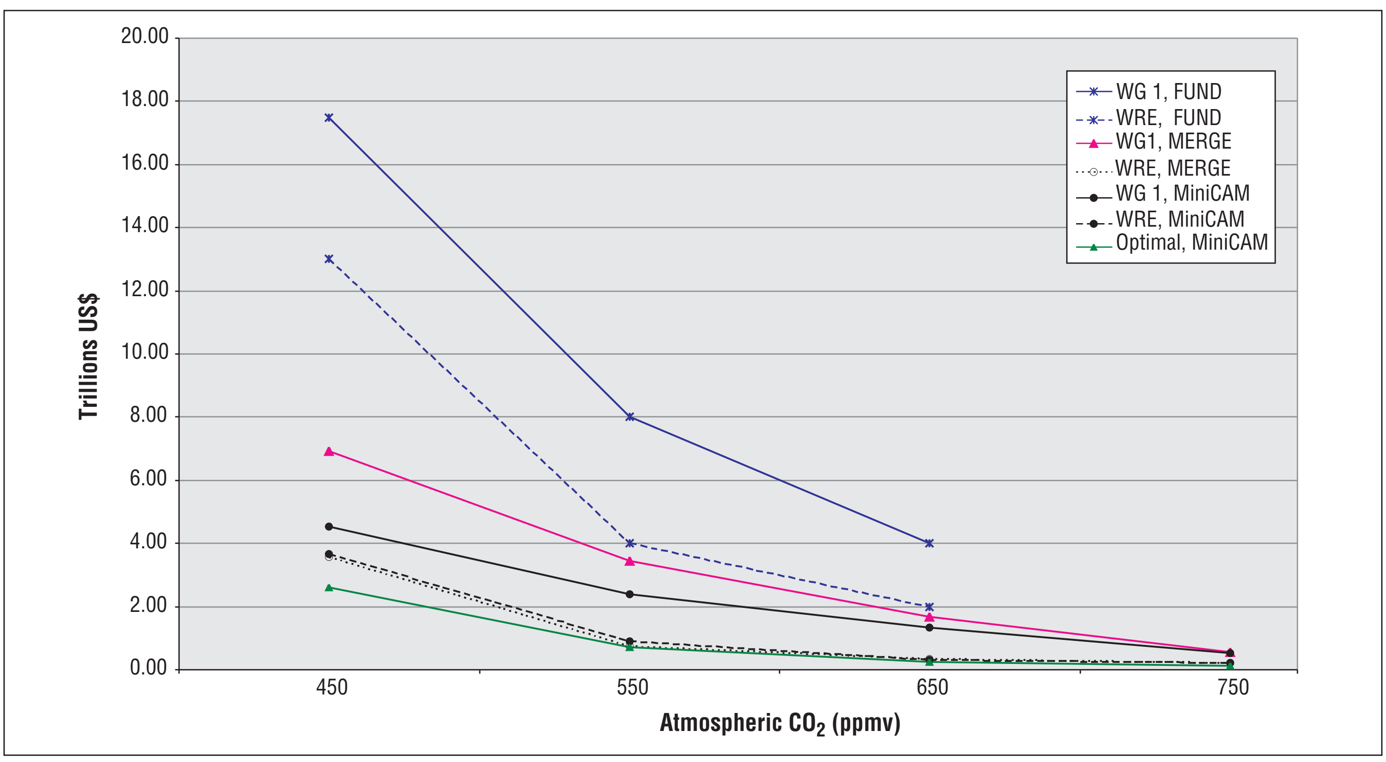

Figure 8.15: *Relationship between present discounted costs for stabilizing the concentrations of CO_2 in the atmosphere at alternative levels.*

