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Article

Simulating the impact of a carbon tax on food in four European countries

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Abstract

Since agriculture is responsible for a considerable share of anthropogenic greenhouse gas emissions (GHGE), this paper examines the impact of various carbon taxes designed to incentivize environmentally friendly food consumption patterns in four European countries: Finland, Italy, Sweden, and the UK. As the proposed fiscal policies are likely to affect food consumption patterns, the study also assesses the consequent changes in diet quality and welfare. The results from this analysis reveal considerable variations in the reduction of GHGE across countries and tax schemes. While most taxation schemes have only a modest impact on dietary quality, these effects differ among nations. Additionally, the welfare cost of the compensated scheme is relatively small but not insignificant. These findings raise questions about the efficacy of a common European fiscal policy for climate mitigation compared to a more flexible approach where each member state calibrates the tax according to its unique circumstances.

Keywords: carbon tax, demand analysis, greenhouse gas emissions, cross-country analysis, environmental policy.

JEL codes:C33, H23, H31, Q18

1. Introduction

Agriculture is estimated to be responsible for a considerable share of global anthropogenic greenhouse gas emissions (GHGE). In particular, the agricultural and food sector contributes significantly to the release into the environment of CO_2 (carbon dioxide), methane, and NO_2 (nitrogen dioxide). These originate from most agricultural activities, including land use, livestock breeding, energy consumption at the farm level, and fertilizer production. Depending on the approach adopted to obtain these estimates (Houghton 2003), researchers have calculated that the agricultural sector's contribution to the gross human-made GHGE could lie between 10 per cent and 25 per cent (Steinfeld et al. 2006; McMichael et al. 2007; Pachauri et al. 2014; Wellesley, Froggatt, and Happer 2015; FAO 2017). Within the European Union (EU), the European Environmental Agency (EEA) places this share at roughly 10 per cent, with meat production alone being responsible for almost 70–80 per cent of total emissions from agriculture, excluding land use and land use change (McMichael et al. 2007; EEA 2019). Given that climate change and environmental issues are now exceptionally high on the political agenda, the EU has set a 55 per cent GHGE reduction goal by 2030 with respect to the levels registered in 1990 (European Commission 2021), as a result of the strategies defined under the EU Green Deal, including the so-called Farm to Fork Strategy, which is specific for the agri-food sector (European Commission 2020).

Along these lines, one of the key strategic objectives of the new EU Common Agricultural Policy (CAP) (2023–27) is to ensure that the agricultural sector will contribute substantially to climate change mitigation by reducing direct GHGE and improving carbon sequestration through appropriate soil management techniques, as well as by lowering fossil fuel intensity and ensuring sustainable energy production (i.e. biomass production) (European Commission 2019). For instance, by tightening conditionality rules to foster carbon sequestration and prevent soil degradation and by increasing the number, targets, and budget of voluntary support measures for carbon farming (i.e. eco-schemes and agri-environmental and climate measures), the new CAP aims at making a significant contribution to upscaling carbon farming across the EU member states and reducing GHGE from the agricultural sector. However, as acknowledged in the Farm to Fork Strategy document, it is necessary to implement both demand and supply measures to ensure that the EU can progress towards a sustainable food system (European Commission 2020; Clora et al. 2021). Therefore, combining the aforementioned supply-side measures with demand-side policies that promote transitioning towards more sustainable diets (e.g. diets with reduced animal-based food content) has the potential to generate greater benefits in terms of GHGE reduction.

Policymakers aiming at curbing GHGE through measures regulating food consumption may choose between three mechanisms: command and control instruments, information provision, and price-based measures. However, whereas command and control instruments have high implementation costs and are scarcely adaptable to situations other than acute threats, information is considered to have limited impact in cases where human health is not straightforwardly involved (Reisch et al. 2013; Edjabou and Smed 2013). Therefore, price-based measures on the demand side are left as the most appropriate and, potentially, most effective way to tackle GHGE.

Even though the literature already presents some empirical studies measuring how carbon taxes on food may contribute to the achievement of the EU GHGE reduction target (Wirsenius et al. 2011; Mytton, Clarke and Rayner 2012; Edjabou and Smed 2013; Caillavet et al. 2016; Jansson and Säll 2018; Bonnet et al. 2018; Tiboldo et al. 2022), the introduction of a 'sin tax' on food, as it was introduced in the past for other goods (e.g. tobacco, alcohol, and sugar-sweetened beverages), may lead to many controversies and oppositions. In detail, some of the controversies are related to the distributional effects of the tax, since this measure might disproportionally affect the most vulnerable groups of the population, such as low-income households (Klenert, Funke, and Cai 2023). Furthermore, the design of such a policy is not a trivial matter, since policymakers should account not only for its effectiveness but also for the potential spillover effects, such as the one on diet quality in the population. Last, even though the aforementioned 'Farm to Fork Strategy' explicitly calls for a unitary fiscal intervention able to help consumers transitioning towards sustainable and healthy diets (European Commission 2020), the tax scheme's effects might be highly heterogeneous among European countries, making the application of a common and unique policy challenging. For these several reasons, Springmann et al. (2017) suggested to accompany a tax on food commodities, which if appropriately designed can have a positive effect both on GHGE reduction and on health, with some compensating policies, such as using the tax revenues for health promotion policies targeting the most vulnerable groups of the population.

Therefore, in our work, we assess the impact of a complex tax scheme on GHGE from food consumption in three EU member states (MS) (Finland, Italy, Sweden) and the UK. In particular, we follow a common approach to data collection and aggregation, demand estimation, and policy simulation, thereby providing comparable results across all four countries. Specifically, we use estimates from four EASI (Exact Affine Stone Index) demand systems (Lewbel and Pendakur 2009) (one for each country) to compute demand elasticities and evaluate the potential impact of fiscal measures on food. In this respect, our set-up consists of two scenarios: We first simulate a zero-revenue (price-compensated)[1] levy where three different groups of food categories are taxed using three different fixed rates, each based on as many estimates of the social cost of CO_2. We then repeat the same scheme but without applying any price compensation. Quantifying the effectiveness of these taxation schemes is achieved through a set of indicators assessing, on the one hand, the reduction in GHGE and, on the other hand, changes in diet quality. In this last respect, we measure both the mean adequacy ratio (MAR) and the mean excess ratio (MER) (Vieux et al. 2013).[2] Finally, we provide welfare implications through the cost-of-living measure proposed in Lewbel and Pendakur (2009). Given that the aim of this policy is to reduce GHGE without negatively affecting dietary quality, we consider the uncompensated scenario as a benchmark. In other words, we are not interested in the tax revenue, but, rather, we explore whether redistributing money by price-compensating less polluting food categories may be beneficial to reducing GHGE, while limiting negative distributional effects. Measuring the impact of 'green' fiscal measures on environmental, health, and welfare indicators goes in the direction of what Bonnet et al. (2020) call 'convergence of impacts': Because taxation mechanisms often have nontrivial and sometimes opposing effects on these dimensions, cross-evaluating results of different nature becomes a key step in policy analysis.

