Chapter 6
CGE Models in Environmental Policy Analysis: A Review and Spanish Case Study

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6.1 Introduction

The publication of the Fifth Assessment Report by the Intergovernmental Panel on Climate Change (IPCC) has underlined once again the serious consequences of failing to act sufficiently to bring down global Greenhouse Gas (GHG) emissions. These consequences include (although are not restricted to) disrupted livelihoods from increased flooding; risks resulting from damage to infrastructure from extreme weather events; increased morbidity and mortality rates from periods of extreme heat and issues of food insecurity resulting from droughts, floods, and precipitation volatility. At the global level, the successor to the Kyoto agreement, the Paris Conference of Parties (COP) of the United Nations Framework Convention on Climate Change (UNFCCC) ratified in December 2015, faces new uncertainty with the United States having pulled out of the agreement. For its part, since the launch of its Emissions Trading Scheme (ETS) in 2005, the European Union (EU) has set its own relatively ambitious unilateral GHG reduction targets to 2020, with mooted GHG reductions of up to 40% (EC 2014) by 2030 (compared with 1990 levels).

The use of computable general equilibrium (CGE) simulation models in the analysis of environmental and energy policy has a long history. In seeking to provide the reader with a broad overview of the key issues currently facing CGE models in environmental policy analysis, part one of this chapter discusses the main modelling-, data- and scenario driven innovations which have occurred in the CGE literature. Thus, the chapter traces back to the early days of general equilibrium models being applied to energy and environmental issues, beginning with coverage of applications examining energy and fossil fuels, during and after the oil shocks of the 1970s. With steady improvements in computational facility and greater availability of secondary data sets, the degree of complexity of the issues tackled by CGE models

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also increased. As a result, more recent CGE studies incorporate a much more intricate representation of (inter alia) land use and production technologies, whilst other research extended the models by including bottoms up engineering estimates to anticipate potential uptake of abatement technologies in response to tighter emissions reductions. Finally, a further area of advancement has been in the modelling and scenario design to examine environmental policy options through different permit allocation schemes, issues of 'carbon leakage' or to explore the so-called 'double dividend' hypothesis.

In part two of this chapter, the focus narrows to examine an application of a single country neoclassical CGE model of the Spanish economy with a particular emphasis on the primary agricultural sectors. In 1990, Spain had the sixth highest GHG emissions of the EU27, although the ensuing period was characterised by aggressive economic growth driven by the construction boom up to the financial crisis. As a result, under a burden sharing scheme, the Spanish emissions target in 2012 was directed toward limiting the rate of increase rather than absolute reductions in Spanish GHG emissions. Nevertheless, under the Climate and Energy Package, the major sources of GHGs not covered by the ETS (waste, transport, buildings and, in particular, primary agriculture) were obliged to reduce emissions by 10% in Spain, whilst reductions in ETS sectors will be dependent on domestic allocations, and on the carbon price determined by the demand for (i.e. economic conditions) and supply of (i.e. EU policy) permits. Under three emissions reductions scenarios and employing some of the methodological innovations discussed in the literature review, a neoclassical single country CGE model of Spain examines the implications for the Spanish primary agricultural sectors and the broader macro-economy.

Part One: Key Issues in CGE Environmental Policy Modelling

6.2 Energy-Economy CGE Models

Hudson and Jorgenson (1974) constructed a model which drew on both the econometric approach developed by Goldberger and Klein (1955) and the Input-Output analysis of Leontief (1941) to project a macroeconomic growth path for the U.S. economy. This study demonstrates three principal uses of CGE in energy/environmental analysis: to project forward a 'business-as-usual' baseline, which allows analysts to explore the possible future structure of the economy in the absence of significant unforeseen changes; to analyse the impact of a given change in policy (in this case, energy taxes); and to estimate the level at which a policy (such as a tax) must be applied in order to meet a given objective (in this case, energy independence). These three uses will be seen repeatedly throughout the papers discussed below, and in this study.

The authors extended their work with an in depth analysis of the dynamic effects of energy policy on economic growth in Hudson and Jorgenson (1978), a subject also touched upon in Hazilla and Kopp (1990) and Adams et al. (2000). The common thread in all three studies is that restrictions on energy use or pollution reduce economic output in the short run, and growth in the long run, by reducing the productivity of labour and capital, as they have less energy to work with. In the short run total output is a function of the stocks of these factors and their productivity, so reducing the latter causes a contraction in the productive capacity of the economy. In the long run, lower capital returns discourage investment, and a lower real wage encourages workers to substitute leisure for labour (assuming an upward sloping labour supply curve). Thus in the long run both factor endowments and their productivities are reduced, resulting in a lower rate of growth than that which would have arisen in the absence of restrictions.

Another set of papers uses dynamic CGE models to explore the idea of 'optimal pathways' for greenhouse gas emissions over time (Nordhaus 1990, 1992; Hamdi-Cherif 2012). These inter-temporal models aim to simulate the optimal level of emissions at any given point in the simulation period. Technological progress means abatement is relatively cheaper in later periods, but an environmental damage module means there is a net present value to avoided emissions in early periods as they do not add to stocks of pollutants. Martin and Van Wijnbergen (1986) use a similar concept to map out an optimal use pathway for natural resource depletion, based on the seminal work on the subject by Hotelling (1931). This maps the rate at which a scarce resource is used up to the development of alternative technologies which do not rely on the resource and the net present values of current and expected future returns to using the dwindling resource in different periods. These studies all have to deal with the question of the discount rate, i.e. the weight which the material welfare of future generations is given relative to that of the current generation. This is a difficult issue for the economics profession as it concerns questions of ethics as well as efficiency. For example, the Stern Report on Climate Change (Stern 2007) controversially used a discount rate of zero.

A key development in the literature by Rutherford and Montgomery (1997), Böhringer (1998) and Böhringer and Rutherford (2008), was to combine the 'bottom-up' detail of an energy model with the 'top-down' interactions of a CGE model. In Rutherford and Montgomery (1997), the CGE model derives energy demands which are an input into the partial equilibrium (PE) model used to derive energy prices, which then feedback into the CGE model—an iterative process which repeats itself until the results of the two models converge. Böhringer (1998) and Böhringer and Rutherford (2008) employ model complementarities within the energy sector such that specific types of plants come online when they become profitable, and a non-zero price for a specific energy source emerges when demand reaches supply, with plant costs and capacities coming from bottom-up energy data.

A further development for characterising energy sectors in a CGE model was through the representation of their production technologies. More specifically, the 'nesting' structures within the production function are arranged, subject to the availability of plausible substitution elasticities, to determine more accurately the production processes which govern output in these industries. An early example is the OECD's GREEN model (Burniaux et al. 1992; Lee et al. 1994), wherein the top nest of energy inputs, firms choose between an electricity composite and non-electrical energy. At the next level down the non-electrical composite divides into coal on one branch, and an oil and gas composite on the other, and at a further level down the oil and gas composite splits into those two fuels. This general approach has filtered into the mainstream literature through its adoption in (inter alia) the GTAP-E model (Burniaux and Truong 2002), the MMRF-Green model (Adams et al. 2000), and the ORANI model (Horridge et al. 1993).

6.3 Different Pollutants and Environment-Economy Feedbacks

In the environmental extension to his Input-Output framework, Leontief (1970) illustrated the importance of how pollution is assigned by taking the data for emissions by industry, and reallocating it on the basis of emissions embodied in final demands. In presenting, if only briefly, this form of analysis, Leontief showed an early form of the so-called ‘farm to fork’ method of measuring total emissions associated with the production of a given agricultural commodity, which has more recently garnered increasingly popular in academic and policy circles (FAO 2010). In the same study he extended the notion of ‘input-output coefficients’ to ‘discharge coefficients’ which attach pollution to output or to the use of certain inputs in specific industries. A similar approach was adopted by Willett (1985), Conrad and Schröder (1991) and numerous studies since.

