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Carbon footprint, municipality size and rurality in Spain: Inequality and carbon taxation

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ABSTRACT

By using an environmentally extended multi-regional input-output model, this paper analyses the Spanish households' carbon footprint for the 2008–2017 period considering the municipality size as well as the urban or rural residential zone where families live. Results show that, on a per capita basis, inhabitants of medium-large municipalities emit fewer carbon emissions than those settled in small ones (between 0.34 and 0.54 tCO_2/cap depending on the year studied). This carbon unbalance is mainly explained by the higher direct carbon footprints of dwellers who reside in small municipalities and, in special, in rural zones. Furthermore, applying inequality measures through a consumption-based carbon footprint Gini coefficient, we show that both income and CO_2 emissions inequality are lower in small municipalities. In the light of the findings, in Spain, the application of a carbon pricing on direct and indirect carbon footprints will be regressive, disproportionally affecting people of small municipalities and rural areas. Accordingly, household carbon inequalities must be contemplated to avoid poorly designed climate change mitigation policies.

1. Introduction

The term "empty Spain" has been coined to reflect how part of the Spanish territory is being depopulated during the last years. This phenomenon mainly affects the small municipalities in the centre of the country, where industrial activity is very depleted, and the population is relatively aged (CES, 2018). The young and middle-aged population leaves these places hoping to find better job opportunities and to reach urban living standards in cities. Although urbanisation is especially incipient in developing countries, it also affects developed nations (UN, 2019b). In Spain, for instance, during the 2008–2017 period, 269 thousand people have moved from small municipalities to other places; while towns and cities have grown by 683 thousand inhabitants [dataset] (INE, 2020a). These demographic changes have an impact on climate change through the differences between urban and rural consumption patterns. Nonetheless, in the literature, there is still an open debate about whether the urbanisation process boosts or saves CO_2 emissions on a per capita basis (Hubacek et al., 2017b; Schubert and Gill, 2015).

On the one hand, cities allow people to take advantage of agglomeration economies obtaining a higher labour division and labour productivity that gives rise to larger average wages (Krugman, 1991; Puga, 2010). As a consequence, these higher per capita earnings in cities could intensify the environmental scale effect which states that as the families' income rises, so does their spending on consumption and, with it, their CO_2 emissions (Chancel and Piketty, 2015; Hubacek et al., 2017a; López et al., 2016), energy requirements (Lenzen et al., 2006; Moll et al., 2005; Reinders et al., 2003) and material uses (López et al., 2017). Apart from the larger purchasing power, in urban zones, there are also a wider variety of products for consumption, which usually leads to high-carbon lifestyles (Gill and Moeller, 2018; Heinonen et al., 2013). All these thoughts have encouraged previous literature to make many efforts on assessing the carbon footprints of big cities given its standing as a hot-spot for fighting against climate change (Chen et al., 2016; Harris et al., 2020; Huang et al., 2018; Moran et al., 2018).

On the other hand, some scholars have argued that urbanisation can contribute to reducing global warming (Glaeser, 2011). Behind this idea underlies the so-called "relief by density" hypothesis, according to which, in per capita terms, city dwellers

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emit fewer CO_2 emissions than those settle in rural zones (Dodman, 2009; Gill and Moeller, 2018; Schubert and Gill, 2015). In this manner, people located in urban areas might save CO_2 emissions thanks to the scale economies linked with public transport and commute smaller distances (Rau and Vega, 2012); the larger number of gas distribution networks, which is more carbon-efficient than other fuels (Poumanyvong and Kaneko, 2010); and the abundance of compact and small houses that save energy (Norman et al., 2006). Meanwhile, rural municipalities are limited in employment, infrastructure, and goods and services, which may force their inhabitants to use private transport (e.g., cars) for travelling to cities to fully meet their needs (VandeWeghe and Kennedy, 2007) as well as to employ more carbon-intensive fuels for heating and cooking (Labandeira et al., 2011; Wang and Jiang, 2017).

A highly relevant decision to address the aforementioned academic discussion is how to identify which are urban and rural households. To date, there is not yet a globally agreed distinction due to national differences in the criterion used to separate urban from rural areas. Traditionally this categorisation has been based on differences in living standards (UN, 2019a). Nevertheless, it has become blurred in developed countries where well-being is pretty widespread, giving way to other criteria relying on municipality size or population density. It has influenced previous works that have estimated carbon footprints of urban and rural households: Gill and Moeller (2018) employed the municipality size as a criterion for calculating rural and urban individuals' carbon footprints in Germany. These authors found that the accumulation of people in cities save some greenhouse gas emissions and pointed out that inhabitants of small municipalities could be hit heavily by carbon taxes on direct energy use; Ottelin et al. (2019) followed the urbanisation degree (i.e., a population density approach) as the criterion for estimating EU households' carbon footprints showing that per capita carbon emissions are slightly lower in cities than in rural areas when income and other household features are controlled; lastly, for Spain, Arce et al. (2017) and Duarte et al. (2012) applied both approaches to defining urbanity and rurality concluding that larger population density or municipality size leads households to adopt more carbon-intensive lifestyles.

In the context of this discussion, this article contributes to the existing body of knowledge in several ways. First, we test the "relief by density" hypothesis in Spain by employing the size of the municipality as the criterion for separating the households. This approach allows connecting the scientific findings with local administrative bodies, i.e., municipal councils, expediting the good design, applicability and effectiveness of mitigation policies. Second, we account for carbon footprints of households located in small and medium-large municipalities from a dynamic perspective (considering both crisis and post-crisis period, i.e., from 2008 to 2017). As the division between municipalities could not reflect rurality as such, we cross municipality size with a control variable that states the urban vs rural residential location of the household to capture those typical nuances of rural life in our analysis adequately. Third, we evaluate how the economic cycle has affected income and carbon footprint inequality within municipalities through the Gini coefficient. And, finally, we go beyond just estimating carbon footprints by simulating carbon pricing scenarios. We view this analysis particularly relevant, given that, lately, the Spanish Government has firmly pledged to boost policy actions to address climate change, among which a green fiscal reform is called to have a prevalent role (Gobierno de España, 2020). However, this kind of measures may cause adverse distributional effects because of poor people spend a large share of income on inelastic and potentially carbon-taxable products (Böhringer et al., 2019). Furthermore, the households' location determines their energy demand significantly (Jain and Kumar, 2018; Reinders et al., 2003; Zhang and Lahr, 2018). Thus, we look at the distributional effects of carbon pricing policies on Spanish households for different settlements' locations and types of products (Bureau, 2011; Callan et al., 2009; Pashardes et al., 2014) to propose some policy recommendations that could pave the way for the acceptance of eventual green tax reforms.

2. Methodology and data

2.1. Estimating household carbon footprints on a per capita basis

The indicator chosen to measure the direct and indirect Spanish families' CO_2 emissions is the household carbon footprint per capita (CF), which is defined in expression [1]:

$$CF = \overbrace{i}^{1.1} CF + \underbrace{dCF}_{1.2}$$

where expression [1.1] represents the indirect household carbon footprint per capita (iCF), i.e., the CO_2 emitted, directly and indirectly, along the global value chains until the households' final demand is satisfied; and expression [1.2] shows the direct household carbon footprint per capita (dCF), i.e., the direct CO_2 emissions associated with the demand for energy goods consumed both within and out of the home.

2.1.1. The indirect household carbon footprint per capita

In order to estimate the iCF of different families, we employ an environmentally extended multi-regional input-output (EEMRIO) model adapted to the households in its consumption based-approach, which has been previously applied in the literature (Arce et al., 2017; Brizga et al., 2017; Gill and Moeller, 2018; Huang et al., 2018; López et al., 2016). The main virtue of this environmental accounting method is its capacity to interconnect local consumption decisions with the total CO_2 emissions embedded along fragmented global production chains and international trade (Hubacek et al., 2014). Regarding the standard EEMRIO model framework presented in Miller and Blair (2009), we undertake the necessary adaptations until achieving the accurate model to estimate the iCF for different types of Spanish households. Let us define the following variables: r is the region under study, in this case, Spain; s is a conglomerate of regions formed by the rest of the countries called "Rest of the World"; $\widehat{f}$ is a vector of CO_2 emissions coefficients diagonalized, which collects the CO_2 per monetary unit of production for all regions and all industries; A is a matrix of domestic and import technical coefficients; $L = (I - A)^{-1}$ is the Leontief inverse matrix, which shows the direct and indirect inputs necessary for an additional monetary unit of output to satisfy the final demand; $\widehat{c}_i^r$ is a consumption vector diagonalized of a type of household with i characteristics of the r region, which can be decomposed into the diagonalized vector of domestic household consumption ($\widehat{c}_i^{rr}$) and the diagonalized vector of imported goods from region s ($\widehat{c}_i^{sr}$); and, P_i^r is the total population that belongs to a determined kind of household i in r region. In accordance with all these definitions, the iCF for a determinate kind of households i of the region r is calculated as follows:

$$iCF_i^r = \frac{\widehat{f}\, L\, \widehat{c}_i^r}{P_i^r} = \left(\overbrace{\frac{\widehat{f}^r L^{rr} \widehat{c}_i^{rr} + \widehat{f}^r L^{rs} \widehat{c}_i^{rr}}{P_i^r}}^{1.1.1} \cdots \underbrace{\frac{\widehat{f}^s L^{sr} \widehat{c}_i^{sr} + \widehat{f}^s L^{ss} \widehat{c}_i^{sr}}{P_i^r}}_{1.1.2} \right)$$

In expression [1.1] two emission sources can be discerned: **a)** the direct and indirect CO_2 emissions associated with the production of the region r that is intended to meet the demand of households in the region r (expression [1.1.1]); and, **b)** the direct and indirect CO_2 emissions linked to the output of region s that ends up supplying the demand of households in region r (expression [1.1.2]).

