eman ta zabal zazu

Universidad del País Vasco Euskal Herriko Unibertsitatea

Facultad de Ciencias Económicas y Empresariales
Departamento de Fundamentos del Análisis Económico I

Doctoral Thesis

Distributional analysis of climate change mitigation policies

---

Author:

Joaquín García González

Supervisors:

Dr. Mikel González Ruiz de Eguino

Dr. Anil Markandya

Bilbao 2016

Publications based on this PhD thesis

Peer-reviewed journal publications

García-Muros, X., Markandya, A., Romero-Jordán, D., González-Eguino, M., 2016 The distributional effects of carbon-based food taxes. *Journal of Cleaner Production*. doi:10.1016/j.jclepro.2016.05.17.

García-Muros, X., Burguillo, M., González-Eguino, M., Romero-Jordán, D., 2016. Local air pollution and global climate change taxes: a distributional analysis for the case of Spain. *Journal of Environmental Planning and Management* 0, 1–18.

Submitted journal publications

Böhringer, C., Garcia-Muros, X., Cazcarro, I. and Arto, I. 2016. The Efficiency Cost of Protective Measures in Climate Policy. A Computable General Equilibrium Analysis for the United States. *Submitted*, 2016

Conference presentations

Garcia-Muros, X., Markandya, A., Romero-Jordán, D. and González.-Eguino, M. 2015. The distributional effects of carbon-based food taxes. *41 Simposio de la Asociación Española de Economía-Spanish Economic Association (SAEe)*. Girona

Böhringer, C., Garcia-Muros, X., González.-Eguino, M. and Rey, L. 2015. US Climate Policy: A Critical Assessment of Intensity Standards. *The European Association of Environmental and Resource Economists (EAERE) 21st Annual Conference*. Helsinki.Garcia-Muros, X., Burguillo, M., Gonzalez-Eguino, M. and Romero-Jordán, D. 2013. Local air pollution and global climate change taxes: a distributional analysis. *Workshop on Economics of Energy Efficiency, Universidad Rovira i Virgili*. Reus

Working papers

Böhringer, C., Garcia-Muros, X., Cazcarro, I. and Arto, I. 2016. The Efficiency Cost of Protective Measures in Climate Policy. A Computable General Equilibrium Analysis for the United States. *Oldenburg Discussion Papers in Economics 392 (16).*

García-Muros, X., Burguillo, M., González-Eguino, M., Romero-Jordán, D., 2014. Local air pollution and global climate change taxes: a distributional analysis for the case of Spain. (WP No. 2014-01). *BC3 Basque Centre for Climate Change*.

ABSTRACT

Inequality has become particularly important on the environmental policy agenda and many studies have focused on the issue of who bears the cost of environmental and climate protection, as well as who is affected by a poor environment. The main objective of this thesis is to contribute to knowledge of distributional analysis through four case studies applied to different climate-related policies, where each of them tries to address different questions related with distributional implications. In our various case studies, we seek to analyse a range of significant measures which currently form part of the policy debate. Chapter 2 addresses the distributional implications of local air pollution tax policies and compares them with climate change taxes. Chapter 3 assesses the implications of levying taxes on the consumption of food products based on their carbon footprint. Chapter 4 focuses on the implications of climate policy in energy-intensive industries and compares the economic implications of four alternative protective measures for preventing carbon leakage. Finally, Chapter 5 examines the distributional implications of different schemes for financing the promotion of renewables in the electricity sector. Each of these case studies tries to address different questions related with who bears the cost of environmental and climate protection.

ACKNOWLEDGEMENTS

Esta tesis doctoral ha sido el resultado de cuatro años de trabajo y esfuerzo. Aunque solo aparece un nombre como autor de este trabajo, sería ingenuo pensar que se ha llevado a cabo de manera solitaria, sin ningún tipo de ayuda. Por eso quiero dedicar las siguientes líneas a agradecer a aquellos que han participado para que esto sea posible y que me han ayudado y apoyado cuando más lo he necesitado.

Para empezar quiero agradecer el apoyo y la confianza que han depositado en mí mis dos directores, Anil Markandya y Mikel González Ruiz de Eguino. Gracias Anil por tu apoyo y por la sabiduría que me has facilitado siempre que te lo he requerido. Mikel, has sido mucho más que un director para mí durante estos años, has sido un mentor y un espejo en el que mirarse. Sin tu ayuda y predisposición estoy convencido de que me hubiese sido imposible llegar hasta donde he llegado, por ello solo puedo dedicarte palabras de agradecimiento.

También quiero dedicar unas palabras de gratitud al Basque Centre for Climate Change (BC3) por financiar mi etapa como doctorando. Del mismo modo, a aquellos que forman parte de esta institución y han sido las personas con las que he convivido y disfrutado durante los últimos años. Gracias especialmente a Mavi Roman y a Guillermo Pardo, ya que nos hemos apoyado y soportado mutuamente.

Me gustaría asimismo agradecer el apoyo de aquellas personas con las que he colaborado durante el proceso de esta tesis: Christoph Böhringer, Desiderio Romero-Jordan, Mercedes Burguillo, Iñaki Arto, Luis Rey y Nacho Cazcarro. Desiderio gracias por el apoyo que me has prestado estos años y por el excelente trabajo que hemos podido hacer juntos a lo largo de esta tesis. Chris, my Oldenburg experience was very fulfilling, I learned and enjoyed a lot working with you and your team. Thank you also to be tough with me and try to bring out the best in me. Espero poder seguir colaborando y participando con todos en distintos proyectos.

Por último y no por ello menos importante, muchas gracias a mis familiares y amigos. Gracias a mis padres y mi hermano por el apoyo que me han dado siempre. Aunque en los últimos años hayamos estados alejados físicamente, siempre les he sentido muy cerca. Gracias también a mis amigos que siempre me han sabido sacar una sonrisa cuando lo he necesitado. Isa para ti solo puedo tener palabras de cariño y agradecimiento. GRACIAS por estar a mi lado y por tu apoyo infinito, no podría imaginarme estos años sin ti.

“The secret of happiness is low expectations.”

Barry Schwartz

Contents

List of Tables

List of Figures

Chapter 1

Introduction

1.1. Motivation and objectives

In recent years there has been increasing interest in inequality and distributional analysis (Piketty and Saez, 2014). Inequality has also become particularly important on the environmental policy agenda and many studies (OECD 2006) have focused on the issue of who bears the cost of environmental and climate protection, as well as who is affected by a poor environment. In this thesis distributional analyses seek to measure the potential effect of climate on individual groups within society. The main objective of this thesis is to contribute to knowledge of distributional analysis through four case studies applied to different climate-related policies.

Distributional analyses of climate mitigation policy are important for at least two reasons: (i) because equity is one of the main objectives in designing policies; and (ii) to examine the political feasibility of environmental measures. Looking first at the principle of equity, Article 3.1 of the United Nations Framework Convention on Climate Change (UNFCCC, 1992) states as follows: "The Parties should protect the climate system for the benefit of present and future generations of humankind, on the basis of equity and in accordance with their common but differentiated responsibilities and respective capabilities." This article refers to the greater responsibility of developed countries in climate mitigation, but the United Nations establish that environmental and mitigation policies should follow the principle of justice and fairness. Moreover from a capabilities approach, policies should be applied to each person according to the capacity of that person to shoulder the cost (Markandya 2011).

As regards political feasibility, climate change mitigation policies are widely perceived to be regressive, i.e. they affect low income groups more. For many countries, regressiveness is a relevant barrier to implementing climate change mitigation and environmental policies. Public acceptability is essential for effective mitigation policies to be adopted, and equity and fairness play an important role in how such measures are regarded by public opinion (Bristow et al. 2010). This has led several studies on public acceptability of climate policy that link regressive implications with lower acceptability (see Berrens et al. 2004, Li et al. 2004 or Wiser 2007). In fact, some researchers conclude that support for environmental taxes can be raised by taking into account distributional consequences, especially by protecting against regressive effects (Ščasný et al 2016).

The distributional impacts of climate protection have been extensively investigated and many studies find that they tend to be regressive[1]. This is observed in early studies such as Poterba (1991) and Pearson and Smith (1991). More recent papers for a panel of European countries (such as Ekins et al. 2011) also find major country-to-country differences. The differences between countries are due mainly to differences in types of policy, consumer patterns, income levels and energy and transport infrastructures. Most studies find regressivity in global climate-change-related measures, such as carbon taxation, but this conclusion cannot be taken as a general rule because it depends on the case study. In fact, there are papers that do not find regressivity: for example Labandeira and Labeaga (1999) for Spain, Sterner (2012) for a panel of European countries and Tiezzi (2005) for Italy.

There are several factors that play important roles in distributional analysis. For example, the degree of substitutability of the taxed goods is an essential factor in explaining welfare impacts. Thus, the regressivity of taxation is due to the possibility of substitution between taxed goods and non-taxed goods. Similarly, the distributional impacts of climate change mitigation measures, such as carbon taxation, also depend on the use of new revenues. As proposed in the literature on double dividends (see Goulder, 1995), the efficiency of the tax system could be improved if other distortionary taxes such as those on capital or labour are reduced. Alternatively, the revenues could be used to fund lump-sum transfers to compensate groups who have been left worse off. However, there could be a trade-off between efficiency and equity (distributional effects) depending on the revenue-recycling scheme selected. For example, in countries with inefficient labour markets a reduction in taxes on labour could reduce unemployment and thus have a positive efficiency impact, but the distributional implications may not be positive.

In short, the literature shows that distributional impacts vary from one case study to another and depend on policy design, so new environmental proposals should always consider analysing their distributional implications. Moreover, to date the leading distributional literature has focused mainly on climate policies and measures related to the energy and transport sectors, but recent studies suggest that indirect related measures can also deliver cost-effective emission reductions, e.g. through local air pollution control policies or the

[1] Climate change mitigation policies can to raise inequality both between countries and within a country. In this dissertation, we focus on the second inequality effect. Thus, we consider that climate protection policies would tend to be regressive if they affect low income households more.

promotion of healthier diets (see Bollen et al., 2009 for local air pollution or Stehfest et al. 2009 for healthier diets). Although we also turn our attention to carbon emissions and the energy sector, our study seeks to shed light on the distributional impacts of alternative measures that affect other sectors.

In our various case studies, we seek to analyse a range of significant measures which currently form part of the policy debate. Chapter 2 addresses the distributional implications of local air pollution tax policies and compares them with climate change taxes. Chapter 3 assesses the implications of levying taxes on the consumption of food products based on their carbon footprint. Chapter 4 focuses on the implications of climate policy in energy-intensive industries and compares the economic implications of four alternative protective measures for preventing carbon leakage. Finally, Chapter 5 examines the distributional implications of different schemes for financing the promotion of renewables in the electricity sector. Each of these case studies tries to address different questions related with who bears the cost of environmental and climate protection. However, they do not consider the consequences of the measures though the changes in environmental quality. Thus, the main limitation of these studies is that impacts are analysed purely from cost perspective without accounting for the monetary health benefits associated with climate change mitigation.

1.2. Methodological objectives

If distributional impacts and incidence analysis are to be investigated effectively different methodologies must be used. In this PhD thesis two methodologies are used and implemented: Microsimulation (MS) models and Computable General Equilibrium (CGE) models. MS models are an appropriate tool for assessing micro-economic effects and distributional impacts of policies and for showing the incidence of climate protection measures on consumers. On the other hand, CGE models provide an economy-wide analysis that it is not captured with microsimulation or other partial equilibrium models. The connection between these methodologies enables the benefits of CGE and MS models to be combined. These two types of model are used here in isolation and in combination, with “soft” and “hard” links between them. A major milestone of this research process has been to apply these methodologies concurrently. To that end it has also been necessary to use econometric techniques and software such as GAMS and Stata, and to develop the ability to work with big databases such as the Global Trade Analysis Project (GTAP) and national consumer surveys.

As pointed out, the degree of substitutability of the goods taxed is an essential factor in explaining welfare impacts. To study the distributional implication of environmental policies we need tools that capture substitution effects between goods. Multi-goods demand models capture the behaviour of households and provide a realistic picture of substitution, demand and income effects. To assess distributional effects, the micro-simulation model based on a multi-good demand system is thus a suitable tool for exploring incidence analysis.

At the same time, economic adjustment to emission regulation climate policy is driven by comprehensive substitution in production, output and income effects across multiple markets following changes in relative prices. In this context, Computable General Equilibrium (CGE) models are a standard tool for economy-wide numerical analysis of policy regulation. One of their main features is that they are able look at the economy as a whole. Therefore, a CGE approach based on empirical data enables the impacts of environmental policies to be assessed from the efficiency and macro-economic perspectives.

The link between CGE and MS models enables us to analyse macroeconomic policy simulations at the microeconomic level. There are various ways of linking macro-micro models that can be summed up basically as “soft” link approaches and “hard” link approaches. Soft link approaches rely on using CGE and MS models sequentially[2]. Under this approach the outputs from the macro model are used as an input in the micro model, making it possible to analyse the distributional impacts. The empirical analysis therefore involves two stages: in the first, the price changes produced by the environmental tax are studied through a macro model. In the second stage a microsimulation model is used to calculate the distributional effects of price changes. A hard link is a recursive approach, where an iterative process enables feedbacks to be introduced between the two methodologies[3].

This thesis is built around four projects, each with a different methodological objective and approach. Chapter 2 uses an Input-Output model in combination with a micro-simulation model to estimate the impact of local air pollution and global climate change taxation on the household. The micro tool used in Chapter 2 was developed by Sanz-Sanz et al. (2003), whereas we develop an input-output (IO) price model. In this chapter we use a soft approach to link the micro and macro models. Chapter 3 sets out a full micro-model from the estimation phase to the simulation phase. In more detail, we use the well-known Almost Ideal Demand System (AIDS) designed by Deaton and Muellbauer (1980) to explore household behaviour when faced changes in food prices caused by levying consumption taxes on food products based on their carbon footprint. Chapter 4 sets out a large-scale computable general

[2] Two types of soft link can be distinguished: the top-down approach and the bottom-up approach (Peichl, 2008). In this thesis only the top-down approach is explored.
[3] An algorithm for using the hard link approach is described in Böhringer and Rutherford (2006).

equilibrium model for the global economy. Finally, Chapter 5 combines the expertise attained in the previous chapters and sets out a computable general equilibrium model and a micro-simulation model for a single economy. The two methodologies are then combined through a hard link approach.

Finally, in our different studies we have tried to incorporate specific indices to compare the distributional impacts in the different policy scenarios. One of the main challenges of distributional analysis is to be able to compare different measures and show whether the different climate policies analysed are regressive, progressive or neutral based on specific criteria. In our studies we compute the Gini, Theil and Atkinson indices among others to analyse the inequality caused by climate measures. Moreover, to investigate regressivity we use the Reynolds-Smolensky and Kakwani coefficients.

1.3. Structure of the thesis.

The dissertation is structured in four different applications, where each of them tries to address different questions related with distributional implications. Next, a summary of the motivation, main objectives and contributions of each of the chapters composing the present thesis will be briefly addressed.

Chapter 2, "Local air pollution and global climate change taxes: a distributional analysis for the case of Spain" explores the distributional implications of local air pollution tax policies and compares them with climate change taxes. Local air pollution and global climate change are two significant, interrelated environmental problems. Some recent papers have dealt with these two problems in combination, but most of the relevant literature has focused solely on climate change or solely on the ancillary benefits of climate change mitigation in terms of air pollution. Similarly, from the distributional perspective, most of the literature has focused on the impacts of climate change-related taxes such as excise duties on CO2, energy or fuels. The air pollution tax scheme is based on the estimated damage associated with the main local air pollutants, while the climate change scheme is based on a CO2 tax. A combination of an Input-Output model with a micro-simulation model is used for the case of Spain. The distributional implications of a revenue-neutral tax reform are also explored. From a methodological perspective in this first chapter we have tried to introduce the use of the two methodologies and how to link both.

Chapter 3, "The distributional effects of carbon-based food taxes" evaluates the implications of levying consumption taxes on food products based on their carbon footprint. We estimate specific elasticities for the food demand system based on a dataset of around 20,000 households, using a demand system model. In this chapter we develop a microsimulation model to analyses food tax policies. We analyse the capacity of this policy to reduce emissions and, at the same time, help to change consumption patterns towards healthier diets. Finally, for the first time in the related literature, we also explore the distributional implications.

Chapter 4, “The Efficiency Cost of Protective Measures in Climate Policy” focuses on climate policy analysis for the United States of America (US) and compares the economic implications of four alternative protective measures for Energy-intensive and trade-exposed (ETIE) US industries: (i) output-based rebates, (ii) exemptions from emission pricing, (iii) energy intensity standards, and (iv) carbon intensity standards. Despite some recent achievements towards a global climate agreement, climate action to reduce greenhouse gas emissions remains quite heterogeneous across countries. EITE industries in industrialized countries are particularly concerned on stringent domestic emission pricing that may put them at a competitive disadvantage with respect to producers of similar goods in other countries without or only quite lenient emission regulation. We quantify how these protective measures can reduce the possible competitiveness cost in the EITE industries and which can be the impact in the rest of the economy. Hence, chapter 4 explores how the mitigation costs are distributed across sectors. Moreover, for the first time in this dissertation, we develop a large-scale computable general equilibrium model for the global economy.

Chapter 5, “Economic and distributional implications of alternative mechanism to finance renewables”, applies a computable general equilibrium (CGE) model in combination with a microsimulation (MS) model to examine the distributional implications of different schemes to finance the promotion of renewables in the Spanish electricity sector. These schemes include exemptions from the RES-E surcharge in the price of electricity for residential or industry consumers and also different financing alternatives where the cost to renewables is not financed through the electricity bill but by other tax sources such as fuel tax, VAT or via transfers. Our integrated modelling approach includes a rich representation of household’s heterogeneity and the inter-sectoral and price-related effects, which are fundamental to analyse the implications of these schemes that are not restricted to the electricity sector.

Finally, Chapter 6 summarizes the main conclusions derived from this thesis, along with some suggestions for possible further research on the subject.

Chapter 2

Local air pollution and global climate change taxes: a distributional analysis for the case of Spain

2.1 Introduction

The Paris Agreement at the 21st Conference of Parties (COP21) introduces voluntary pledges of individual countries – so-called intended nationally determined contributions (INDCs) to reduce GHG emissions. However, the costs and their distribution will make implementation difficult. In this context, local air pollution measures may play an important role in the political agenda since their effects (mainly on health) are felt more immediately by citizens. Indeed, global climate change (GCC) and local air pollution (LAP) are two significant, interrelated causes of environmental concern, whose potential synergies could improve policy design (Swart et al., 2004). Most relevant literature to date has dealt with these two problems separately or has focused mainly on the ancillary benefits of GCC mitigation (see for example OECD 2001 or Barker and Rosendahl, 2000). However, some authors (Xu and Masui, 2009) have recently explored the ancillary benefits of LAP mitigation, given the fact that the health effects of pollution are of more immediate concern to developing countries[4][1]. Finally, Bollen et al. (2009) assess the effects in a cost-benefit analysis framework and find that "LAP control combined with GCC policy creates an extra early-kick-off for the transition towards climate friendly energy supply".

As mentioned, for many countries, distributional implications are one of the difficulties of implementing GCC policies. For example, studies of the distributional impacts of energy and carbon taxes on households reveal that they tend to be regressive, i.e. they affect low income households more. This is observed in early studies such as Poterba (1991) and Pearson and Smith (1991). Poterba (1991) finds regressivity in motor fuel taxes, though only so a low degree when the results are expressed as a proportion of expenditure[5]. Pearson and Smith

[4] According to WHO estimates, LAP is also one of the leading causes of death in developing countries (WHO, 2009).

[5] The differences between the results for annual income and other proxies of lifetime income are due to the fact that many households initially included in the lowest income group do not remain poor permanently (e.g. students). Other papers show that annual income overestimates distributional effects.

(1991) also show that a carbon tax in Europe would be regressive, but that there would be differences from one country to another. More recent papers for a panel of European countries (such as Ekins et al. 2011, and Barker and Köhler 1998), also find major country-to-country differences. These studies find that GCC tax regressivity is caused by home energy use (lighting and heating), but the results become ambiguous when the analysis focuses on motor fuel taxes. Differences between countries are due mainly to differences in the type of tax, consumer patterns, income level and energy and transport infrastructures[6].

Most studies find regressivity in GCC related taxes, but it cannot be concluded that this is a rule, since it depends on the case study; some papers do not find regressivity- for example Labandeira and Labeaga (1999) for Spain; Symons et al. (2002) for Italy and U.K.; Sterner (2012) for a panel of European countries and Tiezzi (2005) for Italy.

The degree of substitutability of the goods taxed is an essential factor in explaining the distribution of the costs of the two taxation schemes analysed. For example, whether or not there is a good public transport network is a basic factor in explaining household motor fuel expenditure. In countries or regions with poor public transport, a tax on motor fuel is more regressive because the lowest income groups in those regions or countries use private transport more than their peers in regions with good public transport infrastructures. Thus, tax regressivity is related to the possibility of substitution between public and private transport. In that regard, the relevant literature also shows that tax impacts are higher in rural areas than in urban ones (see e.g. Labandeira et al. (2004), Wier et al. (2005), Romero et. al (2014)), because urban households have fairly easy access to public transport.

The distributional impacts of these taxes also depend on the use of new revenues. As proposed in the literature on double dividends (see Goulder, 1995), the efficiency of the tax system could be improved if other distortionary taxes –such as those on capital or labour– were reduced. However, other studies, such as Bovenberg and Mooij (1994) show that environmental taxes could increase pre-existing distortions in the tax system. Schöb (1997) shows that this contradiction is caused by the definition of the second-best optimal tax considered. In distributional terms, the revenues could also be used to fund lump-sum

See for example Feng et al., (2010); Metcalf, (1999); Sterner, (2012); Wier et al., (2005). Only Rausch et al. (2011) fail to find evidence that annual income overestimates distributional impacts. Most of these studies look at snapshots of taxes in one year relative to a proxy for lifetime income, which is often current consumption.

[6] Other studies find regressive effects in some countries (e.g. Metcalf et al., 2010, for the US, Wier et al., 2005, for Denmark; Feng et al., 2010, for the UK; Kerkhof et al., 2008, for The Netherlands and Brännlund and Nordström, 2004, for Sweden) because the tax is levied on goods which are consumed in greater proportions by low income households, especially consumption linked to home energy use.

transfers to compensate groups who have been left worse off. Rausch et al. (2011) show (using a CGE model for the US) that lump-sum transfers to households are more progressive than lowering income tax, which proves to be highly regressive[7]. Hence, there is a trade-off between efficiency and equity (distributional effects) depending on the revenue-recycling scheme (Bovenberg 1999). The revenue can only be used to compensate poorer people or to reduce pre-existing distortions. Along these lines, Aigner (2014) shows that the higher the redistribution, the higher the distortions of the tax system. Barker and Khöler (1998) also show that a reduction in taxes on labour is regressive, but recycling via lump-sum transfers is progressive.

To date the relevant literature has concentrated on the distributional implications of GCC policies, but there are a few papers reporting on research into the distributional effects of LAP policies. For example, Parry (2004) assesses the distributional effects of emission permits for CO_2, NOx and SO_2 and finds that CO_2 permits are more regressive than SO_2 permits but less regressive than NOx ones. Metcalf (1999) assesses the distributional effects of various environmental taxes, and finds that an air pollution tax is less regressive than a carbon tax or a motor fuel tax[8]. Due to this shortage of studies[9], it is not yet clear what the effect of LAP tax is on the distribution of the tax burden across income groups.

This chapter investigates the potential regressive impacts of a LAP tax (based on the internalization of the external costs), and compares it in a compressive fashion to a GCC tax (tax on CO_2)[10]. For the comparison, both tax schemes are set to yield the same revenue. The aim is to analyse whether LAP taxes might be easier to implement when the distributional issue is factored into the political agenda. An Input-Output model is used, combined with a micro-simulation model, making it possible to estimate the impact of LAP and GCC on the household disposable income and also on the deadweight loss. The cost and deadweight loss are calculated by expenditure deciles, as are the main progressivity and redistribution indexes such as the Reynolds-Smolensky and Kawani indexes. Finally, this study also explores the

[7] Ekins et al. (2011), Barker and Khöler (1998) and Metcalf (1999) also find that revenue recycling through distortional taxes could be more regressive than other types of revenue recycling. Additionally, Gonzalez (2012) finds that in Mexico and the US, recycling through tax cuts on manufacturing is regressive, while recycling through food subsidies is progressive.

[8] However, the results of Metcalf (1999) are not definitive because if impacts are studied with lifetime income measures the results are different; with lifetime income measures an air pollution tax is more regressive than a motor fuel tax.

[9] However, there are studies assessing the economic effects of the internalization of the external costs of local air pollution. See for example Kiulia et al. (2013)

[10] Although environmental taxes must be complemented with other instruments in the long term, in the short terms, they are successful (del Río González, 2008).

distributional effects of a revenue-neutral recycling scheme through a reduction in taxes on labour (social security contributions paid by employers).

The rest of the chapter is structured as follows: Section 2.2 presents the methodology and data; Section 2.3 describes the different tax scenarios proposed; Section 2.4 presents the results and Section 2.5 sets out the conclusions.

2.2 Methods and data

2.2.1 Methods

A top-down approach is used to link the micro and macro model. In this approach, the outputs from the macro model are used as an input in the micro model, making it possible to analyse the distributional impacts. The empirical analysis therefore involves two stages: in the first, the price changes produced by the environmental tax are studied through an Input-Output (IO) price model. In the second stage a microsimulation model is used to calculate the distributional effects of price changes. This is carried out using the microsimulation tool developed by Sanz-Sanz et al. (2003).

2.2.1.1 Input–Output model

Our tax scenario levies the emissions produced through the production process and inputs used by each sector. Price changes produced by the tax are assessed through an Input–Output (IO) model. This model assumes that the production technology is linear, i.e. that each sector produces a single good or service under fixed coefficients by combining intermediate inputs, primary factors (labour and capital) and imports. This means that there is no possibility of substitution between inputs; taxes on producers are therefore passed on to consumers. Although this is a strong assumption in the long term, it is reasonable for assessing short-term impacts.

Price changes are the result of taxing emissions from industry sectors, which represents the internalization of the externality or cost generated by each pollutant. To internalise the external cost caused by emissions, the study uses emissions from the different production sectors[11] and the cost of the associated externality if a tax is levied on it. The following equation denotes the external cost internalised by each sector (EC_j):

[11] See methodology on Spanish Environmental Satellite Accounts.

$$EC_j = \sum_{z=1} c_z \,.\, E_{zj} \tag{2.1}$$

where E_{zj} is emissions of pollutant *z* from sector *j* and c_z is the price of pollutant *z*. The size of this effect depends on the level of internalization as it is not necessary to include all social costs.

Once the total tax payment by industry sector has been calculated, the input–output approach allows us to estimate total indirect price changes of consumption. In particular, to analyse the price change an input-output model is used based on Leontief's price model with differentiation of imports, so that the taxes proposed do not alter import prices, similar to Buñuel (2011) and Demisse et. al (2014).

The following equation can be used to evaluate the effects on prices by each sector[12]:

$$P_j = \sum_{i=1}^{I} p_i a_{ij} + p_{m_i} a_{m_{ij}} + (1 + s_j) w l_j + r k_j + EC_j \tag{2.2}$$

where P_j is the price of consumption good *j;* a_{ij} stands for the input-output coefficients, and p_i is the price of inputs from sector *i* and *I* is the number of total sectors . The term p_{m_i} represents the price of imports and $a_{m_{ij}}$ is the coefficient that represents imported goods per euro of output. Further l_j, k_j, are, respectively, labour and capital, and the terms w and r are the price of labour (wage) and the price of capital, and s_j is the tax rate of the social security paid per sector. When there is no internalization (i.e. no tax on pollutants) c_z is zero, and therefore EC_j is also zero.

2.2.1.2 Microsimulation model

Households may be expected to alter their spending decisions as a result of price changes. A demand model reveals households' behaviour and provides a realistic picture of the substitution, own-price and income effects. To assess the distributional effects, a micro-simulation model developed by Sanz-Sanz et al. (2003) is used.

The micro-simulation model uses an Almost Ideal Demand System (AIDS) designed by Deaton & Muellbauer (1980). The main advantage of AIDS is that it enables a first-order approximation to be made to an unknown demand system. In addition, this model satisfies the consumer axioms and does not impose constraints on the utility function (Sanz-Sanz et al. 2003). AIDS is

[12] There are 21 sectors in our IO approach.

based on the assumption that the households will alter their spending decisions as result of price changes as per this equation:

$$w_i = \alpha_i + \sum_{J=1}^{n} \gamma_{ij} \ln p_j + \beta_i \ln\left(\frac{G}{P}\right) \tag{2.3}$$

where w_i is the share in expenditure of good *i* for a particular household, p_j is the price per commodity, *P* represents the price consumer index and *G* is total expenditure. Hence, G/P represents real expenditure.

The simulation performed is based on an indirect tax reform which is equivalent to the price change obtained. This price change is the result calculated with the input-output model. The distributional impacts on the short-term effects of the price change are thus examined. The micro-simulation model has 16 different consumption groups, so it calculates the pre and post reform price indexes and the sum of the prices of all individual goods weighted by their contribution to the composite category. The pre-reform price for good *i* is:

$$p_i^0 = \left(1 + t_i^0\right)(x_i) \tag{2.4}$$

where t_i^0 is the initial VAT rate, and x_i represents the price before tax. Hence the price after tax is

$$p_i^1 = \left(1 + t_i^1\right)\left[\frac{p_i^0}{\left(1 + t_i^0\right)}\right] \tag{2.5}$$

where t_i^1 is the post-reform VAT equivalent to price change obtained with the input-output model.

Finally, the cost is assessed through Equivalent Variation (EV), which assumes that households reallocate expenditure as a result of price change. Given a vector of reference price P_r, the equivalent expenditure is defined as the level of expenditure that allows households to achieve a reference level of utility, $v_r(P, G)$, where *P* and *G* are, respectively, the effective price and expenditure.

$$V(P_r, G_e) = v_r(P, G) \tag{2.6}$$

which can be expressed in terms of the expenditure function

$$G_e = e\big(P_r, v_r(P, G)\big) \tag{2.7}$$

The equivalent variation per household is defined as the amount of money that a household would be willing to pay to prevent occurrence of the price change (Deaton and Muelbauer, 1980; Creedy, 1999 among others):

$$\mathrm{EV} = \mathrm{e}(\mathrm{p}^0, \mathrm{v}^1) - \mathrm{e}(\mathrm{p}^1, \mathrm{v}^1) = \mathrm{G}_\mathrm{E}^1 - \mathrm{G} \tag{2.8}$$

where G_e^1 is the final equivalent expenditure. EV measures the effect of a tax reform on household disposable income (impact cost). In other words, a positive value of EV implies that households need extra money to maintain their purchasing power. Regarding efficiency, the well-known equivalent deadweight loss (EF) is used, defined as:

$$\mathrm{EF} = VE - (R_1 - R_O) \tag{2.9}$$

where R_0 and R_1 are revenues in the initial and final tax scenarios. The higher the value of EF the greater the distorting effect of a tax. Comparison of the results obtained in computing expression (2.9) makes it possible to determine which of the two taxes analyzed is worse in terms of efficiency.

2.2.2 Data sources

The IO model is based on the data from the Symmetric Input–Output Table for 2005 (INE, 2013a). The IO table is a representation of the uses and resources of the production sectors of the Spanish production system. Measures for the emission of different pollutants per production sector are obtained from the Environmental Satellite Accounts (INE, 2013b). Information on the damage to society caused by air pollution is obtained from CASES (2006).

The basic data used in micro-simulation come from the Spanish Continuous Household Expenditure Survey, EPCF (INE, 2013c). This database provides micro-data which are used for both the estimation and simulation phases of the demand model. The ECPF provides information on consumption patterns as well as some data on household incomes, taxes and household demographic characteristics. The information is completed with data from TEMPUS, which provides the price of goods and services consumed by households.

2.3 Tax scenarios

The interest of this chapter concentrates on the distributional assessment of LAP tax (based on internalization of the actual external costs), and in comparing it in a compressive way with a GCC tax. The tax scenarios are based on internalization of the external harm through taxes levied on producers and are designed in such a way that the internalised external cost is the same in all scenarios. The tax scenarios used differ in two key dimensions: the environmental issue internalised and the recycling of the revenue. Cross-combination of these two dimension yields four scenarios. Across the four scenarios, the tax is introduced into the input-output model, increasing the cost of production by each sector. The price change calculated with the input-output model is performed in the demand model as an indirect tax reform which is equivalent to the price change obtained.

Table 2.1: Tax Scenarios

All scenarios are based on the internalization of the external harm through taxes which are levied on producers and are designed in such a way that the external cost internalised are the same in all scenarios. The scenarios are distinguished in terms of the environmental harm internalised and the revenues recycling.

		Revenue-Recycling	
		No	Yes
Environmental Issue	GCC	(*GCC tax*)	(*GCC tax with RR*)
	LAP	(*LAP tax*)	(*LAP tax with RR*)

To internalise the external cost caused by emissions, emissions from the different production sectors[6] and the cost of the associated externality are used. The following equation represents the total actual external harm for each scenario.

$$TEC^{LAP} = \sum_{j=1} \sum_{z=1} c_z \,.\, E_{zj} \qquad\qquad TEC^{GCC} = \sum_{j=1} c_{co2} . E_{zj} \qquad (2.10)$$

where TEC is the total external cost for each scenario, E_{zj} is emissions of pollutant *z* from sector *j* and c_z is the price of pollutant *z*. In the case of GCC tax there is only one pollutant, CO_2, while in the LAP scenarios the pollutants (*z*) are NH3, NOx, SO2, NMVOC, and PM10.

The first scenario is a Global Climate Change (GCC) tax based on the external harm caused by the CO_2 emissions. The taxes levied on different countries range from €13.50 per tonne in Denmark to €108 per ton in Sweden[13]. On the other hand, in the EU-ETS, between January 2011 and December 2012 future prices for 2020 fluctuated between €10.50 and €28 per tonne. Finally, this study considers a GCC tax of €25 per ton of carbon. This tax is within the range of carbon taxes levied recently in other countries and is also within the expected price range for the EU-ETS in the future. Moreover, it is also within the range of the current estimations of social cost of carbon averaged over various studies, as calculated by Tol (2005)

[9] See methodology on Spanish Environmental Satellite Accounts.

[13] When it was introduced in 1991, the carbon tax in Sweden was €28/tonne, but it is now estimated to be around €108/tonne, although some sectors are exempted.

and EPA (2013). This tax is also similar to the taxes on CO_2 applied in other studies for Spain (see Buñuel, 2011 and Labandeira and Labeaga, 1999). Considering a CO_2 damage cost of €25 per tonne, the CO_2 tax applied to production sectors would (before any change in the response by producers and consumers is considered) be equivalent to internalising to the tune of €7 billion, 0.86% of GDP.

Table 2.2: Social cost of local air pollution for Spain, 2005.

	Social cost estimated by CASES (€ per ton)	Social cost used in the simulated reform (€ per ton)
Sox	4912.22	2323.44
Nox	3485.07	1648.41
COVNM	797.34	377.14
NH3	5393.91	2551.28
PM10	16037.56	7585.63

The second scenario is an LAP tax. According to the social cost calculated by Markandya et al., (2010)[14] the total external cost caused by local air pollution is much higher than the external cost of the carbon emissions ($\mathrm{TEC}^{\mathrm{LAP}} > \mathrm{TEC}^{\mathrm{GCC}}$). Estimations of the external cost of LAP stand at around €15 billion whereas the external harm of CO2 is €7 billion. The aim is to make a distributional comparison of the costs of a tax on CO2 and a tax on LAP. The way to do this is, initially, to calibrate the taxes in such a way that both yield the same revenue. Thus, the overall cost caused by local air pollution has not been internalised totally; only the proportion equivalent to achieving the same external harm of the carbon tax scenario has been internalised, representing 47.2% of total external harm. Hence, the sum of external cost caused by all emissions is equal in the LAP tax scenario and in the GCC tax scenario ($TEC^{LAP} = TEC^{GCC}$). Then our scenarios yield the same revenues but do not compare two systems with the same emission reduction. Due to the uncertainties associated to any *ex-ante* estimation of the emission reduction, it will be easier for policy makers to introduce a tax according the actual external damage and then revise it depending on the real impact. Table 2 shows the social cost estimated by the CASES project and the social cost used to achieve an internalization equivalent to the GCC tax scenario.