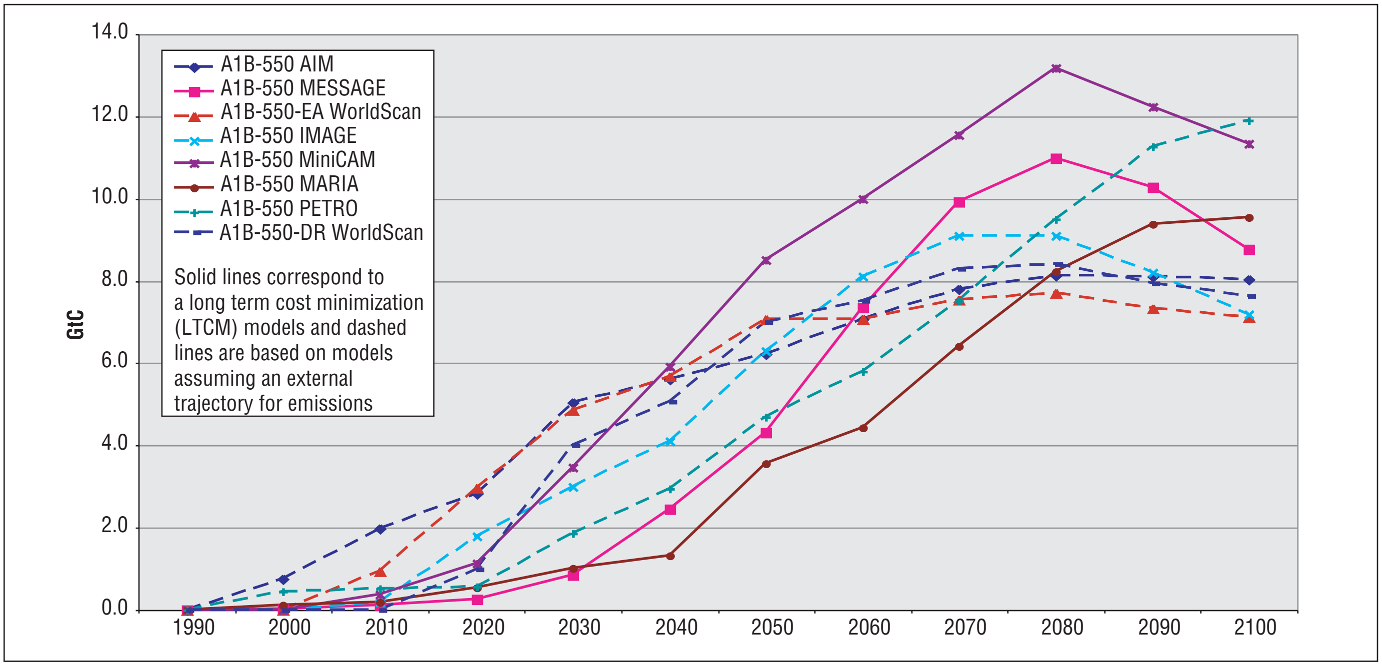

Figure 8.16: *Reductions in carbon emissions for the SRES A1B 550 case.*

450ppmv provides additional insight into the relationship between mitigation and baseline emissions. For the 550ppmv case, there are eight relevant trajectories (see *Figure 8.16*) giving the carbon reductions necessary to achieve a stabilization level of 550ppmv, where the models which impose a long-term cost minimization (LTCM) are represented as solid lines, and the models which use an external trajectory as the basis for their mitigation strategy are presented as dashed lines. The first impression of *Figure 8.16* is that even given common assumptions about GDP, population, and final energy use, and a common stabilization goal, there is still a lot of difference in the model results. A preliminary examination suggests that, in contrast to the non-optimization model results, a common characteristic among the LTCM models is that the near-term emissions pathways departs only gradually from the baseline.

Figure 8.17 clarifies the results by converting the absolute reduction to a percent reduction basis and averages them for the two classes of models. LTCM models show clearly a more gradual departure from the emissions baseline. *Figure 8.17*

