Journal of Cleaner Production 416 (2023) 137973

Contents lists available at ScienceDirect

Journal of Cleaner Production

journal homepage: www.elsevier.com/locate/jclepro

Assessing the environmental impact of diet – Influence of using different databases of foods' environmental footprints

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A R T I C L E I N F O

Handling Editor: Cecilia Maria Villas Bôas de Almeida

Keywords:
Sustainable diet
Foods' environmental impact databases
Environmental indicators
National food survey

A B S T R A C T

Current food systems compromise many environmental impact boundaries, and a dietary transition towards sustainable dietary patterns is needed. The definition of policies to promote the dietary transition implies accurate knowledge of dietary choices' environmental impact. For its assessment, life cycle assessment (LCA) has been employed, but methodological heterogeneity and limited access to LCA data hinder comprehensive comparisons. Several publicly accessible standardized foods' environmental impact databases have been developed to address these challenges. However, variations in indicators included and data sources raise questions about their impact on assessing dietary environmental footprints and the correlation amid various indicators within and between databases. This study aims to evaluate the effect of using different public-access food LCA databases on estimating the individual dietary environmental impact in a nationally representative survey through multiple indicators as well as the correlations between them.

Food-specific environmental impact indicators data from three databases (Poore&Nemececk, SHARP-ID, SU-EATABLE LIFE) were merged with individual food consumption data from the Portuguese Food, Nutrition, and Physical Activity Survey (IAN-AF, 2015–2016) (n = 5811) to estimate the usual environmental impact of diet for each indicator. Food groups' percentual contribution (%) to the environmental footprints and the Pearson correlation between indicators were also estimated.

Our results showed that different databases of foods' environmental impacts led to diverse estimates for common indicators when linked to the same food consumption data (e.g., GHGE – P50(P25–P75) SHARP-ID: 4.42(3.44–5.64)kgCO_2eq, Poore&Nemececk: 6.17(4.46–8.41)kgCO_2eq, SU-EATABLE LIFE: 5.64(4.26–7.36) kgCO_2eq). However, except for water footprints, most indicators from all databases were highly correlated with each other, with meat and other animal-based foods as the top contributors.

In conclusion, the absolute differences observed can compromise the validity of the findings and their comparability with other countries' estimates when different LCA data are used. A standardized, consolidated database covering a judicious selection of indicators across Europe along with official guidelines for assessing dietary environmental impacts would facilitate its assessment benefiting the establishment of food sustainability policies and recommendations at national and pan-European levels.

1. Introduction

The global food system has been linked to climate change by its effects on carbon footprint, arable land use change, and biodiversity loss (Campbell et al., 2017), contributing to crossing several planetary boundaries, levels of environmental perturbations derived by human activities below which the risk of destabilization of the Earth system is low, allowing human societies to develop and thrive (Rockström et al., 2009; Steffen et al., 2015). If current Western consumption patterns expand, the environmental pressure on the food system will increase, and planetary boundaries for several indicators will likely be reached shortly (European Commission, 2020), increasing the risk of

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https://doi.org/10.1016/j.jclepro.2023.137973
Received 4 May 2023; Received in revised form 28 June 2023; Accepted 30 June 2023
Available online 3 July 2023
0959-6526/

compromising processes that may be essential to human life (Campbell et al., 2017; Rockström et al., 2009). Moreover, current dietary patterns, high in red and processed meat and low in vegetables and fruit, are compromising human health, being responsible for a high share of the global burden of disease (Springmann et al., 2018b; Willett et al., 2019). Previous evidence has shown that in the European context (Bryngelsson et al., 2016) and globally (Willett et al., 2019), a dietary transition to decrease red meat consumption, in addition to technological innovations in agricultural productivity and reductions in food losses and waste across the food chain (European Commission, 2020), is essential to meet the climate targets and is expected to benefit human health.

This dietary transition requires robust, widespread, and targeted country-specific policies and public health strategies. For effective policies and public health strategies that promote a transition towards healthy dietary patterns with lower environmental burden, it is fundamental to recognize the environmental impact of the population's dietary choices and information on the individual characteristics associated with it (Garnett, 2011; Mertens et al., 2019b; Strid et al., 2019). Every food product contributes to environmental pressures throughout its life cycle, including effects from food production, processing, distribution, and disposal (ISO 14044:2006, 2006). The extent of these environmental burdens varies significantly depending on the type of food and specific production characteristics (Clark et al., 2022). Therefore, having high-quality data on specific environmental impacts of foods consumed by the population is crucial for guiding policymakers and regulators in their dietary recommendations, the food industry and retailers in their targets and actions and consumers in their informed choices.