The consumption patterns (c_i^r) that represents alternative spending behaviours existing in the Spanish society are derived from a household survey microdata, while the EEMRIO model is based on the principles of the national accounts (NA). Thus, this dataset combination hides many uncertainties, mainly due to the different information origins. The first inexactness comes from compiling the household consumption survey, given that each survey suffers common errors such as defective sampling, recall bias, changes in measurement, inadequate supervision and lack of responses (Amores, 2018; Deaton, 2005). Moreover, the household survey microdata has severe difficulties in accurately measuring the income and expenses of the wealthiest families, whose economic and environmental impacts are high (McCully, 2014; Milanovic, 2013; Piketty and Saez, 2014; Pinkovskiy and Sala-i-Martin, 2016). The second uncertainty emerges from the multi-regional input-output (MRIO) methodology and databases, which assume homogeneous sectors, same price for all the sector supply, linear model, inability to detect structural changes in the economy and accounting and adaptation of MRIO tables and satellite accounts (Peters et al., 2016; Wiedmann, 2009). The remaining challenge is to bridge the household survey microdata (each vector of consumption c_i^r) with the MRIO tables used by the model. This step has been carried out in a non-transparent way by a large part of the previous literature, which calculated household footprints by using data survey on consumption (Min and Rao, 2018). Therefore, we have followed the procedure and materials developed by Cazcarro et al. (2020) to harmonize the information for the case of Spain.

2.1.2. The direct household carbon footprint per capita

In order to fully estimate the total CF of a specific type of family i of Spain, it is necessary to add to the iCF all additional CO_2 emitted directly by families when consuming energy. With this purpose, we have followed the method contemplated by the Intergovernmental Panel on Climate Change (IPCC, 2006) for estimating direct CO_2 emissions in national emission inventories, which is specified below:

$$dCF_i^{re} = \frac{\hat{j}^{re}\, c_i^{re}}{P_i^r}$$

where $\hat{j}^{re}$ is a diagonalized vector of direct emissions factors that shows the CO_2 emissions per unit of quantity consumed for e energy goods in region r; and, c_i^{re} is another vector that collects the basket of the e energy goods consumed by Spanish households with i characteristics expressed in their respective physical units.

2.2. Measuring income and carbon inequality

The Gini coefficient is the most commonly-used indicator when researchers study personal inequality and its evolution over time (Chancel and Piketty, 2015; López et al., 2016; Milanovic, 2013; Palma, 2011; Wiedenhofer et al., 2016). The estimation of the Gini index is based on the Lorenz curve, in which we plot the accumulated percent of the population on the horizontal axis and the accumulated percent of the income or carbon emissions on the vertical axis. In Fig. A1 of the Appendix we illustrate schematically how the indicator can be estimated geometrically by dividing A (the area located above the Lorenz curve and below the line of equality) between A+B (the triangular area below the line of perfect equality). The indicator ranges from 0 to 1, in such a way that when there is maximum equity among individuals the Gini coefficient will be equal to 0, but if the income or carbon distribution is fully unbalanced its value will be 1.

Given that this article analyses economic and environmental inequality we use an income Gini coefficient (Income-Gini) and consumption-based carbon footprint Gini coefficient (CF-Gini) applied to households in small and medium-large municipalities in order to evaluate inequality within each type of settlement (Wiedenhofer et al., 2016). Let us define C as the total income or CO_2 emissions of the household income group j and P as the population size of the household income group j. Based on the expression $c_j = \sum_0^j C_j/C_{j=0\ldots n}$ that shows the proportion of income or CO_2 emitted for each household income group j and the expression $p_j = \sum_0^j P_j/P_{j=0\ldots n}$ that reflects the population share of each household income group j, we built the Gini index that will be applicable for measuring both income and CF inequality as follows:

$$\text{Gini} = 1 - \sum_{j=1}^{n} \left(p_j - p_{j-1}\right)\left(c_j + c_{j-1}\right)$$

2.3. Data sources

In this analysis, the MRIO tables used to feed the EEMRIO model are provided by the World Input-Output Database (WIOD) in its 2016 Release [dataset] (Timmer et al., 2015, 2016). This source includes 44 regions and 56 homogeneous industries and covers the entire study period except for the last three years, i.e., 2015, 2016, and 2017. This data limitation is overcome by using the MRIO table of 2014 for the years without available data under the assumptions of constant technology and fixed commercial structure. WIOD information has been transformed from millions of dollars at current and basic prices to euros, applying annual average euro/dollar exchange rates [dataset] (EUROSTAT, 2020). Environmental information has been obtained from [dataset] Corsatea et al. (2019) who provide CO_2 emissions satellite accounts consistent with the WIOD Release 2016 (44 regions and 56 industries) covering the required period (i.e., 2008–2014) to implement the model according to the assumptions mentioned above. The consumption vectors c_i^r have been created from the Spanish Household Budget Survey (HBS) microdata, that covers the entire period of study [dataset] (INE, 2020b). In order to maximize the impact of the analysis, many types of consumption patterns have been generated, representing alternative spending behaviours in Spanish society. The main criterion used for grouping households is the municipality size, but we also cross it with other control variables such as rural/urban residential area and income level of the household.[1] On the one hand, iCF is calculated using consumption patterns obtained from the HBS in euros at current and purchase prices and distributed into 47 groups of the Classification of Individual Consumption by

[1] The INE provides all the household segmentation variables used in this work. Firstly, the size of the municipality offered splits into five groups: (a) municipality of 100,000 inhabitants or more; (b) municipality with 50,000 or more and less 100,000 inhabitants; (c) municipality with 20,000 or more and less than 50,000 inhabitants; (d) municipality with 10,000 or more and less than 20,000 inhabitants; and, (e) municipality with less than 10,000 inhabitants. We regroup the variable into two categories so that municipalities with 10,000 inhabitants or more are called medium-large municipalities and municipalities with less than 10,000 inhabitants are called small municipalities. Secondly, the area of residence of the family which allow us to identify rural and urban households. And, thirdly, the income level of the household is estimated by deciles (based on the households' per capita income).

Purpose (COICOP).[2] In order to feed a macroeconomic model as EEMRIO, this information must be adapted. In this case, we use the procedure proposed by Cazcarro et al. (2020) through the following main steps: **(1)** Align consumption and population data of the Spanish HBS to NA accounting principles.[3] **(2)** Convert consumption data of the Spanish HBS in NA principles to production-based classifications, concretely in Classification of Products by Activity (CPA) 2008 version.[4] **(3)** Revaluate Spanish HBS data based on NA principles and production-based classification to basic prices.[5] **(4)** Adapt data based on production-based classifications to the WIOD MRIO tables that rely on the industry-based classification.[6] On the one other hand, dCF is calculated using consumption patterns of energy goods obtained from the HBS, which provides data on the number of energy goods consumed in cubic meters (m^3), kilograms (kg) or liters (l), depending on the characteristics of the energy good considered. This information has been aligned with the NA principles taken into account the differences between HBS and HFCE population. The population size contemplated for each type of household by the HBS has been adjusted to the NA ones. Furthermore, direct emissions factors are provided by MITECO (2019) and have been adapted to be combined with energy consumption patterns expressed in physical units. The final results of dCF for all the types of families analysed are calibrated regarding the difference between the total direct household CO_2 emissions calculated via HBS and those provided by the environmental satellite accounts for the whole Spanish household sector (Corsatea et al., 2019).