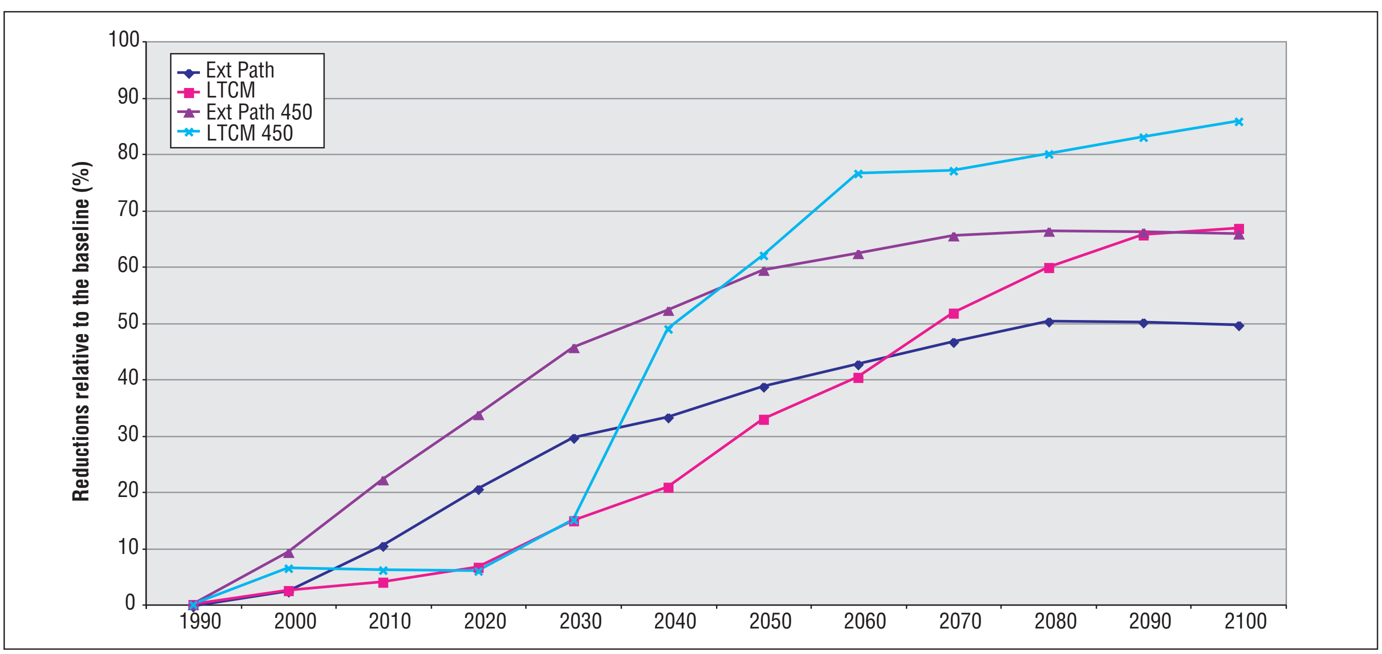

Figure 8.17: *Time path of emissions reductions.*

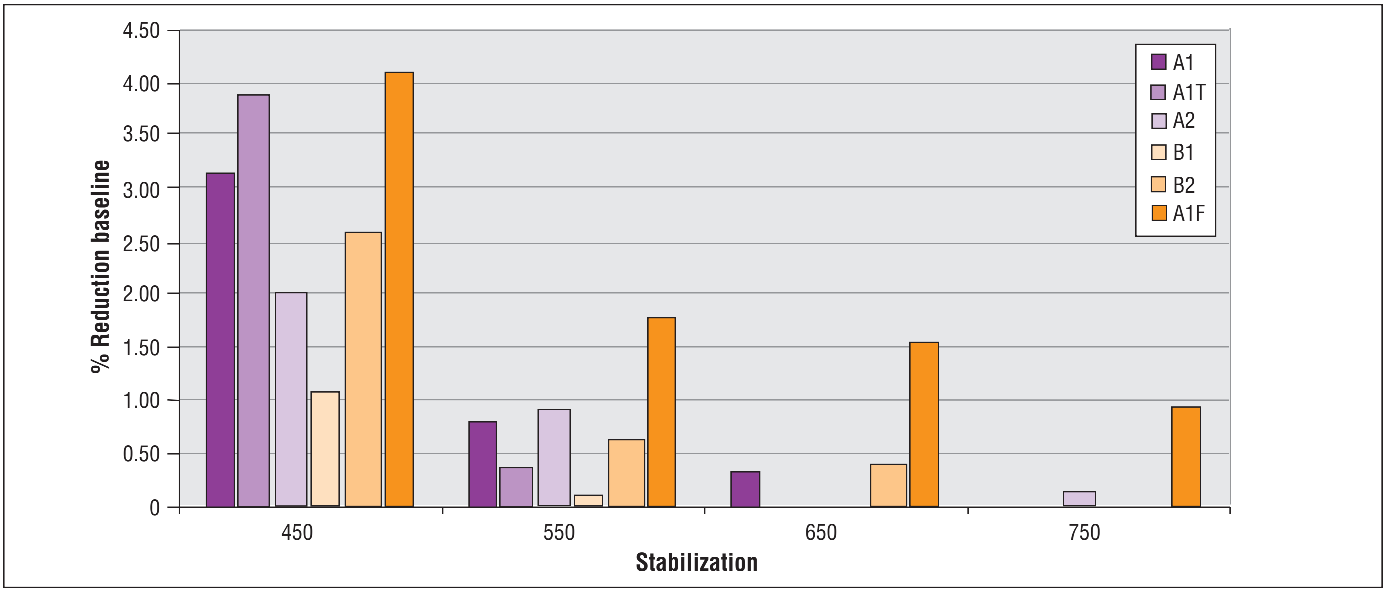

Figure 8.18: *Global average GDP reduction in 2050 for alternative stabilization targets and six SRES reference scenarios.*

also gives comparable results for the four cases with a 450ppmv target. The LTCM show a very similar decoupling until 2030, when this decoupling increases rapidly, and exceeds the other models by 2050, earlier in the 450ppmv case than in the 550ppmv case.

8.4.3 Economywide Impact of CO_2 Stabilization in the Post-SRES Scenarios

The economy-wide impact of stabilizing atmospheric CO_2 concentrations was assessed based on 42 post-SRES stabilization scenarios developed using the AIM, ASF, MARIA, MiniCAM, MESSAGE, and World SCAN models. These scenarios were developed by applying various mitigation policies and measures to the six illustrative scenarios (baselines) presented in the Special Report on Emission Scenarios (SRES).

The economy-wide impact of CO_2 stabilization was assessed on the basis of the difference in GDP in baseline scenarios and corresponding stabilization cases in a given year. This difference is expressed in percent (reflecting a relative GDP loss) and is positive when GDP in a baseline scenario is larger than in a stabilization case and is negative when GDP in a stabilization scenario is larger. Such an approach to measuring effects of stabilization was selected since it better reflects the societal burden of emission stabilization than absolute changes in GDP. For example, a 1% reduction in the 2100 GDP of the SRES A1 world is equal to about US$5.5 trillion and is larger in absolute terms than a 2% or US$5.0 trillion reduction in