In the DICE global climate change model, and its regional counterpart RICE, Nordhaus (1990) and Nordhaus and Yang (1996) include an environmental damage function which translates stocks of greenhouse gases in the atmosphere (which grow each year with emissions) into radiative forcing which provokes a global temperature increase causing economic damage, the severity of which varies between industries. In the latter study, the regional component of the damage function comes from the fact that different industries have different weightings in different regions, not because of any geographical features of the regions in question. By contrast, the GEM-E3 model (Capros et al. 2013) tracks the stocks of a number of different pollutants, and translates them into specific geographical areas and damage functions. Concentration of pollutants causes damages to human health, soils, forests, buildings and territorial eco-systems. Other studies which include feedback mechanisms from the environment to the economy include Vennemo (1997) and Xie and Saltzman (2000). Both include a negative relationship between increasing pollution and factor productivity, and a direct effect of pollution on utility.

6.4 Land Use Change and Forestry

Ahammad and Mi (2005) adapt the Global Trade and Environmental Model (GTEM) to include eighteen different land types based on Agro-Ecological Zones (AEZs). The AEZs distinguish land on the basis of three different climate areas (tropical, temperate and boreal), and 6 different lengths of growing season. The supply of each type of land is fixed, but the production function for agriculture is modified to allow farmers to substitute between the different land types, and between land and fertiliser at a low level of the nest. In addition, the stock of forest area is disaggregated by age, land class and management type, with different carbon densities associated with each. A Constant Elasticity of Transformation (CET) function determines at the first level the movement of land between agriculture and forestry, and then at higher levels the movement of land between different agricultural uses. While most GHG emissions from agriculture are attached to fertiliser use or livestock output, emissions of N_2O from soil disturbance are dependent on the area of land used for agriculture. Net emissions from forestry depend on the change in the carbon stock of forest land, which is a function of the area de- or re-forested, its timber yield, and associated carbon stocking density. Policies to regulate or tax emissions are thus likely to encourage forestry at the expense of agriculture by effectively subsidising land used in forestry and taxing the agricultural sector.

This approach is also used in Golub et al. (2009) with some variations. The paper contains a detailed treatment of the rate at which previously inaccessible forests are accessed depending on the land rents available and the cost of accessing land. The former increases with demand for crop, livestock and forestry products leading to a derived demand for increased land, while the latter increases with the proportion of total land which has been accessed, reflecting the fact that as more land is demanded, the land coming into production is more marginal and so costs more to access. This leads to a Ricardian treatment of land rents whereby inaccessible land will be brought into production when the net present value of the land is equal to the cost of accessing it, so as accessed land increases, rents will rise on previously accessed land. Golub et al. (2009) also explicitly distinguish between the intensive and extensive margins for carbon sequestration in forestry. The extensive margin governs the decision to cut down or plant forests, and is dependent on the land rents and demand. The intensive margin is the potential for a fixed area of forest to hold more carbon through the ageing process, or changes in management practices. This is modelled by increasing the use of forestry products in the forestry sector, thus decreasing net output in order to increase the timber—(and hence carbon) intensity of forests.

Bosello et al. (2010) use a CGE model to analyse the importance of the scheme Reducing Emissions from Deforestation or forest Degradation (REDD) in EU emissions reduction targets for 2020. In their model, the avoidance of deforestation in Latin America, Sub-Saharan Africa and South East Asia generates carbon permits which can be sold on the EU ETS market. This results in a transfer of payments from the EU to those regions, but also reduces land available for agriculture, and timber available for wood products. They find that the inclusion of REDD credits significantly reduces the ETS permit price, but also leads to an increase in the price of land, which is strongest in South East Asia, and the price of timber, particularly in Sub-Saharan Africa.

A number of studies use CGE models to investigate the effects of the re-cent growth in biofuels production on land use and on emissions reduction possibilities. One such paper is Birur et al. (2008), which modifies a version of the GTAP-E model to include biofuels used by both consumers and producers, and land use type by AEZ. The paper distinguishes between cereal- and sugar-based bioethanol and biodiesel from vegetable oil. This distinction is significant as each has different 'feedstock' crops, so each will have different impacts on land use change, as well as having more natural advantages in different geographic areas. Consumers in the model treat each type of biofuel as highly substitutable with petrol, whilst in production, biofuel is treated as a Leontief complement to petrol use. On the supply side, a CET function governs the ease with which land of each AEZ can move between different uses, with a much higher elasticity between different crop types than crops and pasture, or at the most extreme agriculture and forestry. It is this which restricts land use changes, as farmers are seen as relatively indifferent as to which type of land they use, with a high elasticity of substitution between different AEZs in the agricultural production function.

6.5 Marginal Abatement Cost (MAC) Curves in CGE Models

A number of the papers already mentioned above include some approximation of end-of-pipe abatement options. Xie and Saltzman (2000), for example develop an Environmental Social Accounting Matrix (ESAM) for China based on the extended input-output table in Leontief (1970). The ESAM includes intermediate and factor purchases for abatement by each industry in the model, as well as government purchases of pollution cleaning services. Bergman (1991), Conrad and Schröder (1991), Adams et al. (2000) and the GRACE model (Rypdal et al. 2007; Rive 2010) all allow firms to use additional quantities of factor and inter-mediate inputs to reduce pollution, although in none of these papers is such 'cleaning' the focus of the study.

An important early study on the inclusion of what has come to be known as 'end-of-pipe' abatement in CGE models was that by Nestor and Pasurka (1995a, b), who used detailed German data showing expenditure on specific abatement inputs to extend the input-output data to include both those which are internal to the firm (i.e. use the firm's own labour and capital), and intermediate inputs purchased from an abatement sector. They note that CGE models offer a significant advantage in modelling environmental compliance as the costs of pollution reduction may be mitigated for those industries whose output is used in abatement activities. As an example, their results suggest that the (German) abatement sector is relatively energy intensive, such that the direct effects of environmental policy on the energy sectors are reduced by the increase in energy demand from the rest of the economy as abatement increases. In this study a government agency collects all abatement expenditure as a 'tax' and uses it to hire factors and buy inputs from the abatement sector. In recent years, a number of researchers have treated emissions as a necessary input into production. One of the first studies to use this approach as a step towards incorporating MAC curves into a CGE model was Hyman et al. (2003), which treats emissions as an additional input within the production process by characterising CES possibilities between greenhouse gas emissions and the use of a composite input (i.e., intermediate inputs and primary factors). Thus firms can reduce their emissions either by reducing their output, or by increasing their use of all conventional inputs relative to output. The elasticity of substitution between emissions and the conventional inputs composite is then calibrated for each industry to match its MAC curve. The most important implication of this approach, in the light of the current study, is that it implicitly assumes that abatement expenditures will have the same cost structure as the industry's production process. This is a significantly different approach to Nestor and Pasurka (1995a, b), described above. Essentially, comparing across different industries, the cost shares of abatement expenditure following the Nestor and Pasurka approach will be the same, whereas in the Hyman et al. (2003) approach, they are approximated by the production cost shares in each industry.

A number of papers (Dellink 2000; Dellink et al. 2004; Dellink and Van Ierland 2006; Gerlagh et al. 2002) use detailed data on abatement options and their associated costs in the Netherlands to construct a single MAC curve for each environmental 'theme', such as climate change or acid rain. Thus all available technologies for the abatement of any greenhouse gas in any industry are included in the same MAC curve, which avoids the problem of a small number of data points in calibration. Similar to Hyman et al. (2003), pollution is treated as a necessary input into production, and an elasticity of substitution is calibrated to the MAC curve. However, in this case, the elasticity is not at the top level of the nest, but rather between abatement and abatable emissions. These papers also include a maximum technical abatement potential (based on the data on abatement technologies) such that a certain proportion of emissions is classified as 'unabatable'. These are produced in fixed proportions to output, as is the composite of abatable emissions and abatement measures. Akin to the Nestor and Pasurka approach, a single abatement sector provides 'abatement measures' to every industry for each environmental theme. In some respects this approach could thus be seen as an attempt to reconcile the two methods described above.

