Distributional Impacts of Green Taxes on Food Consumption in Catalonia

Wisdom Dogbe[1], Jose Maria Gil[2]

[1] University Polytechnic of Catalonia (CREDA)
Parc Mediterrani de la Tecnologia, Edifici ESAB, C/Esteve Terrades, 8, 08860 Castelldefels

[2] University Polytechnic of Catalonia (CREDA)
Parc Mediterrani de la Tecnologia, Edifici ESAB, C/Esteve Terrades, 8, 08860 Castelldefels

Contribution presented at the XV EAAE Congress, "Towards Sustainable Agri-food Systems: Balancing Between Markets and Society"

August 29th – September 1st, 2017

Parma, Italy

Introduction

Current dietary habits contribute immensely to climate change as a result of the large amount of greenhouse gas produced (Castellón, Boonsaeng, & Carpio, 2015). The food chain contributes to about 30% of greenhouse gas produced in Europe compared to 20% from total fossil fuel production (Audsley et al., 2010). In the European Union, Spain is the sixth largest emitters by accounting for 7.7% of total GHG emissions in 2011. Out of the 55% of diffuse greenhouse emission generated in Spain, the agricultural sector, second to the transport sector, contributes about 14% (Bourne, Childs, Philippidis, & Feijoo, 2012).

(Hedenus, Wirsenius, & Johansson, 2014) showed that emission reduction in the agro-food sector can be achieved by: 1) productivity improvements; 2) technological changes (supply-side measures); and 3) changes in consumption behavior (demand-side measures). Supply side measures such as command-and-control regulations, cap-and-trade systems or Pigovian (corrective) taxes have already been used in the European Union (EU) to correct for the massive GHG emissions arising from the production of consumer goods (Chen et al., 2015). However, the use of command and control measures have been found to be economically inefficient and does not lead to optimal production compared to cap-and-trade measures or Pigovian taxes (Burchell & Lightfoot, 2001).

Pigou (1957) proposed that Governments should influence the behavior of economic agents causing negative (positive) externalities through taxes (subsidies) (Endres, 2010). The Pigovian tax (subsidy) is aimed at internalizing externalities generated by economic activities and it is set equal to the social cost (benefit) of the negative (positive) externality. The relevance of a Pigovian tax on unhealthy/high carbon footprint foods has been justified under the assumption of close to perfect competition food industry. Under such an assumption, the burden of a Pigovian tax is irrelevant whether applied to the supply side or the demand end. For this reason, several studies have shown that imposing Pigovian taxes on food demand rather than on food supply constitute a cost efficient emission reduction strategy. This is because consumers are able to adjust to the tax according to their efficient level of consumption (Edjabou & Smed, 2013). (Mytton, Clarke, & Rayner, 2012) showed that positively changing consumption behavior through food taxes could be attractive not only from the health side but also from the climate perspective. Contrary to taxing production which has always been a delicate issue because of "carbon leakage" (Wirsenius, Hedenus, & Mohlin, 2011) and high monitoring cost (Schmutzler & Goulder, 1997).

According to EU's 2013 progress report on emission, Spain is expected to miss it Kyoto Protocol target of greenhouse gas emission reduction by 3% since current national policies are not sufficient, "the Council recommended that … and Spain shift the tax burden away from labor to taxes less detrimental to growth, including environmental taxes" (*Progress towards achieving the Kyoto and EU 2020 objectives*, 2013:15)

In this regard, the objective of this paper is to evaluate the potential effects of imposing a Pigovian or CO_2 equivalent tax on food consumption in Spain. From food demand elasticities, we showed that levying a CO_2 equivalent tax has two effects: 1) a reduction in greenhouse gas emission and improved nutrient intake ratios; and 2) distributional effects. In spite of the increasing importance of this topic in the policy arena as well as among researchers, up to our knowledge only a very few papers have been published. We highlight the seminal paper by (Säll & Gren, 2015) for Sweden, and that by (Garcia-Muros, Markandya, Romero-Jordán, & González-Eguino, 2016) who also focused on Spain. The former was limited to the meat and dairy sectors and the distributional impact of the Pigovian tax was ignored. Even though, the latter considered 10 food groups and assessed the distributional impact of the taxes. From a methodological point of view, both studies applied the Almost Ideal Demand System (AIDS) model ignoring the impact of household heterogeneity in their demand estimation.

Moreover, both failed to account for substitution effects in food demand and the impact of consumption discount rates on emission reduction strategies. In this paper, we have tried to overcome such limitations by: 1) estimating an incomplete demand system using homescan data; 2) calculating demand elasticities by estimating an approximate Exact Affine Stone Index (EASI) demand model which takes into account household heterogeneity; 3) simulating carbon tax policy scenarios based on current EU medium- and long-term emission reduction objectives; and 4) explicitly considering nutritional as well as welfare effects.

The remainder of the article is structured as follows. Section 2 describes the model and data used in this study. Section 3 shows the main results and brief discussions. The paper ends with some concluding remarks and limitations.

Data and model

Data

This study uses microdata collected by Kantar Worldpanel. It's a homescan panel data from a representative sample of 1146 households in Catalonia (North-East Spain). The data set contained all day-to-day food purchases of Catalonian households in 2012 as well as some household socio-demographic characteristics. From the total of 1146 households, only those who have remained in the sample for at least 45 weeks were considered. Purchased quantities and expenditures for each single food product have been aggregated to the annual level for each household. All food products have been aggregated into 16 food categories (alcoholic drinks are not included while non-alcoholic drinks are included in the residual category for the purpose of this paper). Unit values were obtained by dividing expenditures by the purchasing quantities[1].