the 2100 GDP of the poor A2 world. Nonetheless, the relative level of effort in the latter case would be much greater. It should be also emphasized here that the GDP reduction itself represents a very crude indicator of economic consequences of the CO_2 stabilization. For example, most of the stabilization scenarios reviewed here have not rigorously accounted for the economic effects of introducing new low-emission technologies, new revenue rising instruments or adequate inter-regional financial and technology transfers, all elements which contribute to lower the costs as explained in the rest of the chapter.

The average GDP reduction in most of the stabilization scenarios reviewed here is under 3% of the baseline value (the maximum reduction across all the stabilization scenarios reached 6.1% in a given year). At the same time, some scenarios (especially in the A1T group) showed an increase in GDP compared to the baseline due to apparent positive economic feedbacks of technology development and transfer. The GDP reduction (averaged across storylines and stabilization levels) is lowest in 2020 (0.99%), reaches a maximum in 2050 (1.45%) and declines by 2100 (1.30%). However, in the scenario groups with the highest baseline emissions (A2 and A1FI), the size of the GDP reduction increases throughout the modelling period.

Due to their relatively small scale when compared to absolute GDP levels, GDP reductions in the post-SRES stabilization scenarios do not lead to significant declines in GDP growth rates over this century. For example, the annual 1990-2100 GDP growth rate across all the stabilization scenarios was reduced on average by only 0.003% per year, with a maximum reduction reaching 0.06% per year.