This study contributes to the literature on fiscal policies to mitigate GHGE and to the European debate on demand-side policies to tackle climate change in several respects. First, using a homogenous methodology, the current analysis empirically evaluates the effects of different tax schemes in several European countries (Italy, Finland, Sweden, and the UK), thus allowing a comparison across countries with different food habits. This enables us to assess the extent to which fiscal measures to reduce dietary GHGE need to be adjusted at the national level. Second, for each country and policy scheme, the analysis measures the effects on climate, diet quality, and consumers' welfare. Given the contribution of this study, the results are relevant for European policymakers and the public debate, since they can enlighten and give insights to some of the major questions related to the implementation of this type of fiscal measure. Results show that the effects of these policies are highly heterogeneous among countries and tax designs. These findings open some questions regarding how a common unique European fiscal policy aiming at climate mitigation could be the best option to reach the policy goals, or whether a more flexible design, where the tax is calibrated by each member state, can be more effective.

2. Methodology

2.1 The EASI demand model

We model food purchases using an EASI demand system (Lewbel and Pendakur 2009). We choose EASI over conventional Almost Ideal (AID) and other commonly adopted formulations for a number of reasons. First, the EASI Engel curves for any category of good are completely unrestricted so the model can accommodate high-order polynomial (or splines) in (implicit) expenditure and household characteristics. Second, the error term associated with EASI budget shares represents by construction unobserved heterogeneity in households' preferences or random utility parameters. Third, Lewbel and Pendakur (2009) show that the EASI nests some of most popular demand systems in the literature. Last, an approximate version of EASI is readily available by simply replacing the implicit utility term (see below) with the difference between nominal expenditures and a Stone price index; this approximate version can be straightforwardly estimated using linear least-squares methods such as Seemingly Unrelated Regression (SUR) techniques (Zellner 1962).

The EASI budget share equation is derived from a log expenditure (cost) function of the form (Pendakur and Sperlich 2010):

$$\mathrm{C}\left(\mathbf{p}, u, \mathbf{z}, \mathbf{e}\right) = u + \mathbf{p}^{\mathrm{T}}\mathbf{m}\left(u, \mathbf{z}\right) + \frac{1}{2}\sum_{l=0}^{L} z_l \mathbf{p}^{\mathrm{T}}\mathbf{A}_l\mathbf{p} + \frac{1}{2}\mathbf{p}^{\mathrm{T}}\mathbf{B}\mathbf{p}u + \mathbf{p}^{\mathrm{T}}\mathbf{e}, \tag{1}$$

where $\mathbf{p}$ is a J_c-vector of log prices, $\mathbf{z}$ is an L_c-vector of household characteristics, u indicates utility, $\mathbf{m}(u, \mathbf{z})$ is a J_c-vector valued function in utility and household characteristics, $\mathbf{A}$ and $\mathbf{B}$ are matrices of coefficients, and $\mathbf{e}$ is a zero-mean J_c-vector of error terms encapsulating heterogeneities in preferences. The subscript c indicates that the size of each vector changes according to the country. The term $\mathbf{m}(u, \mathbf{z})$ generates the model's Engel curves. By Shephard's lemma, differentiating equation (1) with respect to $\mathbf{p}^{\mathrm{T}}$ produces Hicksian budget shares of the form

$$\mathbf{w} = \mathbf{m}\left(u, \mathbf{z}\right) + \sum_{l=0}^{L} z_l \mathbf{A}_l\mathbf{p} + \mathbf{B}\mathbf{p}u + \mathbf{e}. \tag{2}$$

Expressing equation (1) as $\mathrm{C}(\mathbf{p}, u, \mathbf{z}, \mathbf{e}) = u + \mathbf{p}^{\mathrm{T}}[\mathbf{m}(u, \mathbf{z}) + 1/2\sum_{l=0}^{L} z_l\mathbf{A}_l\mathbf{p} + 1/2\mathbf{B}\mathbf{p}u + \mathbf{e}]$ implies that $\mathrm{C}(\mathbf{p}, u, \mathbf{z}, \mathbf{e}) = u + \mathbf{p}^{\mathrm{T}}\mathbf{w} - 1/2\sum_{l=0}^{L} z_l\mathbf{p}^{\mathrm{T}}\mathbf{A}_l\mathbf{p} - 1/2\mathbf{p}^{\mathrm{T}}\mathbf{B}\mathbf{p}u$; therefore, plugging equation (2) into equation (1) and solving for u yields the so-called *implicit utility*:

$$y = \frac{x - \mathbf{p}^{\mathrm{T}}\mathbf{w} + 1/2\sum_{l=0}^{L} z_l\mathbf{p}^{\mathrm{T}}\mathbf{A}_l\mathbf{p}}{1 - 1/2\mathbf{p}^{\mathrm{T}}\mathbf{B}\mathbf{p}}, \tag{3}$$

where $x = \mathrm{C}(\mathbf{p}, u, \mathbf{z}, \mathbf{e})$. Equation (3) indicates an affine transformation of (log) nominal food expenditure deflated by a (log) Stone price index ($\mathbf{p}^{\mathrm{T}}\mathbf{w}$). Consequently, we can view y as a close approximation to real food expenditures.[3] Unlike AID systems, this model employs an exact Stone index to deflate x. Finally, substituting y for u in equation (2) provides implicit *Marshallian budget shares:*

$$\mathbf{w} = \mathbf{m}\left(y, \mathbf{z}\right) + \sum_{l=0}^{L} z_l \mathbf{A}_l\mathbf{p} + \mathbf{B}\mathbf{p}y + \mathbf{e}. \tag{4}$$