Table 1 shows some descriptive statistics of the data used. As can be observed, Spanish consumers allocate a significant share of their food budget to meat (fresh and processed) (26%) followed by fish and seafood (14%), milk and dairy products (including cheese) (14%), fruits (10%) and vegetables (8%). Differences among socioeconomic groups are found in Figure 1. In the high social class, the consumption of fruits, vegetables and beef and lamb are significantly more important than the other social classes, while in the case of poultry, pork and grains, differences are not significant. On the opposite side, in lower social class households, the consumption of grains, starchy roots, processed meat and composite dishes are slightly more important (although not significant) relative to the other social classes.

Model

Estimating Food Price Elasticities

Food price elasticities have been calculated by estimating an approximate Exact Affine Stone Index (EASI) demand model (Lewbel & Pendakur, 2009) which incorporates household characteristics. The EASI demand model has several advantages over the traditional Almost Ideal demand System (AIDS) as it derives the Implicit Marshallian demand function which combine desirable properties of both Hicksian and Marshallian demand functions. Moreover, the error terms can be interpreted us unobserved preference heterogeneity among individuals and Engle curves can adopt any shape over real expenditures. Finally, similar to the AIDS model, we can estimate a linear approximation which generates similar results than the full model.

The approximate EASI demand equation expresses budget shares, w^i, as a function of food prices *p*, total household expenditure *y*, and socio-demographic characteristics *z*, as follows:

[1] Zero purchases is not an issue in this study. The highest percentages of zero purchases have been found in Beef, veal and lamb (3.1%) and Snacks and other food (2.7%).

$$w^i = \sum_{r=0}^{5} b_r \bar{y}^r + Cz + Dz\bar{y} + \sum_{p=1}^{k} Ap + Bp\bar{y} + \varepsilon \quad \text{... Equation 1}$$

where b_r, r= 0,1,2,..., 5 are the parameters that control the shape of the Engel curve (up to a fifth-order polynomial); $\bar{y} = x - \acute{p}\bar{w}$ is the log of the Stone index-deflated nominal expenditure (being $\bar{w}$ the mean budget shares); C is a matrix of parameters corresponding to socio-demographic variables excluding the intercept; D is the matrix of coefficients from interaction between income and socio-demographic variables, A is a matrix of prices coefficients, B is a matrix of coefficients from interaction between income and prices, and ε is the error term. For the model to be consistent with theory, the budget share equations w^j are required to satisfy the properties of adding-up, linear homogeneity and Slutsky symmetry.

Taking into account the characteristics of our data set, quality adjusted prices were calculated from unit values *(total expenditure/total quantity)*. The EASI demand system in (1) was estimated using derived quality adjusted prices following (Cox & Wohlgenant, 1986). Moreover, to preserve household heterogeneity, the actual log of the Stone index $y = x - \acute{p}w$ was used in the estimation of system (1) with $\bar{y}$ as an instrument because of income endogeneity. Finally, for parsimony (Castellón et al. (2015), the interaction effects between prices and social demographics were not included for parsimony. Linear 3-Stage least Squares was used to estimate demand parameters.

By deriving (1) with respect to expenditure and log prices, we get the Marshallian demand semi-elasticities. The Hicksian and Marshallian price elasticities (at sample means) as well as the expenditure elasticities can be obtained using the following expressions (Lewbel & Pendakur, 2008)

- Hicksian price elasticities:

$$\frac{\partial q^i}{\partial p^j} = \frac{(A_j + By)}{\overline{w^i}} + \overline{w^j} - \delta \quad \text{... Equation 2}$$

- Marshallian price elasticities:

$$\frac{\partial q^{iM}}{\partial p^j} = \frac{(A_j + By)}{\overline{w^i}} - \delta - \frac{\overline{w^j}}{\overline{w^i}}\left(\left[\sum_{r=1}^{5} b_r r y^{r-1} + Dz + Bp\right]\right) \quad \text{..................... Equation 3}$$

δ =1 where i=j and zero if otherwise

- Expenditure elasticity (the total expenditure is assumed to be constant):

$$E = \left(diag(w)\right)^{-1} \left[\left(I_j + BP\right)^{-1} B\right] + 1_j \text{... Equation 4}$$

W refers to the $J \times 1$ vector of observed budget shares, B is a $J \times 1$ vector whose i_{th} element equals $b_r \bar{y}^r$, P is the $J \times 1$ vector of log prices, and 1_J is a $J \times 1$ vector of ones.

Measuring the impact of CO_2 equivalent (CO_2-Eq) tax on food demand

To measure the impact of CO_2-eq tax on food demand, we need data on CO_2 emissions per kg of food products. Although several studies have provided some figures, there isn't any single study that covers all food categories considered in this study for Spain (Macdiarmid et al, 2012). CO_2 equivalent emissions for major food products consumed in the European Union (EU) were taken from (Hartikainen & Pulkkinen, 2016). In spite of the limitation to use this data due to differences in food production systems in Spain and other EU countries, we consider that the data set will serve the purposes of this study as it uses a common framework to estimate GHG emissions for a large list of food products. Average values for the 16 food groups considered in this study are shown in **Table 1**.

The impact of imposing a carbon/green/Pigovian tax on demand for food has been analyzed taking into account the price/kg of CO_2 equivalent emissions for each of the 16 food categories. Previous studies have used a wide range of values running from 0 USD up to 400 USD (Stern, 2007). To cite only two examples, (Edjabou & Smed, 2013), based on the Stern Report (2006), assumed a social cost of carbon of 85 USD per ton CO_2 equivalent, while Irz, Leroy, Réquillart, Soler, & others (2015) assumed a value of 32 Euro (35 USD), based on the meta analyses carried out by Tol (2012).