The current state of the art in this field is described in Kiuila and Rutherford (2013). The paper compares, on the one hand, sector specific and economy-wide approaches to abatement and, on the other hand, 'traditional' and 'hybrid' approaches. Briefly, the sector specific approach treats abatement as internal to each industry in the model. This can be seen as the optimal method, but can be limited by data availability. The economy-wide approach has an 'abatement sector', from which all other industries purchase abatement services, which assumes the cost structure of abatement technologies is constant across abating industries and gases. Furthermore, the traditional approach has a smooth (CES) production function for abatement, whilst the hybrid approach attempts to integrate stepwise MAC curves from bottom-up data though Leontief functions for specific technologies that become active when the carbon price reaches a certain ‘trigger’ level. The study suggests that at low levels of abatement, a smooth approximation gives similar results to the stepwise function. When abatement options reach their maximum potential though, the step function approaches infinity more immediately than the smooth curve, so at these higher levels of abatement the traditional approach may overestimate abatement potential.

6.6 Emissions Reduction Options

Many of the early studies of environmental policy focussed on standards and restrictions on emissions (see, for example, Blitzer et al. 1994; Ellerman and Decaux 1998; Wang et al. 2009). The results tend to support (or are caused by) the neoclassical assumption that the cheapest options for reducing emissions (the so-called ‘low hanging fruit’) will be exploited first, thus the marginal cost of abatement rises with abatement. This result is found so consistently that it seems generally sound, but a note of caution is needed. Some abatement technologies (specific types of renewable energy, or carbon capture and storage, for example), may require high levels of initial investment to reach a ‘tipping point’, after which the marginal costs of spreading the technology (and the resulting abatement) may be significantly lower. If enough abatement technologies follow this pattern, the effect may be enough to cause a kink in the otherwise smoothly convex cost curve for emissions reductions. These complexities often relate to industry structure, and are difficult to include in a CGE context, but modellers should be aware that they are implicitly assuming perfect knowledge of the total (investment and operating) costs of emissions reduction options, and of their abatement potential.

A number of global CGE models have shown the importance of including non-CO_2 gases by comparing on the one hand, scenarios where temperature or radiative forcing (see footnote 3 above) targets are met solely through reductions in CO_2 emissions with, on the other hand, studies where other GHGs could contribute to meeting the target (Hyman et al. 2003; Bernard et al. 2006; Tol 2006). A significant and consistent finding across the papers was that non-CO_2 gases are likely to contribute a relatively higher proportion of emissions reductions when the total target is less stringent. This is because abatement options for these gases tend to be cheaper than those for CO_2, but technically limited. Thus as emissions reduction targets become more stringent, CO_2 takes more of the burden—though obviously with some variation between regions. All the studies found that a consideration of non CO_2 gases can significantly reduce the cost of meeting overall targets, and this approach has become the normal method in the years since.

Bergman (1991) and Rutherford (1992) were among the first studies to attach permits to fossil fuel combustion emissions and force an endogenous permit price to emerge by exogenously restricting the supply of permits. Bergman (1991) reports that if pollutants are concentrated in a few sectors of the economy, the remaining sectors may actually benefit from pollution controls, as factors of production are released from the constricting sectors, bringing their price down. In contrast, Hazilla and Kopp (1990) note that introducing environmental regulations to only a few industries causes prices to rise, and production to fall, in every sector of the economy, as the regulated sectors are used as intermediate inputs in other industries.

As stated in the introduction to this chapter, a strength of CGE models is that they can simulate multiple policies simultaneously and be used to explore how these different policies interact (and possibly conflict) with each other. Morris (2009) uses a CGE model of the U.S. to examine the effects of a cap-and-trade scheme and a 'Renewable Portfolio Standard' (RPS), which mandates that a minimum percentage of electricity come from renewable sources. Each policy is first modelled in isolation, and then both at the same time to see how each affects the other. Interestingly, the results suggest that in the presence of a cap-and-trade scheme to achieve a given emissions reduction, adding the RPS causes an additional welfare loss with no extra GHG mitigation. By adding the RPS on top of the cap-and-trade policy, one is essentially mandating how a certain portion of the emissions reduction target is to be met (i.e. through carbon-free electricity) as op-posed to allowing all abatement to occur where the marginal cost is lowest. Of course, if switching to renewable electricity was the cheapest way of meeting the emissions target, the RPS would be non-binding and adding it into the policy mix would have no effect on either welfare or the carbon price.

Another issue of interest is how industry- or country-specific targets (as opposed to permit trading schemes) affect industries or countries with low benchmark emissions intensities. Blitzer et al. (1994), for example, find that in the sector-specific case, stringent reductions are infeasible in the services sector due to a lack of substitution possibilities—forcing them to exempt services from reductions in those scenarios. In a similar vein, Paltsev et al. (2004) find that the high level of energy efficiency in Japan means that there are few cheap abatement options available as further efficiency improvements are likely to be expensive. This translates into the highest direct abatement costs of all Annex I regions in terms of $/tCO$_2$ abated. This does not, however, translate into the highest welfare cost as the small size of the energy sector relative to total output means that energy cost increases do not have such a significant effect on the rest of the economy as they do in other Annex I countries, where the energy sector is larger. Hence in the current study there may be some industries with low emissions intensities which need an extremely high carbon tax in order to meet an industry-specific reduction target, though this high tax may not translate into large price increases due to the same low emissions intensity that caused it.

Two further studies (Bye and Nyborg 1999; Edwards and Hutton 2001) merit a mention for their research on optimal permit allocation mechanisms. More specifically, these studies examine permit auctions and 'grandfathering' (i.e., distributed for free on the basis of historical emissions). Both studies find that grandfathering permits acts as a significant barrier to entry to the industries in the permit scheme, as well as provoking windfall profits and a transfer of money from the public to the private sector. This is particularly true in Bye and Nyborg (1999) where the permit scheme replaces existing energy taxes but must be revenue neutral, so payroll taxes must increase to offset the lost tax revenues. The paper's principal contribution is the observation that in the design of policies for environmental taxation (and/or permit schemes), there are two kinds of efficiency that need to be borne in mind. One may be termed 'environmental efficiency' and consists in ensuring that pollution abatement happens where the cost of such abatement is lowest. The other ('tax efficiency' perhaps) concerns the effects of the tax on the general economy. The suggestion is that certain fuels are taxed more heavily than others due to low elasticities of demand. Reducing the tax rates on such fuels thus causes a significant loss in revenue which, ceteris paribus, must be raised by tax increases elsewhere. Of course, the premise that taxes on more inelastic goods are less distorting is moot, and will be discussed further in the analysis of the results presented here—specifically in relation to the effects of emissions policy on globally competitive Spanish export sectors, and the extent to which they should be protected from policy-induced price rises. Finally, Edwards and Hutton (2001) report that when permits are auctioned, and the revenues are recycled as an output subsidy, there may be a 'double dividend', i.e. emissions reductions may be achieved in conjunction with some other policy goal, usually economic growth or increased employment. It is to such possibilities for revenue recycling that we now turn.

6.7 Trade and Carbon Leakage

Devarajan (1989) notes how energy-economy models were used, amongst other things, to look at the phenomenon of oil price rises for exporting countries, including the so-called 'Dutch Disease' problem whereby rising revenues from a natural resource export causes a real appreciation of the currency which is damaging for other export-oriented, or import-competing, industries. Benjamin et al. (1989) construct a CGE model which suggests that this is in fact the case for the export sectors, but that the degree to which import-competing sectors suffer de-pends on the degree of substitutability between the domestically produced goods and imports—a parameter which often carries a degree of uncertainty in economic models.