For the sake of tractability, Lewbel and Pendakur (2009) specify $\mathbf{m}(y, \mathbf{z})$ as a degree-five polynomial in y and linear function of $\mathbf{z}$. The demand system defined in (4) is not restricted by Gorman rank conditions (Pendakur and Sperlich 2010). Moreover, except for the quadratic forms in the expression for y, equation (4) is linear in parameters. One can deal with such non-linarites using non-linear least-squares techniques (or M-estimators) or by replacing y with a computable approximation. In the latter case, demand parameters can be easily estimated by linear system estimators such as SUR or similar. One way to approximate real expenditures consists of deflating nominal expenditures using a Stone price index or, depending on the nature of the data, any other theoretically sound variation of it. For example, $\tilde{y} = x - \mathbf{p}^{\mathrm{T}}\bar{\mathbf{w}}$, where $\bar{\mathbf{w}}$ expresses the average budget share across consumers, can be used in place of equation (3). As a result, equation (4) becomes

$$\mathbf{w} = \mathbf{m}(\tilde{y}, \mathbf{z}) + \sum_{l=0}^{L} z_l \mathbf{A}_l \mathbf{p} + \mathbf{B}\mathbf{p}\tilde{y} + \mathbf{e}. \tag{5}$$

Lewbel and Pendakur (2009) show that there is a little empirical difference between coefficients calculated using the approximate EASI and parameters estimated from the exact non-linear EASI budget share equations. The expenditure function presented in equation (1) is very general as it includes cross-terms in prices and utility as well as a generic shape for Engle curves. Consequently, budget share equations defined in equation (5) nest simpler alternatives of EASI demand systems. In this paper, we further simplify equation (5) by replacing $\mathbf{m}(\tilde{y}, \mathbf{z})$ with a polynomial of degree R in $\tilde{y}$ and a linear interaction between $\tilde{y}$ and $\mathbf{z}$. Finally, we include $\mathbf{z}$ as a simple demand shifter and drop any interaction between prices and household characteristics. Therefore, under the assumption of weak separability, the empirical model boils down to

$$\mathbf{w} = \sum_{r=0}^{R} \mathbf{b}_r \tilde{y}^r + \mathbf{C}\mathbf{z} + \mathbf{D}\mathbf{z}\tilde{y} + \mathbf{A}\mathbf{p} + \mathbf{B}\mathbf{p}\tilde{y} + \mathbf{e}. \tag{6}$$

Since the choice of R and the tackling of specific data issues (i.e. censoring) depend largely on the nature and quality of the data, we discuss such details in Supplementary Appendix 1. We compute conditional Hicksian semi-elasticities ($\mathbf{H}_c$), Marshallian price ($\mathbf{E}_p$), and income elasticities ($\mathbf{E}_x$) using the following formulations, respectively (Lewbel and Pendakur, 2009):

$$\mathbf{H}_c = \mathbf{A} + \mathbf{B}\tilde{y},$$

$$\mathbf{E}_x = \left(\sum_{r=1}^{R} r \mathbf{b}_r \tilde{y}^{r-1} + \mathbf{B}\mathbf{p} \right) \circ \underline{\mathbf{w}} + 1,$$

$$\mathbf{E}_p = \left[\mathbf{A} - \mathbf{w} \left(\sum_{r=1}^{R} r \mathbf{b}_r \tilde{y}^{r-1} + \mathbf{B}\mathbf{p} \right) \right] \circ \underline{\mathbf{w}} - \boldsymbol{\delta},$$

where $\mathbf{H}_c$ is computed holding $\tilde{y}$ constant, $\underline{\mathbf{w}}$ indicates a J_c-vector with typical element $1/w_j$, 1 represents a J_c-vector of ones, and $\boldsymbol{\delta}$ is a $J_c \times J_c$ matrix of Kronecker deltas.

2.2. Tax simulation scenarios and indicators of impact

We evaluate the impact of four GHGE-based tax scenarios using nutrient (and environmental) elasticities (Huang 1996). Starting from unconditional Marshallian demand elasticities, Huang (1996) shows a simple method to supplement price–quantity relationships with information on food nutrients to obtain relative changes in nutrient consumption due to price variations. We extend this framework by introducing data on GHGE and calculating environmental responses to different price variations. Consistent with Caillavet et al. (2016), we refer to these new parameters as environmental elasticities (see Supplementary Appendix 2 for details). Since this paper aims at estimating the benefits of an environmentally sustainable food policy, we design a tax model prioritizing highly polluting food categories. Specifically, food groups receive a fixed price mark-up based on their CO_2 emission per unit of weight. We propose three different social costs for CO_2: (a) 0.05€ per kg CO_2-eq, representing the rate originally devised to achieve the EU GHGE reduction target in the reference period (the EU medium term projection) (Assoumou and Maïzi 2011; Quinet 2009); (b) 0.015€ per kg CO_2-eq, corresponding to the average Emission Trading System (ETS) price for the reference period (European Environment Agency 2016); and (c) 0.2€ per kg CO_2-eq, which reflects

Table 1. Taxation scheme scenarios.

Scheme	Scenario	Food categories	Social cost of CO_2
1	Compensated/ uncompensated	Beef and veal	0.05/0.015/0.2
2	Compensated/ uncompensated	Beef and veal, pork and processed meat, poultry, and eggs	0.05/0.015/0.2
3	Compensated/ uncompensated	All animal-based products	0.05/0.015/0.2

Note: The social costs of CO_2 are expressed in € per kg CO_2-eq.

the price aimed at curbing EU emissions by 60 per cent by 2060 (the EU long-term projection) (Quinet 2009; Assoumou and Maïzi 2011). We next define three distinct schemes, each based on the inclusion of a progressively larger number of food categories. We begin by applying the above-defined carbon taxes to (i) beef products only (i.e. the most polluting categories in terms of CO_2 emissions per kg), and then we extend the mark-up structure to other food groups: (ii) pork, processed meat, poultry, and eggs and (iii) all animal-based products (including dairy and fish). However, since food taxes are usually blamed for being regressive (Caillavet, Fadhuile, and Nichèle 2016; Kehlbacher et al. 2016; García-Muros et al. 2017; Tiboldo et al. 2022), we complement the proposed fiscal measure with a compensation mechanism to reduce GHGE without increasing social inequalities. Following Edjabou and Smed (2013), we reduce the original VAT by a fixed percentage for all food groups that are not subject to the levy (see Supplementary Appendix 2 for details). Therefore, for the most climate-friendly food groups, the price differential becomes negative, while prices of targeted goods will increase proportionally to (a), (b), or (c). We also simulate an uncompensated scenario as a benchmark for the zero-revenue one. Table 1 summarizes the tax model discussed so far. For other methodological details, refer to Supplementary Appendix 2. The last step in this exercise consists of comparing changes in diets to abatements in GHG emissions so the optimal compromise can be identified.