[14] In 2006, the CASES ("Cost Assessment of Sustainable Energy System") project (Markandya et al., 2010), funded by the European Commission, compiled a complete, consistent assessment of the social cost of these emissions for EU Countries. This project assessed the physical damage caused by these pollutants to human health, crops and buildings/infrastructures and converted it into monetary values. Measurements of this type should be treated with some caution, but they enable taxes to be distributed proportionally among the pollutants.

The previous tax scenarios proposed are combined with another two scenarios, where a revenue-neutral tax reform is analysed. This is the second stage in the process to make a comparison of the distributional costs of a tax on CO2 and a tax on LAP. There are different ways of undertaking changes in the tax mix – for example by reducing the burden of direct taxes, or giving lump-sum transfers to the losers. One realistic way is the fiscal devaluation recommended by institutions such as the Bank of Spain (2014), IMF (2014) and the OECD (2011) among others. The aim of such reform is to use the revenues in full to cut labour costs, while at the same time increasing indirect taxes. This is a way of reducing unemployment and increasing the price of imports without changing the price of exports. Evidence shows that fiscal devaluation could be an appropriate policy for a country such as Spain, which has the highest unemployment rate in the European Union (23.6% in 2014), and where boosting employment is the primary economic challenge. For Euro Zone countries, De Moijk and Keen (2012) find that a shift of 1% of GDP from social contributions to indirect taxes would increase net exports by around 0.9 to 4% of GDP. For southern countries, Engler et al (2014) show that a fiscal devaluation of 1% of GDP increases output by 0.9 to 1.5% of GDP. For the Spanish case, Boscá et al (2013) analyze a 3.5 % reduction in the effective contribution to social security paid by the employer and a 2% increase in the effective VAT rate. Their results show that, on average, the Spanish economy would grow each year by 0.74 % while employment would rise by 1.3%. In this context, in this chapter a revenue-neutral tax reform was simulated by implementing a tax of €25 per tonne of CO_2 emitted into the atmosphere. Revenues raised by this new tax are used to cut the social contribution rate paid by the employer by 7.45%.

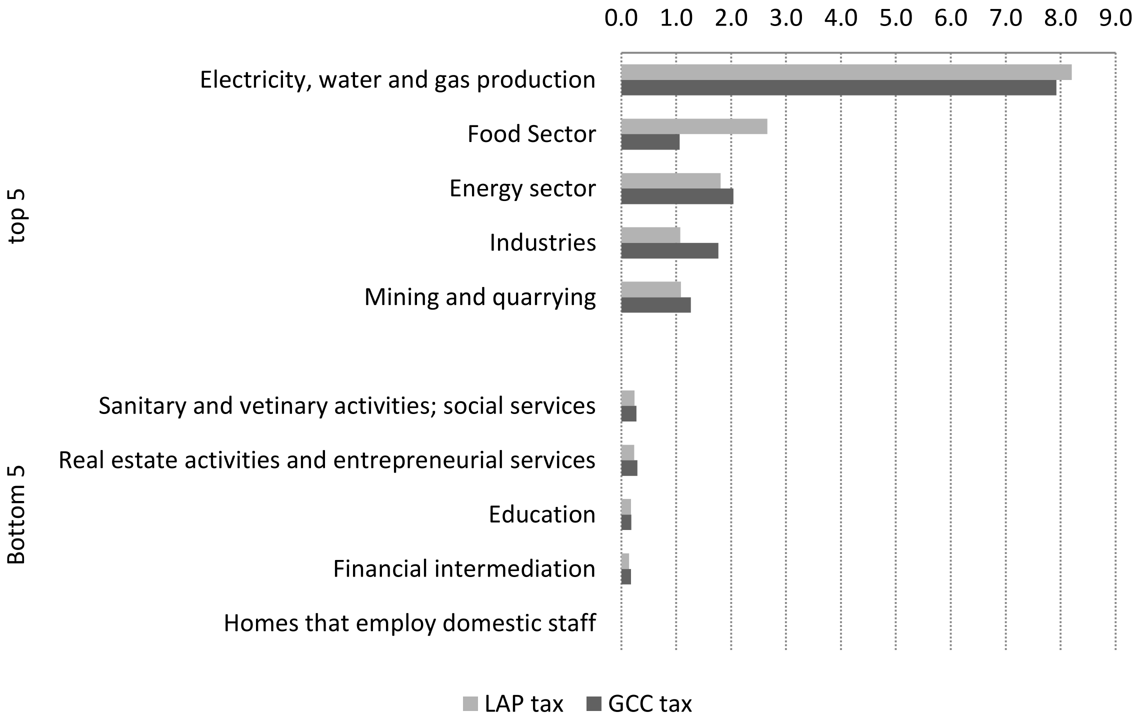
2.4 Results

This section presents the results obtained for the four tax scenarios presented in Section 3. The impacts on prices obtained with the IO model are presented first, then the distributional effects obtained when those price impacts are factored into the demand model are analysed. Thirdly, the implications of "recycling" the revenues, and finally the different aggregate indexes are considered so as to measure the distributional implications consistently and in an overall manner.

2.4.1 Price impacts

Figure 2.1 shows the impact on prices for the five sectors with the highest and lowest impacts on prices changes. Observe that "Electricity, water and gas production", "Energy", "Food", "Industry" and "Mining" are the top five sectors in terms of price impact. These sectors have in common that they are energy-intensive or energy-related. Although all these sectors show similar impacts on prices for the different tax scenarios, there are differences worth mentioning. For example, the price increase for the "Food" sector is higher with an LAP tax than with GCC tax due to emissions of NH_3 produced by animal waste degradation and the use of fertilizer. Similarly, the "Electricity" sector has lower impacts if GCC emissions are considered instead of LAP, due to the large amount of SO_2 emitted by fuel combustion in electricity generation, especially in thermal power stations.

Figure 2.1: Change (%) in production prices. Top and bottom sectors.



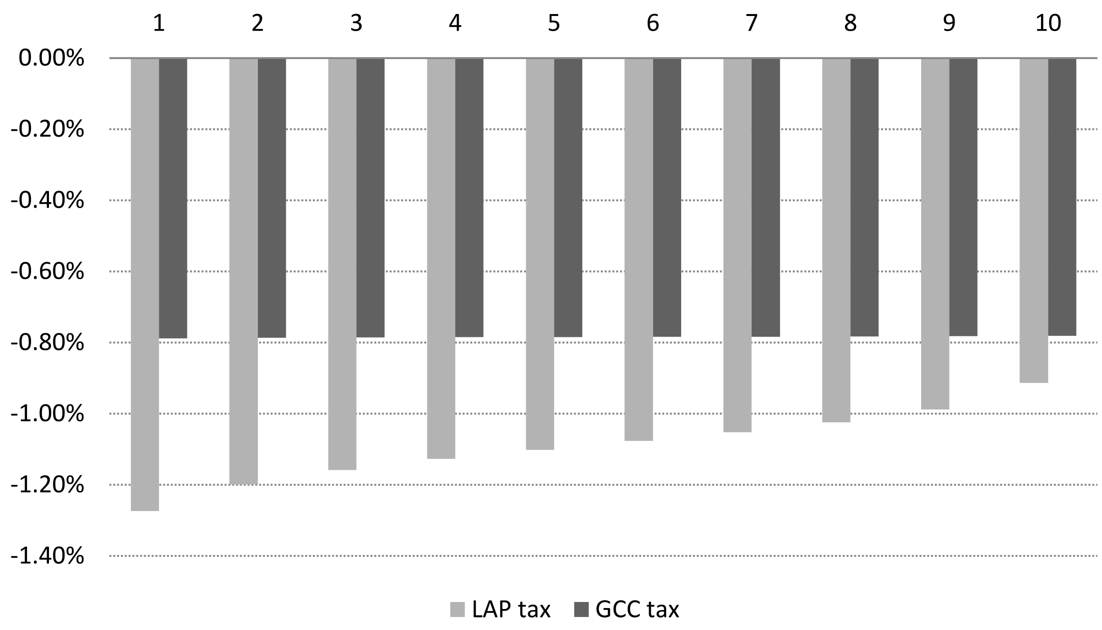
The sectors with the least impact on prices are mainly those that are relatively more labour intensive. "Homes that employ domestic staff", "Education", "Financial intermediation", "Real estate activities" and "Health services" have the lowest price increases, and their impact is almost negligible.

2.4.2 Distributional effects

Figure 2.2 shows the impact of the two taxes analysed on household disposable income by expenditure deciles. The first decile (1) represents the lowest tenth of expenditure and the last one (10) the highest. Cost impacts are measured in terms of equivalent variation (EV) as a percentage of household expenditure. The results show that average cost is €138.17 in the case of the GCC tax and €182.8 for the LAP tax. In other words, the cost is 31% higher with the LAP tax.

Fig. 2.2 shows, firstly, that the cost[15] are below 1.05% for all the expenditure deciles in terms of equivalent variation in expenditure. A wide range of impacts for similar levels of environmental taxes is reported in the relevant literature, but these results are within that range and are similar to those obtained by Wier (2005) or Rausch et al (2011).

Figure 2.2: Cost distribution per expenditure decile.



Secondly, observe that the costs are always lower if the GCC tax is selected and higher with the LAP tax. This can be explained partially by the general price increase that each tax scenario generates.

Thirdly, Fig. 2.2 shows the distributional impacts of the different taxes. Note that the GCC tax shows no regressive effects: in fact it is almost perfectly proportional as the cost is very similar for all expenditure deciles. All income groups lose about 0.8% in terms of equivalent variation in expenditure. These results are similar to those of Labandeira and Labeaga (1999) who also find no evidence of regressivity for a CO_2 tax in Spain. In the case of the LAP tax, the bottom deciles pay a larger share of their expenditure than the top deciles. For example, the lowest decile would lose about 1.27%, whereas the highest decile would only lose around 0.91%. Clearly, the LAP tax is more regressive than the GCC tax in terms of equivalent variation in expenditure. Section 5.4 below uses different standard indexes to measure and confirm this effect more precisely.

[15] It should however be stressed that the benefits of the policy, in terms of increased environmental quality, are not taken into account, and hence the welfare losses only represent the cost side of changes in total welfare.

Consumption patterns are very important if all these results are to be understood. In Spain the low income households spend a larger fraction of their available income than high income households on “food” and “housing”, in relative terms. The budget share accounted for by expenditure on travel, entertainment, restaurants and hotels increases notably with income. For example, the lowest expenditure decile spends 24% on food and 47% on housing, whereas the highest spends only 12% and 27%, respectively. Conversely, expenditure on transport ranges from 3% in the lowest decile to 18% in the highest.

As stated in the previous section, the LAP tax increases the price for food and energy more than for other sectors. That is why this tax is more regressive than GCC. These results can be summarized by saying that LAP taxes are more regressive than GCC taxes because they have a higher impact on basic necessities and goods that are relatively consumed more by “poorer” households. The regressivity of GCC taxes is offset mainly because “richer” households consume more intensively certain goods that also have significant emission factors, such transport.

2.4.3 Effects of revenue recycling on income distribution

This second exercise entails a revenue-neutral tax reform in which the tax revenues from the scenarios are used in full to finance a reduction in taxes on labor, and more precisely a reduction in social security contributions paid by employers. The tax reduction needed to offset the new environmental tax is around 7.5% of social security contributions.

Figure 2.3 shows the further impacts on prices with the revenue-neutral tax reform. The results show that there is still a major increase in energy-intensive sectors: the “Electricity, water and gas production” and the “Energy Sector” undergo large price increases independent of the kind of tax burden imposed, while the “Food Sector” undergoes a large price increase with the LAP tax. However, the important difference now is that those sectors which are non-polluting or “clean” and labor intensive benefit from reductions in their prices. For example, the price changes in “Education” and “Health services” are negative and close to 1%.

Figure 2.3: Impacts on price (%) after revenue recycling. Top and bottom sectors.

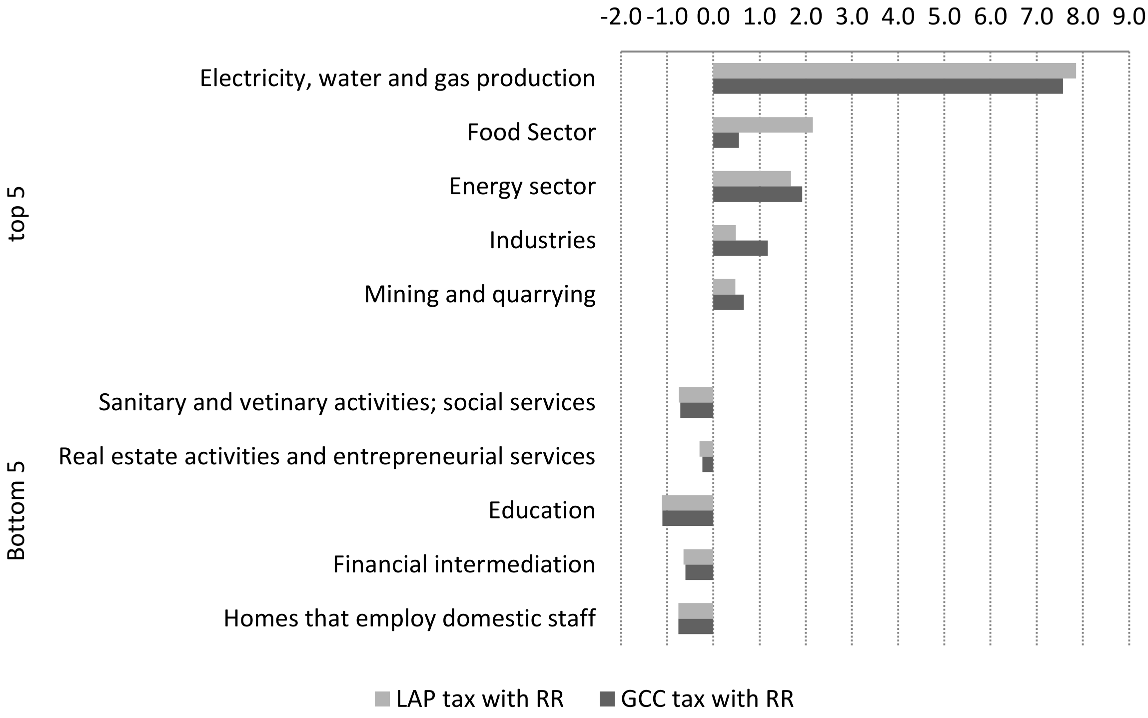
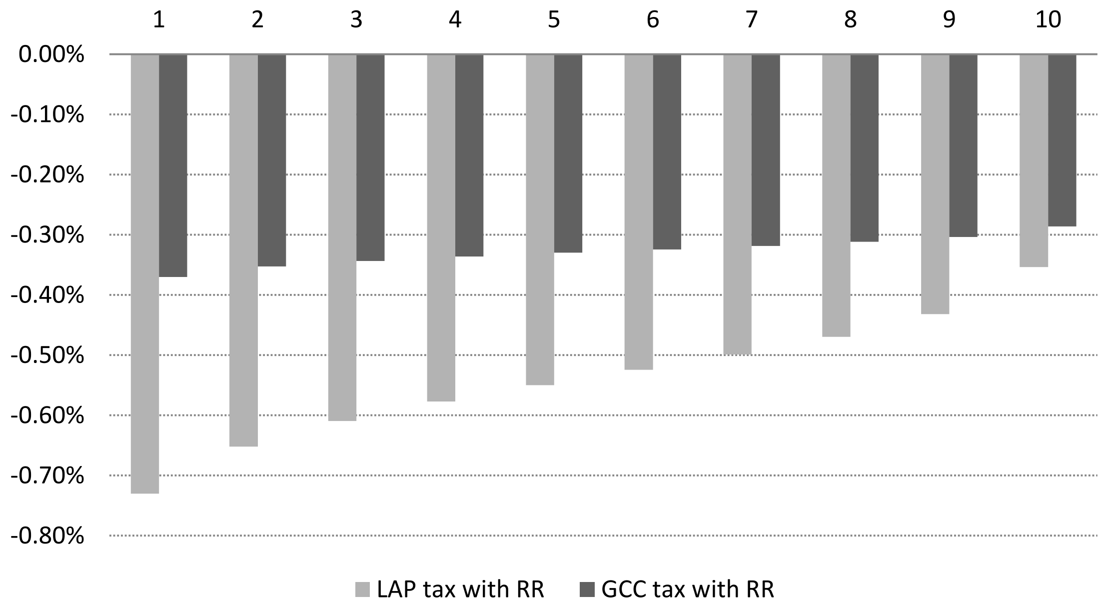


Figure 2.4: Cost distribution after recycling per expenditure decile

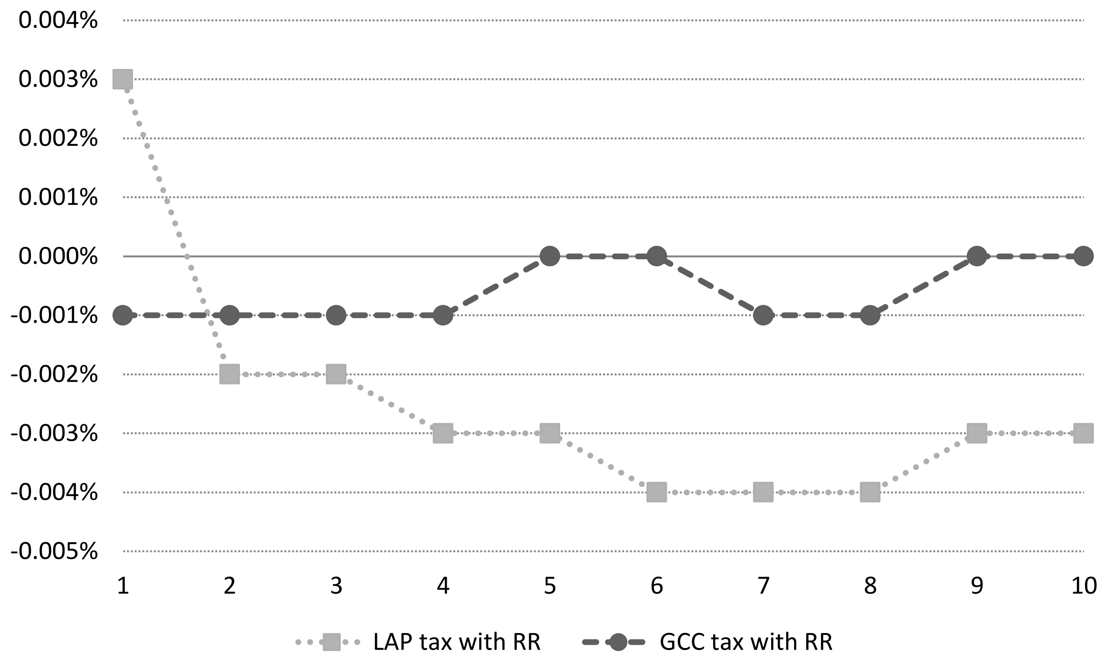


Figures 2.4 and 2.5 show cost impacts and the excess burden[16] per expenditure decile after revenue recycling. Firstly, it is clear that the distributional costs of taxation are lower after

[16] The excess burden is calculated as the difference between equivalent variation (*EV*) and revenue (*R*) generated by households (*h*): $E_{GE} = -\sum_h EV_h - (R_h^1 - R_h^0)$

recycling revenue: they decrease by about 0.5% for all income groups and for both tax scenarios.

Figure 2.5: Relative efficiency impacts after recycling revenue



Revenue recycling through a tax on labor tax can reduce the progressivity of the tax system. Figure 4 reveals that under the GCC tax the differences between different types of household are still very small. However the difference between high and low income groups is larger than before recycling, evidencing that impacts are more regressive with revenue recycling. Under the LAP tax the cost for the highest income group is only 0.35% while that of the lowest group is 0.73%, and the gap between income groups is wider than without recycling.

The effects in terms of efficiency of a revenue-neutral tax reform are as follows: the excess burden is reduced considerably for both the GCC tax and the LAP tax. Thus, the fiscal system is more efficient with revenue recycling for most expenditure deciles. These results are in line with the literature on the double dividend hypothesis, where it is reported that cost decreases if the revenues from environmental taxes are recycled through taxes on labor (Goulder, 1995). Our results show that a trade-off between efficiency and equity (distributional effects) can exist when choosing specific revenue-recycling based on low taxes on labor. Revenue recycling through a distortionary tax has a positive impact on efficiency, but the distributional implications may not be positive.

2.4.4 Indexes for measuring regressivity

We also calculate a set of indexes which can provide information about the overall distributional effect of the taxes proposed. The Reynolds–Smolensky Index (RS Index) provides information about redistribution, and the Kakwani index is used to measure progressivity[17] . All these indexes are estimated relative to total household expenditure.

Table 2.3: Progressivity and redistribution effects

	Marginal Reynolds–Smolensky Index	Marginal tax rate	Marginal Kakwani index
1. Pre-reform index	0.00434	0.11379	0.03855
2. Post-reform indexes Without Revenue-Recycling (NRR)			
GCC tax	0.00440 (+0.00006)	0.12064 (+0.00685)	0.03662 (-0.00193)
LAP tax	0.0039 (-0.00044)	0.12301 (+0.00922)	0.0322 (-0.00635)
3. Post-reform indexes With Revenue-Recycling (WRR)			
GCC tax with RR	0.00419 (-0.00015)	0.1171 (+0.00331)	0.03626 (-0.00230)
LAP tax with RR	0.00381 (-0.00053)	0.1192 (+0.00541)	0.03269 (-0.00586)

(Variation of measures of regressivity with respect to the pre-reform index)

Table 2.3 reports the Reynolds–Smolensky index (RS) and the Kakwani index (K). RS and K indexes are useful to measure the impact of a tax reform in terms of redistribution and progressivity. Variation in absolute terms with respect to the situation in the pre-reform scenario is shown in parenthesis. Table 2.3 shows results for the effects of a reform on GCC and LAP taxes in two cases: (i) without revenue recycling (NRR) and (ii) with revenue recycling (WRR). RS and K indexes have, in the pre-reform and the post-reform scenarios, a positive value. Although positive, the values for both indexes are clearly close to zero in all cases analysed (K<0.04 and RS<0.0045). Therefore, we can say that the tax system tends toward proportionality in both scenarios (pre-reform and post-reform) and regardless of the assumptions used (NRR or WRR). However, there are two issues that deserve to be highlighted. First, (negative) changes in K and RS indexes indicate that progressivity and redistribution are in general worse in the post reform scenario (both in NRR and WRR). The only exception is the redistribute effect of a GCC tax in the case of NRR. Second, in global terms, a GCC tax is superior to an LAP tax in terms of progressivity and redistribution, both under NRR or WRR. Finally, a GCC tax is slightly more progressive and redistributive when a

[17] Both indexes are based in approximations of GINI index.

NRR is used. By contrast, the result is ambiguous in the case of LAP tax. Specifically, it is slightly more progressive under the WRR assumption and more redistributive with NRR.

Finally, indexes show that the changes in redistribution and progressivity are very low, thus the tax system continues to be proportional or even slightly progressive. In the case of GCC tax, the change in the system is negligible, while LAP tax reduces slightly the progressivity of the system.

2.5 Conclusions

Local air pollution (LAP) and global climate change (GCC) are two relevant, interrelated environmental problems. Most of the relevant literature has focused on the distributional impacts of climate change-related taxes such as taxes on CO_2, energy and fuel but to date few papers have investigated the distributional effects of LAP policies. In this study, we conduct a distributional analysis of an LAP tax (based on the internalization of the external costs of several pollutants) and compare it in a compressive way with a GCC tax (tax on CO2), where two tax systems yield the same actual revenue. We use an Input-Output model which calculates the price change caused by these taxes levied on producers, combined with a micro-simulation model that calculates distributional effects on consumers for the case of Spain. We calculate the cost and the deadweight loss by expenditure deciles and also the main indexes such as the Reynolds-Smolensky and Kawani indexes. Finally, we also explore the distributional effects of a revenue-neutral recycling scheme through a reduction on taxes on labor (social security contributions paid by employers)

Our results show that taxes on local pollutants are more regressive than those levied on climate-change pollutants. In fact, the GCC tax tends to be proportional because the energy used in lighting and heating, consumed mainly by low-income households, is offset by the higher spending on transport and energy by high-income households. This is similar to the results obtained by other papers for Spain (see e.g. Labandeira and Labeaga, 1999) and is in line with the emission intensity by income groups in Spain, as show Duarte et al., (2012). LAP taxes tend to be more regressive because they largely affect goods that are consumed by low-income households, such as electricity and food. The increase in food prices is a key factor that explains the regressivity of the LAP tax, because this tax indirectly increases the price of food more and because low income households spend a large proportion of their income on food. The cost in the case of a GCC tax is around 0.8% for all the expenditure deciles, but in the LAP tax the cost decrease ranges from 1.2% for the first decile (the poorest households) to 0.9% for the tenth (the richest households). In any case, the overall effect on distribution in the tax system is very low then the change in the main indexes is compared to the pre-reform situation where no tax is levied.

As far as recycling is concerned, our results show that the overall cost is reduced notably but the distributional implications do not change much. Indeed distributional implications are actually worse, because the average reduction in social security contributions for all sectors reduces the price of some service sectors that are "cleaner" and more labor-intensive because they are consumed relatively more by high-income households. Although the level of progressivity of the tax system does not change much in the LAP tax (where the Kawani index shows better results for progressivity but the Reynolds-Smolensky indexes show worse results for distribution and redistribution), the loss of progressivity is clear for the GCC tax. Finally, recycling also shows that a trade-off may exist between efficiency (of the tax system) and equity (distribution) especially in the GCC tax scenario.

Some caveats should be made in order to put these results into perspective. First, these are empirical results and they can be extrapolated only to countries with similar production and consumption profiles. The distributional implications of taxes on air pollution or climate change depend very much on the structure of the economy, even if revenues are recycled in different forms. Second, we only consider the distributional effect of environmental taxation and not the welfare loss associated with pollution. There are many studies (see for instance Pye et al., 2006 and Walker et al., 2003) that show that LAP affects low income household locations more. Third, our input–output model cannot capture the full effects that a reduction in taxes on labor could have on employment and, therefore, on welfare. The relevant literature suggests that such tax reforms could have a positive effect especially in those countries, such as Spain, that have highly distorted labor markets and high unemployment levels (see for example Markandya et al., 2013). Moreover, input–output model assumes that there is no possibility of substitution between inputs which restricts our analysis to short-run effects. Fourth, our methodology doesn´t incorporate price feedback effects from microsimulation model to IO model, but Rutherford and Tarr (2008) find that this effect is small if the data are reconciled between the national accounts and the household budget survey. Finally, our scenarios yield the same revenues but do not compare two systems with the same emission reduction. Due to the uncertainties associated to any *ex-ante* estimation of the emission reduction, it will be easier for policy makers to introduce a tax according the actual external damage and then revise it depending on the real impact.

The first policy implication of this chapter is that although it was thought that LAP taxes might be easier to implement because their effects (mainly on health) are felt more immediately by citizens and by low-income households than those of GCC taxes, this may not be the case if the distributional issue is factored into the policy maker's equation. The second policy implication is that if it is wished to correct the distributional effect of this type of tax reform the standard approach, i.e. reducing taxes on labor, may not improve the distributional effect. However, and this is the third policy implication, given that the overall regressivity of these taxes is low, various specific combinations of policies could be design to compensate the households or groups that are most affected.

Chapter 3

The distributional effects of carbon-based food taxes

3.1. Introduction

The recent Paris Agreement at the 21st Conference of Parties (COP21) shows that greenhouse gas (GHG) mitigation has been put increasingly on the political agenda. Traditionally, mitigation options for achieving targets have been focused more on the energy and transport sectors, and less attention has been paid to agricultural and food-related sectors. Although energy and transport sectors are the largest contributors, agriculture accounts for 10–12% of global direct GHG emissions. If other indirect emissions are considered, such as those from fertiliser production and land use change, that fraction rises to 20-24% (IPCC 2014). Moreover, when a full life-cycle analysis of emissions is performed according to the EIPRO study (EC 2006), food consumption may account for as much as 31% of the EU-25's total GHG emissions. According to the EIPRO study, 80% of the emissions from food originate from the consumption of meat and dairy products, but these only represent about one-third of the total energy intake (McMichael et al. 2007). Therefore, any policy aimed at mitigating emissions should also consider options that impact on the food system, especially in relation to livestock production and meat consumption.

Many papers have explored the mitigation options, potential and costs throughout the food production chain (see Garnett, 2011). The options can be classified as supply-side measures, such as increasing land productivity via technological or managerial approaches (Webb et al. 2014), and demand-side measures, such as reducing losses in the food supply chain (Godfray et al., 2010) or changing diets (Hedenus et al. 2014 and Hallström et al. 2015). Although demand-driven measures have seldom been considered, yield improvements may not be sufficient to deliver emission reductions and maintain food security without significant expansion of crop or pasture areas (Bajželj et al., 2014). Stehfest et al. (2009) shows that a global food transition to a scenario with less meat in the diet can reduce remarkably[18] the

[18] According to Stehfest et al. (2009), a global transition to a low meat-diet as recommended for health reasons would reduce the mitigation costs to achieve a 450 ppm CO2-eq. stabilisation target by about 50% in 2050 compared to the reference case.

mitigation cost in a 2°C stabilisation scenario. Promoting changes in diet composition may therefore not only play a role in future mitigation policies but also prove to be a cost-effective measure.

Furthermore, dietary changes may be attractive not only from a climate perspective but also from a public health perspective (Mytton et al. 2012). Excess consumption of red meat, sugar and saturated fats increases the risk of various diseases (WHO 2003). On top of this, according to the WHO more than 1.4 billion adults[19] globally are overweight and more than a half a billion are obese. Therefore, recent literature and policies have focused on the health benefits of reducing energy intake and the consumption of meat and dairy products, especially in Northern European Countries (Härkänen et al. 2014). Denmark, for example, introduced[20] a "fat tax" (Jensen and Smed 2013, Gustavsen and Rickertsen 2013), Hungary applied a "junk food tax" and France tried to introduce a tax on sweetened drinks (Villanueva 2011). However, health related food taxation has also been criticised in terms of effectiveness, distributional impacts and acceptability (McColl 2009). Evidence suggests that it can be effective in improving health conditions if taxes are sufficiently high, but changes in nutrients should be considered carefully (Green et al 2015). Most studies also find that health-related food taxation is regressive (Leicester and Windmeijer 2004); that is, poorer income groups pay a greater proportion of their income in tax than richer income groups. But it is not clear whether health gains might be progressive (Nnoaham et al 2009), in which case they would offset the negative effect on income distribution. Although acceptability is generally low and varies widely, support increases when the health benefits are explained and emphasised (MRC 2011). Finally, most studies agree that all these barriers could be ameliorated if taxes on less-healthy foods are combined with subsidies or tax exemptions on fruit and vegetables (Nnoaham et al 2009, Smed et al. 2007, Mytton et al. 2012). In any case, these results suggest that there are potential gains in terms of health and climate change if the consumption of meat and dairy products is reduced.

Additionally, a Mediterranean-style diet has been extensively reported to be associated with a favourable health outcome, with a better quality of life (Sofi et al 2008) and with low carbon emissions (Vidal et al. 2015 and Pairotti et al.2015). The Mediterranean diet comprises eating habits traditionally followed by people in the different countries bordering the Mediterranean Sea, such as Italy and Spain, characterised by a high level of consumption of fruit, vegetables

[19] It is also noteworthy that 40 million preschool children are also overweight : http://www.who.int/features/factfiles/obesity/facts/en/index4.html

[20] The tax on saturated fat (which was accompanied by increased taxes on sugar products, soft drinks and cigarettes) was introduced in 2010 and repealed in 2013.

and legumes, moderate consumption of fish and the consumption of olive oil as the main source of fats. However, this diet has evolved since the early 1960s towards a more animal-protein-rich diet similar to those of Northern Europe and America (Lassaetta et al. 2013). In fact, in Spain about 17% of the population are obese and around 53% are overweight, so food taxation could also be an important tool for recovering the traditional Mediterranean diet there.

This study sets out to evaluate the implications of implementing consumption taxes in Spain on food items depending on their GHG footprint. This route is quite novel and, as far as we know, there are only two studies of this type (Edajabou and Smed 2013 and Abadie et al 2015). These studies show that carbon based food taxation can be effective in reducing GHG emissions and explore the optimal design of different taxation schemes on food. This chapter goes a step further and also evaluates the distributional implications of the policy. The welfare impacts are explored for different income, age and social groups for 14 different food categories, including tax exemptions. The elasticities are estimated with 2002-2013 data from household expenditure surveys, which contain information on around 20,000 households, with the use of a demand system model (AIDS). Finally, it is explored the indirect impacts in terms of nutrients in order to test whether the tax scenarios could also help to move towards a healthier diet.

The chapter is structured as follows. Section 3.2 describes the model and the data used in the analysis. Section 3.3 shows the tax scenarios considered, and Section 3.4 discusses the results of the simulations conducted. Finally, Section 3.5 concludes.

3.2. Methods and data

This section describes the model, the elasticities and the data used in the analysis. Subsection 3.2.1 describes the demand model used to estimate the elasticities of the goods analysed. Subsection 3.2.2 describes the data used in the estimation and simulation stages. Finally, subsection 3.2.3 analyses the price and expenditure elasticities estimated.

3.2.1 Demand model

A two-step approach is followed to assess the welfare effects generated by different tax scenarios. Firstly, a demand model is estimated to provide a set of estimates of the substitution, own-price and expenditure elasticities of the goods analysed. These elasticities are then used in Section 4 to simulate distributional and welfare effects generated by tax rates charged on food. For the first stage, it is used the well-known Almost Ideal Demand System (AIDS) proposed by Deaton and Muellbauer (1980). This model has been widely used in scientific literature to analyse food demand (see Smed et al, 2007; Bouamra-Mechemache et al., 2007; Mergenthaler et al, 2009; Bilgic and Yen, 2013 among others). Its main advantage is that it enables a first-order approximation to be made to an unknown demand system. In addition, the model satisfies the economic consumption theory axioms and does not impose constraints on the utility function. The log-linear approximation (LAIDS) used in this chapter follows an n-good system equation as follows:

$$W_{it} = \alpha_i + \sum_{J=1}^{n} \gamma_{ij} ln\, p_j + \beta_i ln \left({}^{Y_{it}}/_{\widetilde{p_t}} \right) + t + \sum_{i=1}^{3} d_i + e_{it} \qquad [3.1]$$

where W_{it} represents the share associated with good i in period t for each household, α_i is the constant, p_j is the price of commodity j, $\tilde{p}$ stands for the Stone price index, Y is household income (hence, $Y/\tilde{p}$ represents real income), t is a trend variable that captures the role of the time, which takes values equal to 1 in 2002 and 11 in 2013, d_i is a set of dummy variables that controls for the household type[21], the region where the household is located in terms of NUTS1[22] and whether the household is rural or urban, measured through the population density. Finally e_{it} is the idiosyncratic error term. The adding up and homogeneity restrictions of equation [1] are the following:

$$\sum_{i=1}^{n} \alpha_i = 1 \qquad [3.2]$$

$$\sum_{j=1}^{n} \gamma_{ij} = 0 \qquad [3.3]$$

$$\sum_{i=1}^{n} \beta_i = 0 \qquad [3.4]$$

The symmetry condition is given by:

$$\gamma_{ij} = \gamma_{ji} \qquad [3.5]$$

As can be seen in Table 1, the latter restriction does not imply that the cross-price elasticities, ε_{ij} , for the i and j goods are necessarily equal. Finally, the sum of w_i should also satisfy the following:

[21] The household categories used are: adults alone; couple without children; couple with children; single-parent households and other households

[22] The NUTS classification (Nomenclature of territorial units for statistics) is a hierarchical system for dividing up the economic territory of the EU. The NUTS 1 level represents groups of autonomous communities.

$$\sum_{i=1}^{14} w_i = 1 \qquad [3.6]$$

This demand model uses a set of 14 types of foods including cereals, beef, pork, chicken, fish, milk, dairy products, eggs, fruits, vegetables, potatoes and potato-based foods, oils and fats, sugar and sweet products, and other food products. Since the AIDS model is made up of a system of dependent equations, the share equation regarding other food products has been deleted to overcome singularity problems. The elasticity matrix is computed using the following expressions:

Marshallian Own price elasticity: $$\varepsilon_{ii} = \frac{\gamma_{ii}}{w_i} - 1 \qquad [3.7]$$

Marshallian Cross price elasticity: $$\varepsilon_{ij} = \frac{\gamma_{ij}}{w_i} \qquad [3.8]$$

Expenditure elasticity: $$\theta_i = \frac{\beta_i}{w_i} + 1 \qquad [3.9]$$

3.2.2 Data.

The dataset used at both the estimation and simulation stages of this study comes from the Spanish Household Budget Survey (SHBS) (INE, 2014). The SHBS is a representative cross-sectional survey of the whole Spanish population that collects yearly information on consumption patterns as well as socio-economic characteristics. It covers around 20,000 households per year. The estimation stage uses the SHBS data for the 2002 to 2013 period whereas the simulation phase uses data from 2013[23]. In the estimation of equation [3.1], household expenditure is used as a proxy of income, firstly because income is strongly under-reported in household panel surveys (see for example Wadud et al., 2009) and secondly because household expenditure is a good proxy for permanent income (Poterba, 1991). The Food Price Index with a 2002 baseline is used for each item analysed. That index price is divided by the General Price Index per region for 2002. This procedure enables price heterogeneity to be introduced. The emission factor used to calculate tax rates for each item is provided by Hoolohan et al. (2013), who survey some of the main studies of the life cycle emissions of foodstuffs. Finally, the macronutrients provided by Moreiras et al (2011) are used to assess the nutritional impacts generated by the tax scenarios analysed.