Figure 8.18 shows the relationship between the relative GDP reduction, the scenario group, and the stabilization level in 2050. The reduction in GDP tends to increase with the stringency of the stabilization target. But the costs are very sensitive to the choice of baseline scenario. The maximum relative reduction occurs in the A1FI scenario group, followed by the other A1 scenario groups and the A2 group, while the mini-

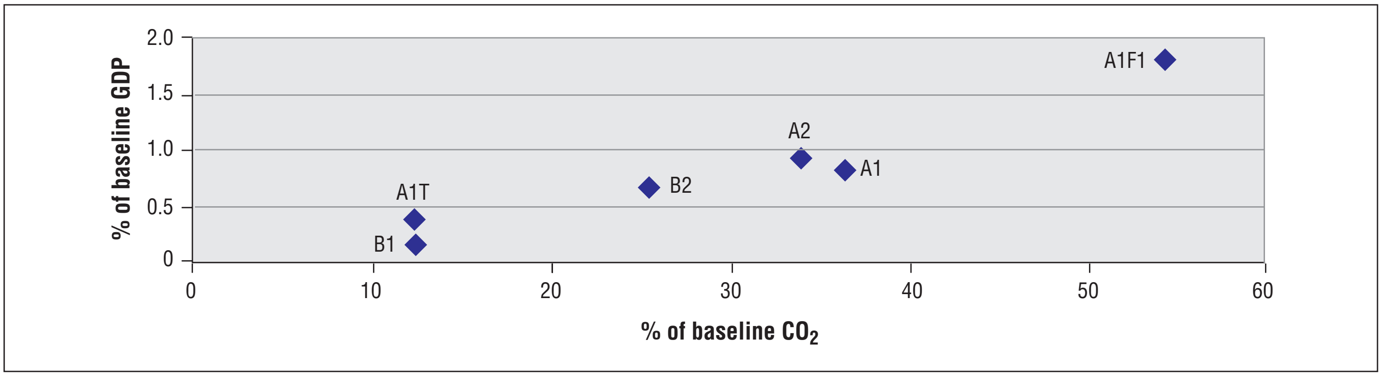

Figure 8.19: *Average GDP and CO_2 reductions in 550ppmv stabilization scenarios: year 2050 (labels identify different scenario groups).*

mum reduction occurs in the B1 group[20]. By 2100, the situation slightly changes with GDP reductions in the A2 scenario group becoming relatively more pronounced.

Differences in relative GDP reductions in different scenario groups are explained by the magnitude of corresponding CO_2 emission reductions needed to achieve a particular stabilization level. The emission reduction is apparently the largest in the A1FI scenario group, which is also associated with the largest GDP loss (*Figure 8.19*). Meanwhile, the smallest relative GDP loss occurs in the A1T and B1 groups, which have low baseline emissions and accordingly require the smallest reductions to reach the CO_2 stabilization.

Regional GDP reduction patterns in the post-SRES stabilization scenarios are also generally explained by corresponding required reductions in CO_2 emissions, which are determined by baseline emissions, the stabilization levels, assumptions about emissions trading mechanisms and about the relative contribution of regions to global CO_2 emissions; reduction and associated financial and technology transfer. In most of the baseline (SRES) scenarios starting from 2020, absolute CO_2 emissions in developing (non-Annex I) regions remain larger for the rest of the 21st century than in the industrialized (Annex I) regions.

8.4.4 Reasons why Energy-economy Models Tend to Favour Gradual Departure from Baseline in the Near-term

There are several reasons why the models tend to favour a more gradual departure from their reference path if used to study a pre-determined concentration level. First, energy using and energy producing capital stock (e.g., power plants, buildings, and transport) are typically long lived. The current system was put into place on the basis of a particular set of expectations about the future. Large emission reductions in the near term require accelerated replacement, which is apt to be costly. There is more opportunity for reducing emissions cheaply at the point of capital stock turnover.

Second, the models suggest that currently there are insufficient low-cost substitutes, on both the supply and demand sides of the energy sector, for deep near-term cuts in carbon emissions. With the anticipated improvements in the efficiency of energy supply, transformation, and end-use technologies, such reductions should be less expensive in the future.

Third, because of positive returns on capital, future reductions can be made with a smaller commitment of today's resources. For example, assume a net real rate of return on capital of 5% per year. Further, suppose that it costs US$50 to remove a tonne of carbon, regardless of the year in which the reduction occurs. To remove the tonne today would cost US$50. Alternatively, it only needs US$19 to be invested today to provide the resources to remove a tonne of carbon in 2020.