This study has considered four tax scenarios taking into account two different sources to calculate the price of CO_2 equivalent emissions: 1) The US Environmental Protection Agency (EPA) which uses a comprehensive procedure for calculating the social cost of emission based on three consumption discount rates (low consumption discount rates means greater emphasis on inter- and intra-generational equity and vice versa). We have chosen two discount rates of 5% and 2.5% which generate a social cost of CO_2/t of \$57 and \$11, respectively; and 2) The European Union´s (EU) medium- and long- term carbon emission reduction objectives: the EU proposes a social cost of CO_2/t equivalent emission of 56 Euros and 200 Euros to reduce carbon emissions by 20% and 60% by 2020 and 2050 across the EU.

Following Baumol & Oates (1975), the taxes imposed on each food category have been calculated in the following way. We have first calculated the average CO_2 equivalent emissions generated by each food category using the data from (Hartikainen & Pulkkinen, 2016). The tax on the i-th food category, (Tax_i) has been calculated as the product of the average CO_2 equivalent emissions per kg of each food category (e_i) and the social cost (SC_m) associated with each of the four scenarios mentioned above (based on EPA and EPA and EU objectives):

$$\boldsymbol{Tax_i = e_i * SC_m} \quad \text{...Equation 5}$$

From (5), the tax rate for the i-th food category *i*, has been calculated as:

$$\boldsymbol{Tax_{rate.i} = \frac{p_1 - p_0}{p_0} * 100} \quad \text{..................................Equation 6}$$

where $p_1 = Tax_i + p_0$, being p_0 the average price before the tax.

Results are shown in

Table 2. We have found tax levels ranging from 0.07% for Starchy roots, legumes, and pulse category to 55.07% for composite dishes, which contained a large extent of meat-based prepared meals. The percentage reduction in the quantities consumed after imposing the tax has been calculated taking into account the demand elasticities:

$$\boldsymbol{\frac{\Delta Q^j}{Q_j} = \sum_k^n \varepsilon_{jk} * \frac{\Delta P^j}{P_j}} \quad \text{.......................................Equation 7}$$

where $\frac{\Delta P}{P} = \frac{p_1 - p_0}{p_0}$ and $\frac{\Delta Q}{Q}$ represent the percentage change in prices and quantities of each food group, respectively (Säll & Gren, 2015)

Finally, the post-tax change in CO_2 equivalent emission is defined as

$$\boldsymbol{\Delta E_i = \sum_j^n e_i * \Delta Q_j} \quad \text{.......................................Equation 8}$$

where e_i, is the average CO_2 equivalent emission in each of the food group.

Estimating the impact of CO_2-eq tax on household´s welfare

In order to calculate the impact of the above mentioned tax on household's welfare, being consistent with previous literature, we have assumed, that the food supply is perfectly inelastic and is not influenced by the CO_2-eq tax. In other words, the burden of the tax is borne solely by the consumer. Deaton (1989, 1997) defined the welfare effects of a price change as the compensating variation expressed as a share of the total household expenditure. Tax effects have been estimated taking into account both first-order and second-order effects. The first order-effect assesses the distributional impact of the tax imposition on each food category as the product of its corresponding budget share by the price change in that food category, while the second order-effect considers how consumers react to price changes.

Welfares effects are based (Lewbel & Pendakur, 2009) log of living cost index which takes into account both first and second order effects:

$$\boldsymbol{C(p_1, u, z, \varepsilon) - C(p_0, u, z, \varepsilon) = (p_1 - p_0)'w_0 + 0.5(p_1 - p_0)'\left(\sum_{k=1}^{K} A + By\right)(p_1 - p_0)} \quad \text{..........Equation 9}$$

The term $(p_1 - p_0)'w_0$ in (9) is the Stone index for the price change while $0.5(p_1 - p_0)'(\sum_{k=1}^{K} A + By)(p_1 - p_0)$ models substitution effects resulting from price changes.

Results and Discussion

Price and expenditure elasticities

Table 3 shows the calculated expenditure as well as Marshallian own price elasticities (Cross price elasticities are available on request). Expenditure elasticity estimates are statistically significant at the one percent level and positive. Nine food groups out of the 16 are expenditure elastic, including grains and grain products, vegetables and vegetable products or plant based fats as well as all categories related to animal sources of protein. Again, in this case, results do not significantly differ from previous studies, taking into account again that sample periods are different as well as food categories. Dhehibi, Gil, & Angulo, (2007) and (Garcia-Muros et al., 2016) also found that the food expenditure elasticity for beef to be greater than one (in the latter case also the demand for fish was also elastic with respect to income).

Impact of a CO_2 tax on household CO_2-eq emissions

Table 4 (first row) shows the total marginal change (decrease) in households´ CO_2 equivalent emissions after the tax imposition under the four tax scenarios mentioned in the previous section while Figure 2 shows the impact on each food category. Taking into account substitution effects, the tax has a more significant effect on those food categories which generate higher CO_2 equivalent emissions: composite dishes; beef, veal and lamb; and milk and dairy products despite their relatively lower share in food expenditure. This is consistent with previous literature (Henchion, McCarthy, Resconi, & Troy, 2014; Säll & Gren, 2015) and also highlight the importance of composite dishes which has been neglected in previous studies. Vegetables and vegetable products category has a relatively low impact on the decline in CO_2-eq emissions despite their relatively large expenditure share as it generates a quite low CO_2 equivalent emission per kg. Thirdly, although there is a high correlation between consumption decreases, the tax level and the policy pursued despite taking into account substitution effects. Results from this study suggest that a reduction in the consumption of red meat and dairy products would contribute to a significant decline in CO_2-eq emissions (Hedenus et al., 2014; Säll & Gren, 2015).