Burniaux et al. (1992) uses the OECD-GREEN model described above to examine how distortions in global energy markets affect policies to reduce CO_2 emissions. These distortions generally take the form of taxes in OECD countries, and subsidies in non-OECD countries, and this has a significant bearing on the results. They find that eliminating all energy market distortions (i.e., subsidies and taxes) globally is sufficient to reduce CO_2 emissions by 18% on the baseline in 2050, and the falling world oil price resulting from reduced demand means even the non-OECD countries (with the exception of energy exporters) witness a welfare improvement from such a liberalisation scenario. This paper highlights the importance of 'joined up thinking' in energy policies, and outlines the potential for the removal of existing energy subsidies to make a significant—if not entirely sufficient—difference to GHG emissions. Indeed, taking a medium-term scenario to 2030, Maisonnave et al. (2012)

explore the impact of (unilateral) EU climate policy on the import cost of oil prices to the EU, as well as the effect that steep in-creases in the oil price have on the costs of EU climate policy. They find that climate policy reduces the cost of the oil price by approximately a third or, alternatively, that a high oil price could reduce the cost of climate policy dramatically—by more than two thirds.

Gerlagh et al. (2002) and Blitzer et al. (1994) both find that when emissions restrictions are applied unilaterally in a single country model, the comparative advantage of the country in question shifts towards less polluting products, and more emissions intensive products are increasingly imported from abroad—otherwise known as 'carbon leakage'. The picture is the most stark in Blitzer et al. (1994), with results suggesting that while oil would still be mined in Egypt in the presence of emissions restrictions, it would be exported to be refined, with the petroleum products then reimported.

Babiker et al. (1997) investigate two options for addressing carbon leakage when emissions restrictions are only applied to OECD countries: Border Tax Adjustments (BTAs) depending on the carbon content of imports, or restricting exports from countries not limiting their emissions. The first seems the most logical approach, and indeed it reduces carbon leakage to zero, and reduces the necessary permit price by around 10%. In welfare terms the losses to the OECD countries from the carbon tax are mitigated, but the result is that the non-OECD countries suffer a welfare loss. Alternatively, non-OECD countries fare better under the export restriction scenarios, although this does not reduce the permit price, or carbon leakage rates by as much. This study reinforces the importance of the carbon leakage issue, as well as the need (and opportunity) to set emissions policy simulations in the context of other policies relevant to the period being studied—trade or agricultural policies for example.

Bosello et al. (2013) also study two options for mitigating carbon leakage, this time from the EU: Border Tax Adjustments (BTAs) on imports to tax them according to carbon content, and the assumption that non-EU countries will also face emissions restrictions. BTAs reduce GDP as the improved competitive-ness of domestic production is balanced by increased costs for firms which import intermediate inputs—dependent on the degree to which imports are substitutes or complements to domestic production. Similarly, the imposition of emissions reduction policies in non-EU regions does not have an unambiguously positive effect in the EU, as the substitution effects towards EU exports is balanced by an income effect as global GDP growth is slowed, reducing trade volumes overall.

Part II Spanish Case Study: Spanish Agricultural Emissions

This case study uses a single country neoclassical CGE model to analyse the effects of agreed emissions reductions on the agricultural sector over the period 2007–2020. The model employs Spanish input-output data for the year 2007. With a starting point in 2007, the study is carefully baselined to 2020 employing a mix of historical observations on the components of aggregate demand and population, and projections data for growth and population in Spain. Where possible, both technological change and taste shifters have been employed to capture as reasonably as possible the trends in the Spanish economy up to the latest available period. To understand the different emissions intensities within different agricultural activities, a detailed agricultural sector split of the parent activity of ‘agriculture, forestry and fishing’ in the national accounts, is required. In addition, to improve the validity of the research, both agricultural factor market and product market (i.e., the Common Agricultural policy) rigidities are modelled explicitly for Spain. Associated emissions data for the Spanish economy is taken from the UNFCCC, which disaggregates emissions of six GHGs (CO_2, CH_4, N_2O, HFCs, PFCs, HS6) into the following categories. In the model these emissions are mapped to the classification of sectors in the model, whilst drivers for each emitting activity (i.e., combustion and non-combustion) are assigned. European Union environmental policy is characterised explicitly through the modelling of the ETS scheme, where an exogenous (projected) permit price is assumed, whilst diffuse sectors (i.e., non ETS sectors) classified as transport, waste and buildings and agriculture, face emissions reductions subject to a carbon tax. All emissions target reductions are set as lower limits, such that a non-binding emissions target results in a zero permit price/carbon tax. A key innovation in this study is the implementation of available ‘end-of-pipe’ reductions discussed in Sect. 6.5 in part 1 of this chapter, through investment in abatement technologies such as precision farming or anaerobic digestion. The MAC curves are discussed further in Sect. 6.9.

6.8 Agricultural Emissions in Spain

In 2007, Spanish agriculture was responsible for 53 million metric tonnes (Mmt) of Carbon Dioxide Equivalent (CO_{2e})—around 12% of Spain’s total of 444 Mmt. Food production adds another 3.75 Mmt—less than 1% of the Spanish total. Agricultural emissions are dominated by methane (CH_4) and nitrous oxide (N_2O). Indeed, when Spanish emissions of non-CO_2 GHG emissions only are considered, the proportion corresponding to agriculture rises dramatically to 59%.

The breakdown of agricultural emissions can be seen in Fig. 6.1. Cattle (including dairy cattle) and sheep contribute over a third of the agricultural total, while the combined livestock emissions are over half the total. Among the crops sectors, emissions from cereals production are significant, but olive growing is the single industry with the largest emissions, with over 10% of the agricultural total.

Another measure of how polluting an industry is the ‘emissions intensity’—the quantity of GHGs emitted per euro of industry output. These figures are presented in Table 6.1, which shows fruit and vegetable growing to be the least emissions intensive agricultural activities, emitting 0.59 and 0.14 $kgCO_{2e}$/€ respectively, compared to 1.72 for cereals, and 3.78 for olives. It should be noted that the fruit aggregate masks some significant differences, as it includes grapes (1.88 $kgCO_{2e}$/€) and citrus (0.27 $kgCO_{2e}$/€). The table suggests cattle and sheep farming are more emissions intensive than pig and poultry farming, but less so than olive growing. These emissions intensities become relevant when examining the results of the scenarios. For example, while fruit and vegetable growers may find it more difficult to reduce the relatively small amount of (predominantly combustion) emissions they do produce,

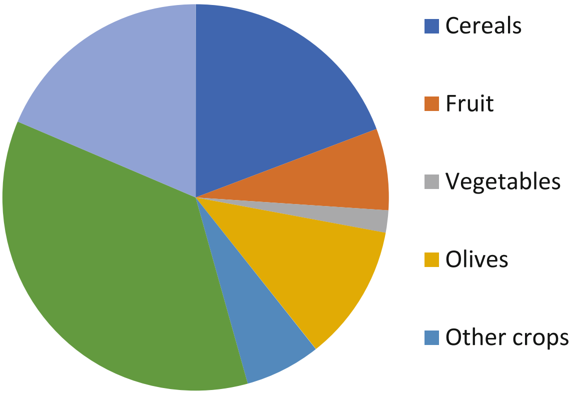


Fig. 6.1 Breakdown of agricultural emissions in 2007

Table 6.1 Emissions intensities of various agricultural activities in 2007

Industry	Emissions ($MmtCO_{2e}$)	Size (€ millions)	$kgCO_{2e}$/€
Cereals	10.24	5966	1.72
Fruit	4.62	6139	0.59
Vegetables	0.99	7039	0.14
Olives	6.07	1606	3.78
Cattle and sheep	19.03	7824	2.43
Pigs, poultry and other animals	9.89	8729	1.13
Agriculture	53.22	42,644	1.25
Spanish industrial total	358.53	2,071,404	0.17

by the same token, the increase in total costs from any tax on emissions will impact less in this sector (in proportional terms) than in an industry with a high emissions intensity (i.e., olives). This brings us to the next section, which discusses the sources of agricultural activity emissions and the degree to which they can be abated.