Life cycle assessment (LCA) is an approach that compiles and quantitatively evaluates all inputs, outputs, and environmental impacts of a product system from its production until its disposal (i.e., life cycle). It is commonly endorsed by international institutions like the European Commission and the United Nations Environment Programme to support sustainability-focused policy decisions (Cucurachi et al., 2019). The environmental impact assessment implies the selection of relevant indicators of environmental performance (ISO 14044:2006, 2006). In the context of this study, the indicators are quantitative variables that can be measured, calculated, or described, representing the environmental impacts or footprints (ISO 14050:2020, 2020). Combining food LCA data for different environmental impact indicators with individual food consumption data can be used for a comprehensive estimate of the environmental footprints of dietary patterns, and it has been done in several studies (Bryan et al., 2019; Capone et al., 2013; Hyland et al., 2017; Meier and Christen, 2013; Mertens et al., 2019b; Saxe et al., 2013; Temme et al., 2015; Vellinga et al., 2019; Vieux et al., 2012) that predominantly focus on carbon footprint (greenhouse-gas emissions) but also some other indicators. Nonetheless, these studies present high methodological heterogeneity due to the flexibility in international standards, different indicators assessed, the lack of an LCA standard database implying differences in the life-cycle stages of the food chain considered, and the use of assumptions, extrapolations, and substitutions for missing LCA data (Garnett, 2011; Heller et al., 2013). Thus, a clear environmental impact comparison between different dietary patterns in diverse settings must be improved. Moreover, most of these studies are conducted using private-access food LCA data, hindering its widespread use by multiple stakeholders, such as scientific researchers, policymakers, and citizens in general.

To overcome these issues, some research groups have compiled harmonized food LCA data into standardized environmental impact databases for public use (Heller et al., 2018; Mertens et al., 2019a; Petersson et al., 2021; Poore and Nemecek, 2018). These datasets present a good effort to consolidate and harmonize food LCA data, offering various stakeholders publicly accessible, trustworthy information on the environmental impacts linked to food commodities. This holds significant importance in countries like Portugal, where comprehensive and reliable LCA data is limited (Morais et al., 2016). Nevertheless, these databases present several differences regarding the indicators covered and the data sources used. The impact of these differences on the dietary environmental assessment using individual-level consumption data still needs to be discovered. Moreover, to what extent multiple environmental indicators measured from the same or different databases correlate must also be clarified.

Accordingly, this study aimed to assess the effect on the dietary environmental impact estimation of applying foods' LCA data from different public-access databases to the same individual food consumption data, the Portuguese Food and Physical Activity Survey 2015–2016 (IAN-AF, 2015–2016). The effects will be assessed through differences in absolute estimates for common and diverse indicators and through the percentual contribution of foods to each indicator's performance. Moreover, the correlation between indicators within and between databases was evaluated.

From a practitioner perspective, this information will be valuable for establishing common ground guidelines enabling a standardized and facilitated dietary environmental impact assessment that should guide the pondering of the multiple trade-offs in the process of establishing food sustainability policies and recommendations, nationally and in a pan-European context, allowing for multi-country dietary environmental impact comparisons. Moreover, a uniform dietary environmental impact assessment framework, with harmonized indicators, will also allow for more robust and fair monitoring and evaluation of various food sustainability policies and recommendations (Sustainable Development Solutions Network, 2015).

