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

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Global climate change measures are difficult to implement. In this context, local air pollution measures may play an important role in the political agenda since their effects are felt more immediately by citizens. Distributional implications are one of the main barriers for implementing environmental policies. This paper explores the distributional implications of air pollution taxes and compares them to climate change taxes. For the comparison, both tax schemes were set to yield the same revenue. Methodologically, the study uses a top–down approach linking a macro model to a micro model. We find that taxes on local air pollutants are more regressive than those levied on CO_2. This is because the goods implicitly taxed have a greater weight in the consumer basket of low-income groups, even if the tax revenues are recycled. Furthermore, the revenue-neutral recycle scheme increases both taxes efficiency, but, at the same time, can increase regressivity.

Keywords: environmental tax reform; distributional impact; cocal air pollution taxes; global climate change taxes; taxes regressivity

1. Introduction

Despite the continuing progress of international climate negotiations, including recent agreement at the 20th Conference of the Parties (COP) in Lima, it is still not known whether the sum of these reduction declarations will be enough to keep any future temperature increase below 2°C, as established in the Copenhagen Accord. Climate action is expected to remain unilateral or fragmented at least for the next decade with high-income countries leading the effort to reduce emissions. However, the costs and their distribution make implementation of unilateral global climate change (GCC) measures difficult. In this context, local air pollution (LAP) measures may play an important role in the political agenda, since their effects (mainly on health) are felt more immediately by citizens. Global climate change and 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

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Masui 2009) have recently explored the ancillary benefits of LAP mitigation, given the slow progress of international agreements on climate change and especially the fact that the health effects of pollution are of more immediate concern to developing countries.[1] Finally, Bollen *et al.* (2009, 19) 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,[2] Pearson and Smith (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 mainly due to differences in the type of tax, consumer patterns, income level and energy and transport infrastructures.[3]

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, Speck, and Proops (2002) for Italy and UK; 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, Labeaga, and Rodríguez (2004), Wier *et al.* (2005), Romero-Jordán, Del Rio, and Burguillo (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 de 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 transfers to compensate groups who have been left worse off. Rausch, Metcalf, and Reilly (2011) show (using a computable general equilibrium [CGE] model for the US) that lump-sum transfers to households are more progressive than lowering income tax, which proves to be highly regressive.[4] 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 Köhler (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, NO_x and SO_2, and finds that CO_2 permits are more regressive than SO_2 permits but less regressive than NO_x 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.[5] Due to this shortage of studies,[6] it is not yet clear what the effect of LAP tax is on the distribution of the tax burden across income groups.

This paper investigates the potential regressive impacts of a LAP tax (based on the internalisation of the external costs), and compares it in a compressive fashion to a GCC tax (tax on CO_2).[7] 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 (IO) 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 (RS) and Kakwani indexes. Finally, this study also explores the 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 paper is structured as follows: Section 2 presents the methodology and data; Section 3 describes the different tax scenarios proposed; Section 4 presents the results and Section 5 sets out the conclusions.

2. Methods and data

2.1. Methods

A top−down approach is used to link the micro and macro models. 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 IO price model. In the second stage, a micro-simulation model is used to calculate the distributional effects of price changes. This is carried out using the micro-simulation tool developed by Sanz *et al.* (2003).

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 IO model. This model assumes that the production technology is linear, i.e. 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 represent the internalisation 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[8] 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):

$$\mathrm{EC}_j = \sum_{z=1} c_z \cdot E_{zj} \tag{1}$$

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 internalisation as it is not necessary to include all social costs.

Once the total tax payment by industry sector has been calculated, the IO approach allows us to estimate total indirect price changes of consumption. In particular, to analyse the price change, an IO 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:[9]

$$P_j = \sum_{i=1}^{I} p_i a_{ij} + p_{m_i} a_{m_{ij}} + (1 + s_j) w l_j + r k_j + \mathrm{EC}_j \tag{2}$$

where P_j is the price of consumption good j; a_{ij} stands for the IO 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 and k_j are labour and capital, respectively, and the terms w and r are the price of labour (wage) and the price of capital, respectively, and s_j is the tax rate of the social security paid per sector. When there is no internalisation (i.e. no tax on pollutants) c_z is zero, and therefore, EC_j is also zero.