6.9 Emissions Factors and Marginal Abatement Cost (MAC) Curves

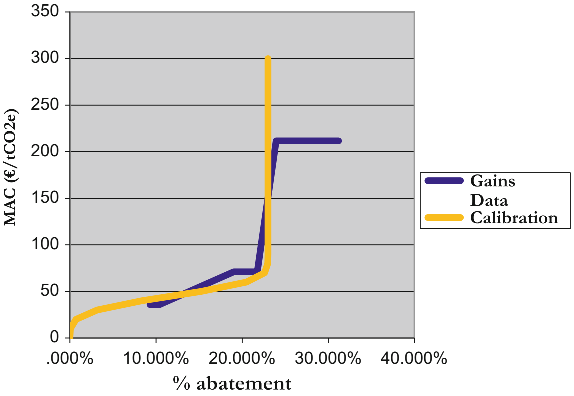
As well as the quantity of emissions associated with each agricultural industry, it is useful to be aware of where those emissions come from, as this has implications for their abatement possibilities. Emissions which come from petrol combustion, for example, are difficult to mitigate as petrol is the only non-electric source of energy used in significant quantities by farmers, so substitution possibilities are limited. The proportion of combustion emissions is very small in the livestock sectors—around 0–6% (not shown). In the crops sectors emissions factors are considerably higher.

Olives have the lowest proportion, at around 13%, whilst for the cereals and fruit and vegetables sector, about one-third of emissions come from fuel combustion, and in the remainder of the crops sectors the average is almost one-half. These emissions cannot be reduced by 'end-of-pipe' abatement measures.

Further evidence suggests that N_2O from manure is impossible to abate. If these N_2O emissions are added to those from fuel combustion, it brings the proportion of livestock sector emissions which are impossible to abate up to around 21%, much closer to the average for crops. For the remainder, the ease of abatement is governed by the MAC curves (Figs. 6.2 and 6.3) which show the ease of the uptake of abatement technologies (governed by the slope) at different carbon prices.

The first thing to notice from these graphs is how much cheaper abatement is in livestock than crops at any point up to the technically feasible maximum (around 25%). Considering the emissions reduction target of 10%, this means end-of-pipe abatement is likely to be heavily concentrated in livestock sectors. Thus, Fig. 6.3 reveals that 20% of livestock methane emissions could be abated for less than €10/tCO_{2e}. This translates to 4.6 MmtCO_{2e}, or 8.6% of total agricultural emissions in the benchmark. If this were the case, the crops sectors would have to contribute relatively little abatement in a scenario where the 10% reduction is an aggregate target applied to the agricultural total. If each agricultural industry must individually meet the 10% target, it implies that the target is likely to be easily met in the livestock sectors, meaning some relatively low-cost abatement opportunities may not be taken up, whilst the crops sectors are forced to engage in relatively expensive abatement options. The expectation is that this will increase the overall cost of an industry-specific target relative to a single one for the agricultural sector.

Fig. 6.2 Calibrated MAC curves for N_2O emissions from fertiliser use



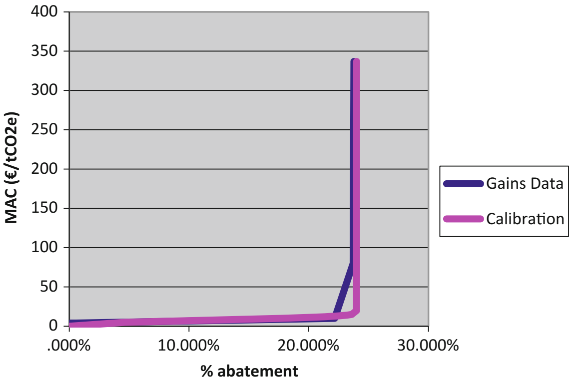


Fig. 6.3 Calibrated MAC curves for CH_4 emissions from livestock

6.10 Scenarios

The baseline, or status quo reference scenario, contains neither restrictions on Greenhouse Gas (GHG) emissions nor any kind of emissions tax. Whilst this is clearly unrealistic, the purpose is to give a counterfactual in order to isolate the effects of environmental policy in the results from all following scenarios. The policy shocks which are employed to characterise the CAP remain unchanged in the baseline and all scenarios, in order to fully isolate the effects of emissions restrictions in agriculture.

The key features of each scenario are shown in Table 6.2. Scenario 1 does not include the calibrated MAC curves for end-of-pipe abatement of agricultural emissions, in order that the effect of these can be isolated in scenario 2. All other features are constant across these two scenarios, with a 10% reduction in aggregate agricultural emissions, and the emergence of a single agricultural emissions price. This could be likened to an emissions trading scheme applied to agricultural emissions in isolation from any other emissions targets or permit trading schemes. Alternatively, it could be seen as a hypothetical exercise in finding the 'optimal' distribution of reductions across agricultural industries, with and without end-of-pipe abatement. Those industries with a cost of abatement higher than the agricultural average will reduce emissions by less than 10%, with the slack taken up by industries with cheaper abatement options. Scenario 3 precludes this possibility by requiring each one of ten agricultural subgroups (Table 6.3) to meet the 10% target. As a result, ten different agricultural emissions prices emerge, although in some cases the 10% reduction may be non-binding, resulting in an emissions price of zero.

Table 6.2 Scenario descriptions

Scenario	ETS emissions	Non-agric diffuse emissions	Agricultural emissions	End-of-pipe abatement in agriculture?
Baseline	Zero ETS price	Unrestricted	Unrestricted	No
Scenario 1	Exogenous non-zero ETS price	Reduced by 10% for each industry	Aggregate emissions reduced by 10%—single carbon price	No
Scenario 2	Exogenous non-zero ETS price	Reduced by 10% for each industry	Aggregate emissions reduced by 10%—single carbon price	Yes
Scenario 3	Exogenous non-zero ETS price	Reduced by 10% for each industry	Emissions of each specific agric industry reduced by 10%—multiple carbon prices	Yes

Table 6.3 Emissions factors 2007–2020 (%)

Industry	Scenario 2 relative to baseline/scenario 1	Scenario 3 relative to baseline/scenario 1
Cereals	−2.6	−5.4
Fruit	−2.6	−21.9
Vegetables	−2.6	−22
Olives	−2.6	−19.5
Other crops	−2.6	−15.8
Cattle and sheep	−21.7	0.0
Raw milk	−23.5	−28.1
Pigs	−21.6	−11.4
Poultry	−22.1	−7.1

6.11 The Distribution of Emissions Reductions

6.11.1 Scenario 1: 10% Reduction in Aggregate Agricultural Emissions, no End of Pipe Abatement

Having discussed the baseline results above, the first thing to notice is that in scenario 1, emissions reductions are concentrated in the cereals and cattle and sheep sectors, with other crops being the only other industry to contribute more than the 10% average across agriculture (Fig. 6.4). A general pattern in moving from the baseline to scenario 1, however, is that the change in emissions between the two scenarios tends to be greater in the crops than in the livestock sectors, with overall fertiliser emissions