2. Methods

2.1. Study participants and food consumption data

We used data from the Portuguese Food and Physical Activity Survey 2015–2016 (IAN-AF, 2015–2016), which methodological approach is fully described elsewhere (Lopes et al., 2017, 2018). IAN-AF 2015–2016 was a cross-sectional study designed to collect detailed individual food consumption data through harmonized and standardized procedures defined by European guidelines from the European Food Safety Authority (EFSA), namely in the EU Menu Project (EFSA, 2014). Correspondingly, a representative sample of the general Portuguese population, aged between 3 months and 84 years old, was selected from the National Health Registry by multistage sampling. The participants ($n = 5811$) then completed two non-consecutive dietary assessment interviews, with an 8–15 days break to avoid interdependence of food consumption. The interviews were led by trained dietitians using a validated specific electronic platform (eAT24) (Goios et al., 2020). Comprehensive dietary intake data, including the quantification of foods, recipes, and supplements, were collected for all participants through two non-consecutive one-day food diaries among children (<10 years old) and two non-consecutive 24-h recalls among adolescents and adults (≥10 years old). All food items IAN-AF 2015–2016 participants reported were coded using the EFSA's FoodEx2 classification system (EFSA, 2017).

2.2. Environmental impact indicators data

In this study, we considered three published and publicly available databases of foods' environmental impact data based on LCA, covering multiple environmental indicators and entailing high data standardization criteria. Table 1 presents a summary of the databases' characteristics. The full description and the complete details regarding the development of the databases are published in the respective methodological papers (Mertens et al., 2019a; Petersson et al., 2021; Poore and Nemecek, 2018). In the sections below, we provide a brief methodological description of each database, including details on the respective structure and indicators covered.

Table 1
Characteristics of the foods'' environmental impact databases used in this study: Poore & Nemecek, SHARP-ID and SU-EATABLE LIFE.

Database	Methodology for impact assessment	Included indicators
Poore & Nemecek (Poore and Nemecek, 2018)	LCA - *System Boundaries considered:* • Primary production • Processing • Packaging • Transport • Retail	• Land use • GHG emissions • Acidifying Emissions • Eutrophying Emissions • Freshwater withdrawals
SHARP-ID (Mertens et al., 2019a)	LCA - *System Boundaries considered:* • Primary production • Packaging • Transport • Retail • Home-preparation	• Land use • GHG emissions
SU-EATABLE LIFE (Petersson et al., 2021)	LCA - *System Boundaries considered*: • Primary production • Processing • Packaging • Transport • Retail	• GHG emissions • Water footprint

2.2.1. Database I – Poore and Nemecek

The first was a consolidated multi-indicator global database developed by Poore and Nemecek (2018), which includes life-cycle information for 40 aggregated food groups representing around 90% of protein and calorie global consumption, compiled using data from an accurate selection of 570 published studies (11 criteria to ensure a standardized methodology from an initial volume of 1530 potential studies) covering around 38,700 commercially viable farms in 119 countries. The system boundaries comprise primary production, processing, transport, packaging, and retail phases. The environmental indicators encompassed in this database are greenhouse-gas emissions (GHGE), land use (LU), water footprint (WF) divided into two measures (freshwater withdrawals, F-WF, and freshwater withdrawals weighted by local water scarcity, S-WF), acidifying emissions (AE), and eutrophying emissions (EE). Regarding water footprint indicators, F-WF refers to the direct calculation inventory items, namely irrigation withdrawals, irrigation withdrawals embedded in feed, drinking water for livestock, water for aquaculture ponds, and processing water. Regarding S-WF, the authors assumed that all irrigation water is evapotranspired or embedded in the product, and none returns to the watershed through percolation, which may be an overestimation.

2.2.2. Database II – SHARP-ID

The second database used in this study was the SHARP indicators database (SHARP-ID) (Mertens et al., 2019a). This dataset provides estimates on two indicators, GHGE and LU per kg of food as consumed for 944 food items. Foods' attributional life-cycle data was compiled from various LCA data sources (*Agri-footprint 2.0*, *Ecoinvent 3.3, CAPRI*) and other scientific publications and attributed through direct mapping ($n = 594$) or proxy values ($n = 350$). The data comprised the stages of primary production, use of primary packaging, transport, food losses and waste, and food preparations. Additionally, the foods described in SHARP-ID are classified using the FoodEx2 system.

2.2.3. Database III – SU-EATABLE LIFE

The third database used was SU-EATABLE LIFE, a multilevel database of carbon (GHGE) and water footprint (WF) values of food commodities from peer-reviewed papers, conference proceedings, public reports or studies where methods of data collection and handling were described, and Environmental Product Declarations (EPDs) (Petersson et al., 2021). This dataset was based on a standardized methodology to assign scientifically meaningful life-cycle footprint values and uncertainties to food commodities. The compiled version of the dataset used in this study includes data on 323 and 320 food items concerning GHGE and WF, respectively. The impacts reflect primary production, processing, packaging, transport, and retail stages. This database was originally not classified with FoodEx2. Nonetheless, for the analysis presented in this study, we coded each item in the SU-EATABLE LIFE database with a FoodEx2 code.