3. Results and discussions

3.1. Overview of the individuals' carbon footprint during the period 2008–2017

Throughout the 2008–2017 period, households emitted between 60% and 78% of the total carbon footprint of Spain. This weight trended upwards over time, mainly as a result of the slowdown in investment made in the construction sector during the economic recession (Zafrilla and López, 2018). Despite this increase in relative terms, the Spanish households' have reduced their absolute carbon footprint in this period from 265 to 236 $MtCO_2$, largely due to the lower consumption during the crisis (López et al., 2016). International trade has also been a relevant driver of this trend. Imports from the Spanish economy are very carbon-intensive, especially those from developing countries where there are many poorly paid workers and lax environmental regulations (López et al., 2014). As a result, the sharp decline in imports since the beginning of the crisis helped to relieve the carbon footprint of Spanish households.

On a per capita basis, the effects of the economic cycle are visible too. Fig. 1 illustrates the evolution of the average per capita consumption and CF in small and medium-large municipalities and splits the CF into the dCF and iCF (the latter, in its turn, is divided into domestic and imported iCF). Results show that the per capita consumption was higher in households located in medium-large municipalities than in the small ones, whereas for the CF the contrary occurs. Indeed, depending on the year studied, an average dweller of a small settlement emits between 0.34 and 0.54 tCO_2/cap more than his/her counterparts living in towns or cities. That means that the "relief by density" hypothesis found in others countries is also fulfilled for the case of Spain (Dodman, 2009; Gill and Moeller, 2018; Schubert and Gill, 2015), and, at the same time, it breaks with previous studies applied to the case of Spain that, unlike us, found a positive relationship between CF and municipality size (Arce et al., 2017; Duarte et al., 2012).

Direct emissions from households, not indirect emissions, explains why the households settled in villages have larger CFs (Fig. 1). As is usual in developed countries, dCF represents only between 25% and 38% of the total households' CFs, but it was decisive for the CF unbalance between municipalities (Schubert and Gill, 2015). Indeed, people of small municipalities could not reduce its dCF between 2008 and 2017 period, on the contrary, they increased it by 1%. It becomes evident that the limitations in small settlements in terms of infrastructure, employment, and goods and services, truly affects the direct energy demand of their dwellers and, as a consequence, their CO_2 emissions (Gill and Moeller, 2018; Jain and Kumar, 2018; Zhang and Lahr, 2018). Looking at the iCF, we find that the larger income and consumption levels in medium-large municipalities lead to slightly higher iCF, both domestic and imported (Arce et al., 2017; Gill and Moeller, 2018).

Although the household's location has a relevant impact on its CF, maybe the distinction between municipalities in terms of population size (smaller and larger than 10.000 dwellers) does not allow to isolate the phenomenon of rurality completely. In order to undertake more in-depth research in this direction, we try to answer the question: how does rurality affect CFs in each type of municipality? In this regard, in Table 1, we cross both variables, municipality size and rural-urban residential zone, and calculate the R/U ratio for both environments. This indicator is equal to 1 when the figures of the urban and rural households are balanced; higher than 1 for larger results of the rural households; and below 1 for the opposite situation.

Focusing the attention on population distribution, we find that the rural dwellers are mostly concentrated in villages, while they are residual in settlements of 10,000 or more inhabitants. For the CFs, regardless of whether the household is rural or urban, they are higher the smaller the size of the municipality, which consolidates the idea showed previously in Fig. 1. Even though the differences between the total CFs of the rural and urban households are minimal, rurality affects the household carbon pattern heavily. In terms

[2] Since 2016, the INE is using the European Classification of Individual Consumption by Purpose (ECOICOP). The affected data has been transformed to COICOP following (INE, 2020b) methodology.

[3] This step requires the use of data on Household Final Consumption Expenditure (HFCE) consistent with the Spanish NA and comparable with the HBS [dataset] (INE, 2020c).

[4] This step requires a bridge matrix that links the COICOP classification with the CPC classification. This kind of bridge matrix shows the share of each COICOP category that is reassigned to each CPC category. The accessibility to this bridge matrices is quite reduced (Amores, 2018). Luckily, Cazcarro et al. (2020) have managed to standardize this type of matrices for all EU-28 countries. These authors build bridge matrices that link information from COICOP (with 47 categories, from CP011 to CP127) with the Classification of Products by Activity (CPA) (CPA, 2008 version, with 64 categories, from CPA_A01 to CPA_U) for 2010. Therefore, the Spanish bridge matrix of 2010 has been used to reclassify all consumption patterns based on COICOP, taking them to CPA 2008 for all years analysed.

[5] This step implies the use of IO tables where the information for each CPA category appears in the total supply to purchase prices, net taxes on products, transport margins, commercial margins, and total supply at basic prices. Given that this information it is not public in Spain, we have used that one estimated by Cazcarro et al. (2020) for this country. In this manner, it is possible to calculate implicit ratios of the net taxes, commercial margins, and transport margins, for each CPA 2008 category. This technique implies starting from the data at purchasers' prices; deducting/adding the net taxes; extracting the trade and transport margins and reassigning them in their respective CPA 2008 categories (following the structure of the Spanish IO tables); finally, the consumption patterns are transformed to basic prices (Amores, 2018).

[6] Even though consumption patterns data are using NA principles, CPA 2008 classification and basic prices, they are not yet ready to be integrated into the WIOD Realise 2016 MRIO tables. For this, it is necessary to move the data from the product-by-product approach (i.e., CPA, 2008) to the industry-by-industry approach, which is the form as WIOD has been built (i.e., ISIC Rev.3 (Timmer et al., 2015; Timmer et al., 2016)). With this purpose, we have applied Model D (fixed product sales structure assumption) (Mahajan et al., 2018) in order to transform the consumption profiles in the same manner as the WIOD Realise 2016 MRIO tables has been build (Amores, 2018).

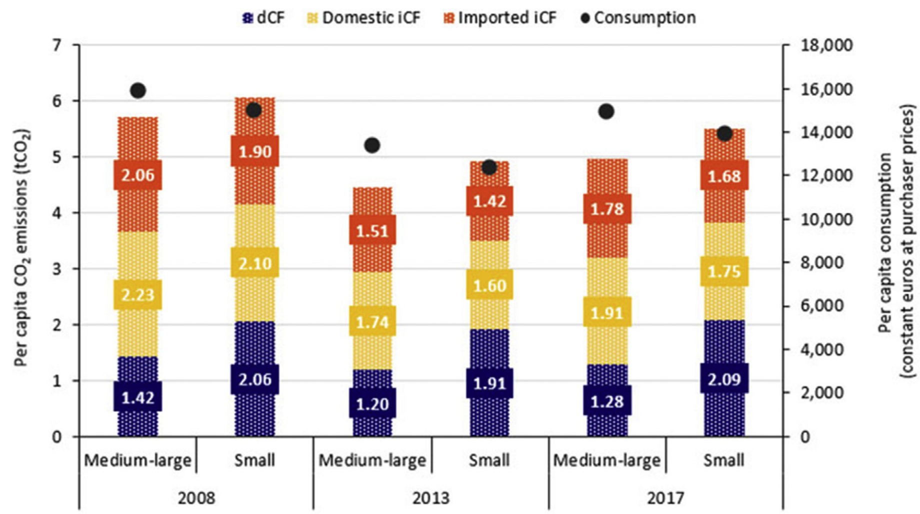

Fig. 1. Spanish households' CF and consumption by municipality size and years.

of dCF, rural households are much more dependent on liquid and solid fuels (e.g., gasoline and coal) for heating and cooking (R/U > 1), whereas the use of natural gas is more common in urban residential zones (R/U < 1). For dCF associated to the use of fossil fuels for private transport, we find that municipality size is the most relevant driver in front of rurality for this matter, as CFs are always larger for households of small municipalities regardless the rurality effect. Finally, looking at the iCF, both domestic and imported, it is observed that urbanity leads to a higher footprint, mainly because urban households, on average, have higher incomes giving rise to lifestyles more inclined towards mass consumption (R/U < 1). Therefore, having a rural or urban lifestyle strongly affects the CF structure of the household, but the quantity of emissions depends on the municipality size: whereas rural households are more dependent on carbon-intensive direct energy sources, urban households tend to have greater purchasing power which leads to more CO_2 emissions associated with the production and distribution of goods and services.

3.2. Measuring income and CF inequality within municipalities

Given the important CF imbalance between households by type of municipality (medium-large vs small) and taking into account that municipalities, as centres of decision-making, are critical for the application of climate policies, we now focus on measuring income and CF inequality at this administrative level. For that, we use the Gini coefficient described methodologically above (section 2.2). The Gini index results showed in Fig. 2 reveals how inequality in terms of income and CF has grown continuously from 2008 to 2013 both in small and medium-large municipalities in Spain, mainly due to the harmful effects of the Great Recession on employment (Anghel et al., 2018; López et al., 2016). The economic recovery has helped to reduce income inequality, but without returning to the pre-recession level of the Income-Gini indicator. At the same time, it must be noted that income inequality is always larger in the medium-large municipalities than in small ones, caused by the concentration of higher salaries in cities as a result of the agglomeration economies generated in these places (Krugman, 1991; Puga, 2010).