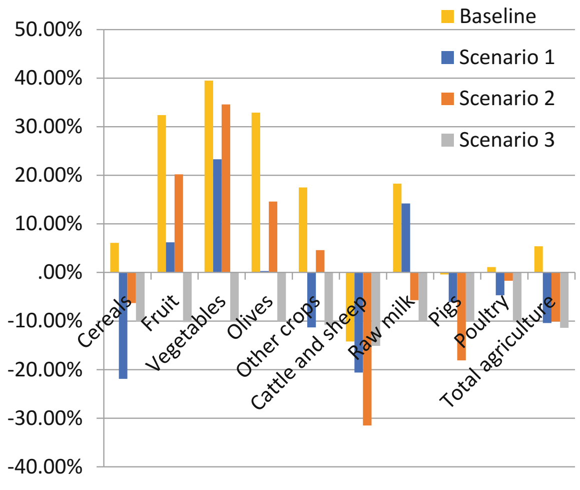


Fig. 6.4 Cumulative changes in emissions 2007–2020, baseline and scenarios 1–3

from the crops sectors 23.5% lower than the baseline in scenario 1 (not shown), and enteric fermentation and manure management emissions from livestock just 5.6% lower. This is because in the absence of end-of-pipe abatement options, the only two ways for emissions to fall are by substituting away toward less polluting inputs and/or a contraction in output. In the model, non-CO_2 emissions from livestock activities are attached to output, so the substitution option is only available to the crops sectors, which have some flexibility to reduce their fertiliser use if they increase their use of other inputs such as land, labour or capital. This extra abatement option explains why the introduction of an emissions tax provokes a bigger emissions reduction in the crops than the live-stock sectors. Taken in isolation, the effect of this substitution would be to increase the pressure on primary factors. However, the substitution effect towards factor use in the crops sectors takes place in the context of agricultural (and other) industries contracting relative to the baseline, so the 'income' effect is to lower factor prices.

6.11.2 Scenario 2: 10% Reduction in Aggregate Agricultural Emissions, with End of Pipe Abatement

The only difference between scenarios 1 and 2 is the inclusion of end-of-pipe abatement options from the calibrated MAC curves, and the effect is to concentrate emissions reductions in the livestock sectors, allowing the crops sectors to increase their emissions relative to scenario 1 such that the overall 10% reduction target for aggregate agricultural emissions is still met. At low levels of abatement, there are cheaper options available in livestock emissions (largely feed changes) than in the crops sectors. Thus, the relatively low emissions price necessary to meet the prescribed target provokes more abatement in the former than the latter. This can be seen in Table 6.3 which shows how emissions factors change in the different scenarios. The first column of Table 6.3 shows how significant the end-of-pipe abatement is in the livestock sectors in scenario 2, with emissions factors around 22% lower in 2020 than they are in the baseline/scenario 1. In contrast, those for the crops sectors fall much less, and are just 2.6% lower than the base-line/scenario 1 in 2020. This explains the result that the inclusion of end-of-pipe abatement places greater emissions reductions in the livestock sectors in the presence of a single 10% target for aggregate agricultural emissions.

6.11.3 Scenario 3: 10% Emissions Reduction in Each Agricultural Sector, with End of Pipe Abatement

The difference between scenarios 2 and 3 is that in the former, emissions reductions can vary between agricultural sectors as long as the overall 10% target is met by the agricultural sector. In scenario 3, however, each individual agricultural activity is forced to meet the 10% target itself. As can be seen in Fig. 6.4 this results in an overall reduction of slightly more than 10%, as for cattle and sheep emissions the target is non-binding, and emissions fall by 14%, whilst all other agricultural emissions fall by 10%. The movement from scenario 2 to 3 is thus beneficial for those industries which were overshooting the 10% target in scenario 2 (cattle and sheep, and pigs), whilst those industries with the highest emissions in scenario 2 (vegetables, fruit and olives) will find the enforced 10% target in scenario 3 the most stringent. To see this reflected in the results, attention now turns to the emissions taxes which emerge in each scenario.

6.12 Emissions Taxes

In the baseline emissions are unrestricted, so the endogenous emissions tax re-mains at zero. In scenarios 1 and 2, the single target for a reduction in aggregate agricultural emissions results in a uniform tax rate per tonne of CO_2 equivalent (€/tCO_{2e}) across all agricultural emissions. In both scenarios this tax rises as the period progresses and the emissions restriction tightens. By 2020 the necessary tax has reached €85/tCO_{2e} in scenario 1, but this is greatly reduced by the addition of end-of-pipe abatement, to €23/tCO_{2e}. It should be noted that this does not mean that meeting the target is 85/23 times cheaper for farmers in scenario 2, since in the modelling, agricultural activities must also meet the investment cost of in-vestment in abatement equipment, which is absent in scenario 1. Nevertheless, the presence of end-of-pipe abatement

Table 6.4 Emissions changes from scenario 2 and taxes from scenario 3

Industry	Scenario 2 cumulative emissions change (%)	Scenario 3 emissions tax in 2020 (€/tCO_{2e})
Cereals	−6.3	30.9
Fruit	20.2	91.2
Vegetables	34.6	259.3
Olives	14.6	63.6
Other crops	4.6	52.4
Cattle and sheep	−31.5	0.0
Raw milk	−5.7	11.1
Pigs	−18.1	7.8
Poultry	−1.7	412.3

options does mean that the emissions tax necessary to bring emissions down to the policy-mandated levels is much lower, as a given tax now provokes a much higher degree of abatement.

Scenario 3 is unique in that each subgroup of agricultural industries faces a specific emissions tax necessary to force each of them to reduce their emissions by 10%. In general it is to be expected that those industries with the highest emissions in scenario 2 will face the highest emissions taxes in scenario 3, as they are the ones for which abatement is most costly, given the baseline economic conditions and the MAC curve data. As shown in Table 6.4, vegetable growing has the largest emissions increase in scenario 2, and the second highest emissions tax in scenario 3, whilst the greatest emissions reduction in scenario 2 is in cattle and sheep, and this is the only industry to face a zero emissions price in scenario 3. In general, the livestock sectors tend to have lower emissions taxes in scenario 3, the exception being poultry farming. The total emissions from the poultry sector are small, but they also include a relatively high proportion of energy emissions, meaning the MAC curves for livestock are barely applicable. As has been noted above, energy emissions are hard to abate, and thus the high emissions tax necessary to force poultry emissions down 10%.

The total direct costs of each scenario to different agricultural groups are shown in Table 6.5. These are calculated as the sum of environmental taxes and abatement expenditure, accumulated over the 13 year simulation period. The results show that the introduction of end-of-pipe abatement dramatically reduces the cost to the agricultural sector as a whole from over €14 billion in scenario 1 (just over €1 billion/year) to just under €4 billion in scenario 2 (approximately €300 million/year)—a fall of around 70%. The activity-specific targets in scenario 3 raise the total cost back up to €6.2 billion, suggesting there are macroeconomic gains to be made from having a single uniform emissions price—a cap-and-trade scheme, as laid out in Weitzman (1974). Only the non-poultry livestock sectors benefit from the activity specific targets for the reasons discussed above. To fill out this emerging picture, the focus now turns to the effects each scenario has on agricultural prices and production.

Table 6.5 Total direct cost of each scenario

€ millions	Scenario 1	Scenario 2	Scenario 3
Cereals	2464	762	1060
Fruit	1057	311	956
Vegetables	323	91	608
Olives	1743	520	1706
Other crops	896	273	684
Cattle and sheep	3827	1021	0
Raw milk	1052	270	230
Pigs	2427	642	358
Poultry	153	42	594
Agriculture	14,064	3964	6246

6.13 Market Impacts

6.13.1 Scenario 1: 10% Reduction in Aggregate Agricultural Emissions, no End of Pipe Abatement

The broad picture from scenario 1 is that in the absence of end-of-pipe abatement measures the price effects of emissions restrictions are heaviest in the most emissions intensive sectors (olives, cereals, cattle and sheep) but production of those commodities with small trade volumes (barley, cattle and sheep) is relatively protected by the price inelasticity of demand. By contrast, those industries with much lower emissions intensities (vegetables, fruit (excluding grapes) and poultry) see relatively little impact from the emissions taxes, with price increases of around 2–3% relative to the baseline, and output falls of similar magnitude.