We estimate changes in diets through an MAR and an MER (Vieux et al. 2013). Both indices quantify the nutritional adequacy of the individual's diet with respect to the nutrient intake recommendations (NIR). However, whereas the MAR focuses on the 'positive' components of a recommended diet (fibre, proteins, vitamins, etc.), the MER concentrates on the less desirable nutrients (saturated fatty acids, free sugars, cholesterol). Therefore, the higher (lower) the MAR (MER) the better the average diet. The building blocks of these two indicators are the so-called nutrient adequacy ratios (NAR), which are computed as the ratio between the NIR and the actual intake of one nutrient. For a set of K nutrients, with Q 'positive' nutrients and $K < Q$ 'negative' nutrients, we define the MAR and MER as follows:

$$\mathrm{MAR} = \frac{\sum_{k=1}^{Q} \mathrm{NAR}_k^A}{Q} = \sum_{k=1}^{Q} \left(\frac{\mathrm{NIR}_k}{\mathrm{c}_k} \right) Q^{-1}, \tag{7}$$

$$\mathrm{MER} = \frac{\sum_{k=\mathrm{Q}}^{K} \mathrm{NAR}_k^E}{\mathrm{K} - \mathrm{Q}} = \sum_{k=Q}^{K} \frac{\mathrm{NIR}_k}{\mathrm{c}_k} (K - Q)^{-1}, \tag{8}$$

where c_k is the actual intake for nutrient k, NAR_k^A is truncated at 1, and NAR_k^E is always larger than or equal to 1. Finally, we quantify the welfare impact of each tax scheme using the log cost-of-living index defined in Lewbel and Pendakur (2009):

$$C(\mathbf{p}_1, u, \mathbf{z}, \mathbf{e}) - C(\mathbf{p}_2, u, \mathbf{z}, \mathbf{e}) = \Delta C = (\mathbf{p}_1 - \mathbf{p}_0)^{\mathrm{T}} \mathbf{w}_0 + 0.5\,(\mathbf{p}_1 - \mathbf{p}_0)^{\mathrm{T}} \left(\sum_{l=0}^{L} z_l \mathbf{A}_l + \mathbf{B} y \right) (\mathbf{p}_1 - \mathbf{p}_0), \quad (9)$$

where $\mathbf{p}_0$ and $\mathbf{p}_1$ represent log-price levels before and after imposing the carbon tax, respectively, and $\mathbf{w}_0$ indicates the vector of pre-tax budget shares, so $(\mathbf{p}_1 - \mathbf{p}_0)^{\mathrm{T}}\mathbf{w}_0$ denotes the Stone index for the price change. Following Lewbel and Pendakur (2009), this class of indices allows for unobserved preference heterogeneity across households through $\mathbf{w}_0$ and small price changes. The second term on the right-hand side of equation (9) represents the matrix of compensated Hicksian semielasticities, $\nabla_{\mathbf{p}^{\mathrm{T}}} \mathbf{w}(\mathbf{p}, y, \mathbf{z}, \mathbf{e})$. Including such information secures the explicit incorporation of substitution effects, and so model large price changes, while also accounting for both observed and unobserved heterogeneity.

3. Data

We estimate price elasticities using household-level consumption data from the four European countries involved in the study. Supplementary Appendix 1 provides details about such information sources and presents a brief overview of the differences between the datasets employed for estimating model equation (6). All the samples discussed therein approximately cover the same time period but, due to the presence of both cross-sectional and (pseudo-)panel structures, some may extend slightly beyond (i.e. from 2003 to 2012). So, although we strived to employ highly comparable household information, the scattered availability of homogeneous data across Europe limits the extent of compatibility. Nonetheless, the datasets we discuss in Supplementary Appendix 1 guarantee the representativeness of the underlying populations, and, importantly, they also include enough household characteristics to estimate the same class of models in all countries.

The choice of including Finland, Italy, Sweden, and the UK reflects the original ERANET SUSDIET[4] project proposal, from which this paper largely results. Out of the nine MS constituting the SUSDIET's consortium, the aforementioned four were directly involved in tasks related to this simulation exercise. In fact, the SUSDIET project effectively kickstarted the whole data collection, aggregation, and homogenization, and prompted the development of a common modelling and simulation framework. Moreover, the diversity of the country group guaranteed the inclusion of heterogeneous dietary habits (i.e. Mediterranean and Nordic diets), thus enabling a rigorous assessment of a hypothetical EU-level carbon tax across Europe. We can appreciate some of these differences in Fig. 1 and Table 2 as discussed later on in this section.

Data were first aggregated into twenty common categories, each defined with the goal of combining detailed household purchase data with external information on GHGE and diets. In particular, we started from the FOODEX2 classification, a system developed by the European Food Safety Authority (EFSA) for a unique and universal identification of food items. This categorization is composed by 20 'level 1' (L1) food groups and 160 'level 2 (L2)' food groups (EFSA 2015). As 25 national food consumptions surveys have already been based on this nomenclature, we deemed this model fit to our data aggregation problem. However, since the FOODEX classification was not specifically designed to address environmental and nutritional issues, some adjustments were needed. For example, at level L1, the food group 'meat and meat products' had to be disaggregated into five sub-categories ('livestock meat', 'poultry', 'processed meat', 'meat imitates', and 'other meat'), 'milk and dairy products' into three ('dairy products', 'cheese', and 'milk and milk products imitates'), 'animal and vegetable fats and oils' into two ('animal fats and oils' and 'vegetable fats and

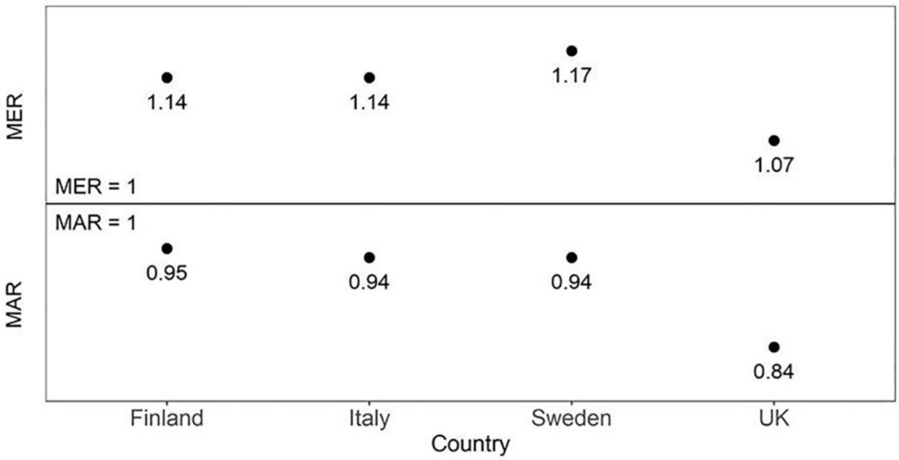

Figure 1. Baseline MAR and MER by country.