Taking into account that income is the main driver of household consumption, and the latter, in turn, determines CO_2 emissions, the trends of the Income-Gini index end up marking the evolution of carbon inequality among individuals (Fig. 2). However, all the inequality indicators evaluated for CF are lower than those applied to income. Behind these results, there are differences in carbon-intensities: poor households allocate a considerable proportion of their income to the consumption of very carbon-intensive goods (e.g., clothing, food or energy) and high-income households direct their marginal consumption to sectors with low CO_2-intensity (e.g., personal services, education, and leisure) (López et al., 2016). Therefore, it could be expected that if carbon inequality measures are calculated by the type of product consumed, the CF-Gini coefficient will be lower for food than services as has been pointed out for other countries (Wiedenhofer et al., 2016).

In addition, we found that from 2008 to 2017 inequality in the CF (Fig. 2b), iCF (Fig. 2c) and dCF (Fig. 2d) have risen in medium-large settlements but tend to remain more stable in villages. Regarding the evolution of inequality in CF, it is important to see

Table 1
Population (millions of people) and CF (tCO_2) by type of household in Spain for 2017.

	Medium-large municipalities				Small municipalities			
	Urban	Rural	Total	R/U	Urban	Rural	Total	R/U
Population	35,30	1,77	37,07	0.05	4,37	5,08	9,45	1.16
Share of population	95%	5%	100%	0.05	46%	54%	100%	1.16
dCF	1.25	1.91	1.28	1.54	1.86	2.28	2.09	1.22
Gas	0.24	0.13	0.23	0.54	0.23	0.17	0.20	0.75
Liquid and solid fuels	0.08	0.65	0.11	7.90	0.43	0.96	0.71	2.22
Private transport	0.93	1.14	0.94	1.23	1.21	1.15	1.18	0.95
iCF	3.72	3.07	3.69	0.82	3.57	3.31	3.43	0.93
Domestic	1.93	1.55	1.91	0.80	1.82	1.68	1.75	0.93
Imported	1.80	1.52	1.78	0.84	1.75	1.62	1.68	0.93
CF	4.97	4.98	4.97	1.00	5.43	5.58	5.51	1.03

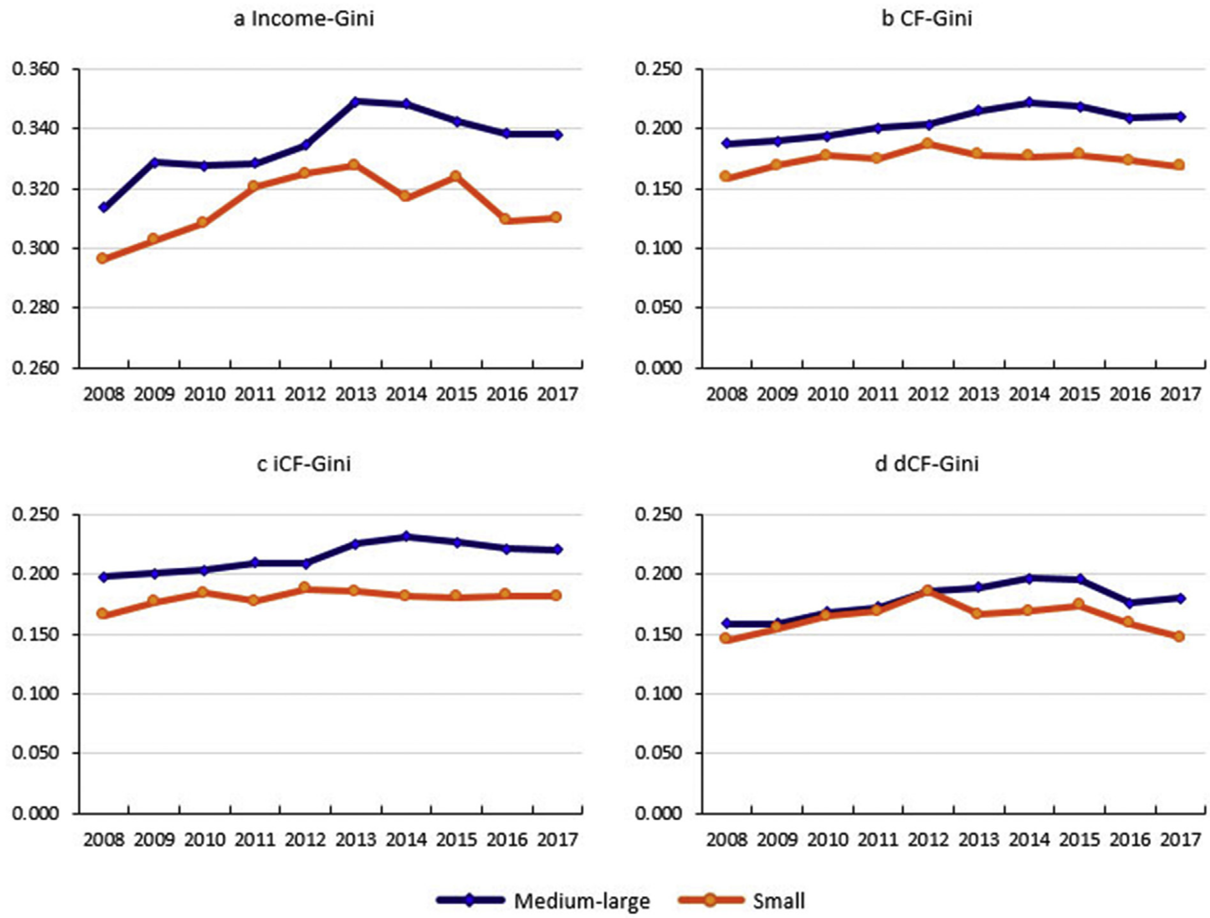

Fig. 2. Quantifying inequality by type of municipality in Spain for the period 2008–2017.

that since 2012 there has been an increase of the gap in inequality between both types of municipalities, which is also observed for the iCF-Gini and dCF-Gini indexes. Also, while the direct CO_2 emissions inequality between individuals located in medium-large and small municipalities is little, especially between 2008 and 2012, the inequality gap between some settlements and others is quite large in the case of iCF. The inequality of CF impacts in direct terms depends on the basket of the energy goods consumed. In this sense, we observe that the inequality in dCF is significantly lower than in iCF. This makes us think that the expenditure on transport, heating, refrigeration, and expenses for preparing meals made by poorer households is not very different to the consumption made by wealthier households in both types of municipalities. However, the imbalance in the iCF is much greater, given that consumption patterns among different income groups vary significantly.

3.3. *Impacts of carbon pricing in urban and rural households*

Carbon pricing serve to capture the external cost of CO_2 emissions into market prices, giving an economic signal to polluters for reducing the environmental harms (Wang et al., 2016; World Bank Group, 2019). We take into account two of the most important instruments currently available to put a price on carbon: emissions trading systems (ETS) and carbon taxes (CT). Both instruments end up having similarly regressive effects because they are quickly transmitted to the final prices of the economic system and share certain similarities (Burtraw et al., 2009; Shammin and Bullard, 2009). Thus, adopting similar approaches used by Feng et al. (2018) and Wang et al. (2019), we develop a hypothetical tax reform scenario based on a carbon pricing of 50 €/tCO_2 applied on the direct and indirect carbon footprint of each consumption category and household type. To this end, we assume: **a)** a carbon price that can be fully passed on to the price paid by consumers; **b)** the demand elasticities and substitution possibilities are ignored; **c)** there is no recycling of the carbon price revenues collected by the government. Therefore, in this section, we evaluate how heavy the carbon pricing burden is in different environments, by employing: **a)** absolute value of per capita carbon payment (i.e., the average cost per person paid for his/her own CO_2 emissions); and, **b)** the per capita carbon payment burden rate (i.e., the per capita carbon payment as share of the per capita expenditure, which is the sum of the pre-tax per capita expenditure and the per capita total carbon payment).