Finally, for higher near-term emissions, the size of the carbon budget (to meet a prescribed emission target) is higher, reflecting that the products of early emissions have a longer time to be removed from the atmosphere, and because the higher concentrations give higher oceanic and terrestrial sinks.

The fact that the least-cost mitigation pathway tends to follow the baseline in the early years has been misconstrued by some analysts as an argument for inaction. Wigley *et al.* (1996) note that this is far from the case: First, all stabilization targets still require future capital stock to be less carbon-intensive than under a business as usual (BAU) scenario. As most energy production and use technologies are long-lived, this has implications for current investment decisions. Second, new supply options typically take many years to enter the marketplace. To ensure sufficient quantities of low-cost, low-carbon substitutes in the future requires a sustained commitment to research, development and demonstration today. Third, any no regrets measures for reducing emissions should be adopted immediately. Lastly, it is clear that one cannot go on deferring emission reductions indefinitely, and that the need for substantial reductions in emissions is sooner the lower the concentration target.

[20] Please note that only one scenario was available for the GDP loss data for A1T-450, A1T-550, A1FI-450, A1FI-650, A2-450, and B2-650.

8.4.5 *Critical Factors Affecting the Timing of Emissions Reductions: The Role of Technological Change*

As pointed out by Grubb (1997), there are several key assumptions imbedded in the energy-economy models that influence the shape of the least-cost mitigation pathway. For a pre-determined target, these relate to the determinants of technical change; capital stock turnover and the inertia in the energy system; discounting; and, the carbon cycle. When the target is uncertain, they include in addition the probability attached to each target and risk aversion (see Chapter 10) which tend to favour a more aggressive departure from current trends.

The discount rate will not be discussed because it is less important in cost-efficiency frameworks (when the target is pre-determined) than in a cost-benefit one (when the discount rate reduces the weight of future environmental impacts, see Chapter 10). Neither are the very few studies discussed which try to assess different benefits in terms of environmental co-benefits of reducing GHG emissions presented. Wigley *et al.* (1996), for example, show pathway-related differentials up to 0.2°C in global mean temperature and 4cm in global mean sea-level (by 2100) for the WGI and WRE 550 stabilization pathways. See Chapter 10 for an elaboration of these timing issues. This part will rather insist on the key features of technical change that are numerically of utmost importance.

To the extent that the cost of reducing emissions is lower in the future than at present, the overall cost of stabilizing the CO_2 concentration is less if emissions mitigation is shifted towards the future. This shift occurs in all models. The extent of the shift that minimizes the cost of limiting the concentration of atmospheric CO_2 depends, at least in part, on the treatment of technological change. Without technological change, the problem is simple and the results of Hotelling (1939) apply. With endogenous technological change, the problem becomes more complex.

This discussion of the determinants of technological change must begin with the acknowledgement that no adequate theory of endogenous technological change exists at present. Many researchers have contributed to the field, but the present state of understanding is such that present knowledge is partial and not necessarily fully consistent. Although no complete theory of technological change exists, two elements have been identified and explored in the literature: induced technological change (ITC) and learning-by-doing (LBD). Work by Ha-Duong *et al.* (1997), Grubb *et al.* (1995), Grubb (1997), and Kypreos and Barreto (1999) examined the implication of ITC, LBD, and inertia within the context of uncertainty and an imperative to preserve the option of concentration ceilings such as 450ppmv. They conclude that emissions mitigation can be shifted from the future towards the present under appropriate circumstances.