oils'), 'non-alcoholic beverages' into two ('tea, coffee, and cocoa' and 'soft drinks'), and 'composite dishes' into another two ('vegetable composite dishes' and 'animal composite dishes'). The L2 level required some specific re-arrangements as well. In fact, foods that comprise each level L2 category needed to be as homogenous as possible from both environmental and nutritional points of view. Therefore, we divided 'livestock meat' into four groups ('beef livestock meat', 'pork livestock meat', 'lamb livestock meat', and 'other livestock meat'), while 'fish meat' was divided into six further L2 categories ('tuna canned', 'tuna not canned', 'salmon', 'cod', 'other fatty fish', and 'other non-fatty fish'). At the end, we came up with a final nomenclature of 163 categories. Starting from this comprehensive inventory, we created five subsets of $M < 163$ food groups (one for each EU country) and used them to create the J_c categories we employ to estimate the demand model in equation (6). Since the goal was to guarantee homogeneity across countries, we performed this two-stage procedure by meticulously checking each national food classification.

We next compute nutrient elasticities using nutritional data from different national statistical sources. This data sources are also described in Supplementary Appendix 1, with a focus on the differences across the four countries. Last, we use the comprehensive review in Hartikainen and Pulkkinen (2016) to collect data on GHGE (expressed in kg of CO_2-eq per kg of ready to eat food) for a wide range of food products. Specifically, the authors estimate GHGE for 151 food categories using Life Cycle Assessment (LCA)-based statistics and existing literature.[5] Furthermore, indications about NIRs are available inside the guidelines for sugar intakes in adults and children (WHO 2015).

Since data sources concerning environmental and dietary indicators contained information at different disaggregation levels, we computed budget-share weighted averages to obtain information about consumption, daily nutrients' intake and GHGE for each of the twenty food groups considered in each country. Details on these groupings as well as category wide GHGE are displayed in Table 2.

A first glance, the average conditional budget shares presented in Table 2 provide an indication on the importance of each food group in the total cost of food. Although these data should be regarded with caution, they nevertheless show that the budget shares of plant-based products represent on average 42–49 per cent of total household expenditure devoted to food among the three EU member countries and the UK. Excluding the UK, the purchase of animal products is characterized by average conditional budget shares ranging from 41 per cent in Finland to 46 per cent in Italy. The remaining aggregate food group

Table 2. Budget shares and GHGE per food group in the five countries.

	Budget shares				GHGE (kg CO_2-eq/kg)			
Food groups	Finland	Italy[(1)]	Sweden[(2)]	UK[(3)]	Finland	Italy	Sweden	UK
Grains and grain-based	0.142	0.145	0.068	0.118	1.2	1.0	1.2	1.2
Vegetables and vegetables products	0.067	0.075	0.091	0.059	1.2	2.0	1.1	1.2
Starchy, legumes, oilseeds	0.023	0.027	0.018	0.099	0.9	0.9	0.8	0.8
Fruit and juices	0.084	0.1	0.066	0.1	0.8	0.7	0.7	0.8
Beef, veal, and lamb	0.039	0.106	0.063	0.044	42	40.1	41.9	38.8
Poultry and eggs	0.05	0.061	0.031	0.011	8.5	4.3	10.2	10.5
Pork	0.045	0.08		0.058	10.2	7.1	4.6	6
Processed and other cooked meats	0.05		0.098	0.046	5.6		7.3	5.6
Fish and seafood	0.042	0.087	0.065	0.058	4.6	5	4.9	4.6
Milk and dairy products	0.088	0.062	0.064	0.042	1.5	2.3	1.5	1.5
Cheese	0.069		0.061	0.032	8.3		8.3	8.3
Sugar, confectionary, and desserts	0.085	0.066	0.093	0.026	3.2	1.6	1.1	3
Soft drinks	0.022		0.038	0.005	0.3		0.6	0.3
Animal fats	0.03	0.067	0.029	0.004	9.5	8.3	5.7	9
Plant based fats	0.011	0.031	0.018	0.006	1.8	3.4	1.2	2
Water, tea, coffee, and other beverages	0.036	0.054	0.034	0.047	0.3	0.3	0.2	0.3
Alcoholic beverages		0.039	0.099	0.032		1.4	1.3	1.4
Composite dishes	0.04		0.012	0.153	3.8		5.3	5
Snacks and other foods	0.007		0.02	0.022	0.9		5.1	0.9
Residual category	0.071		0.032	0.037	1.8		1.3	1.3
Plant-based products	0.47	0.498	0.426	0.46				
Animal-based products	0.413	0.463	0.411	0.295				
Other	0.118	0.039	0.163	0.244				

[1]Italy: 'pork' and 'processed meat', 'milk and dairy' and 'cheese', 'sugar, confectionary, and desserts', and 'soft drinks' are aggregated.
[2]Sweden: 'pork' and 'beef, veal, and lamb' are aggregated.
[3]UK: the aggregate budget share for animal-based products is lower than for other countries partially because many composite dishes are of animal origin.
Source: own elaboration based on Hartikainen and Pulkkinen (2016).

(called 'other') has a more dispersed budget share, which fluctuates between 4 per cent in Italy to 24 per cent in the UK. Among the plant-based products, fruits, grains, sugar and desserts, and vegetables are the most important food items purchased by households in the three EU member countries and the UK. Meat products, when aggregated, have a budget share gravitating around 25 per cent for Italy, while for the other countries this figure is below the 20 per cent benchmark. For the other food items belonging to either the animal product group or the group called 'other', it is difficult to single out systematic patterns of purchases among all countries. In general, the distribution of the budget shares across countries is rather different, especially for specific food groups, as a result of different dietary habits and different relative prices of food items.

A similar pattern is observed for GHGE generated by food production and consumption. The prominent role of animal-based products is undisputed, as the most impacting diets are typically found in countries with the highest rate of CO_2-eq emissions per kg of animal-based food. For instance, the Italian GHGE per kg of average diet amounts to roughly 7 kg of CO_2-eq against emission as high as 13.4 kg of CO_2-eq per kg of animal products. We observe a similar pattern for Sweden, Finland, and the UK.

Information about pre-tax diet quality is displayed in Fig. 1 through the MAR and MER indices. As one can easily infer, in all four countries the average diet is distant from the ideal value of 1, both for the MER and for the MAR. In terms of MAR (content of positive nutritional items), the UK is the farthest from the optimal threshold, being 15 per cent below 1. Vice versa, all other countries seem to have reasonably healthy diets, with MAR values lying 5–6 percentage points below the ideal benchmark. On the other hand, Sweden exhibits the worst performance in terms of MER (content of negative nutritional items), with an index 22 per cent higher than the optimal level. Other countries fluctuate between +7 per cent and +16 per cent, indicating that the excess of 'problematic' nutrients proves more challenging than the lack of 'desirable' substances.