Table 4 suggest that the level of the tax has to be large enough to generate a significant reduction of CO_2 equivalent emissions or, alternatively, to assume significantly lower consumption discount rate.

Lower consumption discount rate indicates that policy makers give relevance to inter- and intra-generational equity and vice versa. Choosing a high consumption discount rate reduces CO_2-eq emission but marginally as shown in **Table 4**. In this context, Stern (2007) proposed a consumption discount rate as low as 1.4 percent on consumption to achieve a higher reduction in Greenhouse gas emissions, although this study has been criticized by researchers like Mendelsohn, (2008) and Nordhaus, (2007). From an EU perspective, the objective for 2050 is more restrictive than that for 2020 and it implies imposing a higher tax level (the kilogram of CO_2-eq emission should be priced at 0.2 Euros). This result is consistent with Bonnet, Bouamra-Mechemache, Corre et al. (2016).

The last three rows in **Table 4** simulate the impact of three alternative policy scenarios. We have concentrated the analysis on those food categories that generate larger CO_2-eq emissions. The following three alternative policy scenarios have been considered: 1) the tax is restricted only to beef, veal and lamb as well as to composite dishes; 2) the tax is restricted to all meat products; and 3) the tax is restricted to beef, veal and lamb, as well as to dairy products as this has been the direction of most studies dealing with environmental taxes.

Table 4 indicate that restricting the imposition of the tax only to beef and dairy products (fourth row) would have the lowest impact in CO_2-eq emission reduction among the four policy scenarios ranging from 0.41% to 8.51%, depending on the four social cost scenarios that we have considered in this study. If all meats are considered, the impact is larger (50% increase in CO2-eq emissions) but still is far from the impact that would be generated by taxing all food categories. This result would imply that studies like Bailey, Froggatt, & Wellesley (2010) and Säll & Gren (2015) would underestimate the potential impact of the tax as in both studies the tax is restricted to meat and dairy products. Finally, results from this study also indicate that a tax policy targeted to beef and composite dishes would have a similar impact like that restricted to all meat and dairy products.

Welfare impacts of CO_2 equivalent taxes

Welfare effects have been calculated using compensated variation: approximate values ignoring substation effects are presented for the individual food groups and Log of the Living Cost Index of (Lewbel & Pendakur, 2009) for all food groups and social classes. **Table 6** shows that the level of compensation is quite heterogeneous across food categories being highly correlated with both their associated CO_2 equivalent emissions and food expenditure elasticities. The needed compensation is higher in the case of composite dishes, followed by beef, veal and lamb and poultry, eggs and other fresh meat categories. The last row on **Table 5** shows that on average, the compensation is higher as the associated social cost (discount rate) of emissions also increases (decreases). For instance, reducing the discount rate from 5% to 2.5% would require an increase in food expenditure from 0.85% to 4.24%.

We further sub-grouped the sample into social classes to analyze the distributional impact of the tax on consumer welfare. **Table 5** shows that the impact of the tax is heterogeneous across social classes. The welfare losses are higher for the lowest social class due to the higher consumption of meat and milk and dairy products and the lower consumption of fish, fruits and vegetables. Nugraha & Lewis (2013) showed that the impact of a tax on consumption or production is more regressive for lower and middle income social groups. However, this study show some differences: the impact decreases as the income level increases. However, for the higher income group the situation worsen in relation to the middle income group. This is due to the fact of the high consumption of ready-to-eat meals (composite dishes) and beef, veal and lamb. Again, as the tax is linear and proportional to the CO_2-equivalent emission, the effect is higher at low consumption discount rates and higher social cost of emission scenarios.

Impact of CO2-eq tax on diet quality

To end with the impact assessment, we have included this section to report the impact of the alternative tax policy scenarios on the diet quality. Although there is a vast literature about alternative measures for diet quality, we have used here a very simplistic approach (the definition of diet quality is out of the scope of this paper). We have considered the World health Organization (WHO) recommendations which suggest that the daily proportion of proteins, lipids and carbohydrates on total energy intake should be 10%-15%, 30%-35% and 50-55%, respectively. In this study we have calculated average per capita adult equivalent values. Figure 3 shows the main results. The first line of the graph corresponds to the current situation while the rest correspond to each of the tax policy scenarios. It is evident that price increases would contribute to lower consumption but, a priori, it was not clear if this reduction would affect or not the diet quality.

Our result indicates that that the current macronutrient intake significantly exceeds the recommended values in the case of lipids (42.71%) and very slightly, in the case of proteins (15.61%). As a consequence, the intake of carbohydrates is lower than the recommended values (41.64%). These results are consistent with previous studies in Spain suggesting an overconsumption of lipids and fats (Moreno, Sarr\ia, & Popkin, 2002), which is one of the main reasons for the rapid increase of the prevalence of obesity and health related diseases compared to other EU countries (Garc'\ia-Goñi & Hernández-Quevedo, 2012).

Any tax policy to reduce CO_2 equivalent emissions would reduce the consumption of the most contaminating products, which will generate a more equilibrated diet. We have not found significant differences here in relation to the magnitude of the tax corresponding to each scenario. In general terms, the consequences on diet are two: a reduction in the intake of lipids and proteins towards an increase in that of carbohydrates. Let us take an example the tax scenario addressed to reduce emissions by 60% in 20250. As can be observed, proteins and lipids intake would decrease by 1.96% and 3.71%, respectively, while that of carbohydrates would increase by 4.55%. Summing up, our results suggest that imposing a carbon tax on all food categories would lead to a decline not only in CO_2-eq emissions but also to a more equilibrated diet. We have also carried out some analyses by social income groups and we have not found any significant difference in relation to the average behavior.