[23] In 2005 the methodology of the survey was changed, so there are two different periods in the estimation data. For 1998 to 2004 the SHBS based on 1997 is used, and for 2006 to 2010 it is the SHBS based on 2006. This change necessitated some adjustments to link the two surveys.

Table 3.1 shows the average consumption of goods analysed by quintiles and household type in Spain in 2013. In relative terms, high-income households consume a slightly greater percentage of beef, fish, dairy and sugar and sweet products than low-income ones. By contrast low-income households allocate a greater share of their expenditure on cereals, pork, poultry, milk, eggs, potatoes and oil than high income ones, which does not necessarily mean the consume more as their total expenditure is lower. There are few differences in the consumption of fruit and vegetables by income levels. Households with children have a greater share of expenditure on cereals, milk, dairy products and sugar and sweet products than other household types. The consumption of fruit (8.9%), vegetables (8.8%) and fish (6.1%) is notably low in relative terms in households of this type.

3.2.3 Elasticities

The price and expenditure elasticities obtained are shown in Table 3.2. The right side of the table has a column with expenditure elasticities. The main diagonal (darker colour) of the matrix shows the own-price elasticities while the remaining elements are cross-price ones. Own-price elasticities have the expected negative sign with a value of less than one in absolute terms. The exceptions are beef (-1.313), fruit (-1.188) and vegetables (-1.128). The goods with the most inelastic demands are oil (-0.224) and milk (-0.296). The expenditure elasticity is positive in all cases, indicating that all the items analysed are normal goods. The results clearly show an expenditure elasticity greater than unity in the case of beef (1.142), fish (1.164) and sugar and sweet products (1.233), indicating such items to be (in economic terms) luxury goods. Dairy, fruit and vegetables show elasticities very close to unity. By contrast, other goods analysed are necessities, with poultry (0.850), eggs (0.845) and milk (0.826) having the lowest expenditure elasticities. Although it is difficult to compare because the food groups and the methods are different, these results are in line with other papers that estimated food elasticities in Spain (see e.g. Angulo et al. 2008, Laajimi et al. 1997).

Table 3.1: Percentage of food expenditure

	Cereals	Beef	Pork	Poultry	Fish	Milk	Dairy	Eggs	Fruits	Vegetables	Potatoes	Oils	Sugar	Others
Average	22.9%	4.3%	3.5%	5.6%	7.6%	5.6%	10.6%	1.9%	10.8%	10.3%	3.1%	3.3%	5.6%	4.4%
Expenditure quintile														
Lower	24.5%	3.9%	3.7%	5.8%	7.0%	6.5%	9.7%	2.3%	10.8%	9.8%	3.1%	3.4%	4.9%	3.6%
Middle low	23.0%	4.5%	3.6%	5.6%	7.5%	5.9%	10.5%	1.9%	10.9%	10.0%	3.0%	3.1%	5.6%	4.1%
Middle	22.5%	4.5%	3.7%	5.5%	7.9%	5.7%	10.6%	1.9%	10.7%	10.3%	3.1%	3.3%	5.6%	4.3%
Middle high	22.3%	4.1%	3.5%	5.6%	7.9%	5.2%	11.2%	1.8%	10.8%	10.5%	3.2%	3.4%	5.7%	4.6%
Upper	22.3%	4.3%	3.1%	5.4%	7.8%	4.8%	11.0%	1.7%	11.0%	10.8%	3.0%	3.1%	6.2%	5.3%
Type of household														
Adults alone	21.8%	3.9%	2.8%	4.8%	7.3%	5.9%	10.7%	2.0%	12.9%	11.1%	2.8%	3.1%	5.2%	3.8%
Couple without children	20.6%	4.6%	3.6%	5.4%	9.2%	5.1%	10.5%	1.8%	12.3%	11.5%	3.0%	3.4%	5.2%	3.6%
Couple with children	25.5%	3.6%	3.2%	5.3%	6.1%	6.2%	11.9%	1.7%	8.9%	8.8%	2.9%	2.7%	6.6%	6.4%
Single-parent family	23.5%	4.1%	3.6%	5.7%	6.9%	5.8%	10.8%	2.0%	10.1%	10.0%	3.2%	3.5%	5.7%	4.5%
Other households	23.4%	4.6%	4.1%	6.2%	7.9%	5.5%	9.8%	2.0%	10.1%	9.9%	3.3%	3.6%	5.4%	4.0%
Age of the breadwinner														
Old	19.8%	4.9%	3.6%	5.4%	9.7%	5.7%	10.0%	1.9%	13.0%	11.0%	3.0%	3.8%	4.5%	3.3%
Adults	24.5%	3.9%	3.5%	5.6%	6.6%	5.6%	11.0%	1.9%	9.7%	9.9%	3.1%	2.9%	6.2%	5.1%
Young	29.2%	2.8%	3.3%	6.0%	3.9%	5.4%	11.0%	1.8%	7.7%	8.8%	3.7%	2.6%	6.7%	5.8%
Location														
Urban	22.0%	4.5%	3.0%	5.5%	7.9%	5.4%	10.8%	1.9%	11.3%	10.7%	3.1%	3.3%	5.7%	4.7%
Semi-urban	23.0%	4.4%	3.4%	5.6%	7.4%	5.6%	11.0%	1.9%	10.3%	10.3%	3.1%	3.2%	5.8%	4.4%
Rural	24.4%	3.8%	4.5%	5.7%	7.4%	6.1%	9.9%	2.0%	10.5%	9.6%	3.0%	3.4%	5.3%	3.9%

Table 3.2: Own and cross price elasticities and expenditure elasticities

	Cereals	Beef	Pork	Poultry	Fish	Milk	Dairy	Eggs	Fruits	Vegetables	Potatoes	Oils	Sugar	Expenditure
Cereals	-0.840*	0.160*	0.059*	-0.036*	0.132*	-0.0286*	-0.162*	0.013*	0.090*	0.050*	-0.033*	-0.035*	-0.116*	0.904*
Beef	0.496*	-1.313*	-0.066*	0.334*	0.091*	0.060*	-0.875*	0.248*	-0.497*	0.448*	0.062*	-0.035*	0.083*	1.142*
Pork	0.252*	-0.091*	-0.735*	-0.262*	-1.048*	0.141*	0.571*	-0.348*	0.093*	0.168*	0.122*	0.287*	-0.035*	0.927*
Poultry	-0.130*	0.384*	-0.220*	-0.675*	-0.537*	0.183*	-0.129*	-0.135*	-0.185*	-0.183*	0.139*	-0.202*	0.210*	0.850*
Fish	0.263*	0.059*	-0.493*	-0.300*	-0.575*	-0.483*	0.127*	0.053*	0.746*	0.048*	-0.096*	-0.333*	0.195*	1.164*
Milk	-0.097*	0.066*	0.113*	0.175*	-0.823*	-0.296*	0.153*	-0.065*	-0.013*	-0.383*	0.023*	0.058*	0.133*	0.826*
Dairy	-0.340*	-0.592*	0.282*	-0.076*	0.134*	0.094*	-0.567*	-0.087*	-0.157*	0.193*	-0.192*	0.006*	-0.036*	1.090*
Eggs	0.120*	0.750*	-0.771*	-0.355*	0.247*	-0.180*	-0.389*	-0.735*	0.720*	-0.229*	-0.061*	-0.076*	0.287*	0.845*
Fruits	0.186*	-0.330*	0.045*	-0.107*	0.770*	-0.008*	-0.154*	0.158*	-1.188*	-0.188*	0.049*	-0.093*	-0.077*	1.018*
Vegetables	0.110**	0.319**	0.087**	-0.113**	0.053**	-0.249**	0.204**	-0.054**	-0.202**	-1.128**	0.043**	0.053**	-0.129**	1.030**
Potatoes	-0.223*	0.133*	0.192*	0.260*	-0.321*	0.059*	-0.611*	-0.044*	0.159*	0.131*	-0.368*	-0.120*	0.105*	0.902*
Oils	-0.169*	-0.055*	0.327*	-0.273*	-0.807*	0.083*	0.014*	-0.039*	-0.219*	0.116*	-0.087*	-0.224*	0.017*	0.943*
Sugar	-0.485*	0.112*	-0.034*	0.246*	0.408*	0.164*	-0.071*	0.128*	-0.157*	-0.245*	0.066*	0.014*	-0.989*	1.233*

*Statistically significant at the 1% level.

** Statistically significant at the 5% level.

3.3. Tax scenarios

A set of three tax scenarios is used to assess the distributional and welfare effects of a carbon tax on food. The reference scenario (REF) represents a tax on carbon emissions embodied in foods, based on a carbon price of €25 per tonne of carbon (CO_2 eq). This new tax is introduced into the model increasing accordingly the VAT[24] tax for each food product. This tax is within the range of carbon taxes levied recently in other countries and is also within the expected price range for the EU-ETS in the future. It is also within the range of the current estimations of social cost of carbon averaged over various studies as calculated by Tol (2005) and EPA (2013). Two additional scenarios are explored: (i) a high carbon price scenario (*HCT*) with a tax of €50 per tonne of carbon; and (ii) a high carbon price scenario with exemptions (ExeHT) on cereals, fruits, milk and vegetables. Exceptions for cereals, fruits and vegetables are introduced because are food products with low emissions factors. Moreover, all this products and milk and dairy products are essential in a healthy diet for kids.

The tax rate, t_i , for each item is defined as follows:

$$t_i = (E_i \ \times pe) \quad [3.10]$$

where E_i is the level of emissions of the i-good *(E_i)* and p_e is the carbon price per tonne. As a result t_i is a cost of carbon emissions charged on each foodstuff. These values are based on life cycle analysis estimates, i.e. emissions from farming, food processing, packaging, transportation and distribution to the point of final consumption are accounted for. The 14 foodstuff groups used in the study are made up of different sub-groups according to their factor emission data[25].

Table 3.3 shows the taxes charged on foodstuff groups in the three scenarios analysed. According to the emission factors used, meat and foodstuffs of animal origin have the highest tax rates. Beef is the foodstuff with the highest emission factor (25 tonnes of CO_2 per kg) and the highest tax rate. On the other hand cereals, fruit, fish and potatoes have the lowest taxation rates. Tax rates are based on the internalisation of the external costs of emissions, which depend on emission factors and total consumption.

[24] It is assumed that the tax passes fully to the consumer.

[25] Annex A shows the subgroups and their emission factors.

Table 3.3: Tax imposed per scenario and foodstuff

Scenario	Cereal	Beef	Pork	Poultry	Fish	Milk	Dairy	Eggs	Fruits	Vegetables	Potatoes	Oil	Sugar
REF	0.015	0.060	0.042	0.022	0.019	0.052	0.044	0.057	0.021	0.044	0.014	0.030	0.021
HCT	0.030	0.121	0.084	0.043	0.035	0.105	0.089	0.115	0.043	0.088	0.027	0.059	0.041
ExeHT	-	0.121	0.084	0.043	0.035	-	0.089	0.115	-	-	0.027	0.059	0.041

3.4. Results and discussion

This section discusses the results obtained for the three scenarios analysed. Subsection 3.4.1 explores the impacts of carbon related food taxation in terms of emissions reduction. Subsection 3.4.2 focuses on welfare effects and distributional implications by income, age and different social groups. Subsection 3.4.3 assesses progressivity and redistributional effects. Finally, subsection 3.4.4 analyses how new consumption patterns modify nutrient intakes.

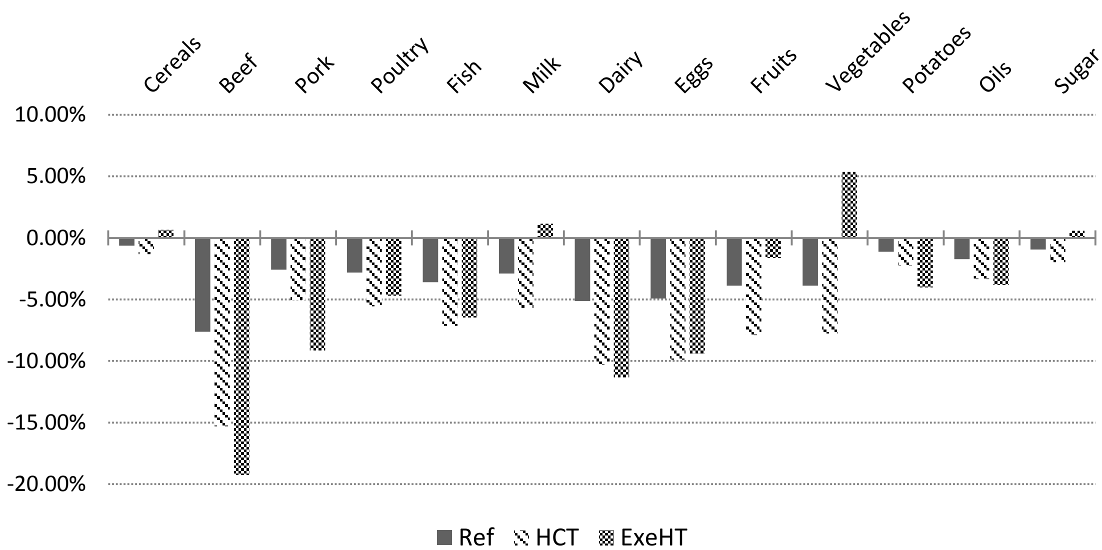
3.4.1 Emissions

The introduction of tax scenarios increases the price of food unevenly. This modifies consumption patterns and thus emissions. The change in consumption depends on elasticities estimated previously. These elasticities represent the consumption change when price level increases by 1%. Thus, the consumption change (CC_i) is represented as follows:

$$\mathrm{CC_i} = \sum_{\mathrm{i=1}}^{14} \mathrm{Tax_i} * \mathrm{elast_{ij}} \qquad [3.11]$$

The emissions and the consumption per food product fall by the same proportion because the emission factors are keep constant in the analysis. For instance, if the beef consumption decreases 1%, the emissions from beef also decrease 1%. Figure 3.1 shows emission impacts caused by changes in consumption of foodstuffs scenario by scenario. Note first that emission reductions are consistent with the tax imposed: the higher the emission factor per food product, the higher the tax and, thus, the greater the emission reduction. As in Edjabou and Smed 2013, it is also found that this tax could be effective in reducing emissions. A tax of €25 per tonne of CO_2 would reduce total emissions by 3.8%. One of the main factors explaining this effect is the striking reduction in beef consumption (7.5 % on the REF scenario). The REF and HCT scenarios show similar patterns, as in the HCT the taxation level has just been doubled. In the HCT scenario total emissions are reduced by 7.6% whereas REF only achieves 3.8%. Finally, due to exemptions in the ExeHT scenario, emissions on foodstuffs with higher emission factors, such as beef, are reduced even more (4.7%) whereas for low-emission foodstuffs such as vegetables there is an increase of 5.3%. Thus, a higher tax with exemptions may reduce meat consumption and at the same time increase the consumption of healthy, low-emission products due to substitution effects.

Figure 3.1: Emission reduction per foodstuff and measure.



3.4.2 Welfare and distributional effects

Welfare effects are reported using the well-known equivalent variation (EV) measure proposed by Hicks (1939). EV assumes that households reallocate expenditure as a result of change in prices. Given a vector of reference price $\mathrm{P_r}$, the equivalent expenditure (G_e) is defined as the expenditure level which allows households to achieve a reference level of utility, $\mathrm{v_r}(\mathrm{P}, \mathrm{G})$, where P and G, respectively, are the effective price and expenditure:

$$G_e = e\big(P_r, v_r(P, G)\big) \tag{3.12}$$

The equivalent variation[26] is then defined as the amount of money that households would be willing to pay to prevent the occurrence of the price change caused by the tax increase:

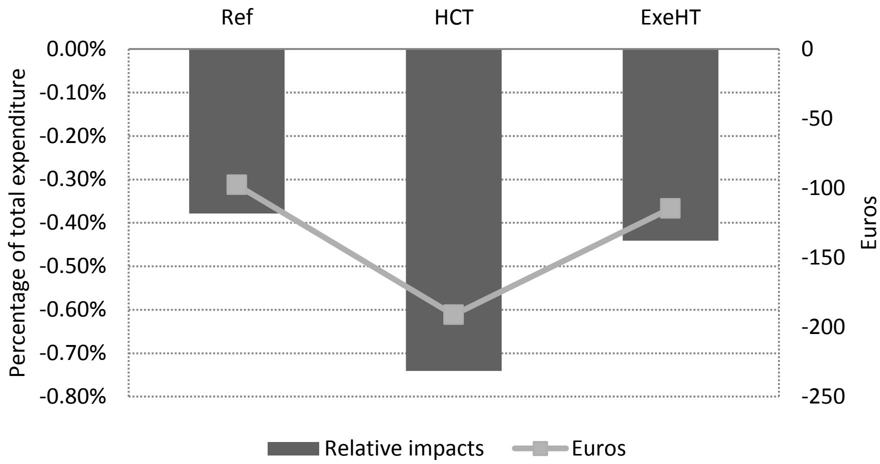
$$\mathrm{EV} = e(p^1, v^1) - e(p^0, v^1) \tag{3.13}$$

Where the after-tax price is computed as:

$$p_i^1 = (1 + \mathrm{t}_i)\, p_i^0 \tag{3.14}$$

[26] Annex B shows how Equivalent Variation (EV) is calculated.

Figure 3.2: Welfare impacts (average).



EV is reported in Figure 3.2 both in Euros and as a percentage of household expenditure. The overall effect is quite low because food products only represent around 12% of total household expenditure. The HCT scenario, as expected, has the greatest welfare loss at around 0.7% of total household expenditure (192€ per household) compared to 0.35% (97€ per household) for REF and 0.42% (115€ per household) for ExeHT. Although the ExeHT scenario has the same carbon price as HCT, the welfare impacts are similar to those obtained in the REF scenario, showing that exceptions can also be a cost-effective policy in promoting more healthy diets.

Figure 3.3: Welfare impacts by income group.

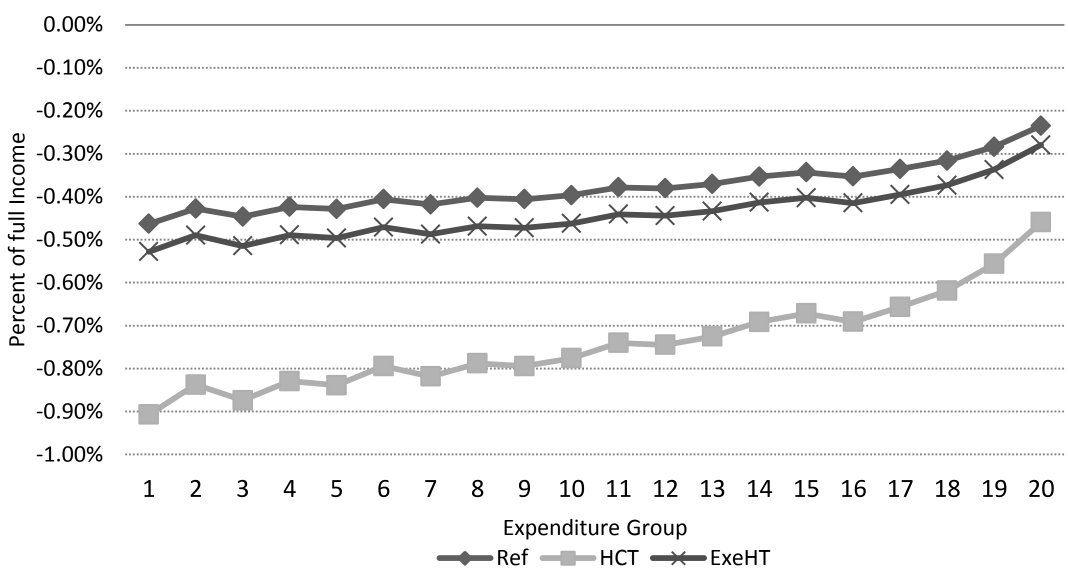
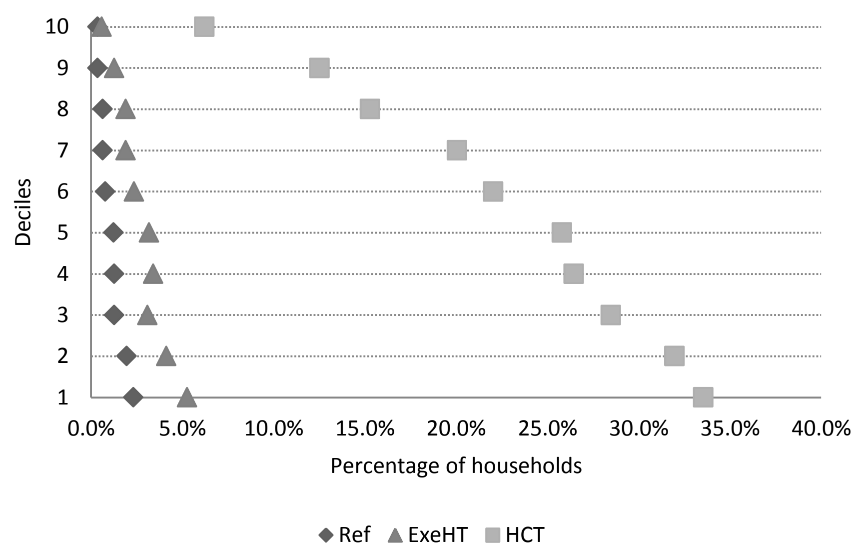


Figure 3.3 shows the welfare impacts by expenditure groups, where group 1 represents the lowest expenditure and 20 the highest. Figure 3.3 shows that carbon-based food taxation leads the poorest people to allocate a greater proportion of their expenditure than the rich. Diet patterns are quite similar across households, but the total average expenditure on food diverges across income groups. The lowest-expenditure households spend 20% of their income on food whereas the wealthiest households only spend around 10%. In line with the average impacts, HCT has the greatest welfare impacts while REF has the lowest for all expenditure groups. Although the carbon price is higher in ExeHT than in REF, ExeHT involves similar welfare impacts per tax scenario.

Figure 3.4: Percentage of households with loss greater than 1%, by income deciles.



To test for heterogeneity within income groups, Figure 3.4 reports the share of households where welfare loss is greater than 1% of annual expenditure per expenditure decile. In the three cases analysed the lowest-expenditure households are found to have the greatest number of households with higher losses. By contrast, few households in the highest expenditure groups have impacts in excess of 1%. The percentage of households with losses greater than 1% increases significantly from the fifth decile to the first. Lower expenditure households who consume a large proportion of meat products are the most affected by the tax reform. As indicated in Figure 3.4, there are more households with high welfare losses in the HCT scenario than in REF or ExeHT. The number of households with high losses is similar in ExeHT and REF because exemptions reduce welfare losses. It is interesting to note that exemptions reduce overall welfare impacts and can also reduce the number of households with higher losses.

Figure 3.5: Welfare impacts per type of household.

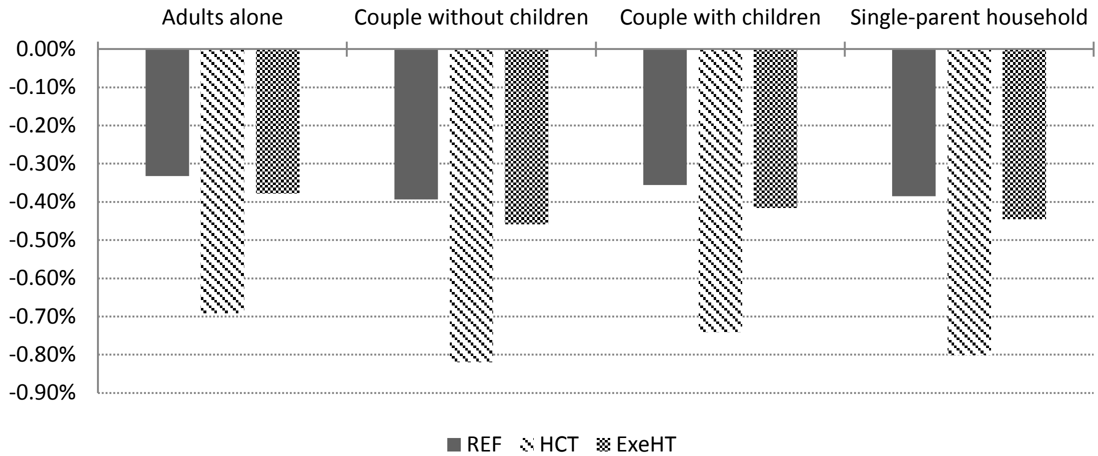


Figure 3.6: Welfare impacts by age of the breadwiner

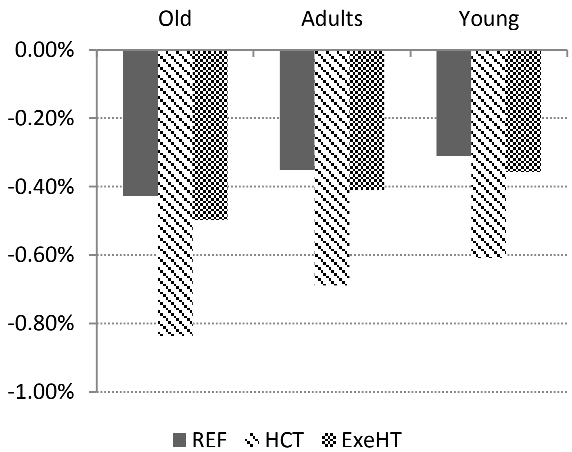


Figure 3.7: Welfare impacts by location and scenario

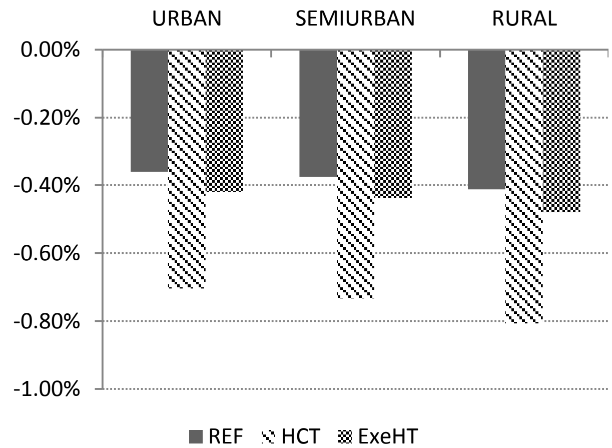


Figure 3.5 reports welfare impacts by social groups to check, for example, whether carbon-based food taxation could have counterproductive effects on households with children. Distinctions can be drawn between couples with children and single-parent households. Single parent households have a greater welfare loss than couples with children because they are normally households in the lower income range, for which the monetary loss represents a higher relative cost. On the other hand, the households where the welfare impact are lowest are those of adults who live alone. This group comprises young people who do not spend much of their income on food (only 10% of their expenditure is food whereas the average across all households is 12%). Given the fact that, according to the data, households have

similar dietary habits, the tax design does not change the welfare losses, so in all scenarios single parent families feel the greatest impact and adults who live alone feel the least.

Figures 3.6 and 3.7 show the welfare impacts by age of breadwinners and location of households. In line with the above results, the youngest households suffer the lowest welfare impacts (figure 3.6). Given the fact that old people spend a higher proportion of their income on food (14% of total expenditure) and consume more protein and climate-unfriendly foods, e.g. dairy products and fish, they have high welfare losses. On the other hand, the consumption patterns of old and rural households are very similar, so their welfare losses are also similar (Figure 3.7). In terms of location, the welfare impact is lowest for urban households, as they spend a lower proportion of their income on food, and highest for rural households.

3.4.3 Progressivity and redistributive effects

The Reynolds-Smolensky and Kakwani indices are useful for measuring the redistribution and regressivity of a tax system. The Reynolds-Smolensky index (1977), Π^{RS}, is used to evaluate the redistributive effects of the three tax scenarios. $L_{\_post}(p)$ is the Lorenz expenditure curve in the post-reform scenarios and $L_{\_pre}(p)$ stands for the Lorenz curve in the pre-reform scenario. As shown in equation 14, the Reynolds-Smolensky index captures the difference between the Gini indices of expenditure in the pre-reform and post-reform scenarios. Redistributive taxes yield $\Pi^{RS} > 0$.

$$\Pi^{RS} = 2\int_0^1 \left[L_{\_post}(p) - L_{\_pre}(p)\right]dp = G_{\_pre} - G_{\_post} \qquad [3.15]$$

The Kakwani index (Kakwani, 1977 a, b) are used to assess the progressivity[27] of three scenarios analysed. As shown below, the Kakwani index, Π^{K}, compares the concentration index of the tax, C_T , and the Gini index of expenditure in the pre-reform scenario, $G_{\_pre}$:

$$\Pi^{K} = 2\int_0^1 \left[L_{\_pre}(p) - L_T(p)\right]dp = C_T - G_{\_pre} \qquad [3.16]$$

where $L_T(p)$ stands for the Concentration tax curve. Tax on food is progressive, regressive or proportional when $\Pi^K > 0$, $\Pi^K < 0$ and $\Pi^K = 0$.

[27] A consumption tax is progressive if the average effective tax rate paid by households increases as the tax base (consumption) grows. Such a tax is regressive otherwise. The average effective tax rate is the ratio between taxes on consumption and income.

Table 3.4: Progressivity and Redistribution indices

Measures	Simulated scenarios		
	REF	HCT	ExeHT
Gini index of expenditure (G_pre)	0.2998	0.2998	0.2998
Gini index of expenditure (G_post)	0.3002	0.3006	0.3002
Concentration index of the Tax (Ct)	0.2001	0.1993	0.2124
Progressivity and Redistribution indices			
Reynolds-Smolensky index	-0.0004	-0.0007	-0.0004
Kakwani index	-0.0998	-0.1006	-0.0874

Table 3.4 shows the Reynolds-Smolensky and Kakwani indices. The Reynolds index values are negative but very close to zero in all scenarios, showing that the taxes on food analysed here have no redistributive effects. In addition, the Kakwani index is negative in all the cases analysed, so the tax system can be said to tend towards regressivity in all scenarios, i.e. they affect low income households more. Figure 3.8 depicts the distribution of average tax rates (ATR) for expenditure groups. As can be seen, the ATR decreases with income level, which confirms regressivity in all three scenarios. In other words, lower income households bear a larger share of taxes relative to their income. This regressivity is stronger in the HCT scenario, where the price per tonne of carbon is the highest. In addition, according to the Kakwani index, the ATR shows that a combination of taxes and exemptions yields better results in terms of distribution regressivity (ExeHT scenario). This is an expected result: as shown in Section 3.2, the exempted goods (cereals, milk, fruit, and vegetables) are consumed in greater proportions by lower income households.

3.4.4 Nutritional impacts

The change in food consumption also involves changes in the consumption of various important nutrients (Figure 3.9). Carbon-based food taxation leads to a notable reduction in cholesterol, saturated fat, sugar and protein intakes. This is consistent with the previous results that show lower consumption of meat and dairy products. A decrease in saturated fat and protein intakes is considered to be positive, because Spanish households currently consume excessive amounts of these nutrients (Lassaetta et al. 2013). Fig. 3.9 further shows that ExeHT also involves an increase in fibre intake due to increased consumption of fruit and vegetables. Thus, the ExeHT scenario is more in line with WHO recommendations (WHO 2015), which encourage increased fibre intake and reductions in saturated fats and animal proteins. This implies that carbon-based food taxation with well-designed tax exemptions can not only reduce emissions but also can make diets healthier.

Figure 3.8: Distribution of average tax rates (ATR).

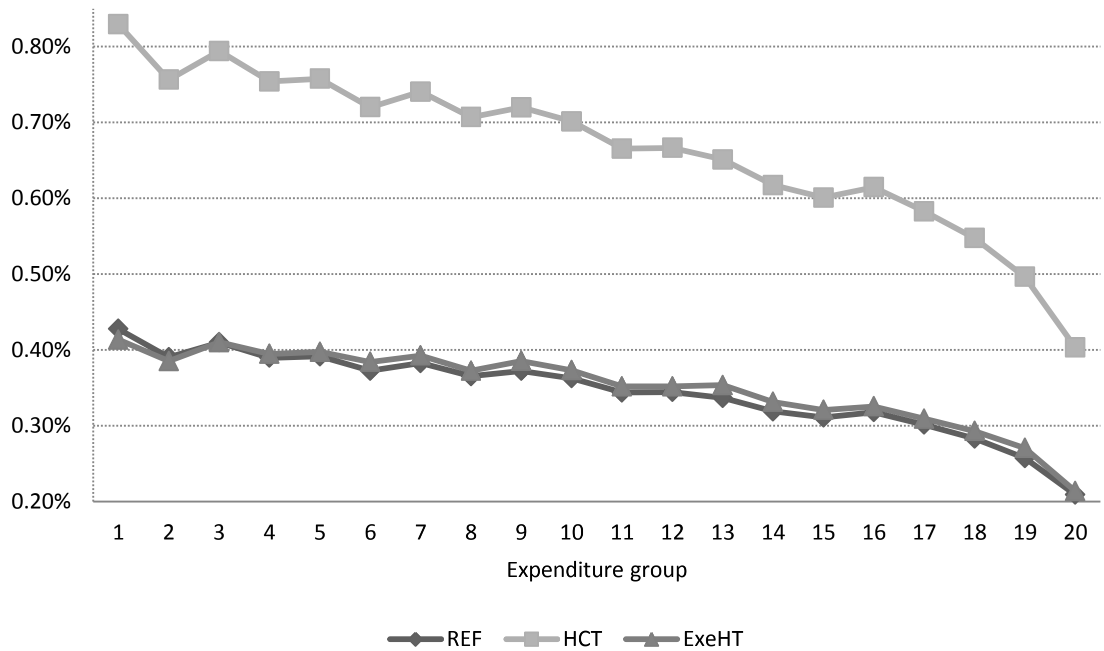
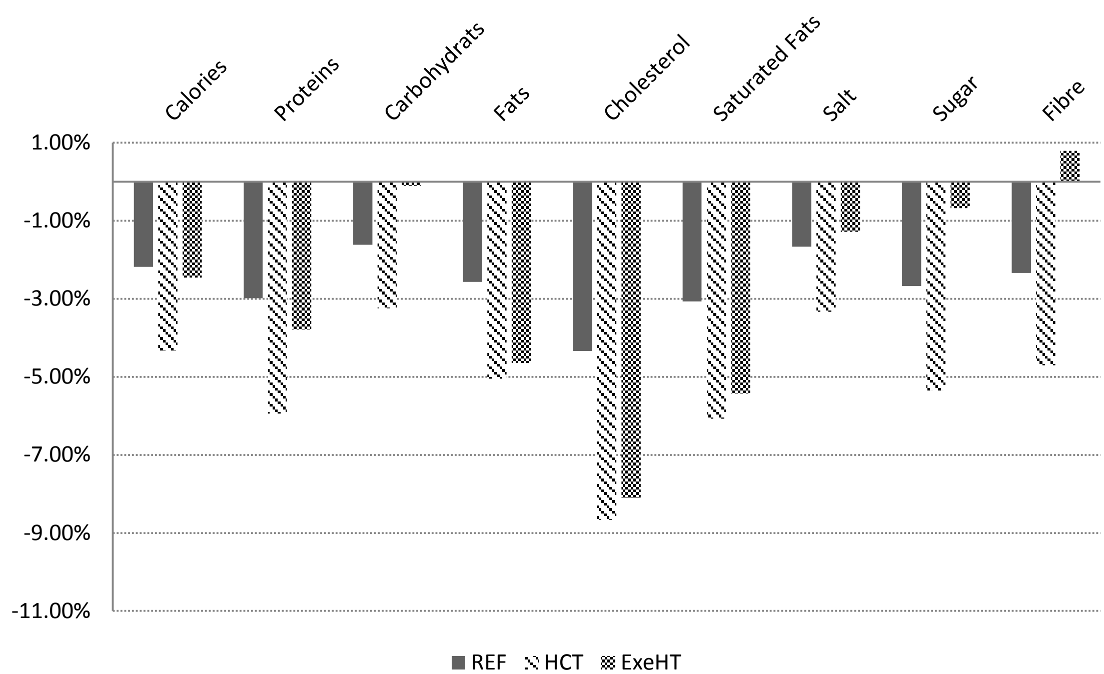


Figure 3.9: Nutrition impacts by macronutrient and scenario.