Goulder and Mathai (1998) also explore how the effect on timing depends on the source of technological change. When the channel for technological change is R&D, ITC makes it preferable to concentrate more abatement efforts in the future. The reason is that technological change lowers the costs of future abatement relative to current abatement, making it more cost-effective to place more emphasis on future abatement. However, when the channel for technological change is LBD, the presence of ITC acts in two opposite directions. On the one hand, ITC makes future abatement less costly but, on the other hand, there is an added value to current abatement because such abatement contributes to experience or learning and helps reduce the costs of future abatement. Which of these two effects dominates depends on the particular nature of assumptions and firms. In recent years, there has been a good deal of discussion about the potential for ITC (e.g., Anderson *et al.*, 1999). Proponents argue that such changes might substantially lower, and perhaps even eliminate, the costs of CO_2 abatement policies. These discussions have exposed very divergent views as to whether technological change can be induced at no cost, or whether a resource cost is involved. For example, in a 1995 article, Porter and van der Linde (1995) contend that properly designed regulation can trigger innovation that may partially or more than fully offset the costs of compliance. Indeed, they argue that firms can actually benefit from more stringent regulation than that faced by their competitors in other countries. However, in an accompanying article, a strongly contrary view is put forward by Palmer *et al.* (1995). Examining available data, they found that such offsets pale in comparison to expenditures for pollution abatement and control.

8.4.5.1 ITC through Dedicated R&D

Including R&D driven ITC in climate mitigation models leads to ambiguous results in terms of time profile and tax level in a cost-benefit framework In a cost-effectiveness framework, the optimal tax is lower in the case with R&D driven ITC and has to be set up early even if the effective resulting abatement shifts from the near-term to the more distant future. If there are market failures in the R&D market (e.g., knowledge spillover), then subsidies for R&D are justified as it enhances social welfare and raises the abatement level (Goulder and Schneider, 1999; Weyant and Olavson, 1999; Goulder and Mathai, 2000).

However, R&D driven-ITC can reduce the gross costs of a carbon tax under special circumstances. Specifically, if R&D has been substantially over-allocated towards the fossil fuel industries prior to the imposition of a carbon tax, the carbon tax can reduce this allocative inefficiency and, as a result, its costs can be quite low or even negative. A substantial prior misallocation towards carbon-intensive industries could occur if there were prior subsidies towards R&D in the fossil fuel industries (with no comparable subsidies in other industries), or if there were substantial positive spillovers from R&D in non-carbon industries (with no comparable spillovers in the fossil fuel industries). Under other plausible initial conditions, however, R&D driven-ITC raises, rather than lowers, the net social costs of a given carbon tax because of the crowding out of R&D from other sectors; to put it clearly the tax level for a given abatement is lower than under the hypothesis of exogenous technical change and part of this decrease is offset when all the general equilibrium effects are accounted.

The same model has been employed to compare the costs of achieving a given abatement target through carbon taxes and R&D subsidies (Schneider and Goulder, 1997; Goulder and Schneider, 1999). If there are no spillovers to R&D, the least-cost way to reach a given abatement target is through a carbon tax alone. The carbon tax best targets the externality from the combustion of fossil fuels related to climate change, and thus is the most cost-effective. However, if there are spillovers to R&D, the least-cost way to achieve a given abatement target is through the combination of a carbon tax and R&D subsidy. If spillovers are present, there is a market failure in the R&D market as well as a (climate change related) market failure associated with the use of carbon. Two instruments (the R&D subsidy and the carbon tax) are needed to address the two distinct market failures most efficiently. In general, a R&D subsidy by itself does not offer the least-cost approach to reducing carbon emissions. Results from this model are highly sensitive to assumptions about the nature and extent of knowledge spillovers. Further empirical work that sheds light on these spillovers would have considerable value.

8.4.5.2 Learning by Doing (LBD)

LBD as a source of technical change was first emphasized by Arrow (1962). Nakicenovic (1996) discussed the importance of LBD in energy technology, and Messner (1995) endogenizes the learning process in energy models. LBD is a happy consequence of those investments in which learning is a result of cumulative experience with new technologies. LBD typically refers to reductions in production cost, in which learning takes place on the shop floor through day-to-day operations, not in the R&D laboratory. The LBD component of change is significant too. Kline and Rosenberg (1986) discuss industry studies that indicate that LBD-type improvements to processes in some cases contribute more to technological progress than the initial process development itself.