Fig. 3 shows that carbon pricing has regressive effects, i.e., the lower-income groups of households have to face a higher burden of carbon pricing than the richest ones. In other words, the per capita carbon payment captures a more significant proportion of their per capita expenditure (Böhringer et al., 2019). Nonetheless, the regressive effect is not equal across municipalities. Indeed, the simulated carbon pricing scenario reveals that families settled in small municipalities will be hit much more at each and every level of income. For example, the carbon payment burden rate on poorest households of small municipalities is 2.15% (€155), whereas it only represents 1.89% (€464) in households with the highest income level. But if we look at the households of the median-large settlements, the burden rates are always lower, varying between 1.73% (€112) and 1.53% (€452). These results are similar in amount to

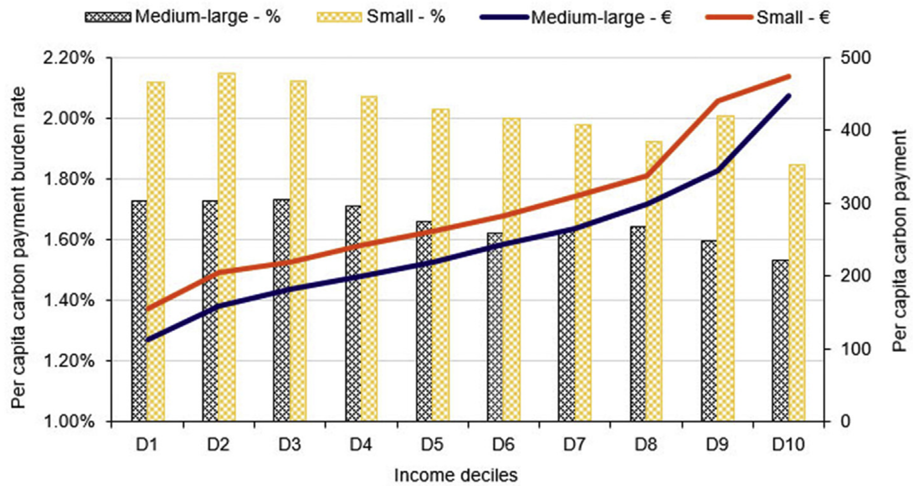

Fig. 3. Per capita carbon payment burden rate and expenditure by median-large and small municipalities (Spain, 2017).

those found by Wang et al. (2019) in a study applied for China, in which they estimated a carbon burdens rates that ranges from 0,5% in some regions to 1,5% in other regions.

A detailed analysis of the carbon pricing scenario, weighting both municipality size and rural-urban residential zone for the year 2017 is showed in Table 2. It allows us to evaluate to what extent the regressive impact generated by the hypothetical green tax reform could affect households. In dCF terms, the carbon payment would be born to a larger extend by rural households, regardless the municipality size where are located (i.e., the R/U is always higher than 1). At the same time, on a per capita basis, the households in small municipalities would face a carbon tax payment of 104 euros (0.42% of burden rate), sharply larger than the 64 euros paid by people in medium-large municipalities (0.76% of burden rate). Thus, the poor population of small municipalities and rural zones is the most vulnerable to the application of carbon pricing, as other previous studies have found, e.g., for Germany (Gill and Moeller, 2018), Ireland (Callan et al., 2009) or China (Wang et al., 2019). The highest regressive impact is found for a carbon tax applied to petroleum (i.e., gasoline or diesel), liquid and solid fuels, which usually are fundamental in the energy pattern of rural households. As a result, a green reform based on carbon pricing could generate, *ceteris paribus*, expected regressive effects on this population which may encourage the depopulation process that is hitting many rural municipalities of "empty Spain".

Looking at the iCF, we observe that the carbon burden rates are very similar for both rural and urban households (i.e., R/U is very close to 1). This is due to the fact that the emissions associated with the purchase of goods and services are driven by the level of consumption, making that the carbon payment burden rate ends up balanced for iCFs in any environment. Also, it should be noted the enormous practical problems in applying taxes on imports, since it involves skipping trade agreements already established between countries and, therefore, requires multilateral negotiations that generally do not end in agreement given the impacts on the competitiveness of the most coal-intensive sectors, frequently relocated in countries with weak labour and environmental regulations (López et al., 2014). Therefore, we find that the regressive impact of carbon pricing applied on total CFs will heavily affect rural households, and will tend to be slightly more regressive in small municipalities. In addition, the regressive impact found for these households will be greatly influenced by their direct emission patterns.

Finally, we go further by analysing the regressive impact by type of consumption product (i.e., the 47 COICOP categories) in Tables A1 and A2 of the Appendix. It allows us to highlight which are the more carbon-intensive goods and services, and, by extension, to point out in which items the carbon burden is more concentrated. We find that 5 of 47 products concentrate the mayor part of the carbon burden (between 66% and 75% depending on the type of household considered) in the following order: operation transport equipment, electricity gas and other fuels, food, transport services and catering services. This is because of these are very carbon-intensive goods (either directly or indirectly) and also have a significant weight in the families' consumption basket. Focusing on these carbon-intensive top five items, we observe several differences in regressive impacts among households. For instance, rural households may be more affected by carbon pricing on electricity, gas and other fuels, operation transport equipment (i.e., private transport) and food (R/U > 1), while families settled in urban zones could be more sensitive to carbon taxes on transport and catering services (R/U < 1). Looking at product carbon burdens, especially for energy goods and private and public transport, municipality size

Table 2
Carbon payment (CP) and carbon burden rate (CBR) by municipality size and residential zone in euros and percentages (Spain, 2017).

		Medium-large municipalities				Small municipalities			
		Urban	**Rural**	**Total**	**R/U**	**Urban**	**Rural**	**Total**	**R/U**
dCF	CP	62.31	95.68	63.91	1.54	93.16	113.94	104.34	1.22
	CBR	0.40%	0.78%	0.42%	1.92	0.65%	0.86%	0.76%	1.32
iCF	CP	186.22	153.30	184.65	0.82	178.42	165.28	171.35	0.93
	CBR	1.21%	1.25%	1.21%	1.03	1.24%	1.25%	1.25%	1.00
Domestic	CP	96.32	77.45	95.42	0.80	90.98	84.23	87.35	0.93
	CBR	0.62%	0.63%	0.62%	1.01	0.63%	0.64%	0.64%	1.00
Imported	CP	89.90	75.85	89.23	0.84	87.44	81.05	84.00	0.93
	CBR	0.58%	0.62%	0.58%	1.06	0.61%	0.61%	0.61%	1.01
CF	CP	248.53	248.98	248.56	1.00	271.58	279.22	275.69	1.03
	CBR	1.61%	2.02%	1.63%	1.26	1.89%	2.11%	2.01%	1.11

also is important.

4. Conclusions and policy implications

By employing an EEMRIO model adapted to the households in its consumption based-approach we have verified that the Spanish households' have reduced their CFs during the crisis (López et al., 2016); however, since 2013, the return of economic growth has boosted CO_2 emissions changing this trend. It has been pointed out that the "relief by density" hypothesis is fulfilled in Spain, given that, on a per capita basis, city and town dwellers emit fewer CO_2 emissions than those settled in small municipalities (between 0.34 and 0.54 tCO_2/cap depending on the year analysed). Such results are mainly due to the sharply unbalance in direct emissions between both types of households, making evident that people living in small settlements are more dependent on private transport and cannot take advantage of some infrastructures as natural gas and public transport networks. On the contrary, we have found that the iCF rises with the municipality size, mainly owing to the existing higher purchasing power in cities. We also observed that significant inequalities arise within municipalities. Indeed, all the Gini indexes calculated (i.e., Income-Gini, CF-Gini, iCF-Gini and dCF-Gini) reveal that inequality is always larger in the medium-large municipalities. However, since 2012, there has been a considerable increase in the gap of inequality between both types of municipalities, which has remained without falling even in spite of the economic growth during the post-crisis years. Finally, because the division based on municipalities size could not reflect rurality as such, we have added rural vs urban residential zone as a control variable, showing that the rural or urban character of the household heavily affects its CF structure, but not especially the figures of CO_2 emissions which largely depends on the municipality size.

At first glance, the statements above could suggest that policy in favour of population migration towards the urban zones would be beneficial for climate change mitigation. Indeed, it could reduce CO_2 emissions through the lower need for transportation to the city as well as to take advantage of environmental economies of scale linked to urban infrastructures. Nonetheless, this urbanisation process would require a vast increase in CO_2 emissions, above all if new infrastructure and homes are needed. On top of that, it could generate other critical problems such as air pollution, waste concentration, or overexploitation of natural resources. In this manner, the urbanisation process and the fulfilment of the "relief by density" hypothesis should not be seen as an acceptable solution to climate change issues linked to the unsustainable consumption pathway of Spanish households. For balancing CFs asymmetries, it is necessary to create and upgrade infrastructure and boost the local economy of the small settlements, making a less polluting lifestyles possible for their inhabitants. In this way, at least, they have the choice to meet their basic necessities with goods and services that are closer to their homes, as well as use more "cleaner" technologies and energy goods.