3.5. Conclusions

In line with other studies, this chapter finds that carbon-based food taxation can be an effective instrument in reducing food-based emissions. The results show that a high carbon tax (€50 per tonne of carbon, HCT scenario) can achieve a reduction of 7.6% in emissions from the food sector, whereas a low carbon tax (€25 per tonne of carbon, REF scenario) achieves a reduction of only 3.8%. However, setting a high carbon tax can also considerably increase welfare losses measured in terms of expenditure (from 0.35% in REF to 0.74% in HCT). Nevertheless, if exemptions are introduced for specific products, such as vegetables and fruit (scenario ExeHT), similar emission reductions can be achieved but with much lower welfare losses (0.42% in ExeHT). The explanation for this 'win-win' situation is that the exemptions force greater reductions in the consumption of the foodstuffs with the highest emission factors and, at the same time, maintain stability in the demand for food that would have had a greater negative impact on welfare, such as food associated with children's basic needs. The main problem may be found in households with low incomes but with other particular social characteristics, as is the case with single parent households. Given that they devote a greater proportion of their expenditure to food, single parent households have highest welfare losses of any of the social groups analysed. Single parent households are more likely to be poor: for example in Spain almost 50% of single parent households are at risk of poverty, so compensation policies need to be explored to correct this undesirable effect of food taxation. Other social groups especially affected by the tax are the old and rural households, because they have diets rich in meat, fish and animal products, which have greater emission factors. On the other hand, young, adults who live alone and urban households suffer the lowest welfare impacts. These households have in common a lower proportion of their income being spent on food. To put it in perspective, however, we need to bear in mind that losses of welfare for any group rarely exceed one percent of income, so any correction to welfare support policies will be minor.

The study also shows that carbon-based food taxation can improve diets, as it reduces the consumption of foodstuff such as meats and animal products, current levels of which are considered to be in excess of recommendations. Especially, the change towards a healthier diet can be observed when exemptions are introduced, as there is a notable reduction in the consumption of beef and an increase in the consumption of vegetables and cereals. The results obtained in terms of changes in nutrient intake also show an increase in the consumption of fibre and a reduction in energy and fat intakes in the ExeHT scenario. These factors are correlated with obesity and impose a high burden on the health systems. For example, in Spain studies highlight that mean cost per patient is 14% higher for the obese than for the non-obese (see Sicras-Mainar et al. 2008). Finally, in Spain, the average of calories intakes is around 2,600 kcal per day (Varela-Moreiras et al. 2013), while the World Health Organization recommends around 2,500 kcal per day. Thus, the moderate reduction on food consumption that it is obtained in the tax scenarios could also have some health benefits.

One novel result of this study is that carbon-based food taxation can be regressive. The largest regressive effects are obtained when a high carbon tax (HCT scenario) is set: the ATR of the lowest income groups is around 0.83% whereas it is only 0.40% in the highest expenditure groups. However, this effect can be substantially reduced if the tax is lower and if exemptions are introduced. In the case of a combination of a high carbon tax with exemptions (ExeHT scenario), the ATR of the lowest expenditure groups is reduced to 0.41% whereas that of the highest expenditure groups is reduced to 0.21%. The regressivity effects are confirmed by the Kakwani index, which shows that the food taxes analysed are regressive in all scenarios, while the Reynolds-Smolensky shows that the redistribution power of such taxes is zero. The regressivity is highest in the HCT scenario and lowest in REF. Given the fact that exempted goods (cereals, milk, fruit, and vegetables) are consumed in greater proportions by low-income households, the ExeHT scenario manages to reduce the regressivity impacts.

One of the main limitations of the study is that it analyses welfare impacts from a pure income perspective and do not account for the monetary health benefits associated with improving diets and how these changes will be distributed among different income or social group. Another limitation is that carbon-based food taxation policies are a challenge from an implementation point of view, although admittedly not more intricate than other health-based food taxes such as those on fats and sugar already implemented in some countries in recent years.

In summary, the results show that carbon-based food taxation can be an effective instrument for cutting emissions and improving dietary habits, especially if exemptions for specific products are implemented. The results also show that the distributional impacts of carbon-based food taxation tend to be slightly regressive, although the effects are very low and well-designed exemptions can ameliorate welfare impacts. Therefore, it does not seem that carbon-based food taxation should be disregarded for distributional reasons. However, further research is needed to elucidate whether these effects can be offset by the health benefits associated with the dietary changes in the lowest income groups.

Chapter 4

The Efficiency Cost of Protective Measures in Climate Policy. A Computable General Equilibrium Analysis for the United States

4.1. Introduction

The 21st Conference of Parties (COP21) to the United Nations Framework Convention on Climate Change in Paris in December 2015 set an important milestone in international climate policy. The so-called Paris Agreement (UNFCCC, 2015) achieved global consensus on keeping the global mean surface temperature increase below 2 degrees Celsius compared to pre-industrial levels. In line with this temperature target not only industrialized countries but also developing countries signalled their willingness to reduce greenhouse gas (GHG) emissions. According to the Paris Agreement, future climate negotiations and emission reduction efforts should be planned in global coordination; however, opposite to the Kyoto Protocol with its legally binding reduction targets for signatory industrialized countries, the Paris Agreement builds only on voluntary pledges of individual countries - the so-called intended nationally determined contributions (INDCs) - to reduce GHG emissions.

Under the Paris Agreement, the United States of America (US) has committed itself to cut domestic emissions by 26% - 28% by 2025 as compared to 2005 emission levels. One contentious issue in domestic US climate policy is the threat of competitiveness losses for US emission-intensive and trade-exposed (EITE) industries if facing more stringent regulation than competitors abroad.

Reflecting such competitiveness concerns, the present study investigates the economic impacts of four alternative protective measures for US EITE industries: (i) output-based rebates, (ii) exemptions from emission pricing, (iii) energy intensity standards (instead of emission pricing), and (iv) carbon intensity standards (instead of emission pricing). Based on simulations with a large-scale computable general equilibrium model (CGE) for the global economy we quantify how these protective measures affect competitiveness of US EITE industries for alternative degrees of climate policy stringency in other OECD countries. We find that while protective measures can substantially attenuate adverse competitiveness impacts, they run the risk of making US climate policy much more costly than uniform emission pricing stand-alone. In fact, the cost increase is associated with negative income effects such that the gains of protective measures for EITE exports may be more than compensated through losses in domestic EITE demand.

The remainder of this chapter is organised as follows. Section 4.2 summarizes the literature on climate policy design in the context of competitiveness concerns. Section 4.3 adopts a simple analytical framework to investigate the competitiveness impacts of alternative protective measures. Section 4.4 provides a description of the CGE model and data underlying our quantitative analysis, presents the policy scenarios, and discusses the simulation results. Section 4.5 concludes.

4.2. Literature review

Concerns on adverse competitiveness effects of asymmetric emission pricing are at the fore of the climate policy debate in many industrialized countries. Energy-intensive and trade-exposed (EITE) industries in countries with stringent emission regulation fear shifts in competitive advantage in favour of other international producers (which could occur under certain conditions). Cost disadvantages would incentivize the relocation of EITE production from domestic sites to abroad thereby amplifying adverse domestic production and employment effects for these industries. In this context, opponents to unilateral emission pricing also point to the risk of counterproductive emission leakage – i.e. the partial offsetting of domestic emission reduction through increases of emissions abroad.

To avoid excessive (and potentially inefficient) structural change against domestic EITE industries, various protective measures for EITE industries which are at risk of carbon leakage are discussed. Principal among these measures are border carbon adjustment, where emissions embodied in imports from non-regulating regions are taxed at the emission price of the regulating region (i.e. "taxing products at the border on their carbon content") and emission payments for exports to non-regulating countries are rebated. From a global efficiency perspective such a combination qualifies as a second-best measure complementing (unilateral) uniform emission pricing (Markusen, 1975; Hoel, 1991; Copeland, 1996). However, border carbon adjustments are quite controversial from the perspective of international trade agreements and their political feasibility (Cendra, 2006; Ismer and Neuhoff, 2007). When border measures are unavailable, differential emission pricing in favour of domestic EITE industries including full exemptions may serve as an alternative protective measure (Hoel, 1996; Böhringer et al., 2014a). Another important strategy for protecting EITE industries involves the allocation of free emission allowances conditional on production (i.e. output-based allocation Fischer, 2001). Contrary to auctioning of emission allowances or unconditional free allowance allocation, an output-based grandfathering system effectively works as a subsidy to production to recover (part) of losses in comparative advantage (Böhringer et al., 1998). A further potential candidate for protection of EITE industries are intensity standards. Instead of being subjected to emission pricing, EITE industries could adopt intensity standards to reduce their emissions as compared to business-as-usual levels. Holland (2012) shows that emission pricing via an emission tax or an emission cap-and-trade system may be an inferior instrument to standards if accounting for emission leakage.

As protective measures for EITE industries are predominantly discussed in the context of competitiveness, there is a need for concepts on the definition and measurement of competitiveness at the sector level. The economic literature provides a broad variety of competitiveness concepts (Oberndorfer and Rennings, 2007; Alexeeva-Talebi and Böhringer, 2012). Among indicators to quantify sector-specific competitiveness effects most common are metrics to measure international trade performance such as relative world trade shares (RWS – see e.g. Balassa, 1962; Ballance et al., 1987; Gorton et al., 2000; Fertö and Hubbard, 2003; Abidin and Loke, 2008) or revealed comparative advantage (RCA – see e.g. Kravis and Lipsey, 1992; Carlin et al., 2001).

The economic impacts of protective measures for EITE industries in unilateral climate policy design have been quantified by numerous simulation studies predominantly based on multi-sectoral multi-regional CGE analysis. The bulk of these studies investigates border carbon adjustments (e.g., Babiker and Rutherford, 2005; Mattoo et al., 2009; McKibben and Wilcoxen, 2009; Dissou and Eyland, 2011; Winchester et al., 2010; Böhringer et al., 2010) and report impacts on EITE industries in terms of change in production output. The general finding is that border carbon adjustment attenuate negative output effects for EITE industries in unilaterally regulated countries (see Böhringer et al., 2012a for a meta-analysis), while, providing only limited gains in global cost-effectiveness of unilateral action and enhancing negative terms-of-trade spillover effects to countries without emission regulation. Output-based allocation or preferential emission pricing for EITE sectors can also help to dampen adverse output effects (Fischer and Fox, 2012) significantly. To date, there are only a few studies which cross-compare alternative protective measures: Böhringer et al. (2014b) show that – as the coalition of unilaterally abating countries increases – border carbon adjustments are consistently more effective than output-based rebates in mitigating relocation of EITE output; Böhringer et al. (2012b) extend the comparison to include also tax exemptions for EITE industries and find that the negative repercussions on domestic EITE production can be strongly reduced for border carbon adjustments whereas tax exemptions and output-based rebates can only achieve a fraction of this alleviation.

This chapter sheds further light on the relative performance of alternative policy measures to protect competitiveness of EITE industries. In our cross-comparison, we deliberately drop border carbon adjustments since their appeal for practical climate policy is limited given international trade law; instead, we include standards on emissions or energy as a potentially attractive measure beyond output-based rebates or tax exemptions. Furthermore, we quantify sector-specific impacts not only in terms of output changes but also adopt more common metrics for competitiveness such as RWS and RCA. Our simulation analysis for US climate policy design provides insights on how protective measures for EITE industries trade-off with other policy objective such as minimizing economy-wide adjustment cost to national GHG emission targets.

4.3. Stylized theoretical analysis

We adopt a simple partial equilibrium setting (see Böhringer et al., 2014b) to show that protective measures improve competitiveness of domestic industries in international trade (as compared to uniform emission pricing stand-alone). While our stylized theoretical analysis illustrates a fundamental cause-effect chain, it neglects potentially important market interaction and income effects and thus must be complemented with more comprehensive computable general equilibrium analysis as provided in section 4 to draw viable policy conclusions.

Consider two countries (regions) which differ only with respect to potential regulatory action: country M with emission regulation and country N without emission regulation. Demand q_{ik} in country *i* for the good produced in country *k* exhibits constant elasticities with respect to prices. We measure competitiveness as the ratio of exports over imports in the regulated region M where export demand and import supply can be stated as:

$$Exports\ of\ \ region\ M{:}\ \ q_{NM} = a p_{NM}^{-\eta_o} p_{NN}^{\eta_x} \qquad [4.1]$$

$$Imports\ of\ region\ \ M{:}\ \ q_{MN} = a p_{MN}^{-\eta_o} p_{MM}^{\eta_x} \qquad [4.2]$$

with:

a denoting benchmark quantities (as initial prices are normalized to unity),

η_o referring to the own-price elasticity, and

η_x referring to the cross-price elasticity.

As both economies are symmetric, a competitiveness loss will occur when a policy regulation involves lower exports than imports. We thus measure competitiveness φ as the ratio of exports over imports in the regulated country:

$$\varphi = {}^{q_{NM}}/_{q_{MN}} \tag{4.3}$$

We assume competitive markets[28], so prices equal marginal costs plus potential taxes. The emission intensity in country *i* is denoted by μ_i. Marginal production cost $c(\mu)$ is constant with respect to output and increasing as the intensity of emissions μ decreases (i.e. $c'<0$). Let $\mu(t)$ denote the cost-minimising emission intensity at emission tax t. Furthermore, given any positive carbon price, $t>0$, producers decrease their emission intensity to lower compliance costs, so $1 + t\mu_0 > c\big(\mu(t)\big) + t\mu(t)$.

In the benchmark without emission regulation $t=0$, with $\mu_0=\mu(0)$ indicating the initial emissions intensity and normalising $p_0=c(\mu_0)=1$; obviously, benchmark competitiveness $\varphi = 1$.

When an emission tax ($t>0$) is set (subscript *T*), the regulated country adjusts emission intensity and prices are equal to marginal costs plus taxes. Thus, $p_{MM}=p_{NM}=c_T+t\mu_T$, where $c_T=c(\mu_T)$ and $\mu_T=\mu(t)$. Exports and imports in the regulated country are given by:

$$q_{NM} = a(c_T + t\mu_T)^{-\eta_o} \tag{4.4}$$

$$q_{MN} = a(c_T + t\mu_T)^{\eta_x} \tag{4.5}$$

Compared to a situation without emission regulation, exports in the unilaterally regulated country decrease while imports increase:

$$q_{NM} = a(c_T + t\mu_T)^{-\eta_o} < 1 \tag{4.6}$$

$$q_{MN} = a(c_T + t\mu_T)^{\eta_x} > 1 \tag{4.7}$$

Competitiveness for the region with a unilateral emission tax will decrease:

$$\varphi_T = (c_T + t\mu_T)^{-\eta_o - \eta_x} < 1 \tag{4.8}$$

[28] Following Böhringer et al. (2008), imperfect competition can amplify structural change of emission regulation at the expense of EITE industries due to changes in economies of scale. Thus, protective measures for EITE sectors may become even more relevant under imperfect competition than under perfect competition.

We now consider protective measures to restore at least partially competitiveness in the regulated country. In our simple partial equilibrium setting with one commodity produced by each region, it is trivial to see that tax exemptions restores competitiveness – in the extreme case of a full exemption, we are back to the benchmark situation. More interesting is the case of output-based rebates or intensity standards. Output-based rebating suppresses the cost increase for domestic producers, so that the playing field does not tilt toward imports or competitors in export markets. Specifically, a rebate is offered to domestic producers in proportion to their production, based on a benchmark that we assume is equal to the average emissions intensity of the sector, multiplied by the emissions tax. As this allocation is updated according to production, the rebate works de facto as a per-unit subsidy $t\mu_T$. The producer price in the regulated country then no longer includes the cost of the remaining embodied emissions, but the emissions intensities (and corresponding production costs) respond to the emission tax so the production cost with output-based rebating equal the production cost for the case of emission taxing stand-alone. Meanwhile in the non-regulated country, $p_{MN}=p_{NN}=c_0=1$.

Holland et al. (2009) show analytically that intensity standards work as implicit emission taxes on the input side where the fictitious tax revenues are recycled as implicit subsidies on the output side. If we set the implicit tax for the case of standards equal to the exogenous emission tax t, the effects of intensity standards and output-based rebating are identical in our simple model framework where we do not have multiple products that differ in emission intensity.[29]

We then can derive exports and imports in the regulated country for the case of protective rebates or standards (subscript RS) as:

$$q_{NM} = a(c_{RS})^{-\eta_0} < 1 \qquad [4.9]$$

$$q_{MN} = a(c_{rS})^{+\eta_x} > 1 \qquad [4.10]$$

and competitiveness as:

$$\varphi_T = (c_{RS})^{-\eta_0-\eta_x} < 1 \qquad [4.11]$$

With the exogenous emission tax t, $c_T = c_{RS}$ and $\mu_T = \mu_{RS}$ so we can readily compare the competitiveness performance of protective measures against our reference case of an emission tax:

[29] With multiple products that differ in emission intensity, output-based rebating is no longer equivalent with intensity standards even if the latter are made tradable since effective output subsidies across sectors in general differ.

$$\varphi_{RS}/\varphi_T = \frac{(c_T + t\mu_T)^{\eta_O+\eta_x}}{(c_T)^{\eta_O+\eta_x}} > 1 \qquad [4.12]$$

Hence, we can see that output-based rebates like intensity standards attenuate the competitiveness losses as compared to emission taxing stand-alone. Moreover, when the intensity standard is equal to the intensity achieved with output-based rebate, the competitiveness changes should be equal in both measures. As we will see in subsection 4.4.6, the results in the CGE are in the same direction than the expected in this analysis.

4.4. Computable general equilibrium analysis

The stylized theoretical analysis provides qualitative insights into the competitiveness effects of different measures for protecting EITE industries under unilateral emission regulation. For an empirical quantitative assessment it is however imperative to account for real-world complexities that are no longer tractable in theoretical analysis. Economic adjustment to emission regulation is driven through complex substitution, output and income effects across multiple markets following changes in relative prices. In this context, computable general equilibrium (CGE) models represent an important complement to theoretical analysis of policy regulation since they allow researchers to conduct counterfactual experiments that are grounded in microeconomic theory and have quantitative content based on empirical data. We therefore undertake numerical simulations with a large-scale CGE model of global trade to quantify the economic impacts of US climate policy design where alternative protective measures for EITE industries are still under debate. We first provide a non-technical summary of the CGE model and describe the data sources used for parameterization. Next, we lay out the criteria for industries to qualify as EITE sectors and recall the definitions of sector-specific competitiveness indicators for international trade performance. We then follow up with a characterization of counterfactual climate policy scenarios and discuss simulation results.

4.4.1 Non-technical model summary

We use a multi-region, multi-sector CGE model of global trade and energy use destined for the impact assessment of climate policies (see Böhringer and Rutherford, 2010 or Böhringer et al., 2014b for recent applications and the detailed algebraic formulation of the core model).

Production of commodities except fossil fuels is captured by constant elasticity of substitution (CES) cost functions describing the price-dependent use of capital, labour, energy, and material in production (see Figure C1). At the top level, a CES composite of material trades off with an aggregate of energy, capital, and labour at a constant elasticity of substitution. At the second level, a CES function describes the substitution possibilities between intermediate demand for the energy composite and a value-added aggregate of labor and capital. At the third level, the value-added composite is formed as a CES function of labour and capital while the energy composite is formed as a CES function of different primary and secondary energy inputs (coal, gas, refined oil, electricity). Production of fossil fuels (coal, gas, crude oil) is characterized by a single-level CES function where the fossil-fuel resource trades off with a Leontief composite of all other inputs (see Figure C2.).

Final consumption demand in each region is determined by the representative household who maximizes utility subject to a budget constraint with fixed investment and exogenous government provision of public goods and services. The household's total income consists of tax revenues and net factor income from primary factors labour, capital and fossil-fuel resources. Final consumption demand is given as a CES aggregate of composite non-energy consumption and composite energy consumption. Both – the non-energy consumption composite and the energy consumption composite – are in itself CES functions of disaggregate non-energy and energy commodities (see Figure C1).

Labor and capital are mobile across sectors within a region but immobile between regions. Fossil-fuel resources are tied to the respective resource production sectors.

Bilateral trade follows the Armington (1969) approach of product heterogeneity where domestic and foreign goods are distinguished by origin. A balance of payment constraint incorporates the base-year trade deficit or surplus for each region. All goods used on the domestic market in intermediate and final demand correspond to a CES (Armington) composite that combines the domestically produced good and the imported good from other regions (see Figure C3).

CO_2 emissions are linked in fixed proportions to the use of fossil fuels, with CO_2 coefficients differentiated by the specific carbon content of fuels. Restrictions to the use of CO_2 emissions in production and consumption are implemented through exogenous emission constraints or (equivalently) CO_2 taxes. CO_2 emission abatement then takes place by fuel switching (inter-fuel substitution) or energy savings (either by fuel-nonfuel substitution or by a scale reduction of production and final demand activities).

4.4.2 Data

As is customary in CGE analysis, base-year data and exogenous elasticities determine the free parameters of the model's functional forms that characterize production technologies and consumer preferences. The base-year data together with exogenous elasticity values calibrate the functional forms such that the Global Trade Analysis Project (GTAP) dataset is consistent with market structure assumptions and optimizing behavior of economic agents. For the calibration we use the most recent data from the GTAP which features detailed accounts of regional production and consumption, bilateral trade flows, energy flows, and CO_2 emissions for up to 140 regions and 57 sectors in the base-year 2011 (Narayanan et al. 2015). Elasticities in international trade and in sector-specific value-added are included in the GTAP database. Interfuel substitution elasticities are based on Narayanan and Steinbuks (2014). The elasticities of substitution in fossil fuel sectors are calibrated to match exogenous estimates of fossil fuel supply elasticities (Graham et al., 1999; Krichene, 2002).

The GTAP database is aggregated toward a composite dataset that accounts for the specific regional and sectoral requirements of our analysis. On the regional dimension, we have depicted important geopolitical players and major trading partners of the US to reflect concerns about competitiveness losses induced by stringent US climate policy regulations. On the sectoral dimension, the composite dataset identifies five primary and secondary energy goods to track differences in CO_2 intensity and the degree of substitutability. We furthermore include all GTAP sectors explicitly that qualify as energy-intensive and trade-exposed (EITE) industries according to the criteria as laid out in section 4.4.3. All remaining sectors are condensed in a composite of “other manufactures and services”. Table 4.2 lists all sectors and regions included in the model (for the sectors we include acronyms in brackets with are used in Figure 4.1 of section 4.4.3.).

4.4.3 Qualification criteria for EITE sectors

For the selection of EITE sectors we adopt the criteria put forward by the EU in the definition of industries at risk of carbon leakage which in turn serves as a proxy for the threat of international competitiveness losses. The EU ETS Directive, Article 10a (EU, 2003) defines that a sector or sub-sector is deemed to be exposed to a significant risk of carbon leakage using two metrics: trade intensity (T) and the additional cost (A) induced by emission regulation. These metrics are formally defined at the sector level as:

$$\text{Trade intensity } (T) = \frac{(X+V)}{(Y+V)} \qquad [4.13]$$

$$\text{Additional Cost } (A) = \frac{(c * e + d) * \varepsilon}{\omega} \qquad [4.14]$$

where:

X, *V*, *Y*	denote exports, imports and output,
c	are the direct emissions,
d	are the indirect emissions of CO_2 from electricity consumption,
e	is the share of emissions that are auctioned,
ε	is the expected carbon price in 2020, and
ω	denotes the gross value added at factor costs.[30]

Article 10a of the ETS Directive classifies a sector to be exposed to a significant risk of carbon leakage if at least one of the three criteria combinations listed in Table 4.1 is met.

Table 4.1: Criteria to qualify as sector at significant risk of carbon leakage (EU 2003)

	Additional costs (A)	Trade intensity (T)
Criteria combination #1	>5%	>10%
Criteria combination #2	>30%	-
Criteria combination #3	-	>30%

As we apply these criteria to the GTAP dataset, the sectors which qualify only contribute 13% to overall gross value-added. To correct for the fact that GTAP only features a relative broad and highly aggregated sector classification (and hence that it may hide some subsectors with high trade intensity or additional costs which we find worth considering) and for the different context of the US compared to the EU (the trade as % of GDP[31] measure of the WB, 2016 was 31% for the US in 2011 and 81% in 2011 for the EU), we have lowered the threshold to the third criteria from 30% to 10%. Additional sectors that meet these relaxed criteria are attributed towards two composite sectors, i.e. trade-intensive agricultural goods (TIA) and trade-intensive manufactured goods (TIM), increasing the share of all EITE sectors in economy-wide value-added to roughly 22%. Figure 4.1 provides a scatter plot in trade intensity (%) and additional cost (%) for the selected EITE sectors.

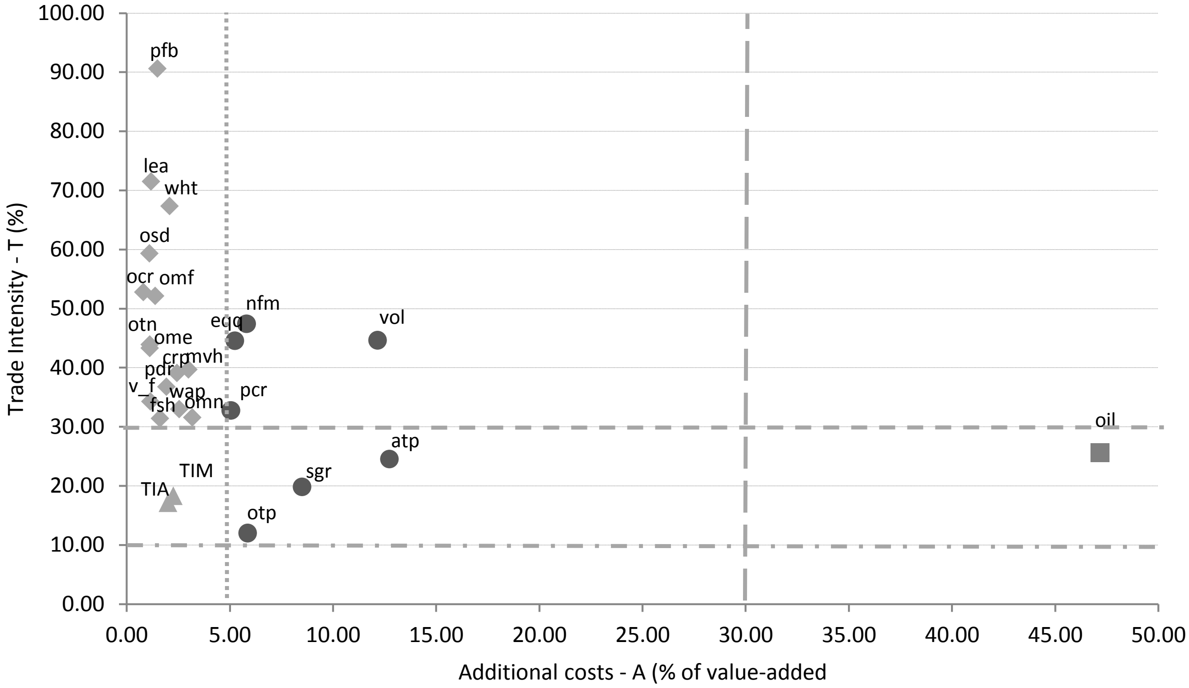
[30] Values for *e* and ε were taken from De Bruyn et al (2013). All other parameters are assigned based on GTAP data.
[31] Trade is the sum of exports and imports of goods and services measured as a share of gross domestic product.

Table 4.2: Model sectors and regions

Sectors and commodities	Regions
Primary and secondary energy goods:	United States of America
Coal (COL)	
Crude oil (CRU)	*Other OECD regions:*
Natural gas (GAS)	EU-28+ EFTA
Refined oil products (OIL)*	Japan
Electricity (ELE)	Canada (CAN)
	Other OECD countries (ROE)
Energy-intensive and trade-exposed (EITE) industries:*	
Air transport (ATP)	*Other geopolitical players:*
Chemical rubber plastic products (CRP)	China (CHN)
Electronic equipment (EEQ)	Russian Federation (RUS)
Fishing (FSH)	Organization of the Petroleum Exporting Countries (OPEC)
Leather products (LEA)	Rest of the World (ROW)
Motor vehicles and parts (MVH)	
Metals (NFM)	
Crops (OCR)	
Machinery and equipment (OME)	
Manufactures (OMF)	
Minerals (OMN)	
Oil seeds (OSD)	
Transport equipment (OTN)	
Transport (OTP)	
Processed rice (PCR)	
Paddy rice (PDR)	
Plant based fibers (PFB)	
Other manufactures and services (ROI)	
Sugar (SGR)	
Vegetable oils and fats (VOL)	
Vegetables fruit nuts (V_F)	
Wearing apparel (WAP)	
Wheat (WHT)	
Trade-intensive agricultural products (TIA)	
Trade-intensive manufactured products (TIM)	
Remaining industries and services:	
Other manufactures and services (ROI)	

*Refined oil products are included in EITE

Figure 4.1: EITE Sectors in the US – trade intensity (%) and additional costs (% of value added)



4.4.4 Sector-specific competitiveness indicators

In this section competitiveness is measured as making use of the data of exports and imports, being good sectoral competitiveness indicators, although not necessarily as good to reveal economy-wide competitiveness (since trade may represent only a fraction of GDP, in particular, it represented a 31% for the US in the year 2011). In order to assess the competitiveness effects on EITE sectors induced by emission regulation then we firstly draw on two widely used competitiveness indicators: relative world trade shares (RWS) and revealed comparative advantage (RCA). These indicators are defined as follows:

$$RCA_{ir} = \frac{X_{ir}/V_{ir}}{\sum_i X_{ir}/\sum_i V_{ir}} \qquad [4.15]$$

$$RWS_{ir} = \frac{X_{ir}/\sum_r X_{ir}}{\sum_i X_{ir}/\sum_i \sum_r X_{ir}} \qquad [4.16]$$

where:

X_{ir} denotes the exports of sector *i* in region *r*, and

V_{ir} denotes the imports of sector *i* in region *r*.

The relative world trade shares (RWS) indicator (which in some other works can be found as Index of Revealed Comparative Advantage or Balasa Index) thus compares the export share of a given commodity or sector in a country with the export share of that commodity (or sector) in the world market. If the index exceeds 1, it reveals a comparative advantage of the trade in the focal product (or sector), and the opposite if it does not exceed 1. If the sectoral export-import ratio is identical to the economy-wide ratio, the RWS index takes the value of one ($RCA_{ir} = 1$).

For a particular region and sector, the RCA index compares the ratio of exports by a specific sector to its imports with the ratio of exports to imports across all sectors of the region. The RCA indicator ranges from $0 \leq RCA_{ir} \leq \infty$ and can be interpreted regarding the range for comparative (dis-)advantage similarly to the RWS indicator.

4.4.5 Policy scenarios

Our Our research interest is in the economic impact assessment of protective measures for EITE industries as a potentially important element of US climate policy design. We distinguish four different protective measures: (i) output-based rebates (*obr*), (ii) exemptions from emission pricing (*exe*), (iii) energy intensity standards instead of emission pricing (*eis*), and (iv) carbon intensity standards instead of emission pricing (*cis*)[32]. Table 4.3 summarizes the characteristics across the five climate policy designs underlying our simulation analysis. The reference policy (*ref*) without any protective measures for EITE industries involves a uniform emission pricing across all segments of the US economy. The protective measures are implemented as follows. In scenario *obr*, US EITE industries pay the same CO_2 emission price on fossil fuel inputs as all other segments of the US economy – however, the emission payments by EITE industries are recycled as an output subsidy (rather than being handed back lump-sum to the representative US household). In scenario *exe*, US EITE industries are fully exempted from emission payments. In the scenarios with energy intensity standards (*eis*) or carbon intensity standards (*cis*), EITE sectors do not face explicit emission taxes but get imposed standards for energy or CO_2 emissions that reflect the energy/emission intensity emerging from the reference policy. Across all scenarios, the CO_2 emissions for the US are reduced to the same level in order to accommodate a coherent cost-

[32] Border carbon adjustment has been also explored in the literature as way to protect EITE sectors. It could be expect, that BCA performs better than the policy analyzed because the abatement efforts are shifted to other economies and not to the remaining segments of the US economy. However, after Paris the use of BCA is quite unlikely, and under BCA, the export rebates policy may constitute a subsidy under the WTO's Agreement on Subsidies and Countervailing Measures. Thus from a policy feasible point of view we have skipped BCA from our analysis.

effectiveness analysis: The CO_2 emission price paid by eligible segments of the US economy adjusts endogenously to achieve the exogenous economy-wide US reduction target.

Table 4.3: Summary of policy scenarios (scenario acronyms in parenthesis)

Reference scenario	**Scenarios with protective measures for EITE industries**
Uniform CO_2 emission pricing across all segments of the economy (*ref*)	Output-based rebates (*obr*) Full exemption (*exe*) Energy intensity standard (*eis*) Carbon intensity standard (*cis*)

In our central case simulation, we assume that the US is committed to a CO_2 emission reduction of 30% compared to business-as-usual emission level. This emission reduction target is roughly in line with the voluntary pledges to reduce GHG emissions submitted by the US under the Paris Agreement. In the reference scenario (*ref*) the emission reduction target is achieved through uniform CO_2 emission pricing across all segments of the economy – an economy-wide CO_2 emission tax is set sufficiently high to achieve a 30% reduction in domestic emissions.[33] When we replace emission pricing in EITE sectors by intensity targets, we set EITE intensity standards at the level achieved in the *ref* scenario while CO_2 emission taxes for all other segments of the US economy are adjusted endogenously to warrant the economy-wide US emission reduction of 30%.[34]

Despite the Paris Agreement on the need of globally coordinated action to achieve the 2°C temperature target, a common global emission price is unlikely in the foreseeable future. Signatory countries intend to contribute with quite different types of pledges.[35] Yet, it is unclear how these voluntary pledges will materialize in effective emission pricing. When accounting for different reference years (1990, 2005, 2030) and metrics (absolute emission caps versus emission intensity targets), our assumption for the medium term remains that industrialized countries lead the way with more stringent emission controls while developing countries refrain from rigorous measures to curb GHG emissions. In our central case simulations, we reflect such cross-country asymmetries in climate action by assuming that non-OECD countries have negligible emission prices while emission pricing in OECD countries is significant. To

[33] Revenues from taxing emissions are recycled lump-sum to the representative US household.
[34] In our central case simulations, we do not impose a global emission constraint to keep global emission for all US climate policy scenarios constant (for some emission price in other OECD countries). The interest of our simulation analysis is on how alternative protective measures in US climate policy design affect the performance of US EITE industries and domestic US welfare rather than investigating global cost-effectiveness.
[35] http://www4.unfccc.int/submissions/indc/Submission%20Pages/submissions.aspx

reflect uncertainties on the level of OECD emission prices which will affect international competitiveness of US EITE industries our central case simulations investigate a price interval of OECD emission taxes outside the US ranging from \$US 0 per ton to \$US 70 per ton of CO_2 in discrete steps of \$US 10.[36] Our sensitivity analysis (see section 4.7) furthermore investigates the robustness of our policy insights when we extend emission pricing to China and the rest of the world (ROW).