2.1.2. Micro-simulation 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 *et al.* (2003) is used.

The micro-simulation model uses an Almost Ideal Demand System (AIDS) designed by Deaton and 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 *et al.* 2003). AIDS is based on the assumption that the households will alter their spending decisions as a result of price changes as per the following equation:

$$w_i = \alpha_i + \sum_{J=1}^{n} \gamma_{ij} \ln p_j + \beta_i \ln\left(\frac{G}{P}\right) \tag{3}$$

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 IO 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; therefore, 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:

$$p_i^0 = \left(1 + t_i^0\right)\ (x_i) \tag{4}$$

where t_i^0 is the initial value added tax (VAT) rate, and x_i represents the price before tax. Hence, the price after tax is:

$$p_i^1 = \left(1 + t_i^1\right)\ \left[\frac{p_i^0}{(1 + t_i^0)}\right] \tag{5}$$

where t_i^1 is the post-reform VAT equivalent to price change obtained with the IO 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 the effective price and expenditure, respectively:

$$V\ (P_r, G_e) = v_r(P, G) \tag{6}$$

which can be expressed in terms of the expenditure function:

$$G_e = e(P_r, v_r(P, G)) \tag{7}$$

The EV 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 Muellbauer 1980; Creedy 1999 among others):

$$\text{EV} = e\left(p^0, v^1\right) - e\left(p^1, v^1\right) = G_E^1 - G \tag{8}$$

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, and it is defined as:

$$\text{EF} = EV - (R_1 - R_O) \tag{9}$$

where R_0 and R_1 are revenues in the initial and final tax scenarios, respectively. The higher the value of EF, the greater the distorting effect of a tax. Comparison of the results obtained in computing expression (9) makes it possible to determine which of the two taxes analysed is worse in terms of efficiency.

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, Non Methane Volatile Organic Compound (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.

3. Tax scenarios

The interest of this paper concentrates on the distributional assessment of LAP tax (based on internalisation of the actual external costs), and in comparing it in a compressive way with a GCC tax. The tax scenarios are based on internalisation 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 dimensions yields four scenarios. Across the four scenarios, the tax is introduced into the IO model, increasing the cost of production by each sector. The price change calculated with the IO model is performed in the demand model as an indirect tax reform which is equivalent to the price change obtained.

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

$$\mathrm{TEC}^{\mathrm{LAP}} = \sum_{j=1}\sum_{z=1} c_z \cdot E_{zj} \quad \mathrm{TEC}^{\mathrm{GCC}} = \sum_{j=1} c_{\mathrm{CO_2}} \cdot E_{zj} \tag{10}$$

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 NH_3, NO_x, SO_2, non methane organic volatile compounds (NMVOC) and PM_{10}.

Tax scenarios are shown in table 1. The first scenario is a 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 tonne in Sweden.[10] On the other hand, in the European Union emissions trading system (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 tonne 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) and EPA (2013). This tax is also similar to the taxes on CO_2 applied in

Table 1. Tax scenarios.

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

		Revenue recycling	
		No	Yes
Environmental issue	GCC	(GCC tax)	(GCC tax)
	LAP	(LAP tax)	(LAP tax)

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.

The second scenario is a LAP tax. According to the social cost calculated by Markandya, Bigano, and Roberto (2010),[11] the total external cost caused by LAP is much higher than the external cost of the carbon emissions ($TEC^{LAP} > TEC^{GCC}$). Estimations of the external cost of LAP stand at around €15 billion, whereas the external harm of CO_2 is €7 billion. The aim is to make a distributional comparison of the costs of a tax on CO_2 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 LAP has not been internalised totally; only the proportion equivalent to achieving the same external harm for 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 with any *ex-ante* estimation of the emission reduction, it will be easier for policy makers to introduce a tax according to the actual external damage and then revise it depending on the real impact. Table 2 shows the social cost estimated by the Cost Assessment of Sustainable Energy System (CASES) project and the social cost used to achieve an internalisation equivalent to the GCC tax scenario.

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 CO_2 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 Eurozone countries, de Mooij 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%–4% of GDP. For southern countries, Engler *et al*. (2014) show that a fiscal devaluation of 1% of GDP increases output by 0.9%–1.5% of GDP. For the Spanish

Table 2. Social cost of local air pollution for Spain, 2005.