The CFs results from this study determine adverse distributional effects among individuals in front of a possible carbon pricing policy. However, the regressive impact is not equal across municipalities, being always higher for families located in small municipalities at every level of income. Rurality also is important for the regressive impact of carbon pricing, given that it determines the households' emissions patterns. Accordingly, mitigation policies based on carbon taxation should have in mind the carbon inequalities between households depending on its location (i.e., the municipality size) as well as its residential nuances (i.e., rural vs urban zone) in order to avoid the lack of social acceptance of green tax reform and the greatest damage to disadvantaged households, especially in small municipalities. It may be especially useful to recycle the carbon pricing revenues for implementing monetary compensation on the vulnerable population as well as fighting against energy poverty. This should be complemented by the development of energy and transport infrastructures in rural environments along with the establishment of subsidies for low carbon household appliances and electric photovoltaic self-consumption.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

CRediT authorship contribution statement

Manuel Tomás: Conceptualization, Methodology, Software, Validation, Formal analysis, Investigation, Resources, Data curation, Writing - original draft, Writing - review & editing, Visualization, Project administration, Funding acquisition. **Luis Antonio López:** Conceptualization, Methodology, Investigation, Resources, Writing - original draft, Writing - review & editing, Visualization, Supervision, Funding acquisition. **Fabio Monsalve:** Conceptualization, Software, Validation, Formal analysis, Resources, Data curation, Writing - review & editing, Funding acquisition.

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Appendix

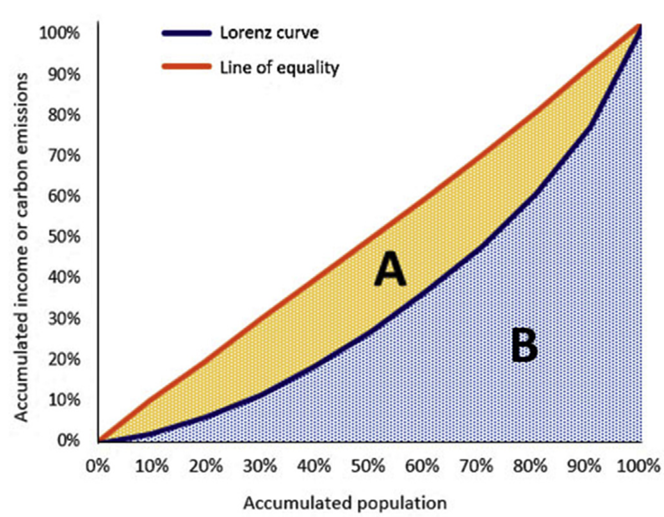

Fig. A1. A schematic diagram of the income or carbon Lorenz curve

Table A1
Carbon payment (CP) by type of product, municipality size and residential zone in euros (Spain, 2017)

	Medium-large municipalities				Small municipalities			
	Urban	Rural	Total	R/U	Urban	Rural	Total	R/U
Food	24.35	22.99	24.28	0.94	24.03	25.21	24.67	1.05
Non-alcoholic beverages	2.00	1.89	1.99	0.95	1.85	1.67	1.76	0.91
Alcoholic beverages	1.57	1.37	1.56	0.87	1.51	1.41	1.46	0.94
Tobacco	3.60	3.63	3.60	1.01	4.08	4.12	4.10	1.01
Narcotics	1.48	1.48	1.48	1.00	1.48	1.48	1.48	1.00
Clothing	9.00	6.99	8.90	0.78	8.33	7.63	7.95	0.92
Footwear	2.78	2.17	2.75	0.78	2.50	2.52	2.51	1.01
Actual rentals for housing	1.54	0.49	1.49	0.32	0.96	0.43	0.67	0.44
Imputed rentals for housing	6.39	5.60	6.35	0.88	6.18	6.24	6.21	1.01
Maintenance and repair of the dwelling	1.46	1.78	1.48	1.22	1.72	1.98	1.86	1.15
Water supply and miscellaneous services relating to the dwelling	2.44	1.14	2.37	0.47	1.51	0.93	1.20	0.61
Electricity, gas and other fuels	44.40	65.59	45.41	1.48	66.18	88.68	78.28	1.34
Furniture and furnishings, carpets and other floor coverings	2.22	1.46	2.19	0.66	1.97	2.01	1.99	1.02
Household textiles	0.88	0.94	0.88	1.07	1.00	0.93	0.96	0.93
Household appliances	1.96	1.45	1.93	0.74	2.00	1.88	1.94	0.94
Glassware, tableware and household utensils	1.22	0.90	1.20	0.74	1.28	0.87	1.06	0.68
Tools and equipment for house and garden	0.47	0.54	0.47	1.16	0.45	0.62	0.54	1.39
Goods and services for routine household maintenance	4.13	3.32	4.09	0.80	3.35	3.45	3.40	1.03
Medical products, appliances and equipment	3.65	2.70	3.60	0.74	3.21	2.81	2.99	0.88
Out-patient services	1.82	1.45	1.80	0.80	1.70	1.63	1.66	0.96
Hospital services	0.36	0.48	0.36	1.34	0.28	0.29	0.29	1.06
Purchase of vehicles	8.43	7.26	8.37	0.86	8.86	7.04	7.88	0.79
Operation of personal transport equipment	62.54	74.96	63.14	1.20	80.31	76.12	78.06	0.95
Transport services	17.30	8.04	16.86	0.46	9.33	6.46	7.79	0.69
Postal services	0.07	0.05	0.07	0.73	0.03	0.04	0.03	1.26
Telephone and telefax equipment	0.53	0.38	0.52	0.73	0.42	0.35	0.38	0.83
Telephone and telefax services	2.07	1.71	2.05	0.83	2.05	1.80	1.91	0.88
Audio-visual, photographic and information processing equipment	1.25	0.82	1.23	0.65	1.09	0.74	0.91	0.68
Other major durables for recreation and culture	0.16	0.10	0.16	0.65	0.30	0.12	0.20	0.40
Other recreational items and equipment, gardens and pets	2.51	2.61	2.52	1.04	2.84	2.36	2.58	0.83
Recreational and cultural services	3.75	2.90	3.71	0.77	3.68	3.13	3.38	0.85
Newspapers, books and stationery	1.14	0.58	1.12	0.50	0.90	0.62	0.75	0.69
Package holidays	1.71	0.89	1.67	0.52	1.30	0.97	1.12	0.75
Pre-primary and primary education	0.14	0.04	0.14	0.27	0.11	0.04	0.07	0.36
Secondary education	0.10	0.03	0.09	0.35	0.06	0.03	0.05	0.58
Post-secondary non-tertiary education	0.10	0.03	0.09	0.35	0.06	0.03	0.05	0.58
Tertiary education	0.26	0.07	0.25	0.28	0.27	0.12	0.19	0.45
Education not definable by level	0.35	0.26	0.35	0.75	0.22	0.21	0.21	0.96
Catering services	15.52	10.61	15.29	0.68	14.12	12.67	13.34	0.90
Accommodation services	1.72	0.67	1.67	0.39	1.29	1.01	1.14	0.78
Personal care	5.75	4.27	5.68	0.74	4.86	4.14	4.47	0.85
Prostitution	0.00	0.00	0.00	1.00	0.00	0.00	0.00	1.00
Personal effects n.e.c.	1.38	1.03	1.36	0.75	0.89	1.12	1.01	1.26
Social protection	1.06	0.87	1.05	0.83	0.56	0.67	0.62	1.19
Insurance	0.60	0.51	0.59	0.86	0.59	0.49	0.53	0.82
Financial services n.e.c.	1.42	1.35	1.42	0.95	1.09	1.08	1.08	0.99
Other services n.e.c.	1.07	0.59	1.05	0.55	0.89	1.11	1.01	1.24

References

Amores, A.F., 2018. The challenge of using consumption surveys to feed macroeconomic models. In: 6th SHAIO Workshop, Madrid, Spain.

Anghel, B., Basso, H., Bover, O., Casado, J.M., Hospido, L., Izquierdo, M., Kataryniuk, I.A., Lacuesta, A., Montero, J.M., Vozmediano, E., 2018. Income, consumption and wealth inequality in Spain. SERIEs 9, 351–387.

Arce, G., Zafrilla, J.E., López, L.A., Tobarra, M.Á., 2017. Carbon footprint of human settlements in Spain. In: Fernández, R.Á., Zubelzu, S., Martínez, R. (Eds.), Carbon Footprint and the Industrial Life Cycle. Springer.

Böhringer, C., Garcia-Muros, X., González-Eguino, M., 2019. Greener and fairer: a progressive environmental tax reform for Spain. Economics of Energy & Environmental Policy.

Brizga, J., Feng, K., Hubacek, K., 2017. Household carbon footprints in the Baltic States: a global multi-regional input–output analysis from 1995 to 2011. Appl. Energy 189, 780–788.

Bureau, B., 2011. Distributional effects of a carbon tax on car fuels in France. Energy Econ. 33, 121–130.

Burtraw, D., Sweeney, R., Walls, M., 2009. The incidence of U.S. Climate policy: alternative uses of revenues from a cap-and-trade auction. Natl. Tax J. 62, 497–518.