4.4.6 Simulation results

In the exposition of simulation results we quantify the effects of alternative US climate policy designs in percentage change from the business-as-usual (*BaU*) without climate policy. We start our results discussion with policy-induced changes in international competitiveness of US EITE industries.

Figure 4.2 visualizes competitiveness effects for US EITE sectors measured in terms of wide-spread indicators RCA and RWS. For both metrics, protective measures help to attenuate adverse trade performance of EITE sectors compared to the *ref* scenario. As expected, the international cost disadvantage for US EITE industries and thus the argument in favour of protective EITE measures are weakened when OECD trading partners adopt increasingly stringent emission pricing.

Figure 4.2: Competitiveness effects on EITE industries (% from BaU) – RCA and RWS

Figure 4.2.a. RCA (% from BaU)

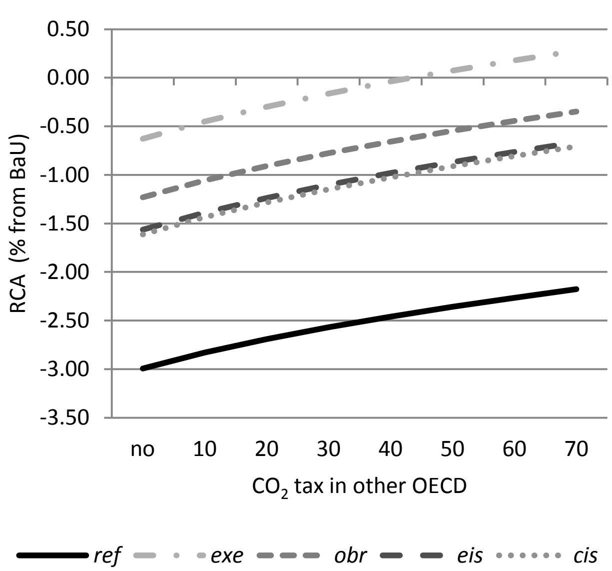
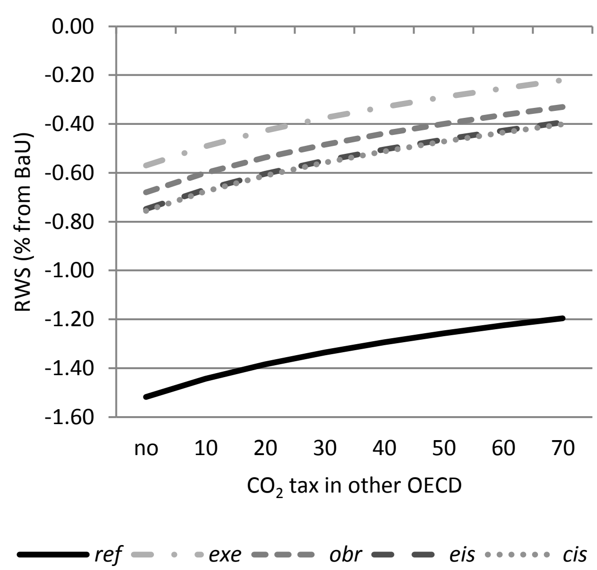


Figure 4.2.b. RWS (% from BaU)

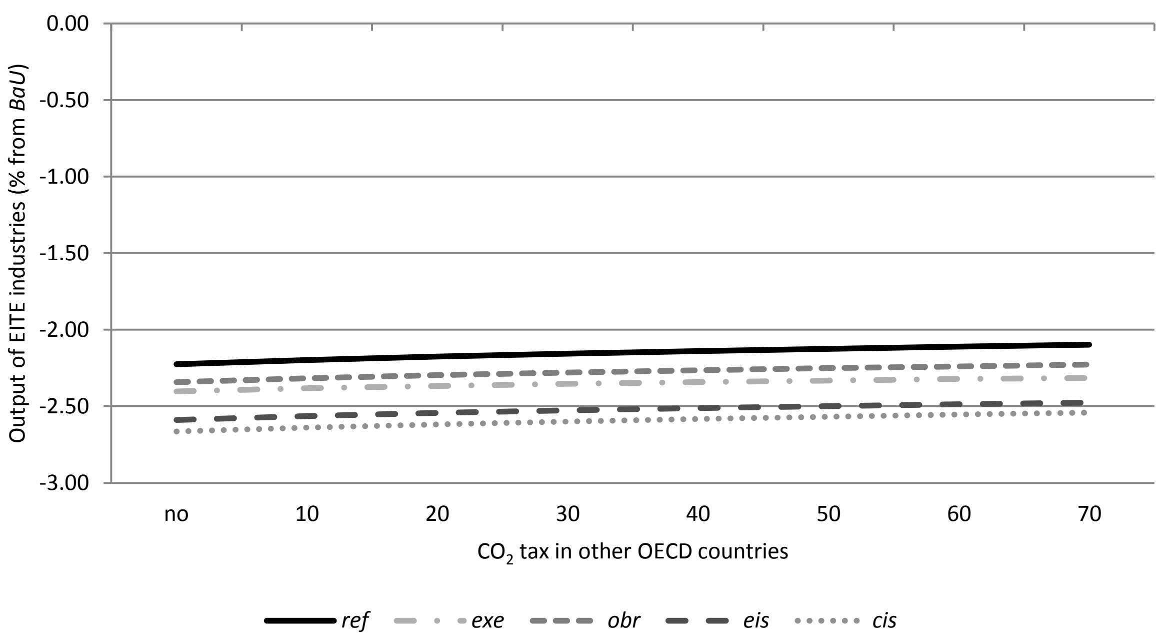


[36] Our price interval is meant to cover some reasonable pricing floor in OECD countries. For example, Canada has announced to levy a CO_2 tax of \$CAD 10 per ton from 2018 onwards, rising by \$CAD 10 per year until it reaches \$CAD 50 in 2022 (https://www.theguardian.com/world/2016/oct/03/canada-carbon-emissions-tax-paris-climate-agreement).

Among protective measures, full exemptions (*exe*) are most effective to preserve competitiveness of US EITE industries followed by output-based rebates (*obr*) and standards (*cis*, *eis*). With full exemptions, US EITE sectors might even slightly gain in competitiveness vis-à-vis the *BaU* provided OECD trading partners adopt sufficiently high emission taxes. Output-based rebates (*obr*) dampen the price increase of EITE industries by recycling sector-specific emission payments via explicit output subsidies. Similarly, standards in scenarios *eis* and *cis* (which are set at the energy/carbon intensities obtained in scenario *ref*) imply subsidization of output and thereby reduce competitiveness losses as compared to the *ref* scenario.

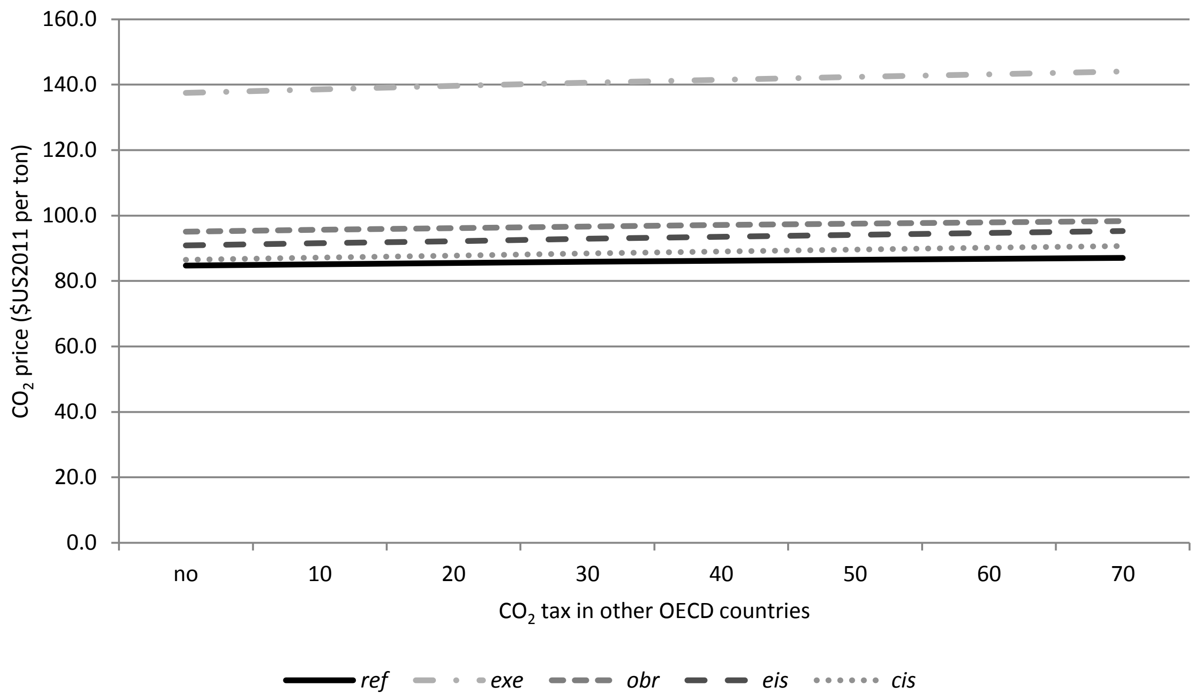
Figure 4.3 reports how alternative US climate policy designs affect composite output of US EITE industries measured as change in output value across EITE industries. The striking insight is that protective measures rather decrease than increase the composite output value via-à-vis the *ref* scenario. This is because protective measures augment overall compliance cost for the US economy (see Figure 6 below) which translates into lower real income and lower domestic demand for US products. US EITE sectors are trade-intensive, but their output still depends mainly on domestic demand. With protective measures, the gains in export supply compared to the *ref* scenario are more than compensated through reductions in domestic demand.

Figure 4.3: Output effects in US EITE industries (% from BaU).



Comparison of Figures 4.2 and 4.3 conveys the warning that sector-specific indicators for international trade performance can be quite misleading as a proxy for sector-specific output performance: While EITE competitiveness in terms of RCA or RWA increases with seemingly protective measures vis-á-vis scenario *ref*, domestic EITE output actually fares worse

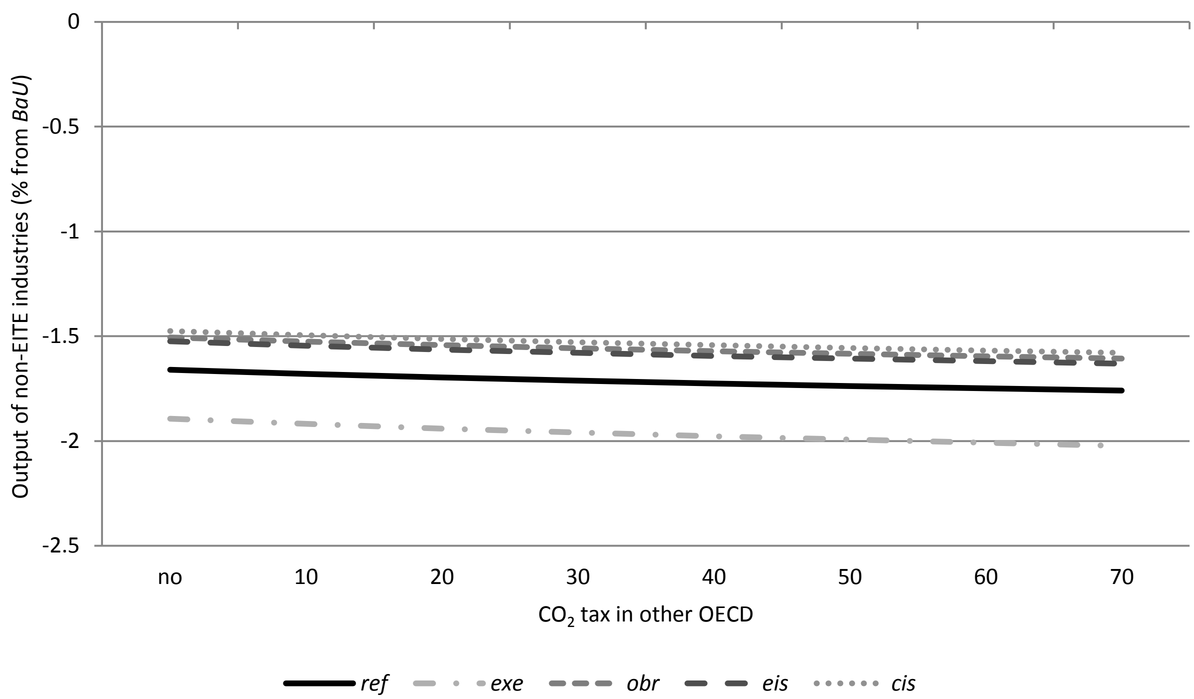
Figure 4.4: CO2 price (in $US2011 per ton)



In our general equilibrium setting, policy measures targeted to EITE industries spill over to non-EITE industries via changes in relative prices and real income. In first place, however, preferential treatment of EITE industries will require higher mitigation efforts by non-EITE sectors to keep with the economy-wide emission reduction target. Figure 4.5 captures the implications of protective EITE measures on the CO_2 emission pricing. Without emission pricing in other OECD countries, the CO_2 emission price in the US to achieve a 30% emission reduction amounts to 85 $US per ton for the *ref* scenario. When EITE industries are fully exempted from emission pricing (*exe*), the rest of the US economy faces a substantially higher CO_2 price, i.e. 137.5 $US per ton, to make up for the reduced abatement contribution by EITE industries. Output-based rebates and standards still involve emission pricing to EITE industries but since these measures subsidize EITE output there is still the need for higher CO_2 emission prices as compared to *ref*: 95 $US, 91 $US, and 86.5 $US per ton under *obr*, *eis,* and *cis* respectively.

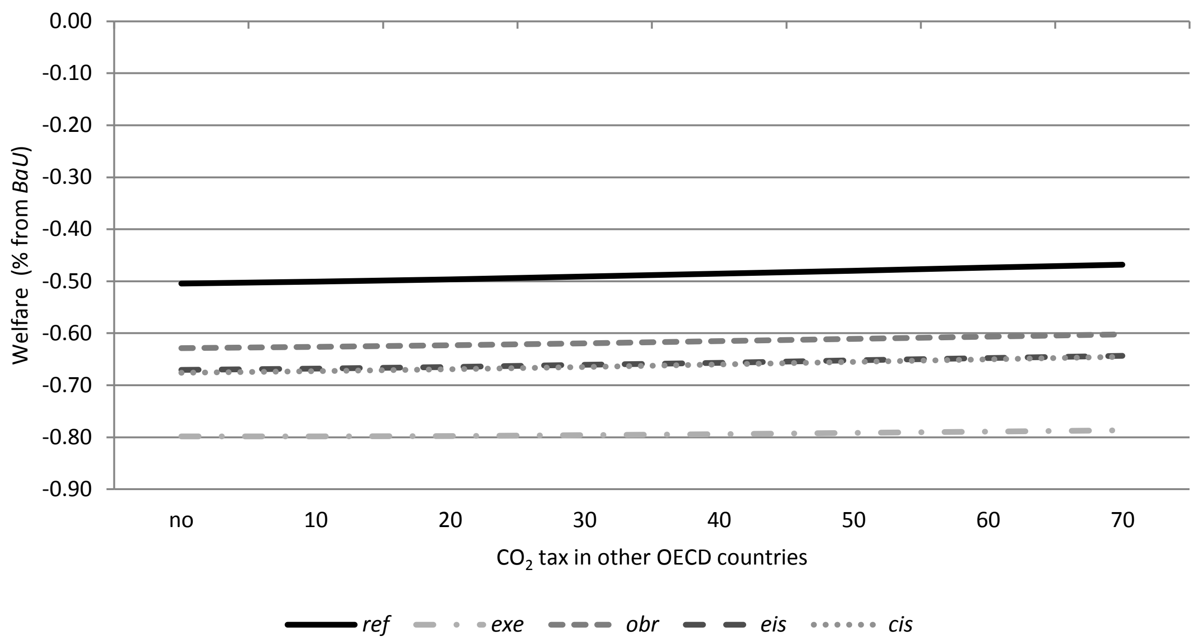
Figure 4.5 displays the implications of alternative US climate policies on the output value of non-EITE industries. In scenario *exe*, only non-EITE sectors are subject to stringent CO_2 emission pricing which depresses their value of output below the *ref* level. For all other protective measures, non-EITE output does not fall as much as in the *ref* scenario due to differential income effects.

Figure 4.5: Output effects in US non-EITE industries (% from BaU)



Our appraisal of protective measures so far has been limited to trade performance and output at the sectoral level. However, a general equilibrium perspective calls for an economy-wide valuation where policy-induced changes are rigorously linked to real consumption (Aiginger, 2006; Grilo and Koopman, 2006). In this vein, we measure the welfare impacts of policy regulations as Hicksian equivalent variation in income. The Hicksian equivalent variation in income denotes the amount necessary to add to (or subtract from) the BaU income of the representative consumer so that she enjoys a utility level equal to the one in the counterfactual policy scenario on the basis of ex-ante relative prices. Figure 4.6 clearly indicates that none of the protective measures yields welfare gains compared to the *ref* policy. Cost-effective reduction of domestic emissions is warranted by uniform emission pricing stand-alone. Subsidies to EITE industries – be it explicit in terms of exemptions and output-based rebates or implicit via standards – induce a deviation from the cost-effective pattern of CO_2-abatement via the combination of substitution and output effects. Most costly is scenario *exe* where EITE industries which typically dispose of relatively cheap emission abatement options do not face any emission price – abatement efforts are shifted to the remaining segments of the US economy which must undertake more costly emission reduction measures. Figure 6 furthermore indicates that the welfare cost of US emission reduction mainly depend on the choice of the domestic policy design. Across all policy variants, welfare cost decrease only slightly when OECD trading partners follow up with emission pricing (thereby reducing the loss in comparative advantage triggered by stringent US emission reduction targets).

Figure 4.6: US welfare impacts (% Hicksian equivalent variation (HEV) in income)



4.4.7 Sensitivity analysis

To test the robustness of our policy insights, we have performed a series of sensitivity analysis. First, we evaluate how more trade stringent criteria for the definition of EITE sectors affect our findings. Second, we investigate the implications of more comprehensive emission pricing outside the US. Table 4.4 summarizes the results of the sensitivity analysis for EITE competitiveness indicators (RCA and RWS) and US welfare. For comparison, the results of our central case simulations are listed with the label *core*.

We start the discussion looking at the effects of tightening the selection criteria for EITE industries to the original prescriptions of the EU Commission, i.e. we re-set the threshold for the trade intensity criterion from 10% to 30%. A more restrictive selection of EITE industries implies that protective measures fare better on improving trade performance while the economy-wide excess cost of such measures as compared to the *ref* scenario decrease. The reason is that protective measures become more targeted and apply to a lower base. As the trade intensity of EITE industries is higher, the preferential EITE treatment is more effective in preserving international competitiveness. At the same time, US EITE industries now only account for 13% of overall US gross value-added which means that the economy-wide cost of preferential treatment decline.

Table 4.4. Sensitivity analysis – competitiveness measures and US welfare

RCA in US EITE industries (% from *BaU*)															
CO_2 tax in other countries	*ref* 10	30	60	*exe* 10	30	60	*obr* 10	30	60	*eis* 10	30	60	*cis* 10	30	60
Core	-2.83	-2.57	-2.26	-0.45	-0.16	0.18	-1.06	-0.78	-0.45	-1.39	-1.10	-0.76	-1.44	-1.15	-0.81
EITE_EU_criteria	-2.74	-2.52	-2.28	0.08	0.33	0.63	-0.56	-0.32	-0.04	-0.82	-0.57	-0.28	-0.85	-0.60	-0.30
OECD+CHN	-2.78	-2.44	-2.05	-0.40	-0.03	0.41	-1.01	-0.64	-0.22	-1.34	-0.97	-0.54	-1.39	-1.02	-0.58
OECD+CHN+ROW	-2.62	-2.05	-1.40	-0.22	0.41	1.15	-0.84	-0.23	0.48	-1.17	-0.55	0.17	-1.22	-0.60	0.13
RWS in US EITE industries (% from *BaU*)															
CO_2 tax in other countries	*ref* 10	30	60	*exe* 10	30	60	*obr* 10	30	60	*eis* 10	30	60	*cis* 10	30	60
Core	-1.44	-1.34	-1.23	-0.49	-0.38	-0.25	-0.60	-0.48	-0.36	-0.67	-0.55	-0.43	-0.68	-0.56	-0.43
EITE_EU_criteria	-1.46	-1.38	-1.31	-0.11	-0.02	0.08	-0.31	-0.21	-0.12	-0.37	-0.28	-0.18	-0.38	-0.29	-0.19
OECD+CHN	-1.43	-1.31	-1.19	-0.48	-0.35	-0.21	-0.59	-0.46	-0.32	-0.66	-0.52	-0.38	-0.67	-0.53	-0.39
OECD+CHN+ROW	-1.37	-1.16	-0.97	-0.41	-0.19	0.02	-0.52	-0.30	-0.09	-0.59	-0.37	-0.15	-0.60	-0.38	-0.16
US welfare impacts (% Hicksian equivalent variation (HEV) in income) (% from *BaU*)															
CO_2 tax in other countries	*ref* 10	30	60	*exe* 10	30	60	*obr* 10	30	60	*eis* 10	30	60	*cis* 10	30	60
Core	-0.50	-0.49	-0.47	-0.80	-0.80	-0.79	-0.63	-0.62	-0.61	-0.67	-0.66	-0.65	-0.67	-0.66	-0.65
EITE_EU_criteria	-0.50	-0.49	-0.47	-0.74	-0.73	-0.73	-0.63	-0.62	-0.61	-0.65	-0.64	-0.63	-0.65	-0.64	-0.63
OECD+CHN	-0.50	-0.48	-0.46	-0.80	-0.79	-0.78	-0.62	-0.61	-0.60	-0.67	-0.66	-0.64	-0.67	-0.66	-0.64
OECD+CHN+ROW	-0.49	-0.46	-0.41	-0.80	-0.78	-0.76	-0.62	-0.60	-0.56	-0.66	-0.64	-0.60	-0.67	-0.64	-0.60

Regarding the regions outside the US that undertake emission abatement in the aftermath of the Paris Agreement we consider two variants with a broader coverage as compared to the central case setting: one variant in which China joins other OECD countries in emission pricing and another variant where we assume that all non-US countries adopt emission pricing (ranging from \$US 0 to \$US 70). As in our central case simulations, we simply treat all countries in the respective abatement coalitions (outside the US) symmetrically. As expected, the international cost disadvantages for US EITE industries and thus arguments in favour of protective EITE measures get weaker the more countries adopt increasingly stringent emission pricing.

To summarize the results of the sensitivity analysis in qualitative terms: All of the key insights from the central case simulation remain robust.

4.5. Conclusion and policy implications.

Despite of the Paris Agreement, international climate policy over the next years will likely be characterized by disparate ambition levels in greenhouse gas abatement where industrialized countries lead the way with more stringent emission reduction targets while developing countries stick to rather lenient emission controls. Against this background energy-intensive and trade-exposed (EITE) sectors in industrialized countries are concerned on competitiveness losses and call for protective measures. The EU has already opted for output-based rebates to EITE industries that are regulated under the EU Emissions Trading System. In other OECD countries such as the US the debate on industry-specific protective measures is still ongoing.

In this chapter we have analysed the economic implications of US climate policy design considering alternative protective measures for US EITE industries: (i) output-based rebates, (ii) exemptions from emission pricing, (iii) energy intensity standards, and (iv) carbon intensity standards. We have compared how these measures perform against a climate policy reference where the US relies solely on economy-wide uniform emission pricing to reduce domestic CO_2 emissions by 30%.

Based on simulations with a multi-sectoral multi-regional CGE model of global trade and energy use we find that protective measures can attenuate adverse competitiveness effects measured in terms of common indicators such as relative world trade shares (RWS) or revealed comparative advantage (RCA). However, protection of domestic US EITE industries distorts the cost-effective pattern of emission abatement inducing substantial excess cost for US climate policy. The associated real income losses depress domestic demand for EITE production which is the key determinant for US EITE industries. As the gains of protective measures in US EITE exports get more than compensated through the losses in domestic EITE demand, protective measures may rather be detrimental than beneficial for US EITE output. We conclude that our simulation results warrant caution against an embracement of protective measures for US EITE industries even from the perspective of EITE lobbyists.

Chapter 5

Economic and distributional implications of alternative mechanism to finance renewables

5.1. Introduction

Promoting renewable energy has become a policy priority for governments around the world[37] because of its positive environmental and socioeconomic effects, such those related to climate change, energy security, “green" jobs, public health, and energy access. In this context, renewable energy deployment is increasing fast[38] (IEA 2016) and in 2015 investments in renewable power capacity accounted for more than half of the new global installed capacity for the first time (FS-UNEP 2016). In the European Union (UE-28), for example, the share of electricity produced with renewable sources (RES-E) grew from 14.4% in 2004 to 27.5% in 2014 (Eurostat 2016), mainly due to the rapid expansion of wind and solar technologies in countries such as Germany or Spain. It is expected to increase more in the future to achieve the energy and climate targets adopted for 2030 and 2050 (EC 2011, 2014).

However, there is also concern about the potential effect of promoting RES-E on the final price of electricity and how this may affect different social groups, firms and competitiveness. Although renewables are already economically competitive in various circumstances and their cost has decreased drastically in recent years[39] (IEA 2016), their average levelized private[40] costs are higher than those of conventional sources (IPCC 2011), especially when the costs of the network infrastructures and the back-up systems needed to cover for the intermittency of renewables are considered. The main instrument[41] for supporting renewables has been technology-specific feed-in tariffs (FITs) in the electricity sector, a mechanism that guarantees a long-term fixed price for RES-E and an obligation to

[37] See IRENA (2016) for a more detailed overview of the arguments used to support the promotion of renewables. More critical analyses can be found in Böhringer et al (2007) and Fischer and Preonas (2010).
[38] The share of RES-E (wind, solar, biomass, geothermal and tide) grew from 1.3% in 1990 to 6.7% in 2015 (IEA 2016).
[39] The cost of solar photovoltaic (PV) energy decreased in five years (2009-2014) by 80% and that of wind turbines by 30% (IEA 2016).
[40] Including the monetary value of the external costs of energy would improve the competitiveness of renewable options.
[41] There are other mechanisms for supporting RES-E (del Río and Mir-Artigues 2014) but they all tend to entail passing on the promotion of renewables to the electricity bill. Other support schemes include feed-in premiums (FIPs), a price premium paid on top of the market price of electricity, and renewable portfolio standards (RPS) a quantity-based instrument that enables generators to issue RES-E certificates that electricity distributors need to surrender as a share of their annual consumption.

purchase all output from renewable sources. The difference between FITs and the wholesale[42] electricity price is accounted for as subsides to renewables, and included in the retail electricity price as a surcharge on renewables. In the EU, for example, the price of electricity increased from 2005 to 2015 by 21% for households and by 32% for industrial consumers (Eurostat 2016). Although the contribution of support for RES-E to the final electricity prices is still under debate (Traber and Kemfert 2009, del Rio et al 2016), it is clear that increases in the price of electricity undermine the social acceptability and political feasibility of policies in support of renewables. Therefore, the issue of the incidence of RES-E promotion and how to finance these schemes to offset its negative impact is now receiving increasing attention among researchers and policymakers (Schmalensee 2012; del Río and Mir-Artigues 2014; Mir-Artigues, Cerdá, and del Río 2015; Neuhoff et al. 2013)

The literature on this incidence shows (Fullerton 2009) that climate and energy policies tend to be regressive as they raise the price of fossil-fuel-intensive products, which typically represent a higher fraction of the expenditure of low-income groups (consumption channel). Also, non-fossil fuel options are usually more capital intensive than fossil fuel options so they induce firms to demand more capital relative to labor, lowering relative wages and negatively affecting low-income groups (income channel). This general finding can also be applied to the promotion of RES. Using household micro data from Germany, Neuhoff et al. 2013 show that the burden of an RES-E surcharge is significantly higher on low income groups. They therefore propose three measures to reduce this effect: reducing the tax on electricity, increasing support for energy efficiency measures and increasing social transfers to low income groups. Using a microsimulation and a computable general equilibrium model, Böhringer et al 2016 show the cost-efficiency losses and regressiveness of RES-E policies in Germany but also show that these effects can be decreased if exemptions to the electricity surcharge are introduced or, alternatively, if the cost of renewables is financed through other tax sources such as value added taxes (VAT).

In this study we apply a computable general equilibrium (CGE) model in combination with a microsimulation (MS) model to examine the distributional implications of different schemes for financing the promotion of renewables in the Spanish electricity sector. These schemes include exemptions from the RES-E surcharge on the price of electricity for residential and industry consumers and also different alternatives where the cost to renewables is not passed on to consumers in the

[42] RES-E technologies have low or close to zero marginal costs which can reduce the wholesale electricity prices due to the so-called merit of order effect (Sáenz de Miera et al 2008; Sensfuss et al 2008, Gelabert et al 2011). This effect depends on the relative slopes of the supply of renewable and non-renewable technologies (Fischer 2010).

electricity bill but financed by other tax sources in the energy sector, such a as fuel tax, or in the overall economy, such as VAT or transfers. Our integrated modeling approach includes a rich representation of household heterogeneity and the inter-sectoral and price-related effects, which are fundamental for analyzing those implications of these schemes that are not restricted to the electricity sector.

Spain provides a relevant case study for two reasons: first, it implemented one of the strongest support schemes for renewable energy in the world through FITs that substantially increased the share of renewables in the electricity sector; and second, it had to reduce them substantially in 2013 due to concerns about their financial implications in the context of a fiscal consolidation process of the government budget in the aftermath of the Great Recession of 2008.

The rest of this chapter is organized as follows. Section 5.2 provides an overview of the case study of Spain. Section 5.3 summarizes the basic structure and parameterization of the CGE and MS models used for the simulation analysis and outlines how the models are linked. Section 5.4 sets out the policy scenarios and discusses simulation results. Section 5.5 concludes.

5.2. RES-E promotion and electricity prices in Spain

The promotion of renewable energy in Spain has been driven historically by the main objective of increasing the share of renewables given the country's high level of dependency on imported fossil fuels. More recently these policies have also begun to be directed at the objective of reducing greenhouse gas emissions (GHG), closely aligned with the European Union's climate and energy targets. Spain is making clear progress[43] towards the binding national target of having renewable energy account for 20% of gross final energy consumption by 2020 and RES-E promotion policies are contributing substantially to that objective. The share of RES-E in Spain doubled in ten years from 20.3% in 2004 to 40.9% in 2014[44] (Eurostat 2016), due to the large-scale expansion of wind[45] and solar capacity.

The RES-E support scheme in Spain has been based mainly on feed-in tariffs and premiums since 1998, with some rather minor reforms of the whole scheme taking place in 2004 and 2007 under the Spanish

[43] The share of renewable energy in gross final energy consumption increased from 12% in 2005 to 17% in 2015.

[44] In 2014 nuclear power was the main source of electricity generation with a share of 20.9%, followed closely by wind power with 19.1% and natural gas with 17.2%. The remainder consists of coal (16.3%), hydropower (14.3%), oil (5.2%), solar (5%) and biofuels and waste (2%). The maximum level of generation from RES-E was on 13 February 2016, when renewables accounted for 67.5% of the day's output.

[45] The shares of both wind and solar power are the fourth-highest. Spain is ranked behind Denmark, Portugal, and Ireland as regards the share of wind in electricity generation and behind Italy, Greece, and Germany as regards that of solar. Spain's wind power capacity is the fourth-highest in the world after China, the United States, and Germany, and practically equal t that of India (IEA 2015, Spain).

Renewable Energy Act. In 2013 this system was replaced by a return-based remuneration system in which renewable operators are guaranteed a rate of return which is based on 10-year Spanish government bonds plus a spread, which was set originally at 300 basis points. The reform was motivated by the need to balance the costs and revenues of the electricity system as cost was increasing much faster than revenues and by 2012 there was a tariff deficit of €26 billion (equivalent to 2.5% of GDP). The cost of support for RES-E was an important component of the regulated cost of electricity, as Figure 5.1 shows. The cost of promoting RES-E increased from €2.9 billion in 2005 to €6.6 billion in 2015. RES-E support costs were high partly because investments in renewables far exceeded[46] those planned by the National Energy Plan for 2015-2020: The targets envisaged total public spending of €5 billion on RES-S for the whole period but that amount was actually spent in 2010 alone. Only after 2012 did the RES-E cost and the tariff deficit start to decrease.

Figure 5.1: Regulated cost in the Spanish electricity system, 2005-2015

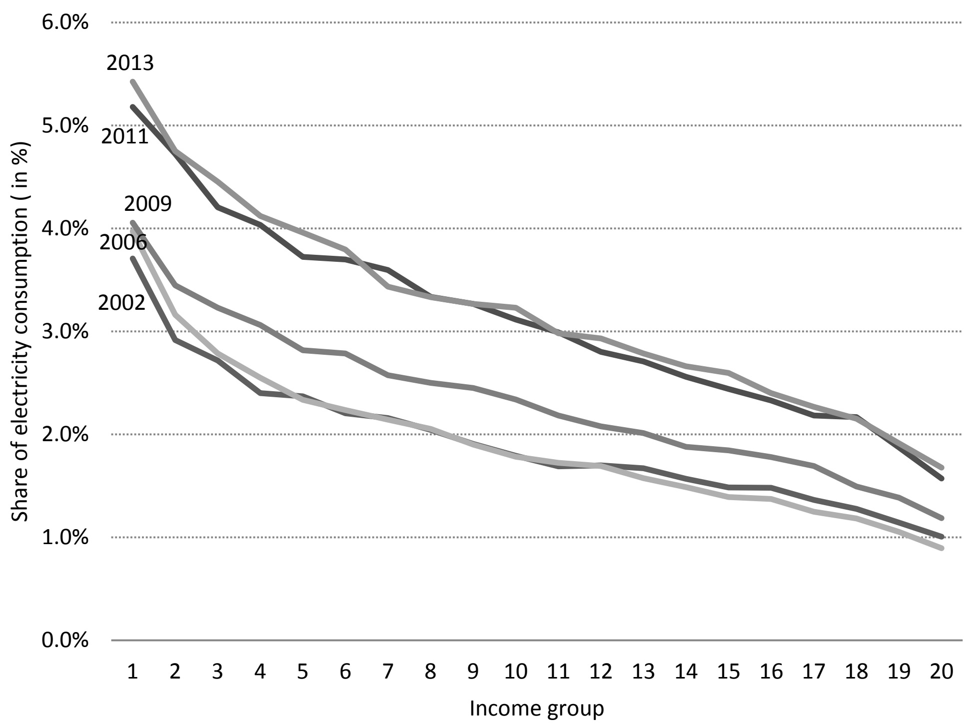
Source: Spanish Ministry of Industry, Energy and Turism

In this context, electricity prices in Spain have increased significantly. The annual average electricity price for households increased from 2004 to 2014 by 109% (from €0.1079 per kWh to €0.2252 per kWh)

[46] This was particularly the case of solar PV which experienced an unprecented investment spike. Solar PV generation capacity increased from 146 MW in 2006 to 3398 MW in 2008 and accounted for 56% of all the support received by renewables despite providing just 12% of Spain´s renewable electricity (Mir-Artigues et al 2015).

and for medium-sized industry by 120% (from €0.0538 to €0.1185 per kWh). This price rise has increased spending on electricity, especially for low income households. Figure 5.2 shows electricity costs as percentages of consumer spending for twenty income groups (ventiles) for various years, using data from the Spanish Income and Expenditure Survey (INE 2016). Spending on electricity as a proportion of disposable income increased in lowest income group (first ventile) from 4% in 2006 to 5.5% in 2013 and in the highest income group (twentieth ventile) from 1% to 1.5% for the same period. This increase reflects the increase in electricity prices but also a decline in real incomes for this period due to the economic crisis.

Figure 5.2: Percentage of total expenditure devoted to electricity per income group and year



Expenditure on electricity is an important fraction of total expenditure on energy, which also includes other components such as expenditure on fuel and gasoline for private transport (ranging between 0.6% and 1.2% for the lowest and highest income groups) and in gas for heating (between 1.3% and 0.5%), as shown in Figure 5.3a. The structure of expenditure by social groups together and by income sources (see Figure 5.3b) is important information for understanding the distributional implications (consumption and the income channel) of the alternative scenarios for RES-E promotion to be assessed in this study, which are presented in the following sections.