	Social cost estimated by CASES (€ per tonne)	Social cost used in the simulated reform (€ per tonne)
SO_x	4,912.22	2,323.44
NO_x	3,485.07	1,648.41
NMVOC	797.34	377.14
NH_3	5,393.91	2,551.28
PM_{10}	16,037.56	7,585.63

case, Boscá, Doménech and Ferri (2014) analyse 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 paper, 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%.

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. Third, 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.

4.1. Price impacts

Figure 1 shows the impact on prices for the five sectors with the highest and lowest impacts on price changes. Observe that 'electricity, water and gas production', 'energy',

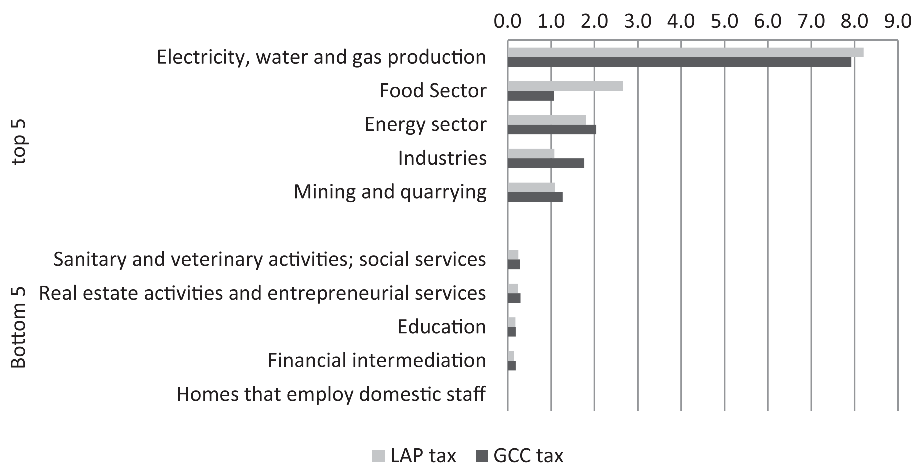

Figure 1. Change (%) in production prices. Top and bottom sectors.

'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 a 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.

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.

4.2. Distributional effects

Figure 2 shows the impact of the two taxes analysed on household disposable income by expenditure deciles. The first decile (1) represents the lowest 10th of the expenditure and the last one (10) the highest. Cost impacts are measured in terms of 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.

Figure 2 shows, first, that the cost[12] is below 1.05% for all the expenditure deciles in terms of EV 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 *et al*. (2005) or Rausch, Metcalf, and Reilly (2011).

Second, 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.

Third, Figure 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 EV in expenditure. These results are similar to those of Labandeira and Labeaga

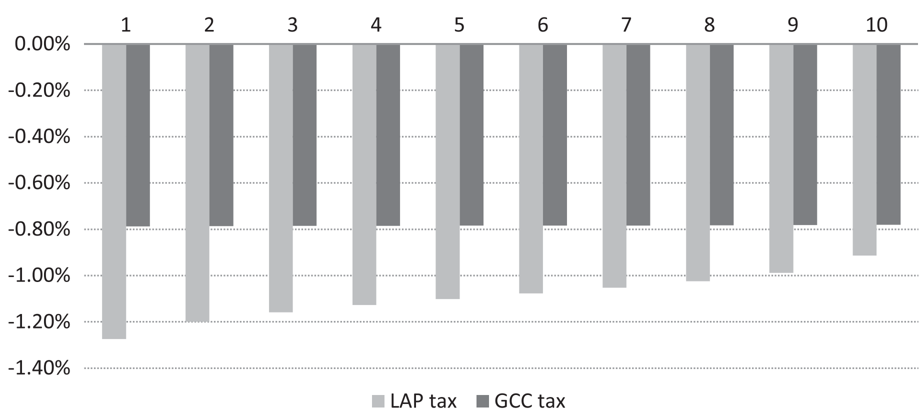

Figure 2. Cost distribution per expenditure decile.

(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 EV in expenditure. Section 4.4 uses different standard indexes to measure and confirm this effect more precisely.

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 summarised 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 as transport.