Callan, T., Lyons, S., Scott, S., Tol, R.S.J., Verde, S., 2009. The distributional implications of a carbon tax in Ireland. Energy Pol. 37, 407–412.

Cazcarro, I., Amores, A.F., Arto, I., Kratena, K., 2020. Bridge matrices for linking multisectoral economic models and consumption surveys for the EU-28 countries. In: Under Review.

CES, 2018. El medio rural y su vertebración social y territorial. Consejo Económico y Social (España), Madrid.

Chancel, L., Piketty, T., 2015. Carbon and Inequality: from Kyoto to Paris. Trends in the Global Inequality of Carbon Emissions (1998-2013) & Prospects for an Equitable Adaptation Fund. Paris School of Economics, Paris.

Chen, G., Wiedmann, T., Wang, Y., Hadjikakou, M., 2016. Transnational city carbon footprint networks – exploring carbon links between Australian and Chinese cities. Appl. Energy 184, 1082–1092.

Corsatea, T.D., Lindner, S., Arto, I., Roman, M.V., Rueda-Cantuche, J.M., Afonso, A.V., Amores, A.F., Neuwahl, F., 2019. World Input-Output Database Environmental Accounts. Joint Research Centre (Seville site).

Deaton, A., 2005. Measuring poverty in a growing world (or measuring growth in a poor world). Rev. Econ. Stat. 87, 1–19.

Dodman, D., 2009. Blaming cities for climate change? An analysis of urban greenhouse gas emissions inventories. Environ. Urbanization 21, 185–201.

Duarte, R., Mainar, A., Sánchez-Chóliz, J., 2012. Social groups and CO2 emissions in

Spanish households. Energy Pol. 44, 441–450.
EUROSTAT, 2020. Bilateral Exchange Rates. European Commission.
Feng, K., Hubacek, K., Liu, Y., Marchán, E., Vogt-Schilb, A., 2018. Managing the distributional effects of energy taxes and subsidy removal in Latin America and the Caribbean. Appl. Energy 225, 424–436.
Gill, B., Moeller, S., 2018. GHG emissions and the rural-urban divide. A carbon footprint analysis based on the German official income and expenditure survey. Ecol. Econ. 145, 160–169.
Glaeser, E.L., 2011. Triumph of the City : How Our Greatest Invention Makes Us Richer, Smarter, Greener, Healthier, and Happier. Penguin Press, New York.
Gobierno de España, 2020. Acuerdo de consejo de ministros por el que se aprueba la declaración del gobierno ante la emergencia climática y ambiental. Gobierno de España retrieved from. https://www.miteco.gob.es/es/prensa/declaracionemergenciaclimatica_tcm30-506551.pdf.
Harris, S., Weinzettel, J., Bigano, A., Källmén, A., 2020. Low carbon cities in 2050? GHG emissions of European cities using production-based and consumption-based emission accounting methods. J. Clean. Prod. 248, 119206.
Heinonen, J., Jalas, M., Juntunen, J.K., Ala-Mantila, S., Junnila, S., 2013. Situated lifestyles: I. How lifestyles change along with the level of urbanization and what the greenhouse gas implications are—a study of Finland. Environ. Res. Lett. 8, 025003.
Huang, R., Zhang, S., Liu, C., 2018. Comparing urban and rural household CO2 emissions—case from China's four megacities: Beijing, Tianjin, Shanghai, and Chongqing. Energies 11, 1257.
Hubacek, K., Feng, K., Minx, J., Pfister, S., Zhou, N., 2014. Teleconnecting consumption to environmental impacts at multiple spatial scales. J. Ind. Ecol. 18, 7–9.
Hubacek, K., Baiocchi, G., Feng, K., Muñoz Castillo, R., Sun, L., Xue, J., 2017a. Global carbon inequality. Energy Ecol. Environ. 2, 361–369.
Hubacek, K., Baiocchi, G., Feng, K., Patwardhan, A., 2017b. Poverty eradication in a carbon constrained world. Nat. Commun. 8, 912.
INE, 2020a. Demography and Population. Population Figures and Demographic Censuses. Population Figures. Spanish National Institute of Statistics.
INE, 2020b. Spanish Household Budget Survey [Encuesta de presupuestos familiares de España]. Spanish National Institute of Statistics.
INE, 2020c. Spanish National Accounts: Household Final Consumption Expenditure (COICOP). Spanish National Institute of Statistics.
IPCC, 2006. General Guidance and Reporting. Intergovernmental Panel on Climate Change.
Jain, S., Kumar, A., 2018. Energy Comsumption Pattern for Indian Households, International Conference on Recent Innovations in Engineering and Technology, India.
Krugman, P., 1991. Increasing returns and economic geography. J. Polit. Econ. 99, 483–499.
Labandeira, X., Labeaga, J.M., López-Otero, X., 2011. Energy Demand for Heating in Spain: an Empirical Analysis with Policy Purposes, p. 2011. WP 6.
Lenzen, M., Wier, M., Cohen, C., Hayami, H., Pachauri, S., Schaeffer, R., 2006. A comparative multivariate analysis of household energy requirements in Australia, Brazil, Denmark, India and Japan. Energy 31, 181–207.
López, L.A., Arce, G., Zafrilla, J.E., 2014. Financial crisis, virtual carbon in global value chains, and the importance of linkage effects. The Spain-china case. Environ. Sci. Technol. 48, 36–44.
López, L.A., Arce, G., Morenate, M., Monsalve, F., 2016. Assessing the inequality of Spanish households through the carbon footprint: the 21st century Great recession effect. J. Ind. Ecol. 20, 571–581.
López, L.A., Arce, G., Morenate, M., Zafrilla, J.E., 2017. How does income redistribution affect households' material footprint? J. Clean. Prod. 153, 515–527.
Mahajan, S., Beutel, J., Guerrero, S., Inomata, S., Larsen, S., Moyer, B., Remond-Tiedrez, I., Rueda-Cantuche, J.M., Simpson, L.H., Thage, B., 2018. Handbook on Supply, Use and Input-Output Tables with Extensions and Applications.
McCully, C.P., 2014. Integration of Micro-and Macrodata on Consumer Income and Expenditures, Measuring Economic Sustainability and Progress. University of Chicago Press, pp. 137–179.
Milanovic, B., 2013. Global income inequality in numbers: in history and now. Glob. Pol. 4, 198–208.
Miller, R.E., Blair, P.D., 2009. Input-output Analysis : Foundations and Extensions, second ed. Cambridge University Press, Cambridge.
Min, J., Rao, N.D., 2018. Estimating uncertainty in household energy footprints. J. Ind. Ecol. 22, 1307–1317.
MITECO, 2019. Informe de inventario nacional de gases de efecto invernadero. Ministerio para la Transición Ecológica.
Moll, H.C., Noorman, K.J., Kok, R., Engström, R., Throne-Holst, H., Clark, C., 2005. Pursuing more sustainable consumption by analyzing household metabolism in European countries and cities. J. Ind. Ecol. 9, 259–275.
Moran, D., Kanemoto, K., Jiborn, M., Wood, R., Többen, J., Seto, K.C., 2018. Carbon footprints of 13 000 cities. Environ. Res. Lett. 13, 064041.
Norman, J., MacLean, H.L., Kennedy, C.A., 2006. Comparing high and low residential density: life-cycle analysis of energy use and greenhouse gas emissions. J. Urban Plann. Dev. 132, 10–21.
Ottelin, J., Heinonen, J., Nässén, J., Junnila, S., 2019. Household carbon footprint patterns by the degree of urbanisation in Europe. Environ. Res. Lett. 14, 114016.
Palma, J.G., 2011. Homogeneous middles vs. Heterogeneous tails, and the end of the 'inverted-U': it's all about the share of the rich. Dev. Change 42, 87–153.
Pashardes, P., Pashourtidou, N., Zachariadis, T., 2014. Estimating welfare aspects of changes in energy prices from preference heterogeneity. Energy Econ. 42, 58–66.
Peters, G.P., Andrew, R.M., Karstensen, J., 2016. Global Environmental Footprints: A Guide to Estimating, Interpreting and Using Consumption-Based Accounts of Resource Use and Environmental Impacts. Nordic Council of Ministers.
Piketty, T., Saez, E., 2014. Inequality in the long run. Science 344, 838–843.
Pinkovskiy, M., Sala-i-Martin, X., 2016. Lights, camera … income! Illuminating the national accounts-household surveys debate *. Q. J. Econ. 131, 579–631.
Poumanyvong, P., Kaneko, S., 2010. Does urbanization lead to less energy use and lower CO2 emissions? A cross-country analysis. Ecol. Econ. 70, 434–444.
Puga, D., 2010. The magnitude and causes of agglomeration economies. J. Reg. Sci. 50, 203–219.
Rau, H., Vega, A., 2012. Spatial (im) mobility and accessibility in I reland: implications for transport policy. Growth Change 43, 667–696.
Reinders, A.H.M.E., Vringer, K., Blok, K., 2003. The direct and indirect energy requirement of households in the European Union. Energy Pol. 31, 139–153.
Schubert, J., Gill, B., 2015. Are Cities Really Smart? the Environmental Impact of Urban and Rural Municipalities According to Different Methodological Perspectives, Handbook of Research Methods and Applications in Environmental Studies. Edward Elgar Publishing.
Shammin, M.R., Bullard, C.W., 2009. Impact of cap-and-trade policies for reducing greenhouse gas emissions on U.S. households. Ecol. Econ. 68, 2432–2438.
Timmer, M.P., Dietzenbacher, E., Los, B., Stehrer, R., de Vries, G.J., 2015. An illustrated user guide to the world input–output database: the case of global automotive production. Rev. Int. Econ. 23, 575–605.
Timmer, M.P., Los, B., Stehrer, R., de Vries, G.J., 2016. An Anatomy of the Global Trade Slowdown Based on the WIOD 2016 Release. Groningen Growth and Development Centre (GGDC) Research Memorandum.
UN, 2019a. Demographic Yearbook 2018, New York.
UN, 2019b. World Urbanization Prospects: the 2018 Revision [key Facts]. United Nations.
VandeWeghe, J.R., Kennedy, C., 2007. A spatial analysis of residential greenhouse gas emissions in the Toronto census metropolitan area. J. Ind. Ecol. 11, 133–144.
Wang, R., Jiang, Z., 2017. Energy consumption in China's rural areas: a study based on the village energy survey. J. Clean. Prod. 143, 452–461.
Wang, Q., Hubacek, K., Feng, K., Wei, Y.-M., Liang, Q.-M., 2016. Distributional effects of carbon taxation. Appl. Energy 184, 1123–1131.
Wang, Q., Hubacek, K., Feng, K., Guo, L., Zhang, K., Xue, J., Liang, Q.-M., 2019. Distributional impact of carbon pricing in Chinese provinces. Energy Econ. 81, 327–340.
Wiedenhofer, D., Guan, D., Liu, Z., Meng, J., Zhang, N., Wei, Y.-M., 2016. Unequal household carbon footprints in China. Nat. Clim. Change 7, 75.
Wiedmann, T., 2009. A review of recent multi-region input–output models used for consumption-based emission and resource accounting. Ecol. Econ. 69, 211–222.
World Bank Group, 2019. State and Trends of Carbon Pricing 2019, State and Trends of Carbon. Pricing World Banck, Washington.
Zafrilla, J.E., López, L.A., 2018. Chapter 2 - booming and stagnation of Spanish construction sector through the extended carbon footprint concept. In: Muthu, S.S. (Ed.), Environmental Carbon Footprints. Butterworth-Heinemann, pp. 19–43.
Zhang, H., Lahr, M.L., 2018. Households' energy consumption change in China: a multi-regional perspective. Sustainability 10, 2486.