4. Results

4.1 Elasticities

Estimated unconditional[6] Marshallian demand elasticities for the different food categories (one table for each country) are reported in the Supplementary Appendix 3 (Tables A1–A4). In line with the results of existing empirical research, all the (diagonal) own-price elasticities reported in the tables are negative, revealing the absence of Giffen goods among the twenty food categories and four countries. Most own-price elasticity estimates are also smaller than unity in absolute value, implying that food demand, even for reasonably disaggregated categories, is inelastic: An increase (decrease) in price results in a less than proportional decrease (increase) in demand, with some exceptions both in terms of product categories and countries. Overall, our findings are consistent with a growing body of literature on fat and sugar taxes suggesting that price incentives need to exceed 10 per cent or even 20 per cent to exert any substantial effect on diets (PHE 2015).

The last column in each table provides expenditure elasticities. For all countries, those elasticities are unconditional and therefore measure responses relative to total household expenditure (as opposed to being conditional on a constant food budget). For all countries and food groups, the estimates are strictly positive and significant, indicating that, as expected, consumption increases with expenditure. We also note that the expenditure elasticities are generally smaller than unity, except for a few categories in Italy and several others in Sweden. This confirms Engel's law, which states that food's budget share (i.e. its relative importance in terms of expenditure) is inversely related to the household's budget.

The patterns of (off-diagonal) cross-price elasticities are more difficult to capture, since differences across countries are quite strong. In general, however, their value is quite low (below 0.2 in most cases) and substitution patterns tend to prevail in almost all countries.

4.2. Simulations

Figures 2–5 report the simulation results for all the variables of interest: GHGE, MAR, MER, and welfare changes. Each map allows the comparison of countries, scenarios (compensated vs uncompensated) and tax rates in a clear and easy-to-read manner. For completeness, however, Tables A5 and A6 in Supplementary Appendix 3 also present the results in tabular form, while Table A7 indicates the percentage change in prices after the application of the carbon tax.

We start the analysis with the compensated tax scenarios, focusing on the impact of primary interest, namely the reduction in dietary GHGE. Figure 2 indicates that those reductions are only modest (less than 5 per cent) for all countries, unless the price of carbon is set at a very high level (€0.2/kg CO_2-eq). Imposing a carbon tax using a carbon price close to the one prevailing on the ETS today (€0.015/kg CO_2) would result in minimal adjustments in GHGE of at most 1.4 per cent, and less than 1 per cent in most cases The result is explained by the relative inelasticity of demand for broad food categories, as documented

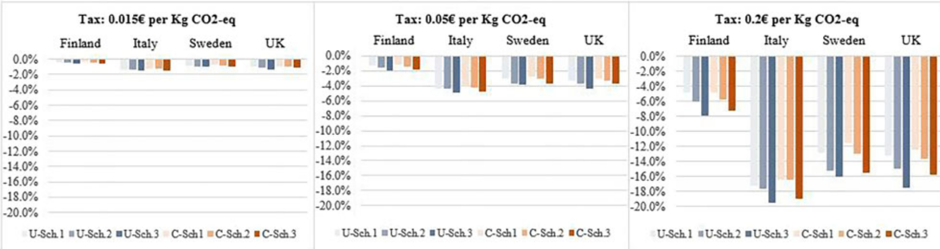

Figure 2. Impact of a fixed-rate carbon tax using three different social costs for CO_2: GHGE. Uncompensated (U-) and compensated (C-) scenarios.
Notes: In Scheme 1 (Sch.1), carbon taxes are applied to (i) beef products only, while in Schemes 2 and 3, the mark-up structure is extended progressively to (ii) pork, processed meat, poultry, and eggs and to (iii) all animal-based products (including dairy and fish), respectively.

in the Supplementary Appendix, and the substitutions among foods that tend to limit the direct effect of a tax. It is also consistent with much of the literature on 'sin taxes' in public health, which considers that non-trivial adjustments in diets require a relatively high tax rate on some food category, usually in excess of 20 per cent (Mytton et al. 2012).

Comparing the results presented in Fig. 2 allows an assessment of the effect of the compensation on GHGE reductions. As expected, compensating consumers for the carbon tax tends to limit the reduction in GHGE due to the income effect of the compensation, but the difference in climate impact between the two scenarios (i.e. compensated vs uncompensated) remains small and usually less than 1 per cent. Thus, for most countries, limiting the regressive impact of the carbon tax does not reduce much the effectiveness of the carbon tax as a GHGE mitigation measure.

In addition to carbon price and compensation, the fiscal schemes differ by the breadth of the tax domain in terms of the food categories included, i.e. from a single category (beef) to all animal products. As was the case for the compensation, Fig. 2 establishes that this characteristic has only limited influence on the climate effect of the tax scheme, with most of the reductions in GHGE being achieved when only beef is subject to the carbon tax. Altogether, our analysis concludes that achieving a substantial reduction in climate effect of the diet requires first and foremost a sufficiently high carbon price, while the breadth of the tax domain and presence of a compensation are relatively less important characteristics.

Beyond those general results, the empirical analysis also reveals a great deal of heterogeneity across countries in terms of response to a given tax scheme. For the compensated scenarios, the GHGE reduction in Italy spans from a minimum of –1.2 per cent (when the social cost of 1 kg of CO_2-eq GHG is set at 0.015€ and only meat products are taxed) to a maximum of –19 per cent (when the social cost is estimated at 0.2€ and we assume that all animal-based products are taxed). A similar range of variations is observed in Sweden and the UK (but with a maximum reduction of around 15–16 per cent), while the maximum and minimum reductions are much closer in Finland (from 0.4 per cent and –7.3 per cent in Finland). This result can be explained by the much lower consumption level, and so, budget share of beef (3.9 per cent) and pork meat (4.5 per cent) in Finland compared to the other countries (i.e. from 4.4 per cent to 10.6 per cent for beef and from 5.8 per cent to 8 per cent for pork meat) (see Table 2). Thus, our results suggest that the first three countries (Italy, Sweden, and the UK) would experience larger cutbacks in GHGE from the application of a fixed-rate carbon tax. While the figures for Finland tend to be in the same order of magnitude as the studies[7] reported in Bonnet et al. (2020), numbers for Italy, Sweden, and the UK are between moderately and considerably larger, especially when the highest social cost for CO_2 is considered (0.2€ per kg of CO_2-eq).