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 labour, 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 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

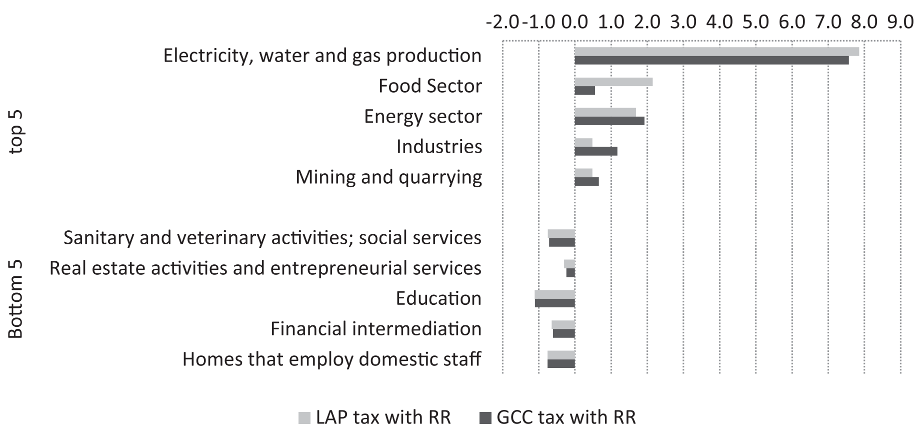

Figure 3. Impacts on price (%) after revenue recycling. Top and bottom sectors.

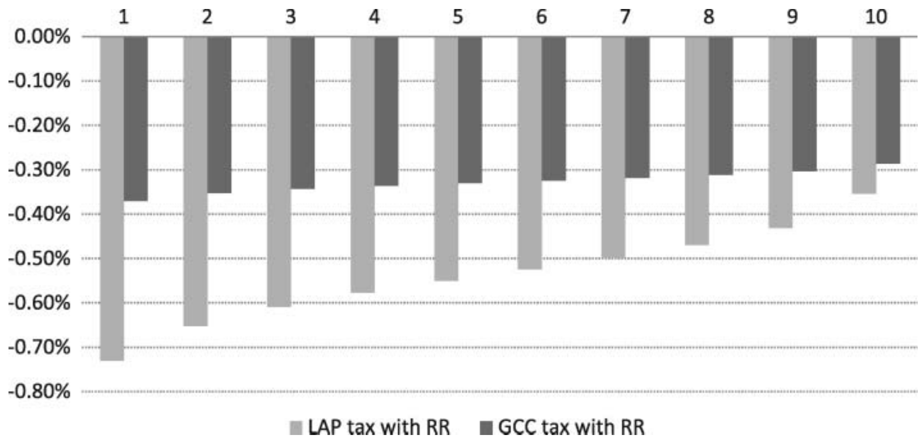

Figure 4. Cost distribution after recycling per expenditure decile.

'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 labour-intensive benefit from reductions in their prices. For example, the price changes in 'education' and 'health services' are negative and close to 1%.

Figures 4 and 5 show cost impacts and the excess burden[13] per expenditure decile after revenue recycling. First, it is clear that the distributional costs of taxation are lower after recycling revenue: they decrease by about 0.5% for all income groups and for both tax scenarios.

Revenue recycling through a tax on labour tax can reduce the progressivity of the tax system. Figure 4 reveals that under the GCC tax, the differences between different types

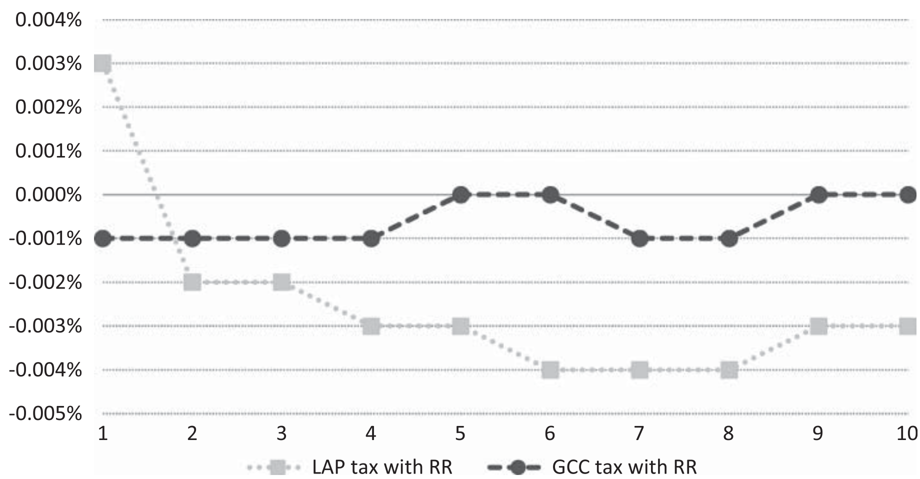

Figure 5. Relative efficiency impacts after recycling revenue.

of household are still very small. However, the difference between high- and low-income groups is larger before recycling, evidencing that impacts are more regressive with revenue recycling (WRR). Under the LAP tax, the cost for the highest income group is only 0.35% while that for the lowest group is 0.73%, and the gap between income groups is wider 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 WRR 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 labour (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 labour. Revenue recycling through a distortionary tax has a positive impact on efficiency, but the distributional implications may not be positive.