Table A2
Carbon burden rate (CBR) by type of product, municipality size and residential zone in percentages (Spain, 2017)

	Medium-large municipalities				**Small municipalities**			
	Urban	**Rural**	**Total**	**R/U**	**Urban**	**Rural**	**Total**	**R/U**
Food	0.16%	0.19%	0.16%	1.19	0.17%	0.19%	0.18%	1.14
Non-alcoholic beverages	0.01%	0.02%	0.01%	1.19	0.01%	0.01%	0.01%	0.98
Alcoholic beverages	0.01%	0.01%	0.01%	1.10	0.01%	0.01%	0.01%	1.02
Tobacco	0.02%	0.03%	0.02%	1.27	0.03%	0.03%	0.03%	1.09
Narcotics	0.01%	0.01%	0.01%	1.26	0.01%	0.01%	0.01%	1.08
Clothing	0.06%	0.06%	0.06%	0.98	0.06%	0.06%	0.06%	0.99
Footwear	0.02%	0.02%	0.02%	0.98	0.02%	0.02%	0.02%	1.09
Actual rentals for housing	0.01%	0.00%	0.01%	0.40	0.01%	0.00%	0.00%	0.48
Imputed rentals for housing	0.04%	0.05%	0.04%	1.10	0.04%	0.05%	0.05%	1.10
Maintenance and repair of the dwelling	0.01%	0.01%	0.01%	1.53	0.01%	0.02%	0.01%	1.25
Water supply and miscellaneous services relating to the dwelling	0.02%	0.01%	0.02%	0.59	0.01%	0.01%	0.01%	0.67
Electricity, gas and other fuels	0.29%	0.53%	0.30%	1.85	0.46%	0.67%	0.57%	1.45
Furniture and furnishings, carpets and other floor coverings	0.01%	0.01%	0.01%	0.83	0.01%	0.02%	0.01%	1.11
Household textiles	0.01%	0.01%	0.01%	1.34	0.01%	0.01%	0.01%	1.01
Household appliances	0.01%	0.01%	0.01%	0.93	0.01%	0.01%	0.01%	1.02
Glassware, tableware and household utensils	0.01%	0.01%	0.01%	0.93	0.01%	0.01%	0.01%	0.74
Tools and equipment for house and garden	0.00%	0.00%	0.00%	1.45	0.00%	0.00%	0.00%	1.50
Goods and services for routine household maintenance	0.03%	0.03%	0.03%	1.01	0.02%	0.03%	0.02%	1.12
Medical products, appliances and equipment	0.02%	0.02%	0.02%	0.93	0.02%	0.02%	0.02%	0.95
Out-patient services	0.01%	0.01%	0.01%	1.00	0.01%	0.01%	0.01%	1.04
Hospital services	0.00%	0.00%	0.00%	1.69	0.00%	0.00%	0.00%	1.15
Purchase of vehicles	0.05%	0.06%	0.06%	1.08	0.06%	0.05%	0.06%	0.86
Operation of personal transport equipment	0.41%	0.61%	0.41%	1.50	0.56%	0.58%	0.57%	1.03
Transport services	0.11%	0.07%	0.11%	0.58	0.07%	0.05%	0.06%	0.75
Postal services	0.00%	0.00%	0.00%	0.92	0.00%	0.00%	0.00%	1.37
Telephone and telefax equipment	0.00%	0.00%	0.00%	0.91	0.00%	0.00%	0.00%	0.90
Telephone and telefax services	0.01%	0.01%	0.01%	1.04	0.01%	0.01%	0.01%	0.95
Audio-visual, photographic and information processing equipment	0.01%	0.01%	0.01%	0.82	0.01%	0.01%	0.01%	0.74
Other major durables for recreation and culture	0.00%	0.00%	0.00%	0.82	0.00%	0.00%	0.00%	0.43
Other recreational items and equipment, gardens and pets	0.02%	0.02%	0.02%	1.31	0.02%	0.02%	0.02%	0.90
Recreational and cultural services	0.02%	0.02%	0.02%	0.97	0.03%	0.02%	0.02%	0.92
Newspapers, books and stationery	0.01%	0.00%	0.01%	0.63	0.01%	0.00%	0.01%	0.74
Package holidays	0.01%	0.01%	0.01%	0.66	0.01%	0.01%	0.01%	0.81
Pre-primary and primary education	0.00%	0.00%	0.00%	0.34	0.00%	0.00%	0.00%	0.39
Secondary education	0.00%	0.00%	0.00%	0.44	0.00%	0.00%	0.00%	0.63
Post-secondary non-tertiary education	0.00%	0.00%	0.00%	0.44	0.00%	0.00%	0.00%	0.63
Tertiary education	0.00%	0.00%	0.00%	0.36	0.00%	0.00%	0.00%	0.49
Education not definable by level	0.00%	0.00%	0.00%	0.94	0.00%	0.00%	0.00%	1.04
Catering services	0.10%	0.09%	0.10%	0.86	0.10%	0.10%	0.10%	0.97
Accommodation services	0.01%	0.01%	0.01%	0.49	0.01%	0.01%	0.01%	0.85
Personal care	0.04%	0.03%	0.04%	0.93	0.03%	0.03%	0.03%	0.92
Prostitution	0.00%	0.00%	0.00%	1.26	0.00%	0.00%	0.00%	1.08
Personal effects n.e.c.	0.01%	0.01%	0.01%	0.94	0.01%	0.01%	0.01%	1.37
Social protection	0.01%	0.01%	0.01%	1.04	0.00%	0.01%	0.00%	1.29
Insurance	0.00%	0.00%	0.00%	1.08	0.00%	0.00%	0.00%	0.89
Financial services n.e.c.	0.01%	0.01%	0.01%	1.20	0.01%	0.01%	0.01%	1.08
Other services n.e.c.	0.01%	0.00%	0.01%	0.69	0.01%	0.01%	0.01%	1.35