Concluding remarks

The study aimed at assessing the impact of introducing a Pigovian or CO_2 equivalent tax on food demand, welfare and diet quality in Spain. Alternative tax policy scenarios have been considered, our results suggest that that the price increase as a consequence of the tax, reduces the consumption of the food products associated with higher CO_2 equivalent emissions. However, the impact on human health is positive as the quality of diet approximates to the WHO recommendations. However, the tax will affect more to lower income groups. It is also evident that the impact increases as the level of the tax also increases, suggesting that the tax level should be large enough to generate significant reduction in CO_2 equivalent emissions. Finally, our results suggest that the impact on the reduction of greenhouse gas emissions will be significant only if the tax is imposed on all food products according to their contamination rate. Governments should be aware about the trade-off between CO_2 equivalent emissions reduction goals and the negative consequences on the citizen's welfare and set up their goals finding out a compromise between these two contradictory goals.

In any case, results from this study only apply to Spain and similar analyses should be conducted in other countries considering all food categories. In spite of the contribution of this study to the policy discussion, we have to recognize that our results should be interpreted with caution for several reasons. The most important is the lack of data. Although there is a lot of studies on life-cycle analysis, most of them are product specific and does not exist any study covering a wide range of products in Spain using a common methodological approach. Second, we have assumed that food supply is perfectly inelastic ignoring potential strategic decisions of firms. Further research could be focused on relaxing this assumption. Finally, authors have assumed, due to data unavailability, strong separability between food and other durable and non-durable goods. On the other hand, this limitation is difficult to overcome as we would need, at least, a composite indicator of greenhouse emission of other non-durable and durable goods. In spite of these limitations, this study provides some evidences about the potential impacts of imposing a CO_2 equivalent tax on food products.

References

Audsley, E., Brander, M., Chatterton, J. C., Murphy-Bokern, D., Webster, C., & Williams, A. G. (2010). How low can we go? An assessment of greenhouse gas emissions from the UK food system and the scope reduction by 2050. Report for the WWF and Food Climate Research Network.

Bailey, R., Froggatt, A., & Wellesley, L. (2010). Livestock—Climate Change's forgotten sector. Global Public Opinion on Meat and Dairy Consumption (London, UK: Chatham House, The Royal Institute of International Affairs, 2014). 1. JE Williams and RJ Price,"Impacts of Red Meat Production on Biodiversity i. *Animal Production Science*, *50*, 723–747.

Baumol, W. J., & Oates, W. E. (1975). The Theory of Environmental Policy: Externalities. *Public Outlays, and the Quality of Life*, 10–13.

Bonnet, C., Bouamra-Mechemache, Z., Corre, T., & others. (2016). *An environmental tax towards more sustainable food consumption: empirical evidence of the French meat and marine food consumption*.

Bourne, M., Childs, J., Philippidis, G., & Feijoo, M. (2012). Controlling greenhouse gas emissions in Spain: what are the costs for agricultural sectors? *Spanish Journal of Agricultural Research*, *10*(3), 567–582.

Burchell, J., & Lightfoot, S. (2001). *Greening of the European Union: Examining the Eu's Environmental Credentials* (Vol. 14). A&C Black.

Castellón, C. E., Boonsaeng, T., & Carpio, C. E. (2015). Demand system estimation in the absence of price data: an application of Stone-Lewbel price indices. *Applied Economics*, *47*(6), 553–568.

Chen, Z.-M., Liu, Y., Qin, P., Zhang, B., Lester, L., Chen, G., … Zheng, X. (2015). Environmental externality of coal use in China: Welfare effect and tax regulation. *Applied Energy*, *156*, 16–31.

Cox, T. L., & Wohlgenant, M. K. (1986). Prices and quality effects in cross-sectional demand analysis. *American Journal of Agricultural Economics*, *68*(4), 908–919.

Deaton, A. (1989). Rice prices and income distribution in Thailand: a non-parametric analysis. *The Economic Journal*, *99*(395), 1–37.

Deaton, A. (1997). *The analysis of household surveys: a microeconometric approach to development policy*. World Bank Publications.

Dhehibi, B., Gil, J. M., & Angulo, A. M. (2007). Nutrient effects in consumer demand systems: Evidence from panel data. *Acta Agriculturae Scand Section C*, *4*(2), 89–102.

Edjabou, L. D., & Smed, S. (2013). The effect of using consumption taxes on foods to promote climate friendly diets--The case of Denmark. *Food Policy*, *39*, 84–96.

Endres, A. (2010). *Environmental economics: theory and policy*. Cambridge University Press.

Garc'\ia-Goñi, M., & Hernández-Quevedo, C. (2012). The evolution of obesity in Spain. *Eurohealth*, *18*(1), 22–25.

Garc'\ia-Muros, X., Markandya, A., Romero-Jordán, D., & González-Eguino, M. (2016). The distributional effects of carbon-based food taxes. *Journal of Cleaner Production*.

Hartikainen, H., & Pulkkinen, H. (2016). *Summary of the chosen methodologies and practices to produce GHGE-estimates for an average European diet* (Vol. 58/2016). Retrieved from http://urn.fi/URN:ISBN:978-952-326-314-7

Hedenus, F., Wirsenius, S., & Johansson, D. J. A. (2014). The importance of reduced meat and dairy consumption for meeting stringent climate change targets. *Climatic Change*, *124*(1–2), 79–91.

Henchion, M., McCarthy, M., Resconi, V. C., & Troy, D. (2014). Meat consumption: Trends and quality matters. *Meat Science*, *98*(3), 561–568.

Irz, X., Leroy, P., Réquillart, V., Soler, L.-G., & others. (2015). Welfare and sustainability effects of dietary recommendations. In *143rd Joint EAAE/AAEA Seminar, March 25-27, 2015, Naples, Italy*.