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 RS index provides information about redistribution, and the Kakwani index is used to measure progressivity.[14] All these indexes are estimated relative to total household expenditure.

Table 3 reports the RS index 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 3 shows results for the effects of a reform on GCC and LAP taxes in two cases: (1) without revenue recycling (NRR) and (2) 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

Table 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	0.12064	0.03662
	(+0.00006)	(+0.00685)	(−0.00193)
LAP tax	0.0039	0.12301	0.0322
	(−0.00044)	(+0.00922)	(−0.00635)
(3) Post-reform indexes with revenue recycling (WRR)			
GCC tax with RR	0.00419	0.1171	0.03626
	(−0.00015)	(+0.00331)	(−0.00230)
LAP tax with RR	0.00381	0.1192	0.03269
	(−0.00053)	(+0.00541)	(−0.00586)

Note: Variation of measures of regressivity with respect to the pre-reform index.

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 and WRR. Finally, a GCC tax is slightly more progressive and redistributive when a 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.

5. Conclusions

LAP and 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 paper, we conduct a distributional analysis of an LAP tax (based on the internalisation of the external costs of several pollutants) and compare it in a compressive way with a GCC tax (tax on CO_2), where two tax systems yield the same actual revenue. We use an IO 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 RS and Kawani indexes. Finally, we also explore the distributional effects of a revenue-neutral recycling scheme through a reduction on taxes on labour (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 shown by Duarte, Mainar, and Sanchez-Choliz (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 1st decile (the poorest households) to 0.9% for the 10th (the richest households). In any case, the overall effect of taxes analysed on distribution 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 labour-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 RS 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, King, and Sturman 2006 and Walker *et al.* 2003) that show that LAP affects low-income household locations more. Third, our IO model cannot capture the full effects that a reduction in taxes on labour 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 labour markets and high unemployment levels (see for example Markandya, González-Eguino, and Escapa 2013). Moreover, the IO model assumes that there is no possibility of substitution between inputs, which restricts our analysis to short-run effects. Fourth, our methodology does not incorporate price feedback effects from micro-simulation 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 with any *ex-ante* estimation of the emission reduction, it will be easier for policy makers to introduce a tax according to the actual external damage and then revise it depending on the real impact.

The first policy implication of this paper 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 labour, may not improve the distributional effect. However, this is the third policy implication, given that the overall regressivity of these taxes is low; various specific combinations of policies could be designed to compensate the households or groups that are most affected.

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Notes

1. According to WHO estimates, LAP is also one of the leading causes of death in developing countries (WHO 2009).
2. 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. See for example, Feng *et al.* (2010), Metcalf (1999), Sterner (2012) and Wier *et al.* (2005). Only Rausch, Metcalf, and Reilly (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.
3. Other studies find regressive effects in some countries (e.g. Metcalf, Mathur, and Hassett 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 Brannlund and Nordstrom 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.
4. Ekins *et al.* (2011), Barker and Köhler (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.
5. 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.
6. However, there are studies assessing the economic effects of the internalisation of the external costs of local air pollution. See for example Kiulia *et al.* (2013).
7. Although environmental taxes must be complemented with other instruments in the long term, in the short term, they are successful (del Río González 2008).
8. See methodology on Spanish Environmental Satellite Accounts.
9. There are 21 sectors in our IO approach.
10. 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.
11. In 2006, the CASES (‘Cost Assessment of Sustainable Energy System’) project (Markandya, Bigano, and Roberto 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.
12. 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.
13. The excess burden is calculated as the difference between equivalent variation (EV) and revenue (R) generated by households (h): $E_{GE} = -\sum_h \mathrm{EV}_h - (R_h^1 - R_h^0)$.
14. Both indexes are based on approximations of the Gini index.

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