LBD models use the installed capacity or cumulative use as an indicator of accumulating knowledge in each sector. The abatement costs are represented by the specific investment costs in US$/kWh. The models are global and therefore the diffusion process is not represented. The optimization problems are non-convex, which raises a difficult computational problem to find an optimum. However, pioneering work at the International Institute for Applied Systems Analysis (IIASA) on the MESSAGE model and additional developments based on models like MARKAL and ERIS; (MATSSON), Kypreos and Barreto (1999), Seebregts *et al*. (1999a), (SKFB), Tseng *et al*. (1999), and Kypreos *et al*. (2000) demonstrate progress in this direction. They show that several technologies are likely to play a prominent role in reducing the cost of abatement, if ITC is indeed taken into account when computing the equilibrium. A problem with modelling endogenous technological change is that the traditional baseline scenario versus optimal policy run argumentation is not feasible. This follows directly from the path dependence. The most important results are: greater consistency of model results with the observed developments of technological change;

- new technologies first appear in niche markets with rising market shares;
- the time of breakthrough of new technologies can be influenced by policy measures (taxes and R&D) if they are strong enough;
- identification of key technologies, like photovoltaic modules or fuel cells, for public R&D investments is difficult; and
- technological lock-in effects depend on costs.

The most important conclusion for the timing of a mitigation policy is that early emissions-reduction measures are preferable when LBD is considered. This is confirmed unambiguously by a macroeconomic modelling study (van der Zwaan *et al*., 1999/2000) which finds also lower levels of carbon taxes than those usually advocated.

These findings must be tempered by the fact that the models are not only highly non-linear systems, and therefore potentially sensitive to input assumptions, but also the quantitative values employed by modellers are typically drawn from successful historical examples. Furthermore, the empirical foundations of LBD are drawn from observations of the relationship between cumulative deployment and/or investment in new technology and cost. This relationship is equally consistent with the hypothesis that a third factor reduced costs, in turn leading to increases in demand. The authors restrict their findings to more qualitative assertions, because of the limitations of current models (Messner, 1997; Grübler and Messner, 1998; Barreto and Kypreos, 1999; Seebregts *et al*., 1999a, 1999b). The research so far has been limited to energy system models and ignored other forms of endogenous, complex changes that are important for emissions, like changes in lifestyles and social institutions.

8.4.5.3 The Distinction Between Action and Abatement

The key message from this discussion about technical change is that a clear distinction has to be made between the timing of action and the timing of abatement. As a result of inertia in technological innovation, short-term action is required to abate more in the future, but a given amount of abatement at a given point in time is not a good measure of the effort. The necessity of this distinction is reinforced by the consideration of inertia in capital stocks. Mitigation costs are influenced by assumptions about the lifespan of existing plants and equipment (e.g., power plants, housing, and transport). Energy-related capital stock is typically long lived and premature retirement is apt to be costly. For example, an effort to change the transportation infrastructure will not reduce carbon emissions significantly for two decades or more. Hence, a drastic departure from the current trend is impossible without high social costs and a delay of action in this sector will require higher abatement costs in the more flexible sectors to meet a given target. Lecocq *et al*. (1999) found that these costs would be increased by 18% in 2020 for a 550ppmv target and by 150% for a 450ppmv target.

This irreversibility built into technological change is far more critical when the uncertainty about the ultimate target is considered. In this case indeed, many of the parameters that legitimize the postponing of abatement play in the opposite direction. If indeed the concentration constraints turn out to be lower than anticipated, there may be a need for abrupt reduction in emissions and premature retirement of equipment. In other words, even if the permanent costs of an option (in case of perfect expectation) are lower than those of an alternative option, it may be the case that its transition costs are higher because of inertia. For example, two ideal transportation systems can be envisaged, one relying on gasoline, the other on electric cars and railways, both with comparable costs in a stabilized situation; however, a brutal transition from the first system to the second may be economically disruptive and politically unsustainable. These issues are examined in more depth in Chapter 10 because the selection of the ultimate target depends upon the decision-making framework and upon the nature of the damage functions. But, it matters here to insist on the fact that the more inertia is built into the technical system, and the less processes of learning by doing and induced technical change have operated, the more costly corrections of trajectories in hedging strategies will be, for example, moving from a 550ppmv concentration goal to 450ppmv (Ha-Duong *et al.,* 1997; see also Grubb *et al*., 1995; Grubb, 1997). This possibility of switching from one objective to another is supported by current material regarding climate damages, in particular (Tol, 1996) if the rate of change is considered in the analysis and the delay between symptoms and the response by society (see Chapter 10).

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