Figure 5.3: Spanish Households. Income and consumption patterns by income group

Figure 5.3 a: Consumption patterns by income group

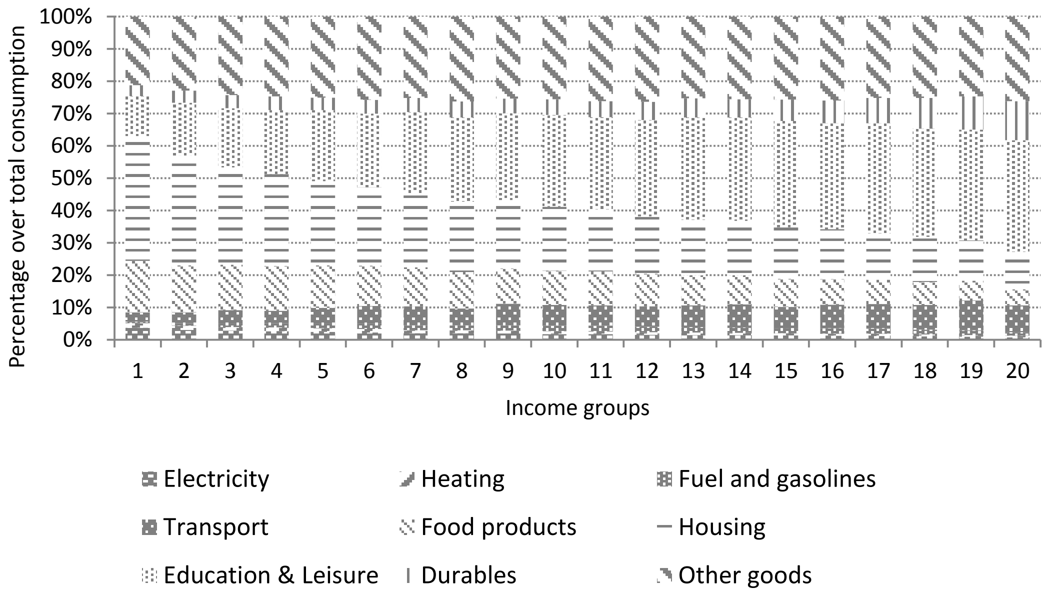
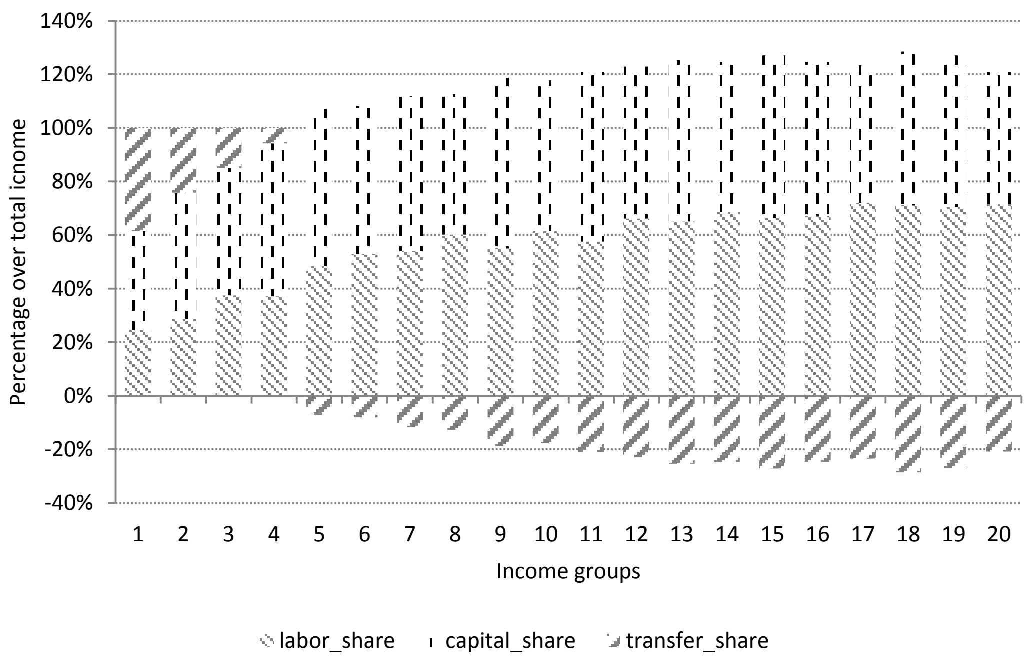


Figure 5.3 b: Income sources by income group.



5.3. Methodology and data

5.3.1 Methods

This study seeks to shed further light on the relative performance of alternative financing measures for RES-E promotion. To that end, we set out a computable general equilibrium model (CGE) and a micro-simulation (MS) model for Spain. The link between CGE and MS models enables us to analyze macroeconomic policy simulations at the microeconomic level. We use a hard link approach that links the micro and macro models using a recursive or iterative process that enables us to capture feedbacks between the two models.

5.3.1.1 Summary of the Computable General equilibrium model

We use a multi-sectoral CGE model to capture the economy-wide assessment of RES-E promotion. For a detailed algebraic formulation of the core model and recent application see Böhringer et al. (2016).

Production of commodities other than fossil fuels is captured by constant elasticity of substitution (CES) cost functions describing the price-dependent use of capital, labor, energy, and material in production. At the top level, a CES composite of intermediate material demands trades off with an aggregate of energy, capital, and labor. At the second level, a CES function describes the possibilities of substitution between intermediate demand for the energy aggregate and a value-added composite of labor and capital. Finally, at the third level, a CES function captures the possibilities of capital and labor substitution within the value-added composite, while different energy inputs (coal, gas, oil, and electricity) enter the energy composite subject to a CES. In the production of fossil fuels all inputs except the sector-specific fossil fuel resource are aggregated in fixed proportions; this aggregate trades off with the sector-specific fossil fuel resource at a CES.

Final demand for consumption is determined by a representative household, which maximizes utility subject to a budget constraint with fixed investment and exogenous government provision of public goods and services. The representative agent receives income from three primary factors: labor, capital, and fossil fuel resources (coal, gas and crude oil). Labor and capital are mobile across sectors. Fossil-fuel resources are fixed to the respective resource production sectors. Final demand for consumption is given as a CES aggregate of composite non-energy consumption and composite energy consumption. Both the non-energy consumption composite and the energy consumption composite are in themselves CES functions of disaggregate non-energy and energy commodities.

Given the paramount importance of the electricity sector with respect to the promotion of renewable power generation we break power generation down into two composite production technologies: conventional power generation and renewable power generation. These two power generation technologies produce electricity by combining technology-specific capital with inputs from labor, fuel, and materials. Electricity from different technologies is treated as a homogeneous good. Power generation technologies respond to changes in electricity prices according to technology-specific supply elasticities (for details on calibration see Rutherford 2002).

Bilateral trade follows the Armington (1969) approach of product heterogeneity, where domestic and foreign goods are distinguished by their origins. A balance of payment constraint incorporates the base-year trade deficit or surplus. All goods used on the domestic market in intermediate and final demand correspond to a CES (Armington) composite that combines domestically produced goods and the goods imported from other regions.

The model links carbon dioxide (CO_2) emissions in fixed proportions to the combustion of fossil fuels with fuel-specific CO_2 coefficients. Emission intensity or energy intensity within a sector can be reduced in two ways: by inter-fuel switching or by substituting away from fuels to non-fuel inputs. The cost of reducing intensity thus depends on the substitution elasticities and benchmark production cost shares. Total domestic emissions and energy use can also be reduced by structural shifts in production and consumption patterns.

5.3.1.2 Demand Model

A demand model captures the real behavior of households and provides a realistic picture of the substitution effects using econometric techniques. We estimate a demand model to provide a set of estimates of the substitution, own-price and expenditure elasticities of the goods analyzed. Accordingly, we use the well-known Almost Ideal Demand System (AIDS) proposed by Deaton and Muellbauer (1980). Its main advantage is that it enables a first-order approximation to be made to an unknown demand system. In addition, the model satisfies the economic consumption theory axioms and does not impose constraints on the utility function. The log-linear approximation (LAIDS) used in this study follows an n-good system equation as follows:

$$W_{it} = \alpha_i + \sum_{J=1}^{n} \gamma_{ij} ln\, p_j + \beta_i ln\left({}^{Y_{it}}/_{\widetilde{p_t}}\right) + t_t + \sum_{i=1}^{3} d_i + e_{it} \qquad [1]$$

where w_i represents the share associated with good i for each household, p_j is the price of commodity *j*, $\tilde{p}$ stands for the Stone price index, Y is household income (hence, $Y/\tilde{p}$ represents real income), t_t is a trend variable that captures the role of the economic cycle, d_i is a set of dummy variables that controls for the type of household[47], the region where the household is located in terms of NUTS 1, whether the household is in property, the number of rooms, the age of the breadwinner, whether the breadwinner is unemployed or retired, the number of active members in the household, whether the house is equip with heating and the type of house[48]. Finally e_{it} is the idiosyncratic error term. The adding up and homogeneity restrictions of equation [1] are the following:

$$\sum_{i=1}^{n} \alpha_i = 1 \qquad [2]$$

$$\sum_{j=1}^{n} \gamma_{ij} = 0 \qquad [3]$$

$$\sum_{i=1}^{n} \beta_i = 0 \qquad [4]$$

The symmetry condition is given by:

$$\gamma_{ij} = \gamma_{ji} \qquad [5]$$

Finally, the sum of w_i should also satisfy the following:

$$\sum_{i=1}^{14} w_i = 1 \qquad [6]$$

In this study we use a set of 9 consumption categories including food, housing, durables, heat, electricity, fuel, transport, leisure and education, and other products. Since the AIDS model is made up of a system of dependent equations, the share equation regarding other products is deleted to overcome singularity problems (Annex D reports the regression results)

5.3.2 Model linking

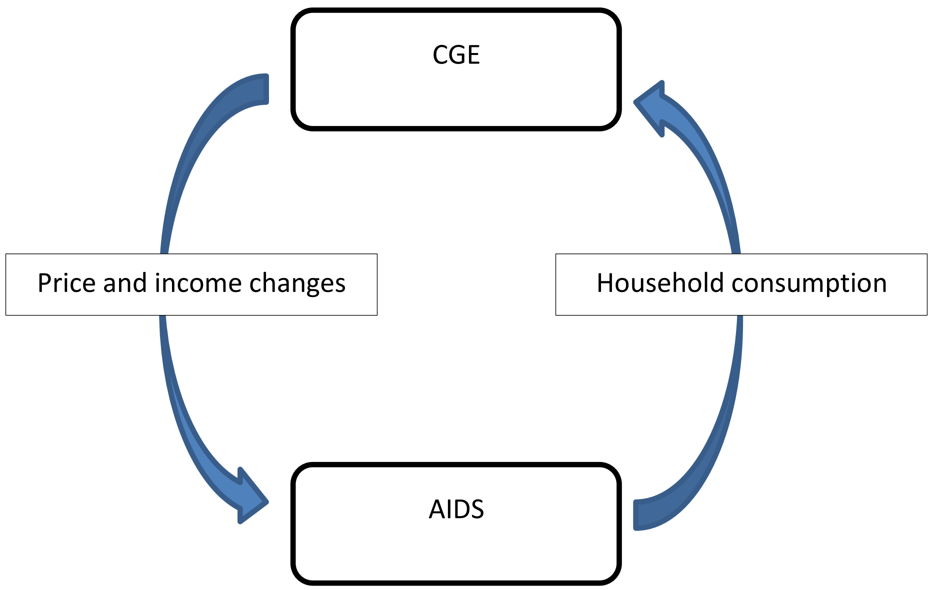
The link between CGE and MS models enables us to analyze macroeconomic policy simulations at the microeconomic level. We use a hard link, which is a recursive approach with an iterative process that enables us to include feedbacks between the two models. We follow the decomposition method used by Rutherford and Tarr (2008). This recursive approach (illustrated in Figure 5.4) is subdivided into three

[47] The household categories used are the following: adults alone; couple without children; couple with children; single-parent households, and other households

[48] The house categories used are the following: luxury, high class in urban area, middle class in urban area, low class in urban area, rural industrial, rural fishing and rural agriculture

different steps. First, we solve the CGE model for the new equilibrium in the representative agent model. Second, the price and income outputs from the CGE model are used as an input in the MS to recalibrate the preferences of the representative consumer. Third, we solve the CGE model again using the new preferences of the representative agent model calibrated using the MS. This last step creates a new imbalance in markets for consumer goods. Subsequent iterations involve carrying out the first step to the third until the two models converge (see Rutherford and Tar, 2008, or Rausch and Rutherford, 2007, for the detailed description of the model recalibration).

Figure 5.4: Recursive approach to link CGE and MS models



In order to implement our integrated model, we need to rescale expenditure and demand data to ensure consistency between IO data and microsimulation data. To achieve the required match, we scale up the total expenditures of households from the microsimulation data to match total household expenditure according to national accounts. Similarly, on the income side, we also scale capital and labor income from the MS model to match total income according to the IO table. Due to a lack of information on savings in the survey, we decided to distribute saving decisions among households in proportion to their capital income. Finally, government transfers are equivalent to the residual between the income factor and savings.

5.3.3 Data

The CGE model is calibrated against the Spanish Input–Output Table for 2007 (INE, 2016a). The IO table is a representation of the uses and resources of the production sectors of the Spanish production system. Output per production sector is linked to consumption by private households in terms of consumption expenditure categories using the so-called "Z-matrix" created by the IPTS Joint Research Centre (Arto et al. 2012). The electricity sector is broken down into two power generation technologies: conventional electricity and electricity from renewables, according to technology-specific production shares provided by Eurostat (2016). Measures for the carbon emission per production sector and fossil source are obtained from the WIOD database (WIOD 2012). At the sectoral level, we identify primary and secondary energy carriers (coal, gas, crude oil, refined oil products, and electricity) which are essential for distinguishing energy goods by CO_2 and energy content as well by their degree of inter-fuel substitutability

The elasticities of substitution used in the CGE are based on empirical estimates by Koesler and Schymura (2015). The elasticities of substitution in fossil fuel sectors are calibrated to match exogenous estimates of fossil fuel supply elasticities (Graham et al. 1999, Krichene 2002, and Ringlund et al. 2008). The price elasticities of electricity supply per technology are calibrated to match the changes in power generation shares across technologies following the subsidies for renewables over the period between 2007 and 2015.

For the Microsimulation model, the dataset used is from the Spanish Household Budget Survey (SHBS) (INE, 2016b). The SHBS is a representative cross-sectional survey of the whole Spanish population that collects yearly information on consumption patterns as well as socio-economic characteristics. It covers around 20,000 households per year. In the estimation stage we use SHBS data for 2006 to 2013. In the estimation of equation [1], household expenditure is used as a proxy of income, firstly because income is strongly under-reported in household panel surveys (see for example Wadud et al., 2009) and secondly because household expenditure is a good proxy for permanent income (Poterba, 1991). The income sources of households are completed by the Living Conditions Survey[49].

Table 5.1: Model sectors and commodities.

Sectors	
Agriculture (Agr)	Gas and distribution (Gas)
Mining (Min)	Manufacturing (Man)
Coal (Coa)	Energy intensity (Ein)
Crude oil and gas (cru)	Services (Ser)
Petroleum products (Oil)	Transport (Trans)
Power electricity sector (Ele)	
Commodities	
Food products (Food)	Housing (House)
Transport (Tran)	Education and leisure (E&L)
Electricity (Elec)	Durables goods (Dura)
Heating (Heat)	Other goods and services (Oth)
Diesel and gasoline (Fuel)	

5.4. Results and discussion

5.4.1 Scenarios

Our research interest is in assessing the distributional impact of different schemes for financing the promotion of RES-E. The scenarios implemented in this study seek to capture two main ways of financing that promotion: i) through a surcharge on electricity prices; and ii) through an increase in other tax sources (Table 5.2).

Table 5.2: Summary of policy scenarios (scenario acronyms in parentheses)

Surcharge on electricity prices	**Alternative financing measures:**
Electricity surcharge *(BaU)*	Value added taxes (*vat*)
Electricity surcharge with an exemption on all producers *(exe_prod)*	Oil taxes (*fueltax*)
Electricity surcharge with an exemption on all households *(exe_house)*	Lump sum (*lsm*)

[49] We use a proxy method to match the information from the two surveys see Rutherfor and Tarr (2004).

The main channel for supporting renewables is a surcharge on the price of electricity for both producers and households (the *BaU* scenario). However, distributional impacts also depend on how the surcharge is shared between them. Therefore, we propose two scenarios that include exemptions from the surcharge for renewables on the price of electricity for households (*exe_house*) and for production sectors (*exe_prod*). These scenarios are two extreme situations where we explore the consequences of exemptions on either all producers or all households.

Alternatively, we also explore options where the cost of renewables is financed by increasing other taxes. The three scenarios analyzed in this study are an increase in i) valued added tax (*vat*); ii) oil taxes in the energy sector (*fueltax*); and iii) lump-sum transfers to consumers (*lsm*). These options have been proposed recently by different institutions. For example, the Spanish employers' organization, CEOE, has proposed that electricity costs not related to the cost of supply should be financed from other tax sources (CEOE 2014). The International Energy Agency (IEA 2015) has also recommended to the Spanish government that it maintain a strong long-term commitment to balancing costs and revenues in the electricity system, and has pointed out that oil taxation in Spain is quite low (e.g. the tax component on the total diesel price is only 51% whereas in the United Kingdom it is 67% and in Italy it is 62% (IEA 2015)).

5.4.2 Cost effectiveness results

This sub-section presents the overall economic effects of the different scenarios in terms of percentage point changes from the business-as-usual scenario (*BaU*), considering that each scenario achieves a similar supply of renewables.

Table 5.3 shows the cost-effectiveness of each scenario. The results show that the macroeconomic effects of the different scenarios are quite low. These results are not surprising, not only because each scenario uses similar revenues to finance the promotion of RES-E but also because the amounts are not highly significant compared to the total output of the economy or to GDP. However, they show that efficiency concerns alone would not provide a strong reason to deviate from financing the promotion of RES-E by increasing electricity prices. From a policy perspective, policy-makers may choose between the different financing designs without efficiency concerns.

Although the overall economic results for each scenario with respect to *BaU* are quite low, there are some differences which deserve to be highlighted. As expected, lump sum transfers (*lsm*) and value added taxes (*vat*) are the most effective financing designs, followed by household exemptions (*exe_house),* producers' exemptions (*exe_prod*) and oil taxation *(fueltax)*. The excess burden is higher under those tax systems where the tax base is narrower and the substitution options are also lower. When *exe_prod* and *fueltax* are set, the ability of consumers to substitute other energy-goods for electricity or fuel is more limited, and thus the welfare results are worse.

Table 5.3: Overall economic effects per policy design.

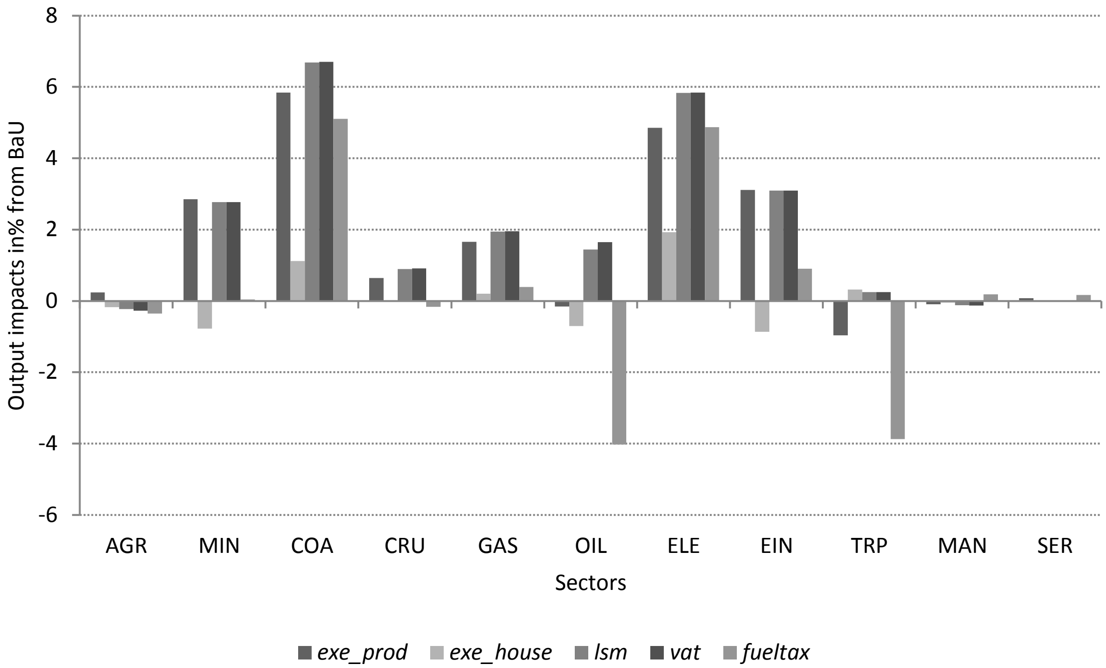
Scenarios	***exe_prod***	***exe_house***	***lsm***	***vat***	***fueltax***
Welfare (in % compared to BaU)	-0.018	0.001	0.063	0.063	-0.025
CO_2 (in % compared to BaU)	2.23	1.02	3.09	3.15	-4.61
Subsidy on renewables (in €bn)	5.40	5.63	5.38	5.32	5.28
Share of renewables (% total electricity)	38.09	39.21	37.74	37.74	38.09
Supply of renewables	14.92	14.92	14.92	14.92	14.92

All scenarios have as a common feature the fact that they modify or eliminate the surcharge for financing the promotion of RES-E. The reduction of the electricity surcharge leads to greater electricity supplies, and thus greater CO_2 emissions. Similarly, a higher electricity demand reduces the target level of renewables achieved, even if the different scenarios achieve the same supply of renewables. Thus, to achieve the pre-scenario target for renewables -equivalent to 40% of the total electricity supply- higher subsidies on renewables would be needed. Under *exe_house* the electricity supply is closer to *BaU* levels (see Figure 5.5 below), so *exe_house* is the most effective mechanism for achieving target levels of renewables without increasing CO_2 emissions.

5.4.3 Sectoral impacts

Figure 5.5 shows how alternative financing designs affect production per sector of the economy. The main argument used by producers to defend exemptions (*exe_prod*) is the avoidance of an excessive increase in energy costs that might affect their competitiveness, especially in energy-intensive sectors. However, producers' exemptions call for greater efforts from the rest of the economy, i.e. from households. The result shows that in general the output in the *exe_prod* scenario increases with respect to *BaU*, and more markedly in the energy-intensity sectors (ein), in the electricity industry, and in those sectors that are most closely related to the electricity sector. By contrast, *exe_house* requires greater financing efforts from economic sectors. Consequently *exe_house* reduces output with respect to *BaU*, mainly in energy intensity industries (ein). Finally, in all the scenarios the production of electricity increases but the *exe_house* scenario is the one where it increases the least, because the higher demand for electricity from households is offset by lower demand from production sectors.

Figure 5.5: Impacts on output per sector and scenario (in % compared to *BaU*).

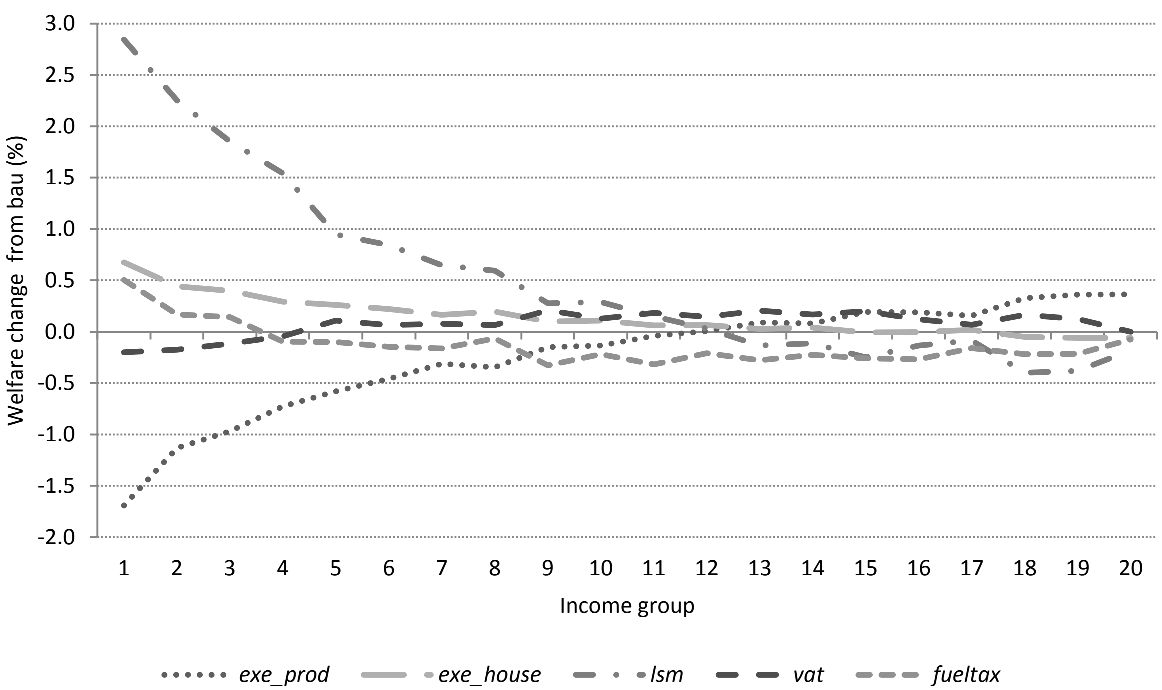


The alternative scenarios all promote RES-E with no surcharge on electricity prices. Therefore, as happened under *exe_prod*, the lower the cost of electricity inputs is the lower the impacts on sectoral output will be. *Lsm* and *vat* financing designs confirm the positive impacts of economic sectors when the effort to finance the promotion of RES-E is not defrayed by industries. On the other hand, *fueltax* shows that the beneficiaries of eliminating the electricity surcharge are mainly the electricity industry, those sectors related to electricity production (such as coal or mining), and energy-intensive industries. However, under *fueltax* the oil sector and the sectors related to oil production and consumption, such as crude oil (cru) and transport (trp), suffer higher cost impacts. All in all, our results show general benefits when the effort to finance the promotion of RES-E is not defrayed by electricity prices.

5.4.4 Distributional impacts

The argument for introducing exemptions on producers (*exe_prod*) is to avoid any loss of competitiveness, but exemptions on households are aimed at avoiding excessive welfare impacts and reducing possible regressive impacts. In this vein, we present the results for the distribution impacts of the scenarios in terms of welfare (measured in terms of Hicksian equivalent variation in income[50]). Figure 5.6 shows welfare impacts by expenditure groups, where group 1 represents the lowest expenditure and group 20 the highest. Figure 5.6 clearly indicates that there are welfare gains when financing efforts are shifted from households to production sectors. This is consistent with the results obtained for the overall welfare effect (Table 5.3 above). Thus, exemptions on households (*exe_house)* can substantially relieve the welfare impacts and correct the undesirable regressive effects that renewable surcharge can have on the poorest households. On the other hand, exemptions on producers (*exe_prod)* comprise the most regressive of all the options. This reflects the excessive welfare impacts caused by financing the promotion of RES-E through an electricity surcharge paid by households, and also the regressive impacts of industrial exemptions. Under *exe_prod* the higher residential electricity price leads the poorest people to allocate a greater proportion of their expenditure to energy than the rich. A comparison of Figures 5.5 and 5.6 reveals an interesting trade-off between economic output and distributional impacts.

Figure 5.6: Welfare impacts per income group (% of Hicksian equivalent variation (HEV) in income).



[50] The Hicksian equivalent variation in income denotes the amount that must be added to (or subtracted from) the BaU income of the representative consumer so that he/she enjoys a utility level equal to that in the counterfactual policy scenario on the basis of ex-ante relative prices .

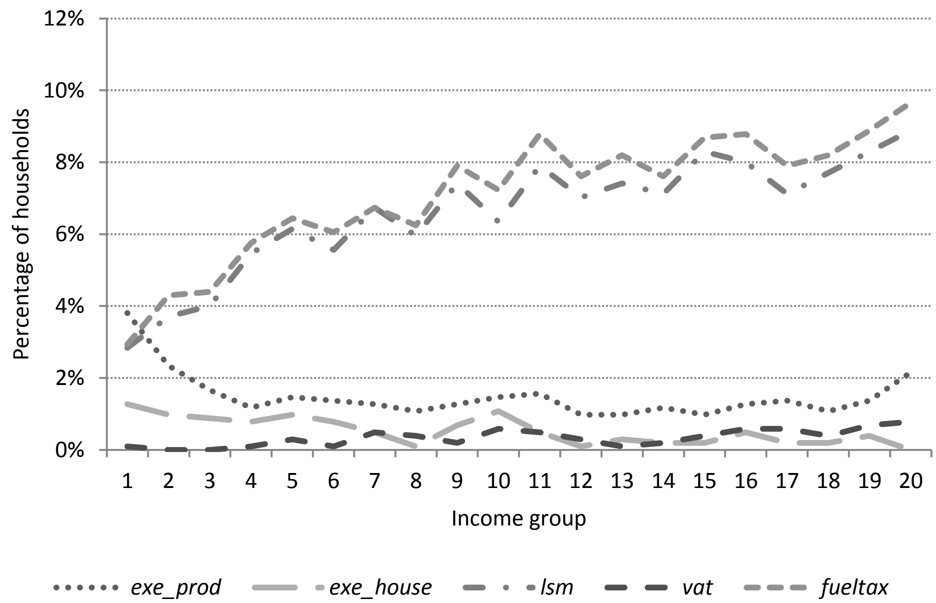
Among alternative taxes sources, *lsm* are the most effective in safeguarding against welfare losses in low income groups, followed by *vat* and *fueltax.* Lump-sum transfers (*lsm*) and value added taxes (*vat*) confirm that there are welfare gains in most income groups from eliminating the promotion of RES-E via an electricity surcharge. When the promotion of RES-E is financed through *lsm* welfare increases in the lowest income households but decreases in higher income households. On the other hand, *vat* and *fueltax* have neutral impacts from a distributional perspective. *Fueltax* results are consistent with the consumption pattern, considering that fuel consumption is similar in the different income groups. Similarly, although value added tax tends to be regressive the differentiation of tax rates for different goods in Spain offsets these regressive effects (Sanz-Sanz and Romero-Jordan, 2012). The main result that emerges from using alternative taxes sources for financing the promotion of RES-E is that the trade-off between sectoral output effects and regressiveness with the electricity surcharge can be overcome and avoided. In fact, *lsm* and *vat* show that both households and production sectors can achieve gains from the promotion of RES-E without increasing electricity bills.

Table 5.4: Impacts on consumer prices and income sources (% compared to BaU)

Scenarios	***exe_prod***	***exe_house***	***lsm***	***vat***	***fueltax***
		Impact on consumer prices			
Food	-0.92	0.27	0.02	0.02	-0.36
Education and Leisure	-0.79	0.26	0.20	0.19	-0.12
Electricity	57.31	-19.16	-16.27	-16.25	-15.33
Fuel	-0.86	0.26	0.12	0.12	8.15
Heat	-0.89	0.34	0.08	0.09	13.02
Housing	-0.81	0.26	0.18	0.18	-0.20
Durables	-0.86	0.27	0.13	0.13	-0.21
Transport	-0.79	0.26	0.19	0.19	1.52
Other goods	-0.80	0.26	0.18	0.18	-0.20
		Impact on income sources			
Labor	-0.52	0.26	0.77	-0.32	0.44
Capital	-0.38	0.10	0.55	-0.55	-0.99
Transfer	-0.84	0.23	3.63	-1.17	-0.01

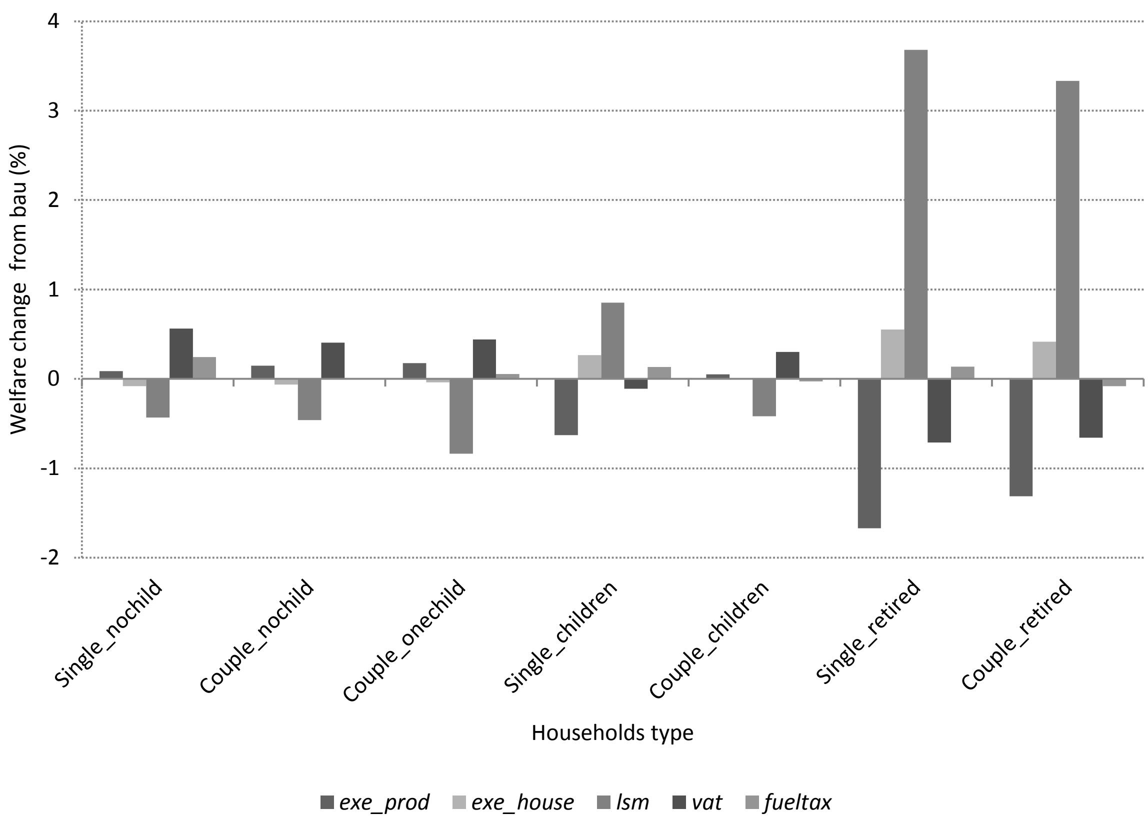
Impact on consumer prices and income sources are key drivers in explaining the above-mentioned welfare and incidence effects. Greater impacts on goods or income sources more related to low income households would tend to lead to greater losses in the poorest households. Table 5.4 shows impacts on consumer prices and on income sources. Industrial exemptions (*exe_prod*) involve higher electricity prices for consumers and thus greater impacts on welfare (Figure 5.6 and Table 5.3). Otherwise, as expected, when *exe_house*, *lsm*, *vat* or *fueltax* are set household electricity prices fall as a consequence of the reduction in the electricity surcharge. In general, impacts on welfare and their incidence are dominated by the electricity price because the rest of the price effects are quite modest and distributed more evenly across different goods. Only under *fueltax* does the fuel price increase notably. Secondly, the impacts on income sources are also quite modest (table 5.4), with the only noteworthy case being the transfer impacts when *lsm* is set. As shown in Figure 5.3.b, the poorest households have net benefits from transfers whereas the middle and upper classes are net transfer donors. Thus, an increase in transfers entails gains for the poorest households and welfare losses for the richest.

Figure 5.7: Percentage of households with losses greater than 5% compared to *BaU* per income group



One of the main advantages of including multiple levels of households in our CGE approach is that we can then zoom in on those households that are more affected. Hence, to test for heterogeneity within income groups, Figure 5.7 reports the share of households where welfare loss is greater than 5% of annual income per income group. As the households where electricity consumption accounts for the largest proportions are in the low income groups, under *exe_prod* the lowest-expenditure households are found to have the greatest number of households with higher welfare losses. By contrast, few households in the highest expenditure groups have impacts greater than 5%. On the other hand, as expected, when *lsm* is set the welfare losses increase with the income of the households. Similarly, when *fueltax* is used to finance the promotion of RES-E the highest-expenditure households are found to have the greatest number of households with higher welfare losses. Although in average terms the impact of *fueltax* is neutral (see Figure 5.6) when we focus on the households with the greatest welfare losses *fueltax* seems to have progressive impacts. By contrast, *exe_house* and *vat* follow a similar trend in average impacts on welfare and income groups with a large proportion of households with higher welfare losses.