Lewbel, A., & Pendakur, K. (2008). Estimation of collective household models with Engel curves. *Journal of Econometrics*, *147*(2), 350–358.

Lewbel, A., & Pendakur, K. (2009). Tricks with Hicks: The EASI demand system. *The American Economic Review*, *99*(3), 827–863.

Mendelsohn, R. (2008). Is the Stern Review an economic analysis? *Review of Environmental Economics and Policy*, *2*(1), 45–60.

Moreno, L. A., Sarr\ia, A., & Popkin, B. M. (2002). ORIGINAL COMMUNICATION The nutrition transition in Spain: a European Mediterranean country. *European Journal of Clinical Nutrition*, *56*, 992–1003.

Mytton, O. T., Clarke, D., & Rayner, M. (2012). Taxing unhealthy food and drinks to improve health. *BMJ*, *344*(7857), e2931.

Nordhaus, W. D. (2007). A review of the Stern review on the economics of climate change. *Journal of Economic Literature*, *45*(3), 686–702.

Nugraha, K., & Lewis, P. (2013). The impact of taxation on income distribution: Evidence from Indonesia. *The Singapore Economic Review*, *58*(4), 1350024.

Progress towards achieving the Kyoto and EU 2020 objectives. (2013). Brussels. Retrieved from https://ec.europa.eu/clima/sites/clima/files/strategies/progress/docs/com_2013_698_en.pdf

Säll, S., & Gren, M. (2015). Effects of an environmental tax on meat and dairy consumption in Sweden. *Food Policy*, *55*, 41–53.

Schmutzler, A., & Goulder, L. H. (1997). The choice between emission taxes and output taxes under imperfect monitoring. *Journal of Environmental Economics and Management*, *32*(1), 51–64.

Stern, N. H. (2007). *The economics of climate change: the Stern review*. cambridge University press.

Sustainable diets for the future: can we contribute to reducing greenhouse gas emissions by eating a healthy diet? (2012). *American Journal of Clinical Nutrition*, *96*.

Tol, R. S. J. (2012). A cost--benefit analysis of the EU 20/20/2020 package. *Energy Policy*, *49*, 288–295.

Wirsenius, S., Hedenus, F., & Mohlin, K. (2011). Greenhouse gas taxes on animal food products: rationale, tax scheme and climate mitigation effects. *Climatic Change*, *108*(1–2), 159–184.

Table 1. Description data for analysis

Food Category	Budget Share (%)	Quantities (kg/capita)		Unit Values (€/kg)		Expenditures (€/capita)		**kg CO2-eq/kg food	
		Mean	SD*	Mean	SD*	Mean	SD*	Mean	SD*
Grains and grain-based products	5.47	54.13	44.8	2.69	0.95	136.95	113.55	1.1	0.3
Vegetables and vegetable products	8.03	124.04	109.51	1.77	0.6	201.06	168.22	1.2	0.7
Starchy roots, tubers, legumes, nuts and oilseeds	1.86	13.51	13.03	4.45	3.31	46.89	45.30	0.4	0.5
Fruit and fruit products	10.3	191.75	139.38	1.38	0.33	258.06	191.86	0.9	0.7
Beef, veal and lamb	5.52	16.14	15.77	8.65	3.27	142.73	165.25	18.9	11.7
Pork	4.71	17.41	15.28	7.48	3.14	119.17	99.79	5.8	0.2
Poultry, eggs, other fresh meat	5.62	39.99	28.63	3.61	1.00	141.23	103.37	5.9	1.7
Processed meat products	10.2	32.76	24.62	7.95	2.63	255.98	214.33	5.4	0.4
Fish and seafood	13.55	42.63	32.29	8.09	2.26	339.69	271.10	5.3	2.3
Milk and dairy products	8.14	183.41	128.01	1.27	0.72	203.93	140.44	1.5	0.1
Cheese	5.84	20.79	15.61	7.33	1.96	146.73	107.14	8.2	0.05
Sugar and confectionary and prepared desserts	7.71	47.45	36.73	4.44	1.66	193.18	144.43	1.2	0.5
Plant based fats	2.07	23.29	19.53	2.42	1.24	53.52	44.44	2.6	1.0
Composite dishes	5.10	31.14	29.99	4.58	1.88	129.27	121.87	12.5	8.6
Snacks and other foods	1.49	6.22	5.85	6.21	2.19	38.28	37.18	1.9	0.2
Residual category	4.39	39.78	38	3.87	4.09	110.21	109.64	1.3	0.3

*SD= standard deviation, ** Own elaboration from (Hartikainen & Pulkkinen, 2016)

Table 2. Tax simulation scenarios (%)

	EPA Discount rate 5%	EPA Discount Rate 2.5%	EU objective: reducing emissions by 20% in 2020	EU objective: reducing emissions by 60% in 2050
Social Cost of Emission	9.62 Euros	48.98 Euros	56 Euros	200 Euros
Grains and grain-based products	0.37	1.88	2.46	7.69
Vegetables and vegetable products	0.61	3.13	3.58	12.78
Starchy roots, tubers, legumes, nuts and oilseeds	0.07	0.37	0.43	1.53
Fruit and fruit products	0.61	3.11	3.56	12.71
Beef, veal and lamb	2.14	10.88	12.43	44.41
Pork	0.67	3.40	3.89	13.89
Poultry, eggs, other fresh meat	1.58	8.06	9.22	32.92
Processed meat products	0.64	3.27	3.74	13.36
Fish and seafood	0.59	3.02	3.45	12.34
Milk and dairy products	1.05	5.35	6.11	21.83
Cheese	1.05	5.37	6.14	21.92
Sugar and confectionary and prepared desserts	0.24	1.21	1.38	4.92
Plant based fats	0.97	4.93	5.64	20.13
Composite dishes	2.65	13.49	15.42	55.07
Snacks and other foods	0.31	1.58	1.81	6.45
Residual category	0.27	1.35	1.55	5.52