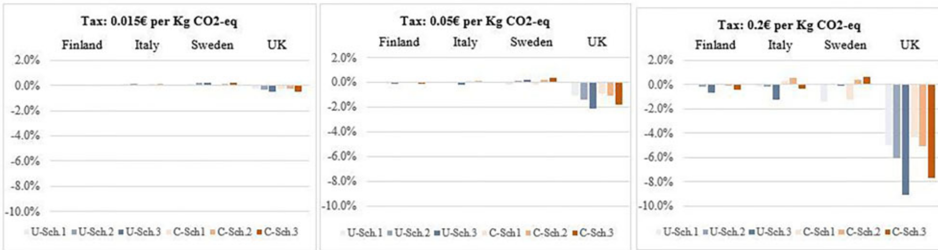

Figure 3. Impact of a fixed-rate carbon tax using three different social costs for CO_2: diet quality through MAR. Uncompensated (U-) and compensated (C-) scenarios.
Notes: In Scheme 1 (Sch.1), carbon taxes are applied to (i) beef products only, while in Schemes 2 and 3, the mark-up structure is extended progressively to (ii) pork, processed meat, poultry, and eggs and to (iii) all animal-based products (including dairy and fish), respectively.

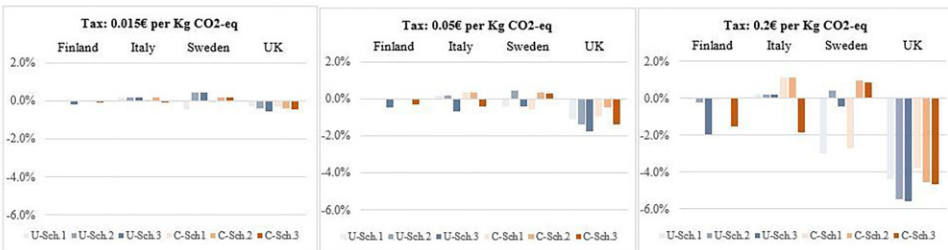

Figure 4. Impact of a fixed-rate carbon tax using three different social costs for CO_2: diet quality through MER. Uncompensated (U-) and compensated (C-) scenarios.
Notes: In Scheme 1 (Sch.1), carbon taxes are applied to (i) beef products only, while in Schemes 2 and 3, the mark-up structure is extended progressively to (ii) pork, processed meat, poultry, and eggs and to (iii) all animal-based products (including dairy and fish), respectively.

From a nutritional point of view, the impact of the carbon tax on the MAR and MER tends to be small, irrespective of whether a compensation is introduced (Figs 3 and 4). As for our observations on GHGE, however, the patterns of change differ quite substantially across countries. For both Italy and Finland, the impact of all taxation schemes on diet quality is almost negligible (less than 1 percentage point), and those small changes follow a similar pattern: there is a slight worsening of the MAR (i.e. less beneficial nutrients in the diet) combined with a slight improvement in the MER (i.e. less undesirable nutrients in the diet), resulting in an ambiguous change in diet quality. We also observe small changes in Sweden (from 1 to 2 percentage points), but in most scenarios both the MAR and the MER tend to improve slightly. The impact of taxation on nutritional quality is much more pronounced for the UK diet: In the heaviest taxation scenarios (0.2€ per kg CO_2-eq and taxing all animal-based products), the MAR drops by more than 7 percentage points, while the MER improves by over 5 percentage points. This is because the UK experiences the largest percentage fall in the quantity demanded of many animal-based products (e.g. cheese and milk), as well as, of fruits and vegetables (see Figure A8 in Supplementary Appendix 3), leading to a substantial fall (up to –23 per cent) in many beneficial nutrients found in these foods (e.g. calcium, retinol, zinc, fibre). On the other hand, the sharp decline in the consumption of animal products including animal fats, which are a major source of some unhealthy nutrients such as cholesterol and saturated fats, is also responsible for the improvement in MER.

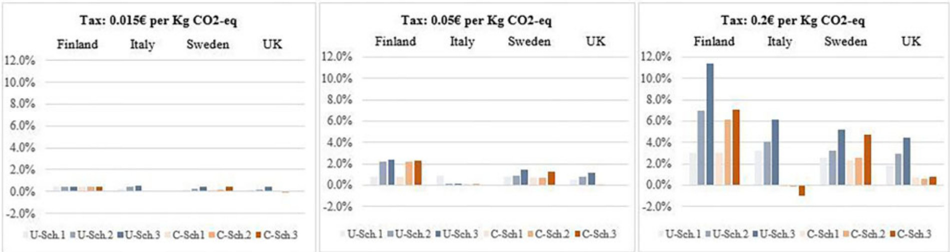

Figure 5. Impact of a fixed-rate carbon tax using three different social costs for CO_2: change in cost-of-living index. Uncompensated (U-) and compensated (C-) scenarios.
Notes: In Scheme 1 (Sch.1), carbon taxes are applied to (i) beef products only, while in Schemes 2 and 3, the mark-up structure is extended progressively to (ii) pork, processed meat, poultry, and eggs and to (iii) all animal-based products (including dairy and fish), respectively.

Finally, our results show that the welfare changes (measured in terms of percentage points increase in the food budget) due to the implementation of different taxation setups, hinge on both the scenario (uncompensated vs compensated) and the scheme (either 1, 2, or 3 in Table 1) (Fig. 5). As expected, the uncompensated scenario impacts more seriously the households' cost structure: When the taxation is heavier (i.e. the social cost of CO_2 estimated at 0.2€ per kg and the tax is levied on all animal-based products), the changes in the cost-of-living index can be as large as nearly 12 percentage points in Finland. Food expenditure increases to a higher extent in Finland than in the other countries analysed especially under Schemes 2 and 3 as it experiences the highest percentage price increase for most food categories subject to carbon taxes (i.e. pork, poultry and eggs, cheese, fish and seafood, and animal fats) (for more details, see Table A7 in Supplementary Appendix 3). In the compensated scenario, the impact is much lower, but again differences remain striking. The cost of the food budget can increase up to 7 percentage points in Finland and roughly 5 points in Sweden, while in the other countries the welfare cost is almost negligible (less than 1 percentage points). If we consider the welfare cost as a measure of the potential social acceptability of the carbon tax, then only the compensated scenario can have an acceptable social cost, although such cost may still be quite high in some countries.

5. Conclusions

In this paper, we study the effect of several taxation schemes on the level of GHGE from food consumption in three EU member states (Finland, Italy, Sweden) and the UK. Along with the environmental aspects of the outcomes, we provide evidence of changes in diets through mean adequacy/excess ratios and welfare effects via a cost-of-living index.