Figure 5.8. Welfare impacts per household type (in % of Hicksian equivalent variation (HEV) in income).



Another important issue is that of the implications for energy poverty. According to some estimations in Spain, 21% of households are at risk of energy poverty (see ACA 2016), with the most vulnerable being those with elderly/retired people and those with children. Figure 5.8 reports the impacts of welfare per social group to check for possible counterproductive effects on vulnerable households. Under *exe_prod,* households of retired persons suffer the greatest welfare loss because they tend to have greater electricity expenses. This result shows that the group most vulnerable to changes in electricity prices is that of households of elderly (retired) persons. At the same time, households with elderly persons are net transfer recipients, which explains their welfare gains when *lsm* is set. Single parent households also have greater welfare losses. Such households are normally in the lower income range, for which the monetary loss represents a higher relative cost. In conclusion, measures that increase electricity prices (such as *exe_prod*) lead to greater welfare losses and regressive impacts (Figure 5.6), and increase welfare losses in vulnerable households at risk of energy poverty.

5.5. Conclusions

Renewable energy promotion has become a policy priority for governments around the world because of its positive environmental effects. However, there is also concern about the effect that the entry of RES-E may have on the total costs of electricity production and how this is going to affect different social groups, firms and competitiveness. In this study we apply a computable general equilibrium (CGE) model in combination with a microsimulation (MS) model to examine the distributional implications of different schemes for financing the promotion of renewables. The schemes considered include exemptions from the RES-E surcharge on the price of electricity for producers or households and also alternatives where the cost of renewables is not financed through the electricity bill but from other tax sources such as fuel tax, VAT or transfers.

Our results provide evidence against the use of a surcharge on electricity prices to promote renewables. We show the consequences of including exemptions from the surcharge for producers and households. Despite the obvious gains for the agent exempted, both scenarios involve greater losses for the rest of the economy. These scenarios also show a trade-off between protecting sectoral output effects and protecting low income households. The exemptions on producers increase the negative effect on low income households (with respect to *BaU*). This can be alleviated with exemptions for consumers, but at the expense of doing more harm to energy-intensive industries. Moreover, both scenarios show the possible regressive impacts of increasing surcharges on electricity prices. The greater the financing efforts from households are when electricity surcharges are increased (*exe_prod*), the higher the welfare and regressive impacts are. However, exemptions on households (*exe_house*) relieve welfare impacts and correct undesirable regressive effects.

The change in the electricity sector plays a decisive role in explaining performance at sectoral and household levels. Hence, under the exemption for households electricity-intensive sectors are more severely affected as they get higher electricity costs. On the other hand, under exemptions for producers the ability of consumers to substitute other goods for electricity is lower, and thus the welfare impacts are worse. Given that low income households devote a greater proportion of their expenditure to electricity, higher electricity prices also entail greater regressive impacts.

Finally, our simulation results show the possible benefits of alternative ways of financing the promotion of RES-E. Lump-sum transfers and value added taxes can significantly attenuate adverse effects on production sectors (especially in energy-intensive industries) and at the same time reduce the regressive effects found in the other options. As the cost of promoting RES-E is not passed on to producers, both scenarios show an increase in output. Similarly, the excess burden is lower because the tax base is larger and thus, at the same time, the substitution options are greater. However, the option of increasing the price of fuel is less clear. All in all, our results show that there are general benefits when efforts to finance the promotion of RES-E is not defrayed by the electricity supply.

Chapter 6

Conclusions and Further Research

This dissertation has sought to contribute to the literature on the distributional implications of environmental and climate policies. Overall, we have analysed the distributional effects of different climate change mitigation measures in four case studies which are relevant to current policy debates. Here, we present the main conclusions of this analysis and suggest some areas for further research.

6.1. General conclusions

Although particular conclusions are presented in each of the four previous chapters, the following general conclusions can also be drawn:

(i) In general, climate mitigation policies reduce welfare and tend to be regressive. However, their overall welfare impacts and regressive effects are quite modest.

According to our results, climate mitigation policies increase the cost of inputs and goods, and thus reduce the overall welfare[51] of the economy. However, their welfare impacts are not as great as might be expected. In the foregoing chapters it can be observed that the welfare impacts are below 1% in average terms of equivalent variation. Even in chapter 4, where an ambitious climate policy in the US is introduced (with a 30% reduction in domestic emissions), the welfare impacts are around 0.5% in terms of equivalent variation.

Although the differences between low and high income groups are quite modest, it is shown that climate change mitigation measures tend to be regressive because they affect low-income households more. The chapters above incorporate specific indices for comparing the distributional impacts of climate mitigation policies. These statistics, such as the Gini coefficient or the Kakwani index, reduce the complexity of the analysis and make it possible to compare different measures. The results of these indices show that climate policies tend to be slightly regressive. The slight differences in the indices before and after climate measures are set show that there is a risk of regressivity but is typically very low. For example, in Chapter 3, before the measure analysed is set the Gini index is 0.2998, while afterwards it is 0.3002 (0.13% higher). From a policy viewpoint, the main barriers to climate mitigation

[51] In our various analyses welfare effects are reported using the well-known equivalent variation (EV) measure proposed by Hicks (1939).

policies include their perceived high welfare and regressive effects. However according to our results the welfare impacts are not as great as might be expected, and although climate change mitigation tends to be regressive the effect is quite modest.

(ii) Regressive impacts depend on economic structure and on the type of policy. Although climate change mitigation tends to be regressive, this conclusion cannot be taken as a rule.

As can be observed in our various analyses, distributional impacts depend on the following: (i) economic structure; and (ii) the policy analysed. In our applications, we show that economic structure is a key factor in distributional impacts. Chapters 2, 3 and 5 show that consumption patterns also play an important role in distributional impacts. Households that devote a large proportion of their expenditure to taxable goods lose more welfare. On the other hand, Chapter 4 shows that the economic structure is an essential factor in explaining the impacts of climate change mitigation on production. The industries impacts hinges predominantly on domestic demand, and thus the gains of protective measures may be more than compensated through losses in domestic. Production and industrial impacts in turn are important in understanding changes in income sources, and thus the welfare impacts per household (Chapter 5).

The various chapters of this thesis analyse the distributional impacts of different measures and reveal that the results differ considerably depending on economic structure and the type of policy. This idea is clear in Chapter 2, where two environmental issues are compared: local air pollution mitigation and global climate change mitigation. In that chapter, our results show that taxes on local pollutants are more regressive than those on greenhouse gas emissions. Chapter 3 shows that promoting a climate-friendly diet has regressive impacts because the measure taxes products and goods consumed more by low-income households. Finally, Chapter 5 sets some different scenarios for financing subsidies on renewables with different impacts in terms of incidence. Chapter 5 shows that renewable financing measures which do not increase the electricity price can significantly attenuate adverse industrial impacts and also reduce the welfare and regressive effects.

To conclude, it can be seen that climate policies tend to be regressive, but their effects can differ depending on economic structure and the type of policy. For example Chapter 2 shows for the case of Spain that carbon taxation tends to be proportional because the energy used in lighting and heating, consumed mainly by low-income households, is offset by higher spending on transport and energy by high-income households. Hence, the distributional implications depend very much on the structure of the economy and the type of policy, so distributional analysis should be considered in every policy proposal.

(iii) Policy design is a key factor in distributional implications and even can reduce potentially regressive impacts.

As with the type of policy, the way in which the various measures are introduced is an essential factor in distributional analysis. Some climate change mitigation measures tend to be regressive, but policy makers can modify the design of policies and even alter other taxes at the same time to increase the overall progressivity of the tax system. Throughout this thesis we have shown the significance of different mechanisms for reducing the negative impact of mitigation policies, such as revenue recycling, exemptions on critical goods and alternative financing measures. Chapter 2 shows that the overall welfare loss caused by global climate change and local air pollution taxes can be reduced notably when the revenues of environmental taxes are recycled. Chapter 3 demonstrates that the regressive effects of levying taxes on the consumption of food products can be relieved if exemptions on some basic food products are introduced. Moreover these modifications to the policy do not have a major impact on its main objective – i.e. of reducing GHG emissions. Finally, Chapter 5 shows that for the promotion of renewable energy, alternative financing designs can involve different welfare and distributional impacts. This conclusion is significant in terms of policy implications. We show that despite the possible regressive impacts of climate change mitigation measures well-designed policies can relieve welfare impacts and correct undesirable regressive effects.

(iv) Alternative measures are available for tackling climate change, but their distributional implications should be considered before they are implemented

As pointed out in the introduction and in Chapters 2 and 3, mitigation options for achieving targets have traditionally focused mostly on the energy and transport sectors and on global climate change pollutants. Although the energy and transport sectors are the largest contributors to climate change emissions, other sectors also account for large-scale emissions. On the other hand, environmental measures not directly related to carbon emissions can also play an important role in climate mitigation. For example, local air pollution seems to be linked to global climate change. Moreover, according to the relevant literature, some alternative measures can also deliver cost-effective emission reductions. Therefore, any policy aimed at mitigating emissions should also consider options that impact on other sectors and pollutants.

Chapters 2 and 3 explore the distributional consequences of two alternative measures in the form of local air pollution mitigation and the promotion of climate friendly diets, for the first time in this literature in the latter case. It is pointed out that these policy options can be effective instruments in reducing emissions related to climate change. However, our results show that these measures may tend to be regressive. One of the main reasons for introducing alternative measures is that they might be easier to implement because their effects (mainly on health) are felt more immediately by citizens than those implemented to reduce climate change. Our results reveal that this may not be the case if the distributional issue is factored into the policy maker's equation. However, as shown in our previous conclusion, the potential regressive effects are quite modest and could be reduced with suitable policy design, as observed with exemptions in Chapter 3 and revenue recycling in Chapter 2.

(v) The trade-off between efficiency and equity is one of the main challenges in the climate debate. The linking of methodologies may be an appropriate approach for investigating that trade-off.

Traditionally, economists have considered that policies makers must choose between efficiency and equity, which means that efficiency and equity goals cannot be achieved simultaneously. Although traditional literature asserts that climate protection leads to a trade-off between equity and efficiency (see for example Goulder and Parry, 2008), more recent studies have found that the trade-off between efficiency and equity can be overcome with appropriate policy design (see Imhof 2012). In this dissertation, most of our analyses show regressive impacts, highlighting this trade-off between equity and efficiency. However in chapter 2 we find that an efficient measure such as carbon taxation[52] can be distributionally neutral. Although there is normally a trade-off between efficiency and equity in mitigation measures, this conclusion cannot be taken as a general rule because it depends on the case study. Most of the literature has focused on efficiency criteria, although some authors find that equity matters to people as much as efficiency does in the design and delivery of environmental policy (Dietz and Atkinson, 2010). Therefore, even if there is a trade-off between efficiency and equity some level of efficiency loss in favour of equity may be advisable to promote climate policies.

Most of the literature that explores the trade-off between equity and efficiency is based on a first-best setting. However, this view might be overly simplistic for the real world (Labandeira and Linares, 2010). Numerical frameworks usually incorporate initial tax distortions such as factor taxes, intermediate input taxes, production taxes and subsidies, value-added taxes, import tariffs and export duties. In this

[52] From a first best perspective, the best policy instrument for reducing carbon emissions can be expected to be a carbon tax (see, for example, Newell and Pizer, 2008).

context, linking macro and micro models may be an appropriate approach for evaluating the trade-off between equity and efficiency. Micro models provide detailed information about households and the heterogeneity of the different economic agents. They thus enable us to increase the distributional analysis and to focus on those sectors and households most affected by policies. On the other hand, macro models enable the impacts of environmental policies to be assessed from efficiency-based and macro-economic perspectives. Therefore, the linking of macro and micro models enables equity and efficiency perspectives to be considered at the same time, as explained in Chapters 1 and 5. The linking of methodologies makes the analysis more complex, but it also enables policy measures to be analysed from multiple perspectives.

6.2. Further research

According to Fullerton (2009) there are six main different distributional effects of any policy, which can be summed up as follows: (i) cost to consumers; (ii) cost to producer or factors; (iii) benefits of scarcity rents; (iv) benefits of protection; (v) cost of transition; and (vi) effects on asset prices. Climate policies reduce the emissions produced through the production process, increasing the cost of production (ii). Similarly, this cost of production increases the prices of carbon intensive products, thus also entailing higher costs for consumers (i). Moreover, the cost of climate policies promotes an energy transition with which other costs and benefits are associated (iv). Although climate change mitigation measures can induce costs, there are also some co-benefits that should be taken into account, such as gains from environmental protection (iv) and the allocation of scarcity rents from a restricted number of permits. Finally, the aforesaid distributional effects can be capitalised into asset prices (vi) which also have distributional consequences. In our approaches we have tried to shift from analyses with only one distributional effect to a more complex analysis with multiple distributional impacts. Hence, Chapters 2 and 3 address the questions of the cost to consumers (i) and of who bears that cost. Chapter 4 analyses distributional effects on cost to producers (ii) and how costs are distributed across sectors. Finally, Chapter 5 seeks to combine the analysis of cost to consumers (i) and cost to producers (ii).

The benefits of scarcity rents (iii), cost of transition (v) and effects on asset prices (vi) are relevant effects in distributional analysis, but in regard to our case studies we consider the benefits of protection to be a key distributional effect that should be considered in future studies. The importance of introducing benefits from climate policies into future research is made clear in Chapters 2 and 3. The main limitation of these studies is that welfare impacts are analysed purely from an income perspective without accounting for the monetary health benefits associated with improving diets or local air pollution. Thus, although we find that these alternative measures are quite regressive, different findings could emerge if health benefits are factored in. In both cases there are many studies (see for instance Pye et al., 2005 for local air pollution and Martin et al. 2008 for diets) which show that air pollution and bad dietary habits affect low income households more and can thus benefit much more from these policies Thus, in future research factoring the benefits of regulation into distributional studies could help to enrich the analysis.

The different distributional effects of climate policy actions make distributional analysis much more difficult, and to quote Fullerton (2009) "a single study to incorporate all effects simultaneously would be very difficult, complex and likely infeasible". As shown here, linking methodologies can combine different distributional effects in a simple analysis (Chapter 5). In further studies, it would be good to expand the potential benefits of this approach and include all distributional impacts. In addition, linking methodologies enables environmental protection to be analysed from different perspectives, such as equity and efficiency. As pointed out in our conclusions, there is no clear consensus about the trade-off between equity and efficiency of climate policy actions. Thus, the linking of methodologies may be an appropriate approach for investigating climate protection in greater depth from both perspectives and for finding measures with progressive effects at a reasonable loss of efficiency.

6.3. Final Remark

The 21st Conference of Parties (COP21) to the United Nations Framework Convention on Climate Change in Paris in December 2015 marked an important milestone in international climate policy. The so-called Paris Agreement (UNFCCC, 2015) achieved a global consensus on keeping the global mean surface temperature increase to no more than 1.5 or 2 degrees Celsius compared to pre-industrial levels. In line with this temperature target not only industrialised countries but also developing countries signalled their willingness to reduce greenhouse gas emissions. Therefore, in the coming years, most countries are expected to introduce various new measures to tackle climate change.

In this context policy makers should ensure that environmental protection does not have adverse distributional impacts. In fact, only if policies follow the principle of justice and fairness will it be feasible and efficient to achieve environmental and climate protection. Otherwise, climate mitigation measures will be rejected by public opinion and attempts to tackle climate change will be unsuccessful. Finally, we believe that economists and environmental scientists should play an important role in finding these measures. Throughout this dissertation we have sought to make a small contribution to this important topic.

ANNEX A. FACTOR EMISSIONS

Food group	Emission factor (CO_2 per kg)	Food group	Emission factor (CO_2 per kg)
Cereals:			
Rice	3.82	Fruits:	
Bread and Other bread products	1.4	Citrus	0.9
Pasta	1.63	Bananas	0.6
		Apples and pears	0.66
Meats:		Fruit with stone	1.8
Beef	25.13	Olives	2.1
Pork	10.29	Berries	3.42
White meat	4.05	Other fruits	2.9
		Dried fruits	4.26
Fish:		Other processed fruits	2.46
Fresh Fish	2.68		
Frozen fish	12.2	Vegetables:	
Dried, smoked and salted fish	12.2	Vegetables	2.13
Other processed fish and shelfish	9.62	Tomatoes	4.2
		Mushroom	4.4
Milk	1.64	Tinned vegetables	2.34
		Frozen vegetables	3.06
Dairy products:		Prepared vegetables	2.73
Yogurt	3.87		
Cheese	13.65	Potatoes:	
Other dairy products	5.77	Potatoes	0.52
		Potato-based products	2.73
Eggs	4.9		
		Sweets and Sugar:	
Fats and oils:		Sugar	3.27
Butter	11.08	Chocolate	4.47
Margarine and other vegetable fats	2.43	Ice Cream	2.36
Oils	3.1	Other sugar products	3.33
Other animal fats	4.3		

ANNEX B. EQUIVALENT VARIATION

According to the consumer theory employed in the AIDS demand system and following Baker et al. (1989), the equivalent expenditure in logarithm form is:

$$lnG_e = \frac{b(p_r)}{b(p)} \ [lny - lna(p)] + lna\ (p^R) \qquad \text{[B.1]}$$

Where lny is the logarithm of food expenditure, and $b(p)$ and $lna(p)$ are:

$$b(p) = \ \prod_{i=l}^{14} p_i^{\beta_i} \qquad \text{[B.2]}$$

$$lna(p) = \alpha_i \sum_{i=l}^{14} lnp_i + \frac{1}{2}\sum_{i=l}^{14}\sum_{j=l}^{14} \gamma_{ij} lnp_i lnp_j \qquad \text{[B.3]}$$

Thus, the equivalent variation after the tax is:

$$lnG_e^1 = \frac{b(p_0)}{b(p_1)} \ [lny - lna(p_1)] + lna\ (p_0) \qquad \text{[B.4]}$$

As equivalent variation is defined as the amount of money that households would be willing to pay to prevent the occurrence of the price change caused by the tax increase. Equivalent variation is defined as:

$$VE = e^{lnG_e^1} - G \qquad \text{[B.5]}$$

ANNEX C: ALGEBRAIC SUMMARY OF THE COMPUTABLE GENERAL EQUILIBRIUM MODEL

The computable general equilibrium model is formulated as a system of nonlinear inequalities. The inequalities correspond to the two classes of conditions associated with a general equilibrium: (i) exhaustion of product (zero profit) conditions for producers with constant returns to scale; and (ii) market clearance for all goods and factors. The former class determines activity levels, and the latter determines price levels. In equilibrium, each variable is linked to one inequality condition: an activity level to an exhaustion of product constraint and a commodity price to a market clearance condition.

In our algebraic exposition, the notation Π_{ir}^{z} is used to denote the unit profit function (calculated as the difference between unit revenue and unit cost) for production with constant returns to scale of sector *i* in region *r*, where *z* is the name assigned to the associated production activity. Differentiating the unit profit function with respect to input and output prices provides compensated demand and supply coefficients (Hotelling's lemma), which appear subsequently in the market clearance conditions. We use *g* as an index comprising all sectors/commodities *i* (*g*=*i*), the final consumption composite (*g*=*C*), the public good composite (*g*=*G*), and investment composite (*g*=*I*). The index *r* (aliased with *s*) denotes regions. The index *EG* represents the subset of energy goods coal, oil, gas, electricity, and the label *FF* denotes the subset of fossil fuels coal, oil, gas. Tables C1–C6 explain the notations for variables and parameters employed within our algebraic exposition. Figures C1–C3 provide a graphical exposition of the production structure. Numerically, the model is implemented in GAMS (Brooke et al. 1996)[53] and solved using PATH (Dirkse and Ferris 1995)[54].

Zero Profit Conditions:

1. Production of goods except fossil fuels ($g \notin FF$):

$$\Pi_{gr}^{Y} = p_{gr} - \left[\theta_{gr}^{M} p_{gr}^{M(1-\sigma_{gr}^{KLEM})} + \left(1-\theta_{gr}^{M}\right) \left[\theta_{gr}^{E} p_{gr}^{E\,(1-\sigma_{gr}^{KLE})} + \left(1-\theta_{gr}^{E}\right) p_{gr}^{KL(1-\sigma_{gr}^{KLE})} \right]^{(1-\sigma_{gr}^{KLEM})/(1-\sigma_{gr}^{KLE})} \right]^{1/(1-\sigma_{gr}^{KLEM})} \leq 0.$$

[53] Brooke, A., D. Kendrick, and Meeraus, A. (1996). GAMS: A User's Guide. GAMS Development Corporation: Washington DC.
[54] Dirkse, S., and M. Ferris (1995). The PATH Solver: A Non-monotone Stabilization Scheme for Mixed Complementarity Problems. *Optimization Methods & Software* 5: 123–56.

2. Sector-specific material aggregate:

$$\prod_{gr}^{M} = p_{gr}^{M} - \left[\sum_{i \notin EG} \theta_{igr}^{MN} {p_{igr}^{A}}^{1-\sigma_{gr}^{M}} \right]^{1/(1-\sigma_{gr}^{M})} \leq 0.$$

3. Sector-specific energy aggregate:

$$\prod_{gr}^{E} = p_{gr}^{E} - \left[\sum_{i \in EG} \theta_{igr}^{EN} \left(p_{igr}^{A} + p_{r}^{CO_2} a_{igr}^{CO_2} \right)^{1-\sigma_{gr}^{E}} \right]^{1/(1-\sigma_{gr}^{E})} \leq 0.$$

4. Sector-specific value-added aggregate:

$$\prod_{gr}^{KL} = p_{gr}^{KL} - \left[\theta_{gr}^{K} v^{(1-\sigma_{gr}^{KL})} + \left(1-\theta_{gr}^{K}\right) w^{(1-\sigma_{gr}^{KL})} \right]^{1/(1-\sigma_{gr}^{KL})} \leq 0.$$

5. Production of fossil fuels (*g* $\in$ *FF*):

$$\prod_{gr}^{Y} = p_{gr} - \left[\theta_{gr}^{Q} q_{gr}^{1-\sigma_{gr}^{Q}} + (1-\theta_{gr}^{Q}) \left(\theta_{gr}^{L} w_{r} + \theta_{gr}^{K} v_{r} + \sum_{i \notin FF} \theta_{igr}^{FF} p_{igr}^{A} \right)^{1-\sigma_{gr}^{Q}} \right]^{1/(1-\sigma_{gr}^{Q})} \leq 0.$$

6. Armington aggregate:

$$\prod_{igr}^{A} = p_{igr}^{A} - \left(\theta_{igr}^{A} p_{ir}^{1-\sigma_{ir}^{A}} + (1-\theta_{igr}^{A}) {p_{ir}^{IM}}^{1-\sigma_{ir}^{A}} \right)^{1/(1-\sigma_{ir}^{A})} \leq 0.$$

7. Aggregate imports across import regions:

$$\prod_{ir}^{IM} = p_{ir}^{IM} - \left[\sum_{s} \theta_{isr}^{IM} \left(p_{is} \right)^{1-\sigma_{ir}^{IM}} \right]^{1/(1-\sigma_{ir}^{IM})} \leq 0.$$

Market Clearance Conditions:

8. Labor:

$$\overline{L}_{r} \geq \sum_{g} Y_{gr}^{KL} \frac{\partial \prod_{gr}^{KL}}{\partial w_{r}}.$$

9. Capital:

$$\overline{K}_{gr} \geq Y_{gr}^{KL} \frac{\partial \Pi_{gr}^{KL}}{\partial v_{gr}}.$$

10. Fossil fuel resources (g $\in$*FF*):

$$\overline{Q}_{gr} \geq Y_{gr} \frac{\partial \Pi_{gr}^{Y}}{\partial q_{gr}}.$$

11. Material composite:

$$M_{gr} \geq Y_{gr} \frac{\partial \Pi_{gr}^{Y}}{\partial p_{gr}^{M}}.$$

12. Energy composite:

$$E_{gr} \geq Y_{gr} \frac{\partial \Pi_{gr}^{Y}}{\partial p_{gr}^{E}}.$$

13. Value-added composite:

$$KL_{gr} \geq Y_{gr} \frac{\partial \Pi_{gr}^{Y}}{\partial p_{gr}^{KL}}.$$

14. Import composite:

$$IM_{ir} \geq \sum_{g} A_{igr} \frac{\partial \Pi_{igr}^{A}}{\partial p_{ir}^{IM}}.$$

15. Armington aggregate:

$$A_{igr} = Y_{gr} \frac{\partial \Pi_{gr}^{Y}}{\partial p_{igr}^{A}}.$$

16. Commodities (*g=i*):

$$Y_{ir} \geq \sum_{g} A_{igr} \frac{\partial \Pi_{igr}^{A}}{\partial p_{ir}} + \sum_{s \neq r} IM_{is} \frac{\partial \Pi_{is}^{IM}}{\partial p_{ir}}.$$

17. Private consumption composite (*g=C*):

$$Y_{Cr} p_{Cr} \geq w_r \overline{L}_r + \sum_{g} v_{gr} \overline{K}_{gr} + \sum_{i \in FF} q_{ir} \overline{Q}_{ir} + p_r^{CO_2} \overline{CO_2}_r + \overline{B}_r \quad .$$

18. Public consumption composite (*g=G*):

$$Y_{Gr} \geq \overline{G}_r \; .$$

19. Investment composite (*g=I*):

$$Y_{Ir} \geq \overline{I}_r \; .$$

20. Carbon emissions:

$$\overline{CO_2}_r \geq \sum_{g} \sum_{i \in FF} E_{gr} \frac{\partial \Pi_{gr}^{E}}{\partial \left(p_{igr}^{A} + p_r^{CO_2} a_{igr}^{CO_2} \right)} a_{igr}^{CO_2} \; .$$

Table C1. Indices (sets)

G	Sectors and commodities (*g=i*), final consumption composite (*g=C*), public good composite (*g=G*), investment composite (*g=I*)
I	Sectors and commodities
r (alias *s*)	Regions
EG	Energy goods: coal, crude oil, refined oil, gas, and electricity
FF	Fossil fuels: coal, crude oil, and gas

Table C2. Activity Variables

Y_{gr}	Production of item *g* in region *r*
M_{gr}	Material composite for item *g* in region *r*
E_{gr}	Energy composite for item *g* in region *r*
KL_{gr}	Value-added composite for item *g* in region *r*
A_{igr}	Armington aggregate of commodity *i* for demand category (item) *g* in region *r*

IM_{ir}	Aggregate imports of commodity *i* and region *r*

Table C3. Price Variables

p_{gr}	Price of item *g* in region *r*
p_{gr}^{M}	Price of material composite for item *g* in region *r*
p_{gr}^{E}	Price of energy composite for item *g* in region *r*
p_{gr}^{KL}	Price of value-added composite for item *g* in region *r*
p_{igr}^{A}	Price of Armington good *i* for demand category (item) *g* in region *r*
p_{ir}^{IM}	Price of import composite for good *i* in region *r*
w_{r}	Price of labor (wage rate) in region *r*
v_{ir}	Price of capital services (rental rate) in sector *i* and region *r*
q_{ir}	Rent to fossil fuel resources in region *r* ($i \in$ FF)
$p_{r}^{CO_2}$	Carbon value in region *r*

Table C4. Endowments and Emissions Coefficients

$\overline{L}_r$	Aggregate labor endowment for region *r*
$\overline{K}_{ir}$	Capital endowment of sector *i* in region *r*
$\overline{Q}_{ir}$	Endowment of fossil fuel resource *i* for region *r* ($i \in$FF)
$\overline{B}_r$	Initial balance of payment deficit or surplus in region *r* (note: $\sum_r \overline{B}_r = 0$)
$\overline{CO_{2r}}$	Endowment of carbon emissions rights in region *r*
$a_{igr}^{CO_2}$	Carbon emissions coefficient for fossil fuel *i* in demand category *g* of region *r* ($i \in FF$)

Table C5. Cost Shares

θ_{gr}^{M}	Cost share of the material composite in production of item *g* in region *r*
θ_{gr}^{E}	Cost share of the energy composite in the aggregate of energy and value-added of item *g* in region *r*

θ^{MN}_{igr}	Cost share of the material input *i* in the material composite of item *g* in region *r*
θ^{EN}_{igr}	Cost share of the energy input *i* in the energy composite of item *g* in region *r*
θ^{K}_{gr}	Cost share of capital within the value-added of item *g* in region *r*
θ^{Q}_{gr}	Cost share of fossil fuel resource in fossil fuel production ($g \in FF$) of region *r*
θ^{L}_{gr}	Cost share of labor in non-resource inputs to fossil fuel production ($g \in FF$) of region *r*
θ^{K}_{gr}	Cost share of capital in non-resource inputs to fossil fuel production ($g \in FF$) of region *r*
θ^{FF}_{igr}	Cost share of good *i* in non-resource inputs to fossil fuel production ($g \in FF$) of region *r*
θ^{A}_{igr}	Cost share of domestic output *i* within the Armington item *g* of region *r*
θ^{M}_{isr}	Cost share of exports of good *i* from region *s* in the import composite of good *i* in region *r*

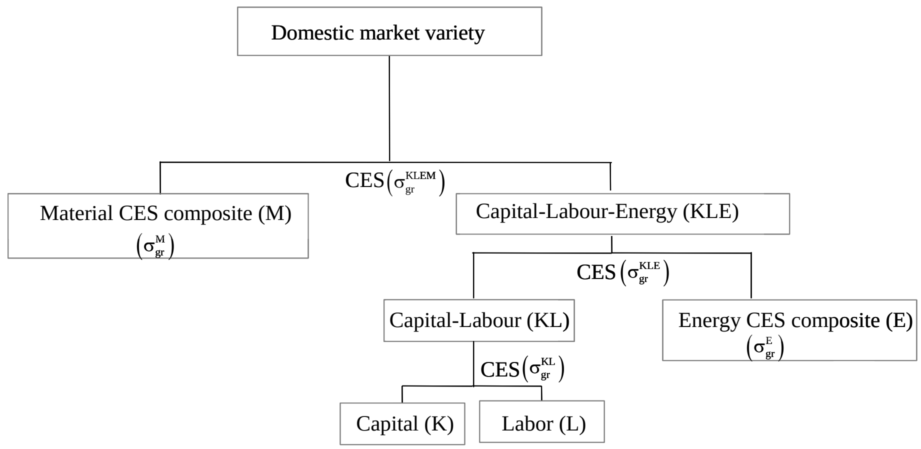
Table C6. Elasticities

σ^{KLEM}_{gr}	Substitution between the material composite and the energy value–added aggregate in the production of item *g* in region *r**
σ^{KLE}_{gr}	Substitution between energy and the value-added nest of production of item *g* in region *r**
σ^{M}_{gr}	Substitution between material inputs within the energy composite in the production of item *g* in region *r**
σ^{KL}_{gr}	Substitution between capital and labor within the value-added composite in the production of item *g* in region *r**
σ^{E}_{gr}	Substitution between energy inputs within the energy composite in the production of item *g* in region *r* (by default: 0.5)
σ^{Q}_{gr}	Substitution between natural resource input and the composite of other inputs in fossil fuel production ($g \in$ FF) of region *r* (calibrated consistently to exogenous supply elasticities)
σ^{A}_{ir}	Substitution between the import composite and the domestic input to Armington production of good *i* in region *r***
σ^{IM}_{ir}	Substitution between imports from different regions within the import composite for good *i* in region *r***

*See Okagawa, A., and K. Ban. 2008. Estimation of Substitution Elasticities for CGE Models. Mimeo. Osaka, Japan: Osaka University.

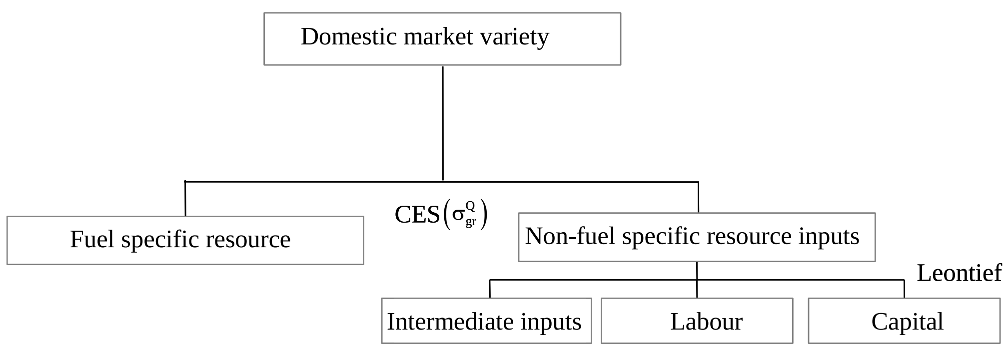
**See Narayanan, G.,B., Aguiar, A., and Robert McDougall, Eds. 2015. Global Trade, Assistance, and Production: The GTAP 9 Data Base, Center for Global Trade Analysis, Purdue University.

Figure C1. Nesting in Production (Except Fossil Fuels)



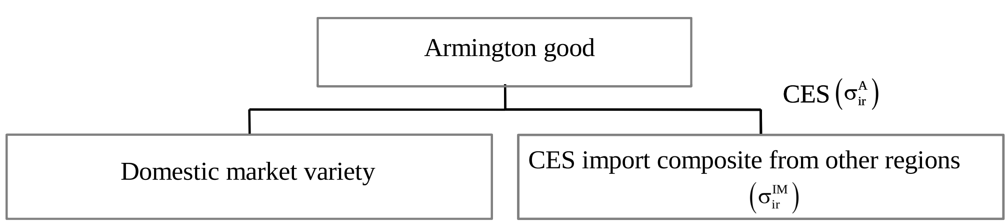
Note: CES=constant elasticity of substitution.

Figure C2. Nesting in Fossil Fuel Production



Note: CES=constant elasticity of substitution.

Figure C3. Nesting in Armington Production



Note: CES=constant elasticity of substitution.

Annex D: Almost Ideal Demand System, estimated as a seemingly unrelated regression

Table D1: Almost Ideal Demand System, estimated as a seemingly unrelated regression, estimates rounded to 3 digits.

	food	Housing	Fuel	Electricity	Heat	Transport	Education and Leisure	Durables
ln (p_food)	0.030**	-0.007	-0.018*	-0.001	0.001*	0.014	-0.057*	-0.011
ln(p_housing)	-0.007	0.176*	-0.012*	0.011*	0.001	-0.003	-0.052*	-0.110*
ln(p_fuel)	-0.018*	-0.012*	0.029*	-0.001*	-0.001*	-0.014*	-0.017*	0.037*
ln(p_electricity)	-0.001	0.011*	-0.001*	0.015*	-0.001*	-0.004*	-0.008*	-0.007*
ln(p_heat)	0.001*	0.001	-0.001*	-0.001*	0.006*	-0.001*	-0.003*	-0.002*
ln(p_transport)	0.014	-0.004	-0.014*	-0.003*	-0.001*	0.042*	0.013	-0.017
ln(p_leisure & education)	-0.057*	-0.052*	-0.017*	-0.008*	-0.002*	0.013	0.131*	-0.014
ln(p_durables)	-0.011	-0.110*	0.037*	-0.007*	-0.002*	-0.017	-0.015	0.097*
ln(p_other goods)	0.048*	-0.001	-0.003	-0.006*	-0.001*	-0.029*	0.008	0.028**

*Statistically significant at the 5% level.

** Statistically significant at the 10% level

Bibliography

Abadie, L.M., Galarraga, I., Milford, A.B., Gustavsen, G.W., n.d. Using food taxes and subsidies to achieve emission reduction targets in Norway. *J. Clean. Prod*. doi:10.1016/j.jclepro.2015.09.054

Abderraouf Laajimi , Azucena Gracia & Luis Miguel Albisu, (1997). The Demand for Food in Spain. *Journal of International Food & Agribusiness Marketing*, 9:2, 1-17, DOI: 10.1300/J047v09n02_01

Abidin, M.Z. and Loke, W.H. (2008). Revealed Comparative Advantage of Malaysian Exports: The Case for Changing Export Composition. *Asian Economic Papers*, 7 (3), 130–147.

ACA (2016). 3er Estudio Pobreza Energética en España - Nuevos Enfoques de Análisis. *Asociación de Ciencias Ambientales*, ACA.