6.13.2 Scenario 2: 10% Reduction in Aggregate Agricultural Emissions, with End of Pipe Abatement

Introducing end-of-pipe abatement options in scenario 2 reduces the price in-crease from the emissions restriction in every agricultural industry compared to scenario 1 (Fig. 6.5). This is intuitive as emissions taxes are lower in scenario 2, and the value of the tax saving is instead invested in abatement equipment.

Thus while the immediate costs do not change between the two scenarios, in the first scenario they are lost completely to farmers as they go entirely to government, whilst in the second scenario a portion is converted into capital, and thus remains on the farm, lowering the emissions factor of future production, and hence the rate of future emissions taxes. The production results follow from those for prices, with

Fig. 6.5 Price changes relative to the baseline (%)

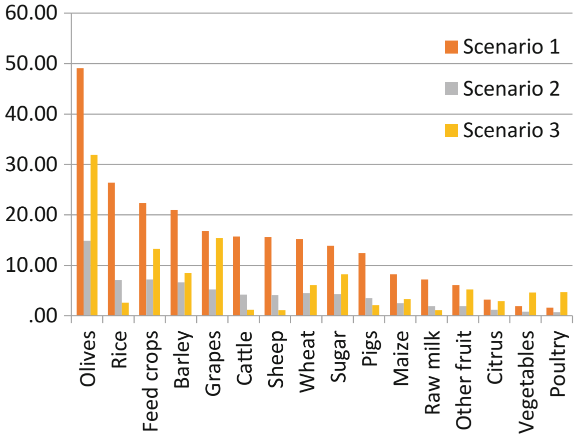
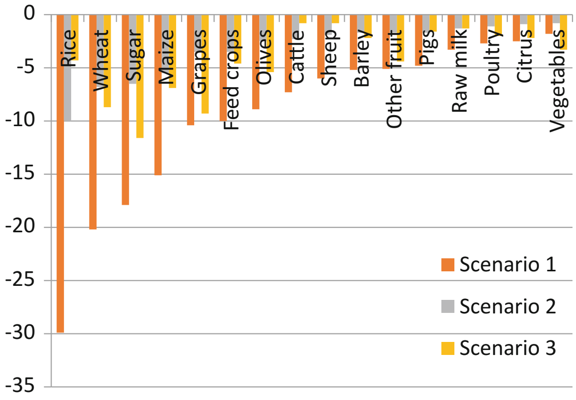


Fig. 6.6 Output changes relative to the baseline (%)



the falls in production in all sectors smaller than they were in scenario 1 (Fig. 6.6). On aggregate, the change in scenarios is not enough to reverse the pattern seen previously that composite crop production falls by more (5.7% in scenario 1) than that for livestock (4.7%). In scenario 2 these reductions in output have become 1.9% and 1.6% respectively.

6.13.3 Scenario 3: 10% Emissions Reduction in Each Agricultural Sector, with End of Pipe Abatement

Scenario 3 changes the picture quite significantly compared to that presented in the other two scenarios. The first thing to notice is that for the livestock sectors the effect of this scenario is a very small increase in prices relative to the baseline (Fig. 6.5). For two of these industries (cattle and sheep) this is because their 10% reduction target is non-binding (as noted above), meaning an emissions tax never emerges for these activities. They are thus able to take advantage of the falling cost of inputs resulting from other agricultural industries' shrinking production. This is true also of dairy cattle and poultry, the difference being that in these sectors the emissions target is binding.

In Spanish vegetable production, this activity exhibits a relatively low emissions intensity, although a high proportion of emissions comes from energy use, and thus is unable to benefit from end-of-pipe abatement. As a result a high emissions tax to force it to meet its target, and this does have an impact on price and output in this industry. The same is true of the fruit sectors which (with the exception of grapes) under scenarios 1 and 2 witnessed the smallest price and output effects. This has implications for Spanish policy-makers as fruit and vegetables are important export sectors—between them fruit and vegetables account for 30% of Spanish agrifood exports.

In the cereals sectors by 2020 the emissions tax generated by the cereals target in scenario 3 is higher than the uniform agricultural emissions tax in scenario 2. One consequence of this is that the cereals sectors undertake more end-of-pipe abatement than they did in scenario 2. As a result, the emissions factor attached to fertiliser use in these sectors falls even more in scenario 3, and the effect of emissions taxes on industry prices and output become less, as the emissions intensity of the industry falls. Thus by 2020, despite a cereals emissions tax in scenario 3 of €30/tCO_{2e}—higher than the agricultural emissions tax of €23/tCO_{2e} in scenario 2—the overall price increases of all cereals are smaller in scenario 3 than they are in scenario 2, as are the reductions in output. Emissions from cereals fall by more in scenario 3 than 2 as well, which suggests that over an extended time period, in this particular case, deeper emissions cuts are not necessarily more costly. Particularly if they are implemented at an early stage they may provoke abatement investment which, by reducing emissions factors, reduces the extent to which producers are penalised by emissions restrictions in later periods.

The overall effect of scenario 3 is to significantly reduce the burden of abatement in the livestock sectors, and share it evenly among all agricultural activities. Of course this means that the stringency of the policy is felt more keenly in those activities with either strong baseline growth or high costs of abatement.

Given the agricultural focus of this study, the focus here is on food prices, after noting from Table 6.6 that in scenario 1 the overall Consumer Price Index (CPI) rises 2% relative to the baseline, and this increase is 1.5% in scenario 2 and 1.6% in scenario 3. The same story is magnified in the aggregate food price index (Table 6.7), which (in comparison with the baseline) rises 6.1% in scenario 1, 2% in scenario 2 and 3.2% in scenario 3. The fact that food prices rise by more than the general price index, even when agricultural emissions benefit exclusively from end-of-pipe abatement options, is indicative of the high emissions intensities of most agricultural activities relative to the Spanish average. Looking at Table 6.7, in scenario 1 the biggest price increases are in the most emissions intensive sectors, namely olives and processed red meat (derived from cattle and sheep, which are both emissions intensive), whilst vegetables have a much smaller price increase. As noted above, the livestock sectors

Table 6.6 Macroeconomic results

Cumulative results in 2020	Baseline	Scenario 1	Scenario 2	Scenario 3
	% Change 2007–2020	% Relative to baseline		
Real GDP	1.8	−1.2	−0.9	−1.0
Real private consumption	−3.0	−0.8	−0.7	−0.7
Real investment	−39.8	−2.8	−2.5	−2.4
Real government spending	5.8	0.2	0.1	0.2
Real exports	64.3	−1.1	−0.7	−0.9
Real imports	−0.3	−0.5	−0.7	−0.6
Consumer price index	−0.9	2.0	1.5	1.6

Table 6.7 Household food prices relative to the baseline in 2020 (%)

	Scenario 1	Scenario 2	Scenario 3
Olives	28.0	8.9	18.7
Lamb	10.0	3.1	1.4
Beef	5.9	1.9	0.9
Poultry	4.3	1.7	2.2
Potatoes	4.2	1.8	2.8
Pork	4.0	1.5	1.8
Alcohol	4.0	1.7	2.9
Other fruit	3.9	1.5	3.5
Dairy	2.5	1.0	1.1
Other food	2.5	1.2	1.9
Citrus	2.3	1.1	2.2
Other crops	2.3	1.2	1.7
Vegetables	1.6	0.9	3.1
Sugar	0.7	0.6	0.7
Food index	6.1	2.0	3.2

benefit most from the addition of end-of-pipe abatement technologies, so a relative fall in the lamb and beef price when moving from scenario 1 to 2 is observed. Olives undergo a dramatic reduction in price be-tween the two scenarios, though they maintain the greatest price increase of all food commodities—indeed the general ranking of price increases is largely unchanged. This is not the case in scenario 3 where, although olives still show the greatest price increase by some distance, that for the red meat sectors in particular is greatly reduced (because cattle and sheep face no emissions tax in this scenario), whilst low emissions intensive products like fruit and vegetables now show the greatest price increases after olives. This is because of the high emissions taxes needed to force these expanding sectors to reduce their emissions by 10% in scenario 3.