While we present estimations for both uncompensated and compensated tax scenarios, we consider the former as a benchmark for the compensated framework, which seems more realistic. First, our analysis indicates that, given the inelasticity of food demand, a sufficiently high carbon price is the primary requirement for achieving a substantial reduction in the climatic effect of the diet. Our results also show that the reduction in GHGE differs substantially across countries, tax schemes, and rates. In particular, the adoption of 0.2€ per kg of CO_2-eq rate and the choice of levying all animal-based products (Scheme 3) may generate a GHGE reduction as high as 19 per cent in Italy, 16 per cent in the UK, and 13 per cent in Sweden. In Finland, however, the same taxation scheme may generate a much smaller GHGE reduction of 7.5 per cent. Then, in terms of nutrients' intake, virtually all taxation schemes have a rather small impact on the quality of the diets; however, differences are found between countries (in some cases, we obtain a slight improvement of the MAR/MER, in some others a slight worsening). The UK is a notable exception, since the diet quality can change substantially by several percentage points. Finally, the welfare cost of the compensated scheme is small but not negligible, since it can reach up to 7 percentage point increase in the food budget in Finland and 4 points in Sweden. Then, given an average EU food budget share of roughly 12 per cent (Eurostat 2019), the incidence of the compensated tax would approximately float around 1 per cent of the total household expenditure.

While this geographical heterogeneity is not surprising (Slimani et al. 2002), it does clearly depend on country-specific characteristics: As diets typically hinge on local habits and customs, cultural differences influence food consumption patterns across member states (Tiu Wright et al. 2001). These differences are naturally captured by structural demand parameters, which, in turn, change considerably from country to country. Specifically, such topological heterogeneity is eventually captured by the magnitude and the spread of unconditional own-price elasticities as well as the distribution of cross-price parameters. For instance, while the Italian demand for beef is quite elastic, the own-price elasticity values for the other are always below the unit value.

This rather differentiated impact of the same tax schemes has important policy implications. Having country-specific tax schemes, calibrated on the specificity of the diets and on structural demand features, seems the most viable option. However, it is also important to recognize that such policy would require detailed knowledge of consumer response to carbon taxes on food and may need to be recalibrated over time. In this area, a common EU policy is likely to create strong distortions across countries and food sectors. At the same time, the approach of introducing a compensated tax scheme, in which the revenue obtained by taxing high-emission products is used to subsidise consumption of low-emission items, seems the best way to approach a very politically sensitive topic such as introducing a carbon tax on food. Consequently, a sensible alternative would consist in setting a common EU target for dietary-related GHGE, and let MS implement tailored tax measures within a joint fiscal framework. In the spirit of the 'Farm to Fork' strategy, which explicitly allows '[…] Member States to take more targeted use of rates […]' (European Commission 2020), this approach would exploit the structural differences across MS, while coordinating the efforts through a shared EU objective.

Nevertheless, these results also emphasize the challenges behind the design of a fiscal intervention aimed at reaching a public common goal. These challenges are first given by the heterogeneity and persistence of dietary habits, which make the overall effectiveness of such intervention difficult to evaluate. While this paper analyses a possible scenario on the implementation of such fiscal policy in some European countries, many questions still remain open.

First, policymakers need to consider whether price policies can be sufficient or 'optimal' to drive changes in food consumption, since, in the long run, the price adjustments among different product categories may require a continuous update of the tax rate. In order to be effective and stable in the long run, a fiscal policy needs to be included in a more general set of policy tools in which consumers are also educated and informed on the value of changing their habits. Second, other market forces influencing consumption patterns need to be evaluated and monitored. For example, the effect of a carbon tax on meat may be offset by the strategic reactions of producers and retailers, such as price cuts and promotions, which may strongly reduce its potential impact.[8] Overall, the results of the current analysis should be interpreted as an upper bound estimate of the GHGE mitigation potential of carbon taxes on food given the assumption of a perfectly inelastic supply curve. Furthermore, the time horizon of the tax must be considered: Is a carbon tax a permanent intervention or only an 'ice-break' policy to induce a turnaround on dietary habits? In this paper, we evaluate the short-run impact of introducing such a fiscal measure, but these potential long-run effects require a more sophisticated dynamic model to be analysed, which we leave for further research. Moreover, given the deterministic nature of our simulation approach, the robustness of the estimated results should be further investigated in future research to address the many complexities and uncertainties that characterize agricultural and food markets. Finally, a systemic policymaking approach needs to evaluate the impact and the costs of this policy on the supply side and consider the adoption of other measures to support the stakeholders, such as farmers and processors, in the transition to a different market environment.

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We dedicate this work to the memory of Professor Yves Surry, who was the leader of our research group and whose contribution to this study was invaluable. We want to acknowledge his passion, intelligence, and dedication to research in our field, as well as his warmth and friendship, which always enriched those who had the privilege to know and work with him.

Supplementary material

Supplementary data are available at *Q Open* online.

Author contribution

All the authors contributed equally to the realization of this paper.

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Conflict of interest

The authors report that there are no competing interests to declare.

Data availability

Restrictions apply to the availability of the data that support the findings of this study, which were used under licence for the current study and so are not publicly available.

End Notes

1. In the revenue-neutral or price-compensated scenarios, carbon taxes on the most polluting food categories are complemented by a VAT reduction on the most climate-friendly food groups to reduce the regressivity of the tax.
2. We define the MAR and MER more explicitly in Section 2.2.
Lewbel and Pendakur (2009) show that y is very highly correlated with the log of nominal expenditure deflated by the Stone price index.
4. SUSDIET is a research project funded within the framework of the ERANET SUSFOOD Call. Its main goals involve (i) identifying sustainable diets compatible with consumers' preferences in Europe and (ii) analysing public and private policies which could foster their adoption. More information at: https://www6.inrae.fr/sustainablediets.

5. The estimated GHGE are mainly related to the primary production, processing, packaging, and storage of food products. On the other hand, GHGE related to transportation and consumer travel were excluded by Hartikainen and Pulkkinen (2016) due to the lack of data and associated uncertainties. However, it is important to acknowledge that GHGE from transport generally make a small contribution to total dietary GHGE.
6. We transform conditional elasticities to unconditional using the approach in Carpentier and Guyomard (2001).
7. These include some aforementioned papers such as Edjabou and Smed (2013), Caillavet et al. (2016), and Bonnet et al. (2018).
8. We do not address these potential effects in the present study, which only models a complete transfer of the tax to the final consumers.

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