EPA: Environmental Protection Agency (USA)

Table 3. Expenditure and Marshallian own-price elasticities (t-values in brackets)

Food Category	Own-price Elasticity	Expenditure Elasticity
Grains and grain-based products	-0.50 (-4.25)	1.35 (28.06)
Vegetables and vegetable products	-1.03 (-16.31)	1.17 (24.99)
Starchy roots, tubers, legumes, nuts and oilseeds	-0.65 (-12.26)	0.49 (6.85)
Fruit and fruit products	-0.87 (-10.89)	0.99 (23.11)
Beef, veal and lamb	-0.65 (-14.60)	1.20 (18.75)
Pork	-0.65 (-9.12)	1.55 (26.99)
Poultry, eggs, other fresh meat	-0.53 (-9.26)	1.16 (26.82)
Processed meat products	-0.39 (- 5.71)	1.02 (23.61)
Fish and seafood	-0.72 (-8.39)	1.04 (24.89)
Milk and dairy products	-0.69 (-15.02)	1.32 (28.84)
Cheese	-0.74 (-11.1)	0.89 (18.82)
Sugar and confectionary and prepared desserts	-0.85 (-14.91)	0.80 (17.55)
Plant based fats	-0.72 (-0.87)	1.06 (16.33)
Composite dishes	-0.89 (-21.53)	0.84 (13.19)
Snacks and other foods	-0.72 (-15.49)	0.66 (8.65)
Residual category	-4.09 (-0.62)	0.45 (1.53)

Table 4. Average percentage reduction in CO2-eq emission under alternative tax scenarios (%)

Tax simulation scenario	EPA Discount rate 5%	EPA Discount Rate 2.5%	EU objective: reducing emissions by 20% in 2020	EU objective reducing emissions by 60% in 2050
All food categories	1.24	6.31	7.23	25.77
Beef and composite dish	0.62	3.18	3.63	12.96
Tax on all meat and dairy	0.60	3.05	3.49	12.47
Tax on beef and dairy	0.41	2.08	2.38	8.51

Table 5. Log of cost living index for different income groups and different policy options

Income Class	Tax Scenario			
	EPA Discount rate 5%	EPA Discount Rate 2.5%	EU objective: reducing emissions by 20% in 2020	EU objective: reducing emissions by 60% in 2050
Low	0,93	4,56	5,20	17,11
Lower middle	0,89	4,43	5,06	16,70
Middle	0,85	4,22	4,82	15,95
High	0,88	4,38	5,01	16,58
All social classes	0,85	4,24	4,84	16,00

Table 6. Approximate compensating variation (CV) as a percentage of initial expenditure for 5 different policy scenarios and carbon tax based on CO2 equivalent emission

Food Category	**Tax Scenario**			
	EPA Discount rate 5%	EPA Discount Rate 2.5%	EU objective: reducing emissions by 20% in 2020	EU objective: reducing emissions by 60% in 2050
Grains and grain-based products	0.02	0.10	0.13	0.40
Vegetables and vegetable products	0.05	0.24	0.27	0.90
Starchy roots, tubers, legumes, nuts and oilseeds	0.00	0.01	0.01	0.03
Fruit and fruit products	0.06	0.31	0.36	1.17
Beef, veal and lamb	0.11	0.52	0.58	1.68
Pork	0.03	0.16	0.18	0.59
Poultry, eggs, other fresh meat	0.09	0.42	0.48	1.47
Processed meat products	0.06	0.32	0.36	1.21
Fish and seafood	0.08	0.38	0.44	1.45
Milk and dairy products	0.09	0.44	0.50	1.57
Cheese	0.06	0.31	0.35	1.10
Sugar and confectionary and prepared desserts	0.02	0.10	0.11	0.38
Plant based fats	0.02	0.10	0.11	0.36
Composite dishes	0.14	0.65	0.73	1.92
Snacks and other foods	0.00	0.02	0.03	0.10
Residual category	0.01	0.06	0.07	0.23

EPA: Environmental Protection Agency (USA)