Table 6.8 Household food demands relative to the baseline in 2020 (%)

	Scenario 1	Scenario 2	Scenario 3
Olives	−21.9	−8.1	−15.7
Lamb	−8.8	−2.9	−1.3
Beef	−4.5	−1.5	−0.7
Potatoes	−4.1	−1.7	−2.6
Poultry	−3.9	−1.6	−2
Pork	−3.6	−1.4	−1.6
Othfruit	−2.9	−1.1	−2.6
Ocrops	−2.6	−1.2	−1.7
Dairy	−1.8	−0.7	−0.8
Other food	−1.8	−0.9	−1.3
Citrus	−1.6	−0.8	−1.5
Vegetables	−1.4	−0.8	−2.7
Alcohol	−0.7	−0.3	−0.5
Sugar	−0.1	−0.1	−0.1
Food index	−4.1	−1.5	−2.1

The responses of household consumption to these price increases are shown in Table 6.8, and offer few surprises, with the biggest reductions in demand in olives and red meat, and the smallest in sugar. Calculating the ratio of percentage changes in household consumption by percentage changes in price—both relative to the baseline—gives an estimate of the 'general equilibrium' elasticities (Table 6.9). The generally higher elasticities in scenario 2 compared to scenario 1 are to be expected as price increases are smaller in the former. Of even greater interest though is the fact that the two commodities with the lowest elasticities of demand are alcohol and sugar—both of which have certain addictive qualities and are generally considered to be price inelastic.

6.14 Conclusions

This review of the use of computable general equilibrium (CGE) models in environmental policy analysis highlights four important strengths. Firstly, CGE models are versatile in that their macroeconomic grounding is ideally tailored to the analysis of economy wide environmental policy analysis and the assessment of different environmental policy options in terms of economic efficiency (for example, with and without revenue recycling), real incomes or even other sustainable development goals. Secondly, by incorporating dynamic economic mechanisms (i.e., savings-investment, capital accumulation, labour market adjustments), the temporal dimension of the model is improved. From the perspective of environmental policy, this enhances the analysis to accommodate the gradual introduction or withdrawal of policies (e.g.,

Table 6.9 Estimated price elasticities of demand of food products

	Scenario 1	Scenario 2	Scenario 3
Olives	−0.78	−0.91	−0.84
Lamb	−0.88	−0.94	−0.93
Beef	−0.76	−0.79	−0.78
Poultry	−0.91	−0.94	−0.91
Potatoes	−0.98	−0.94	−0.93
Pork	−0.90	−0.93	−0.89
Alcohol	−0.18	−0.18	−0.17
Other fruit	−0.74	−0.73	−0.74
Dairy	−0.72	−0.70	−0.73
Other food	−0.72	−0.75	−0.68
Citrus	−0.70	−0.73	−0.68
Other crops	−1.13	−1.00	−1.00
Vegetables	−0.88	−0.89	−0.87
Sugar	−0.14	−0.17	−0.14

switch from grandfathering to auctioning of permits; CAP and trade effects) and the indirect cumulative period-by-period impacts said policies may have on investment decisions and economic growth. A third advantage is the ability of CGE to deal with multiple pollutants (CO_2, CH_4, N_2O etc.) and policies. Given the well documented potential for reducing radiative forcing through abatement of these gases, and in particular their dominance in total agricultural emissions, these are crucial for a full analysis of abatement potential in the agricultural sector. Finally, we have also seen that such models are able to incorporate (in admittedly a stylized way), induced technical change in relation to end-of-pipe abatement options in the agricultural sector. The inclusion of marginal abatement cost curves calibrated to bottom-up data on the costs and abatement potentials of various technologies is a significant advance in improving the realism of climate change mitigation analysis. In the context of the agricultural sector, it enables a full picture to emerge of how emissions reductions may be distributed among agricultural sectors based on the abatement options available to them. Omitting this abatement potential could lead to an overestimation of the cost of achieving the mandated reductions in greenhouse gases.

The second part of the chapter employs a single country CGE model of the Spanish economy which incorporates (in different degrees) each of these analytical advantages to examine the impacts on the agricultural sectors under the auspices of the EU-mandated GHG emissions reductions targets for 2020. The scenarios examine (inter alia) how the incorporation of marginal abatement cost (MAC) curves affect the costs of GHG reductions in the Spanish agriculture sector. The inclusion of MACs induces a modest reduction in the macroeconomic cost of the emissions restrictions to Spain in terms of real GDP (1.2 and 0.9% lower than the baseline in 2020 without and with MAC curves, respectively). Focusing on the agricultural sector, the addition of MAC curves tend to concentrate emissions reductions in the livestock sectors as the data suggests they have access to more low-cost abatement options when compared with crops sectors. The emissions tax necessary to meet the 10% reduction target for agriculture as a ‘diffuse’ sector falls from €85/tCO$_e$ without the MAC curves to €23/tCO$_{2e}$ with, and the projected total direct cost to farmers of the policy (emissions taxes plus the cost of abatement equipment) falls by around 70%.

Policy-induced price increases and output reductions are reduced fairly evenly across all agricultural sectors, as the single emissions target for aggregate agricultural emissions means reductions can still be focused where they are cheapest. Thus the fall in output relative to the baseline is around 20% greater in livestock than that in crops, and this is a consistent result with or without the MAC curves.

In addition, the model is used to analyse two policy options for ensuring the agricultural emissions reduction target is met. The first (scenario 2) sets a single target for aggregate agricultural emissions, with a uniform emissions tax rate, and allows reductions to be distributed depending on the relative costs of abatement—analogous to a cap-and trade scheme among agricultural industries, with all permits auctioned at the market price. The second (scenario 3) divides agriculture into 10 subsectors and forces each of them to reduce their emissions by 10%. The results suggest that in scenario 2, as noted above, emissions reductions are concentrated in the livestock sectors, which allows certain key Spanish export commodities such as fruit, vegetables and olives, a degree of slack to increase their production. In scenario 3 this is no longer the case, and they become the agricultural industries for whom meeting the 10% target is the most costly. Indeed, a consistent pattern is that those industries which reduce their emissions by more than the average (10%) in scenario 2 face a less than average emissions tax (€23/tCO$_{2e}$) and vice versa. At the most extreme, for cattle and sheep farming, which has the largest reduction in emissions of all agricultural sectors in scenario 2, the 10% reduction target in scenario 3 is non-binding, resulting in a zero emissions tax.

In general the costs of the emissions restrictions in terms of welfare, real GDP and, particularly, agricultural output, are smaller in scenario 2 than scenario 3, lending support to the idea that there are efficiency gains from using a cap-and-trade scheme to focus emissions reductions where they can be made at the lowest cost. An important caveat is that the model does not account for the administration costs of running such a scheme, though it is a point of contention as to whether these would be significantly greater than those associated with ensuring each agricultural activity meets a specific emissions reduction target. Such a cap-and-trade scheme appears to work in conjunction with the trend in Spanish agriculture of a moderate expansion in certain key crop sectors relative to livestock. These crop sectors—particularly fruit and vegetables—are among the least emissions intensive agricultural products, so their expansion is likely to help Spain to meet its GHG targets more easily—though it may raise other environmental concerns beyond the reach of this study.

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