Aiginger, K. (2006). Competitiveness: From a Dangerous Obsession to a Welfare Creating Ability with Positive Externalities. *Journal of Industry, Competition and Trade*, 6(2), 161–177.

Aigner, R. (2014). Environmental Taxation and Redistribution Concerns. FinanzArchiv / *Public Finance Analysis* 70, June 2014, pp. 249-277(29)

Alexeeva-Talebi, V., and Böhringer, C. (2012). Unilateral Climate Policy and Competitiveness: The Implications of Differential Emission Pricing. *The World Economy*, 36 (2), 121–154.

Angulo, A.M., N. Mtimet and J.M. Gil, (2008). Análisis de la demanda de alimentos en España considerando el impacto de la dieta sobre la salud. *Economía Agraria y Recursos Naturales*, 8, 2, pp. 3-30.

Armington, P.S. (1969). A Theory of Demand for Producers Distinguished by Place of Production. *IMF Staff Papers*, 16 (1), 159–78.

Arto, I., Genty, A., Rueda-Cantuche, J.M, Villanueva, A., and Andreoni, V. (2012). Global resources use and pollution, volume 2/ Country Factsheets. *JRC scientific and policy reports*. Report EUR 25462. Joint Research Centre.

Babiker, M.H., and Rutherford, T.F. (2005). The economic effects of border measures in subglobal climate agreements. *The Energy Journal*, 26 (4), 99–126.

Bajželj, B., Richards, K.S., Allwood, J.M., Smith, P., Dennis, J.S., Curmi, E., and Gilligan, C.A. (2014). Importance of food-demand management for climate mitigation. *Nature. Clim. Change* 4, 924–929.

Balassa, B. (1962), Recent Developments in the Competitiveness of American Industry and Prospects for the Future. In: Joint Economic Committee – *Congress of the United States* (eds.), Factors Affecting the United States Balance of Payments, Washington, 27-54.

Ballance, R.H., Forstner, H. and Murray, T. (1987). Consistency Tests of Alternative Measures of Comparative Advantage. *The Review of Economics and Statistics*, 69 (1), 157–61.

Bank of Spain (2014). Informe anual 2013. Madrid. *Bank of Spain*

Barker, T., and Köhler, J. (1998). Equity and ecotax reform in the EU: achieving a 10 per cent reduction in CO2 emissions using excise duties. *Fiscal Studies* 19 (4), 375–402.

Barker, T., and Rosendahl, K.E. (2000). Ancillary benefits of GHG mitigation in Europe: SO2, NOX and PM10 reductions from policies to meet Kyoto targets using the E3ME model and externe valuations. Paris: *Organization Economic Cooperation & Development*.

Berrens, R.P., Bohara, A.K., Jenkins-Smith, H.C., Silva, C.L., and Weimer, D.L. (2004). Information and effort in contingent valuation surveys: application to global climate change using national internet samples. *Journal of Environmental Economics and Management* 47, 331–363. doi:10.1016/S0095-0696(03)00094-9

Bilgic, A., and Yen, S.T. (2013). Household food demand in Turkey: A two-step demand system approach. *Food Policy* 43, 267–277. doi:10.1016/j.foodpol.2013.09.004

Böhringer, C., Balistreri, E.J., and Rutherford, T.F. (2012a). The Role of Border Carbon Adjustment in Unilateral Climate Policy: Overview of an Energy Modeling Forum Study (EMF29). *Energy Economics* 34, 97-110.

Böhringer, C., Carbone, J.C., and Rutherford, T. F. (2012b). Unilateral Climate Policy Design: Efficiency and Equity Implications of Alternative Instruments to Reduce Carbon Leakage. *Energy Economics* 34, 208-217.

Böhringer, C., Ferris, M., and Rutherford, T.F. (1998). Alternative CO2 Abatement Strategies for the European Union. C*limate Change, Transport and Environmental Policy* (edited by J.B. Braden and S. Proost), Northampton, Massachusetts: Edward Elgar Publishing, 16–47.

Böhringer, C., Fischer, C., and Rosendahl, K.E. (2014b). Cost-effective unilateral climate policy design: Size matters. *Journal of Environmental and Economic Management*, 67, 318–339.

Böhringer, C., Fischer, C., and Rosendahl, K.E. (2010). The Global Effects of Subglobal Climate Policies. The B.E. *Journal of Economic Analysis&Policy*, 10 (2) (Symposium), Article 13.

Böhringer, C., Hoffmann, T., and Rutherford, T.F. (2007). Alternative Strategies for Promoting Renewable Energy in EU Electricity Markets. *Applied Economics Quarterly.* 58 Supplement, 9-26.

Böhringer, C., Landis, F., and Tovar-Reaños, M.A (2016). Cost-Effectiveness and Incidence of Renewable Energy Promotion in Germany (SSRN Scholarly Paper No. ID 2854687). *Social Science Research Network*, Rochester, NY.

Böhringer, C., Lange, A., and Rutherford, T.F. (2014a). Optimal Emission Pricing in the Presence of International Spillovers: Decomposing Leakage and Terms-of-Trade Motives, *Journal of Public Economics*, 110, 101–111.

Böhringer, C., Löschel, A., and Welsch, H., (2008). Environmental Taxation and Induced Structural Change in an Open Economy: The Role of Market Structure. *German Economic Review* 9, 17–40. doi:10.1111/j.1468-0475.2008.00421.x

Böhringer, C. and Rutherford, T. (2006). Combining Top-Down and Bottom-up in Energy Policy Analysis: A Decomposition Approach. *ZEW Discussion Paper* No. 06-007.

Böhringer, C., and Rutherford, T.F. (2010). The Costs of Compliance: A CGE Assessment of Canada's Policy Options under the Kyoto Protocol. *The World Economy*, 33 (2), 177–211.

Bollen, J., van der Zwaan, B., Brink, C., and Eerens, H. (2009). Local air pollution and global climate change: A combined cost-benefit analysis. *Resources and Energy Economics*, 31, 161–181.

Boscá, J.E., Doménecha, R., and Ferria, J. (2014). Fiscal Devaluations in EMU. Hacienda Pública Española / *Revista de Economía Pública*, 206, 27-57.

Bouamra-Mechemache, Z., Réquillart, V., Soregaroli, C., and Trévisiol, A. (2007). Demand for Dairy Products in the EU (IDEI Working Paper No. 454). *Institut d'Économie Industrielle* (IDEI), Toulouse.

Bovenberg, A. and R. de Mooij (1994). Environmental levies and distortionary taxation. *American Economic Review* 84, 1085–1089.

Bovenberg, A. (1999). Green tax reforms and the double dividend: an updated reader's guide. *International Tax and Public Finance* 6, 421–443.

Brannlund, R., and Nordstrom, J. (2004). Carbon tax simulations using a household demand model. *European Econonomic Review*, 48, 211–233.

Bristow, A.L., Wardman, M., Zanni, A.M., and Chintakayala, V.P.K. (2010). Public acceptability of personal carbon trading and carbon tax. Ecological Economics doi:10.1016/j.ecolecon.2010.04.021

Buñuel M. (2011) El precio de la electricidad y la política de cambio climático: ¿qué papel puede jugar un impuesto sobre el carbono en España? /The Price of Electricity and Climate Change Policy: What Role May a Carbon Tax Play in Spain? *Estudios de Economía Aplicada*. 29.

Carlin, W., Glyn, A. and Reenen, J.V. (2001). Export Market Performance of OECD Countries: An Empirical Examination of the Role of Cost Competitiveness. *The Economic Journal*, 111 (468), 128–62.

CASES (2006). External Cost per unit emission. *Cost Assessment of Sustainable Energy Systems*. www.feem-project.net/cases/index.php

Cendra, J.D. (2006). Can Emissions Trading Schemes be Coupled with Border Tax Adjustments? An Analysis vis-à-vis WTO Law. *Review of European Community & International Environmental Law*, 15: 131–145. doi: 10.1111/j.1467-9388.2006.00518.x

CEOE. (2014) Publicaciones CEOE: Energía y competitividad : propuestas del sector empresarial. *CEOE* 2014.

Copeland, B.R. (1996). Pollution content tariffs, environmental rent shifting, and the control of cross-border pollution. *Journal of International Economics*, 40 (3), 459-476.

De Bruyn, S., Nelissen, D. and Koppman, M. (2013). Carbon leakage and the future of the EU ETS market - Impact of recent development in the EU ETS on the list of sectors deemed to be exposed to carbon leakage. *Delft*, CE Delft, April 2013.

De Mooij.R and Keen, M. (2012). 'Fiscal Devaluation' and Fiscal Consolidation: The VAT in Troubled Times. NBER Working Papers17913*, National Bureau of Economic Research*, Inc.

Deaton, A. and Muellbauer, J. (1980): An almost Ideal Demand System. *American Economic Review*, 70, 312–326.

Del Río González, P. (2008). Policy implications of potential conflicts between short-term and long-term efficiency in CO2 emissions abatement. *Ecological Economics*, 65, 292–303.

Del Río, P., and Mir-Artigues, P. (2014). Combinations of Support Instruments for Renewable Electricity in Europe: A Review. *Renewable and Sustainable Energy Reviews* 40 (December): 287–95. doi:10.1016/j.rser.2014.07.039.

Del Rio, P., Mir-Artigues, P. and Trujillo-Baute, E. (2016) Analysing the Impact of Renewable Energy Regulation on Retail Electricity Prices. *IEB Working Paper* 2016/19

Demisse Gemechu, E; Burtnar, I.; LLop, M. and Castells, F. (2014). Economic and Environmental Effects of CO2 taxation; an Input-Output analysis for Spain. *Journal of Environmental Planning and Management*, 57, 751-768.

Dietz S., and Atkinson G. (2010) The equity-efficiency trade-off in environmental policy: evidence from stated preferences. *Land Economics* 86:423-443

Dissou, Y., and Eyland, T. (2011). Carbon control policies, competitiveness, and border tax. *Energy Economics*, 33 (3), 556–564.

Duarte, R., Mainar, A., and Sanchez-Choliz, J. (2012). Social groups and CO2 emissions in Spanish households. *Energy Policy* 44, 441–450.

EC (2006). Environmental Impact of Products (EIPRO): Analysis of the Life Cycle Environmental Impacts Related to the Total Final Consumption of the EU 25 (2006). *European Commission Technical Report* EUR 22284 EN.

EC (2011). Roadmap for moving to a low-carbon economy in 2050, COM/2011/0112, *European Commission*, Brussels.

EC (2014). A policy framework for climate and energy in the period from 2020 up to 2030, COM/2014/15, *European Commission*, Brussels.

Edjabou, L.D., and Smed, S. (2013). The effect of using consumption taxes on foods to promote climate friendly diets – The case of Denmark. *Food Policy* 39, 84–96. doi:10.1016/j.foodpol.2012.12.004

Ekins, P., Pollitt, H., Barton, J., and Blobel, D. (2011).The implications for households of environmental tax reform (ETR) in Europe. *Ecological Economics*, 70, 2472–2485.

Engler. P, Ganelli. G, Tervala. J, and Voigts. S, (2014). Fiscal Devaluation in a Monetary Union. *IMF Working Papers* 14/201, International Monetary Fund

EPA (2013). Technical Update of the Social Cost of Carbon for Regulatory Impact Analysis*. United States Environmental Protection Agency*, Washington DC

EU (2003). European Commission, (Directive 2003/87/EC of the European Parliament and of the Council of 13 October 2003 establishing a scheme for greenhouse gas emission allowance trading within the Community and amending Council Directive 96/61/EC. (ETS Directive, Article 10a)

Eurostat (2016) Renewable energy statistics. 2016. http://ec.europa.eu/eurostat/statistics-explained/index.php/Renewable_energy_statistics

Feng, K., Hubacek, K., Guan, D., Contestabile, M., Minx, J., and Barrett, J. (2010). Distributional Effects of Climate Change Taxation: The Case of the UK. *Environmental Sciecnce. Technology*. 44, 3670–3676.

Fertö, I. and Hubbard, L.J. (2003). Revealed Comparative Advantage and Competitiveness in Hungarian Agri–Food Sectors. *The World Economy*, 26 (2), 247–59.

Fischer, C. (2001). Rebating Environmental Policy Revenues: Output-Based Allocations and Tradable Performance Standards. *RFF Discussion Paper 01-22*, Washington, D.C., Resources for the Future.

Fischer, C. (2010). Renewable Portfolio Standards: When Do They Lower Energy Prices? *The Energy Journal* 31 (1): 101–19.

Fischer, C., and Fox, A.K. (2012). Comparing policies to combat emissions leakage: border carbon adjustments versus rebates. *Journal of Environmental Economics and Management*, 64 (2), 199–216.

Fischer, C. and Preonas, L. (2010), Combining Policies for Renewable Energy: Is the Whole Less Than the Sum of Its Parts? *International Review of Environmental and Resource Economics*: Vol. 4: No. 1, pp 51-92. http://dx.doi.org/10.1561/101.00000030

FS and UNEP (2016) Global Trends in Renewable Energy Investment 2016. *Frankfurt School-UNEP Centre*/BNEF, Frankfurt am Main.

Fullerton, D. (2009). Distributional Effects of Environmental and Energy Policy: An Introduction. Working Paper 14241. *National Bureau of Economic Research*. http://www.nber.org/papers/w14241.

Garnett, T. (2011). Where are the best opportunities for reducing greenhouse gas emissions in the food system (including the food chain)? *Food Policy* 36, Supplement 1, S23–S32.

Gelabert, L., Labandeira, X. and Linares, P. (2011). An Ex-Post Analysis of the Effect of Renewables and Cogeneration on Spanish Electricity Prices. *Energy Economics*, Supplemental Issue: Fourth Atlantic Workshop in Energy and Environmental Economics, 33, Supplement 1 (December): S59–65. doi:10.1016/j.eneco.2011.07.027.

Genty, A., Arto, I., and Neuwahl, F. (2012) Final database of environmental satellite accounts: technical report on their compilation. *WIOD Documentation*.

Godfray, H.C.J., Beddington, J.R., Crute, I.R., Haddad, L., Lawrence, D., Muir, J.F., Pretty, J., Robinson, S., Thomas, S.M., and Toulmin, C. (2010). Food Security: The Challenge of Feeding 9 Billion People. *Science* 327, 812–818.

Gonzalez, F. (2012). Distributional effects of carbon taxes: The case of Mexico. *Energy Economics*, 34, 2102–2115.

Gorton, M., Davidova, S. and Ratinger, T. (2000). The Competitiveness of Agriculture in Bulgaria and the Czech Republic vis-a`-vis the European Union. *Comparative Economic Studies*, 42 (1), 59–86.

Goulder, L. (1995). Environmental taxation and the double dividend: A reader's guide. *International Tax and Public Finance* 2, 157.

Goulder, L. and Parry, I. (2008). Instrument Choice in Environmental Policy. *Review of Environmental Economics and Policy*, 2:152-174

Graham, P., Thorpe, S., and Hogan, L. (1999). Non-competitive market behaviour in the international coking coal market. *Energy Economics*, 21 (3), 195–212.

Green, R., Milner, J., Haines, A., Chalabi, Z., Markandya, A., Spadaro, J., and Wilkinson, P. (2015). The potential to reduce greenhouse gas emissions in the UK through healthy and realistic dietary change. *Climatic Change*. 1-13.

Grilo, I. and G. J. Koopman (2006). Productivity and Microeconomic Reforms: Strengthening EU Competitiveness, Special Issue on Competitiveness. *Journal of Industry, Competition and Trade*, 6 (2), 67–84.

Gustavsen, G.W. and Rickertsen, K. (2013). Adjusting VAT rates to promote healthier diets in Norway: A censored quantile regression approach. *Food Policy* 42, 88–95. doi:10.1016/j.foodpol.2013.07.001

Hallström, E., Carlsson-Kanyama, A., and Börjesson, P. (2015). Environmental impact of dietary change: a systematic review. *J. Clean. Prod*. 91, 1–11. doi:10.1016/j.jclepro.2014.12.008.

Härkänen, T., Kotakorpi, K., Pietinen, P., Pirttilä, J., Reinivuo, H., and Suoniemi, I. (2014). The welfare effects of health-based food tax policy. *Food Policy* 49, Part 1, 196–206. doi:10.1016/j.foodpol.2014.07.001

Hedenus, F., Wirsenius, S., and Johansson, D.J.A. (2014). The importance of reduced meat and dairy consumption for meeting stringent climate change targets. *Climatic Change* 124, 79–91. doi:10.1007/s10584-014-1104-5

Hicks, J.R. (1939), 2nd ed. (1946). Value and Capital: An Inquiry into Some Fundamental Principles of *Economic Theory*. Oxford: Clarendon Press.

Hoel, M. (1991). Global Environmental Problems: The Effects of Unilateral Actions Taken by One Country. *Journal of Environmental Economics and Management*, 20, 55–70.

Hoel, M. (1996). Should a carbon tax be differentiated across sectors? *Journal of Public Economics*, 59, 17–32.

Holland, S. P., Hughes, J. E., and Knittel, C. R. (2009). Greenhouse Gas Reductions under Low Carbon Fuel Standards? *American Economic Journal: Economic Policy*, 1 (1), 106–46.

Holland, S.P. (2012). Emissions Taxes Versus Intensity Standards: Second-Best Environmental Policies with Incomplete Regulation. *Journal of Environmental Economics and Management*, 63 (3), 375–387.

Hoolohan, C., Berners-Lee, M., McKinstry-West, J., and Hewitt, C.N. (2013). Mitigating the greenhouse gas emissions embodied in food through realistic consumer choices. *Energy Policy* 63, 1065–1074. doi:10.1016/j.enpol.2013.09.046

IEA (2015) Energy Policies of IEA Countries: Spain2015 Review, OECD/IEA, *International Energy Agency*, Paris.

IEA (2016) Key Renewables Trends, IEA, *International Energy Agency*, Paris.

Imhof, J. (2012). Fuel Exemptions , Revenue Recycling , Equity and Efficiency : Evaluating Post-Kyoto Policies for Switzerland. *Swiss Society of Economics and Statistics* 148(1), 197–227.

INE (2013a), Contabilidad Nacional de España. Base 2000. Marco input-output. Tabla simétrica, Año 2005. *Instituto Nacional de Estadística*. www.ine.es.

INE (2013b). Cuentas satélite sobre emisiones atmosféricas. Año 2005. *Instituto Nacional de Estadística*. www.ine.es.

INE (2013c). Encuesta continua de presupuestos familiares, base 1997. *Instituto Nacional de Estadística*. www.ine.es.

INE (2014). Encuesta continua de presupuestos familiares, base 1997. *Instituto Nacional de Estadística*. www.ine.es.

INE (2016a). Contabilidad Nacional de España. Base 2002. Marco input-output. Tabla Input-Output, Año 2007. *Instituto Nacional de Estadística*. www.ine.es.

INE (2016b). Encuesta continua de presupuestos familiares, base 1997. *Instituto Nacional de Estadística*. www.ine.es

International Monetary Fund (2014). Spain: 2014 Article IV Consultation Concluding Statement of the Mission Madrid, May 27, 2014. http://www.imf.org/external/np/ms/2014/052714.htm

IPCC (2012), Renewable Energy Sources and Climate Change Mitigation: Special Report of the Intergovernmental Panel on Climate Change. *Cambridge University Press*, Cambridge.

IPPC (2014). Agriculture, Forestry and Other Land Use (AFOLU). In: Climate Change 2014: Mitigation of Climate Change. Contribution of Working Group III, *Cambridge University Press,* Cambridge.

IRENA (2016), Renewable Energy Benefits: Measuring The Economics. *International Renewable Energy Agency*, Abu Dhabi.

Ismer, R. and Neuhoff, K. (2007). Border tax adjustment: a feasible way to support stringent emission trading. *European Journal of Law and Economics*, 24 (2), 137-164.

Jensen, J.D., and Smed, S. (2013). The Danish tax on saturated fat – Short run effects on consumption, substitution patterns and consumer prices of fats. *Food Policy* 42, 18–31. doi:10.1016/j.foodpol.2013.06.004

Kakwani, N. C. (1977a), Applications of Lorenz curves in economic analysis, *Econometrica*, 45, 719-727.

Kakwani, N. C. (1977b), Measurement of Progressivity: An International Comparison. *The Economic Journal*, 87, 71-80

Kerkhof, A.C., Moll, H.C., Drissen, E., and Wilting, H.C. (2008). Taxation of multiple greenhouse gases and the effects on income distribution - A case study of the Netherlands. *Ecological Economics*. 67, 318–326.

Kiulia,O., Markandya, A., Scasny,M. and Menkyna Tsuchimoto, F. (2013). The Economic and Environmental Effects of Taxing Air Pollutants and CO2: Lessons from a Study of the Czech Republic. *Working Paper. Charles University Environment Centre*.

Koesler, S., and Schymura, M. (2015). Substitution Elasticities in a Constant Elasticity of Substitution Framework – Empirical Estimates Using Nonlinear Least Squares. *Economic Systems Research* 27, 101–121. doi:10.1080/09535314.2014.926266

Kravis, I.B. and Lipsey, R.E. (1992). Sources of Competitiveness of the United States and of its Multinational Firms. *The Review of Economics and Statistics*, 74 (2), 193–201.

Krichene, N. (2002). World crude oil and natural gas: a demand and supply model. *Energy Economics*, 24 (6), 557–576.

Labandeira, X., and Labeaga, J. (1999). Combining input-output analysis and micro-simulation to assess the effects of carbon taxation on Spanish households. *Fiscal Studies*, 20, 305–320.

Labandeira, X., Labeaga, J.M., and Rodríguez, M. (2004). Green tax reforms in Spain. European. Environment, 14, 290–299.

Labandeira, X., and Linares, P. (2010). Second-best instruments for energy and climate policy. Working Papers 06-2010, *Economics for Energy*.

Lassaletta, L., Billen, G., Romero, E., Garnier, J., and Aguilera, E., (2013). How changes in diet and trade patterns have shaped the N cycle at the national scale: Spain (1961–2009). *Reg Environ Change*.

Leicester, A. and Windmeijer, F. (2004). The ‘fat tax’: economic incentives to reduce obesity. *Institute for Fiscal Studies*, London.

Li, H., Berrens, R.P., Bohara, A.K., Jenkins-Smith, H.C., Silva, C.L., and Weimer, D.L. (2004). Would developing country commitments affect US households’ support for a modified Kyoto Protocol? *Ecological Economics* 48, 329–343. doi:10.1016/j.ecolecon.2003.10.010

Markandya, A. (2011). Equity and Distributional Implications of Climate Change. *World Development*, Microfinance: Its Impact, Outreach, and Sustainability Including Special Section (pp. 983-1060)

Markandya, A., Bigano, A., Prochina, and Roberto (2010). The Social Cost of Electricity. Scenarios and Policy Implications. *Fondazione Eni Enrico Mattei, Milan*.

Markandya, A., González-Eguino, M., and Escapa, M. (2013). From shadow to Green: linking environmental fiscal reforms and the informal economy. *Energy Economics*, 40, S108- S118.

Markusen, J. R. (1975). International externalities and optimal tax structures. *Journal of International Economics*, 5 (1), 15–29.

Martín, A.R., Nieto, J.M.M., Ruiz, J.P.N., and Jiménez, L.E. (2008). Overweight and obesity: The role of education, employment and income in Spanish adults. *Appetite* 51, 266–272. doi:10.1016/j.appet.2008.02.021

Mattoo, A., Subramanian, A., van der Mensbrugghe, D., and He, J. (2009). Reconciling Climate Change and Trade Policy. *CGD Working Paper 189*, Washington, D.C., Center for Global Development.

McColl, K. (2009). “Fat taxes” and the financial crisis. *The Lancet* 373, 797–798. doi:10.1016/S0140-6736(09)60463-3

McKibben, W. and Wilcoxen, P.J. (2009). The economic and environmental effects of border carbon adjustments for climate policy. *Working Paper in International Economics No. 1.09*, Lowy Institute for International Economics.

McMichael, A.J., Powles, J.W., Butler, C.D., and Uauy, R. (2007). Food, livestock production, energy, climate change, and health. *The Lancet* 370, 1253–1263.

Mergenthaler, M., Weinberger, K., and Qaim, M., (2009). The food system transformation in developing countries: A disaggregate demand analysis for fruits and vegetables in Vietnam. *Food Policy* 34, 426–436. doi:10.1016/j.foodpol.2009.03.009

Metcalf, G. (1999). A distributional analysis of green tax reforms. *National Tax Journal*, 52, 655–681.

Metcalf, G.E., Mathur, A., and Hassett, K.A. (2010). Distributional Impacts in a Comprehensive Climate Policy Package. *National Bureau of Economic Research*.

Mir-Artigues, P., Cerdá, E., and del Río, P. (2015). Analyzing the Impact of Cost-Containment Mechanisms on the Profitability of Solar PV Plants in Spain. *Renewable and Sustainable Energy Reviews* 46 (June): 166–77. doi:10.1016/j.rser.2015.02.005.

Moreiras, O., Carbajal, Á., Cabrera, L., and Cuadrado, C. (2011). Tablas de composición de alimentos. *Ediciones Pirámide SA*. Madrid. Ed. 15ª.

MRC (2011). Using natural experiments to evaluate population health interventions: guidance for producers and users of evidence. *Medical Research Council*.

Mytton, O.T., Clarke, D., and Rayner, M., (2012). Taxing unhealthy food and drinks to improve health. BMJ 344, e2931. doi:10.1136/bmj.e2931

Narayanan, B. G. and Steinbuks, J. (2014). Fossil Fuel Producing Economies have Greater Potential for Interfuel Substitution. *GTAP Working Paper No. 73*, Center for Global Trade Analysis, Purdue University.

Narayanan, B. G., Aguiar, A. and McDougall, R. (2015). Global Trade, Assistance, and Production: The GTAP 9 Data Base. *Center for Global Trade Analysis,* Purdue University.

Neuhoff, K., Bach, S., Diekmann, J., Beznoska, M., and El-Laboudy. T. (2013). Distributional Effects of Energy Transition: Impacts of Renewable Electricity Support in Germany. *Economics of Energy & Environmental Policy* 2 (1). doi:10.5547/2160-5890.2.1.3.

Newell, R.G., and Pizer W.A. (2008) Indexed regulation. *J Environ Econ Manage* 56:221–233

Nnoaham, K.E., Sacks, G., Rayner, M., Mytton, O., and Gray, A., (2009). Modelling income group differences in the health and economic impacts of targeted food taxes and subsidies. *Int. J. Epidemiol*. 38, 1324–1333. doi:10.1093/ije/dyp214

Oberndorfer, U. and Rennings, K. (2007). Costs and Competitiveness Effects of the European Union Emissions Trading Scheme. *European Environment*, 17, 1–17.

OECD (2006). The Distributional Effects of Environmental Policy - *OECD*. URL http://www.oecd.org/env/tools-evaluation/thedistributionaleffectsofenvironmentalpolicy.htm (accessed 12.1.16).

OECD (2001). Ancillary benefits and costs of greenhouse gas mitigation. In: Proceedings of an IPCC Co-Sponsored Workshop, 27–29 March 2000. *Organization for Economic Co-Operation and Development*, Washington, DC.

OECD (2011). OECD Perspectives*: Spain Policies for a sustainable recovery. Organization* for Economic Co-Operation and Development, Paris

Pairotti, M.B., Cerutti, A.K., Martini, F., Vesce, E., Padovan, D., and Beltramo, R., (2015). Energy consumption and GHG emission of the Mediterranean diet: a systemic assessment using a hybrid LCA-IO method. *J. Clean. Prod*. 103, 507e516.

Parry, I. (2004). Are emissions permits regressive? Journal of Environmental Economics and Management, .47, 364–387.

Pearson, M., and Smith, S., (1991). The European Carbon Tax: An Assessment of the European Commission's Proposals. *Institute for Fiscal Studies*, London.

Peichl, A. (2008). The benefits of linking CGE and Microsimulation Models - Evidence from a Flat Tax analysis *FiFo Discussion Papers* No. 08-6. University of Cologne, FiFo Institute for Public Economics.

Piketty, T., and Saez, E. (2014). Inequality in the long run. *Science* 344, 838–843.doi:10.1126/science.1251936

Poterba, J.M. (1991). Is the Gasoline Tax Regressive? *National Bureau of Economic Research*.

Pye S, King K, and Sturman J (2006). Air Quality and Social Deprivation in the UK: an environmental inequalities analysis, Undertaken on behalf of UK Department of Environment, Food and Rural Affairs, June 2006,

Rausch, S. and Rutherford, T. F. (2007). Computation of Equilibria in OLG Models with Many Heterogeneous Households. *Ruhr Economic Paper* No. 15.

Rausch, S., Metcalf, G.E., and Reilly, J.M. (2011). Distributional impacts of carbon pricing: A general equilibrium approach with micro-data for households. *Energy Economics*. 33, S20–S33.

Reynolds, M., and Smolensky, E. (1977), Public Expenditure, Taxes and the Distribution of Income: The United States, 1950, 1961, 1970, New York.

Ringlund, G.B., Rosendahl, K.E. and Skjerpen, T. (2008). Do oilrig activities react to oil price changes? An empirical investigation. *Energy Economics* 30: 371–396.

Romero, D., Del Rio, P., and Burguillo, M. (2014). Modelling Fuel Demand of Passenger Cars in Spain: A Dynamic Panel Data Analysis Using the Generalised Method of Moments. *Journal of Transport Economics and Policy*, 48, 315-332.

Rutherford, T. (2002). Lecture notes on constant elasticity functions

Rutherford, T. and Tarr, D. (2004). Regional household and poverty effects of Russia's accession to the world trade organization. *The World Bank*, Washington.

Rutherford, T. F. and Tarr, D. G. (2008). Poverty effects of Russia's WTO accession: Modeling "real" households with endogenous productivity effects. *Journal of International Economics*, 75(1):131{150.

Sáenz de Miera, G., del Río González, P., and Vizcaíno, I. (2008). Analysing the Impact of Renewable Electricity Support Schemes on Power Prices: The Case of Wind Electricity in Spain. *Energy Policy* 36 (9): 3345–59. doi:10.1016/j.enpol.2008.04.022.

Sanz-Sanz, J.F., Romero, D., Castaner, J., Prieto, J., and Fernandez, J.F. (2003). Microsimulación y comportamiento en el análisis de reformas de imposición indirecta. El simulador de Imposición Indirecta del Instituto de Estudios Fiscales (SINDIEF). *Instituto de Estudios Fiscales*, Madrid.

Sanz-Sanz, J.F. and Romero-Jordán, D. (2012). Efectos recaudatorios y distributivos en 2012 de algunas opciones de reforma del IVA. Impacto sobre el sector de los hogares*. Cuadernos de Información Económica* nº 228 29 de junio de 2012

Ščasný, M., Zvěřinová, I., Czajkowski, M., Kyselá, E., and Zagórska, K. (2016). Public Acceptability of Climate Change Mitigation Policies: A Discrete Choice Experiment (Working Paper No. 2016–11). *Faculty of Economic Sciences, University of Warsaw*

Schmalensee, R. (2012). Evaluating Policies to Increase Electricity Generation from Renewable Energy. *Review of Environmental Economics and Policy* 6 (1): 45–64. doi:10.1093/reep/rer020.

Schöb, R. (1997). Environmental taxes and pre-existing distortions: The normalization trap. *International Tax and Public Finance* 4, 167–176.

Sensfuß, F., Ragwitz, M., and Genoese, M. (2008). The Merit-Order Effect: A Detailed Analysis of the Price Effect of Renewable Electricity Generation on Spot Market Prices in Germany. *Energy Policy* 36 (8): 3086–94. doi:10.1016/j.enpol.2008.03.035.

Sicras-Mainar, A., Navarro-Artieda, R., Rejas-Gutiérrez, J., and Blanca-Tamayo, M. (2008). Relationship between obesity and antipsychotic drug use in the adult population: A longitudinal, retrospective claim database study in Primary Care settings. *Neuropsychiatric Disease and Treatment* 4, 219–226.

Smed, S., Jensen, J.D., and Denver, S. (2007). Socio-economic characteristics and the effect of taxation as a health policy instrument. *Food Policy* 32, 624–639.

Sofi, F., Cesari, F., Abbate, R., Gensini, G.F., and Casini, A. (2008). Adherence to Mediterranean diet and health status: meta-analysis. *BMJ* 337, a1344. doi:10.1136/bmj.a1344

Stehfest, E., Bouwman, L., Vuuren, D.P. van, Elzen, M.G.J. den, Eickhout, B., and Kabat, P. (2009). Climate benefits of changing diet. *Climatic Change* 95, 83–102.

Sterner, T. (2012). Distributional effects of taxing transport fuel. *Energy Policy*, 41, 75–83.

Swart, R., Amann, M., Raes, F., and Tuinstra, W. (2004). A Good Climate for Clean Air: Linkages between Climate Change and Air Pollution. An Editorial Essay. *Climate Change*, 66, 263–269.

Symons, E.J., Speck, S. and Proops, J.L.R. (2002). The distributional effects of carbon and energy taxes: the cases of France, Spain, Italy and U.K. *European Environment*, 12, 203-212.

Tiezzi, S. (2005). The welfare effects and the distributive impact of carbon taxation on Italian households. *Energy Policy* 33, 1597–1612.

Tol, R. (2005). The marginal damage costs of carbon dioxide emissions: an assessment of the uncertainties. *Energy Policy*, 33, 2064–2074.

Traber, T., and Kemfert, C. (2009). Impacts of the German Support for Renewable Energy on Electricity Prices, Emissions, and Firms. *The Energy Journal* 30 (3): 155–78.

UNFCCC (1992) United Nations Framework Convention on Climate Change

UNFCCC (2014). Lima call for climate action, United Nations Framework Convention on Climate Change.

UNFCCC (2015). Adoption of the Paris Agreement. Proposal by the President. Conference of the Parties Twenty-first session Paris, 30 November to 11 December 2015.

Varela-Moreiras, G., Ruiz, E., Valero, T., Ávila, J.M., and Pozo, S. (2013). The Spanish diet: an update. *Nutrición Hospitalaria*, 28(Supl. 5), 13-20.

Vidal, R., Moliner, E., Pikula, A., Mena-Nieto, A., and Ortega, A. (2015). Comparison of the carbon footprint of different patient diets in a Spanish hospital. *J Health Serv Res Policy* 20, 39–44. doi:10.1177/1355819614553017

Villanueva, T. (2011) European Nations launch tax attack on unhealthy foods. *CMAJ*, 183:E1229-30.

Wadud, Z., Graham, D. J. and Noland, R. B. (2009). Modelling fuel demand for different socio-economic groups, *Applied Energy*, 86, 2740–9.

Walker G. P., Mitchell G, Fairburn, J. and Smith, G. (2003). Environmental Quality and Social Deprivation. Phase II: National Analysis of Flood Hazard, IPC Industries and Air Quality. R&D Project Record E2-067/1/PR1.*The Environment Agency*, Bristol.

WB (2016) Trade (% of GDP) 1960-2015. World Bank national accounts data, and OECD National Accounts data files. *The World Bank*.

Webb, J., Audsley, E., Williams, A., Pearn, K., and Chatterton, J. (2014). Can UK livestock production be configured to maintain production while meeting targets to reduce emissions of greenhouse gases and ammonia? *J. Clean. Prod*. 83, 204–211. doi:10.1016/j.jclepro.2014.06.085.

WHO (2003). Diet nutrition and the prevention of chronic diseases. *World Health Organization Technical Report Series* 916, 1–149.

WHO (2009). Global health risk. Mortality and burden of disease attributable to selected major risks. *World Health Organization*

WHO (2015). Healthy diet. *Fact sheet* N°394 Updated January 2015 http://www.who.int/mediacentre/factsheets/fs394/en/

Wier, M., Birr-Pedersen, K., Jacobsen, H., and Klok, J. (2005). Are CO2 taxes regressive? Evidence from the Danish experience. *Ecological Economics*, 52, 239–251.

Winchester, N., Paltsev, S., and Reilly, J. (2010). Will border carbon adjustments work? *MIT Joint Program on the Science and Policy of Global Change Report No. 184*.

Wiser, R.H. (2007). Using contingent valuation to explore willingness to pay for renewable energy: A comparison of collective and voluntary payment vehicles. *Ecological Economics* 62, 419–432. doi:10.1016/j.ecolecon.2006.07.003

Xu, Y., and Masui, T. (2009). Local air pollutant emission reduction and ancillary carbon benefits of SO2 control policies: Application of AIM/CGE model to China. *European Journal of Operational Research*, 198, 315–325.