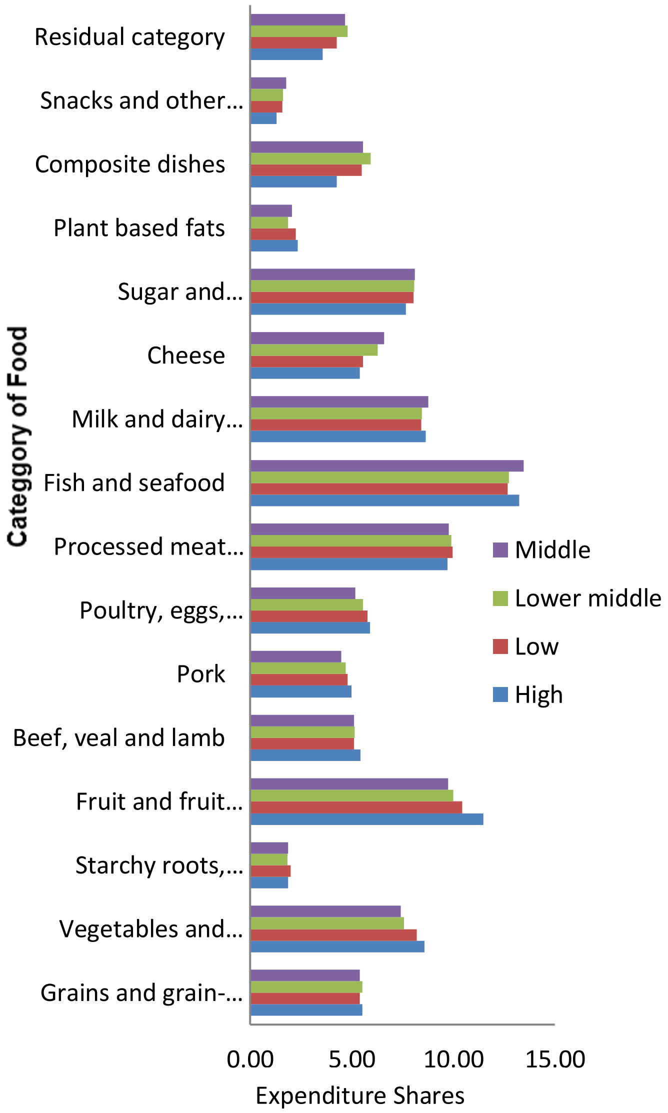

Figure 2. Food Expenditure shares by different social classes in Spain

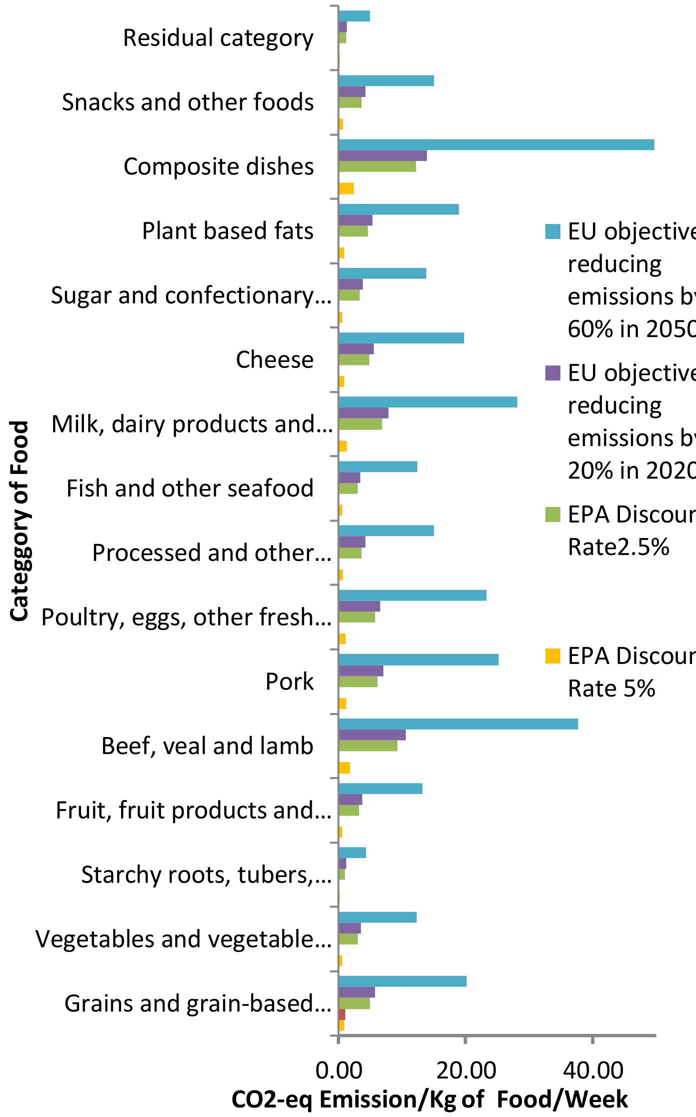

Figure 1. Percentage decrease in CO2-eq emission /year /household for all food categories after the introduction of CO2-eq emission tax

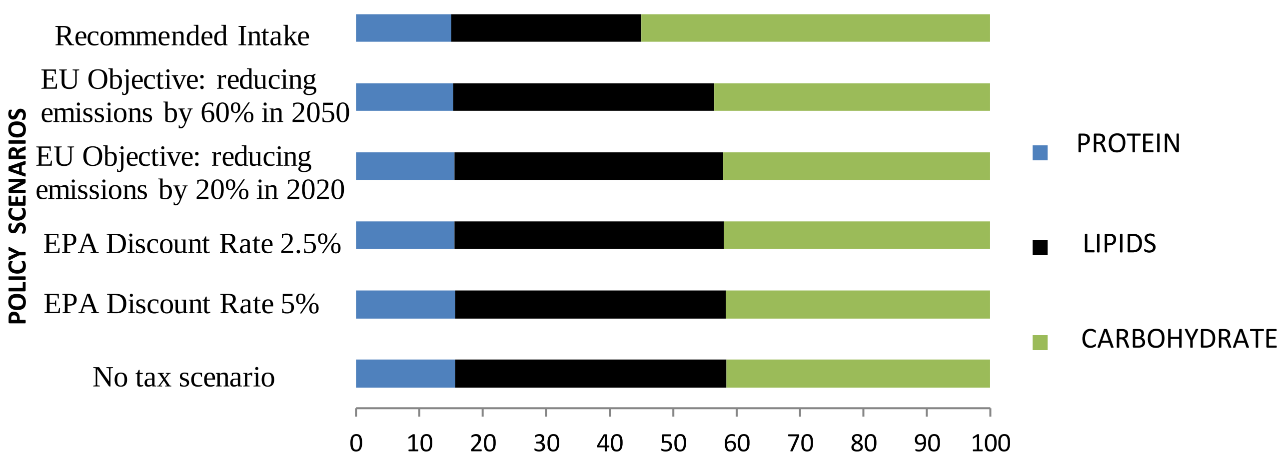

Figure 3. Impact of CO2-eq tax on diet quality