2.3. Assessment of the environmental impact of the diet

Each database was merged with IAN-AF 2015–2016 data and multiplied by the amount consumed per individual to estimate the environmental impact of the Portuguese diet. As the databases presented different structures, the methodology to merge them with the consumption data had to be database-specific. A schematic representation of the methods used for combining the food consumption data with each database is depicted in Fig. 1, and the details are described below.

We merged SHARP-ID and SU-EATABLE LIFE databases with the IAN-AF 2015–2016 data using the FoodEx2 code as the linkage key. However, if a food reported in IAN-AF 2015–2016 was non-existing in these environmental impact databases, we attributed the average value of the hierarchical closest items. The share of foods from IAN-AF 2015–2016 attributed via exact FoodEx2 code match was 62% for SHARP-ID, and for SU-EATABLE LIFE, it varied with the indicator: 30% for GHGE and 43% EM for WF). Then the indicator values were multiplied by the amount consumed per person. The food amount used depended on the database's methodological characteristics. For SHARP-ID, we used the edible cooked amount, as the values are presented for the food as consumed. For SU-EATABLE life, we used the raw edible amount.

Poore & Nemecek database presented data for 40 aggregated food groups. Thus, we considered that the FoodEx2 matching was not suitable. Instead, for this database, each IAN-AF food item was manually linked to a Poore & Nemecek group by similarity. For instance, for the IAN-AF 2015–2016 item "Chicken, breast, roasted", we attributed the values of the Poore & Nemecek "Poultry Meat" group. Furthermore, this attribution was not possible for some food items reported in the survey, namely processed foods, representing around 20% of the total energy intake. After merging, we multiplied the indicator values by the raw edible amount of food reported per participant.

2.4. Statistical analysis

For each indicator assessed from the diverse databases, the overall individual daily impacts were obtained by aggregating the impacts from all foods consumed, in their respective amounts, within each day. Then, the distribution of the usual daily dietary environmental impact considering each available indicator was estimated from the two-day assessment using the 1-Part model for daily-consumed foods or food components as a fractional polynomial of age from the SPADE software (Dekkers et al., 2014). This software statistically corrects for within-person variation, resulting in a shrunken distribution. We calculated the results for the total population and by sex and age group (children: <10 years old, adolescents (10–17 years old), adults (18–64 years old), and elderly (≥65 years old).

Furthermore, we identified the primary dietary contributors to the environmental impacts considered. To do so, we evaluated the percentual contribution (%) of each food group to the environmental impact for each indicator at the individual level, and then we calculated the weighted average for the population.

Finally, the correlation coefficients amid indicators within and between the three databases were estimated using the Pearson correlation, and a heatmap was created to represent the correlations graphically. The

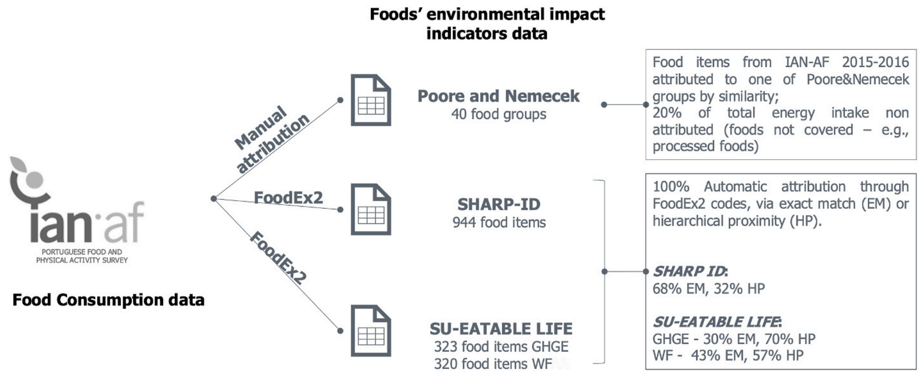

Fig. 1. Schematic overview of the methodological approach of the present paper, namely for linking food consumption data (IAN-AF, 2015–2016), with different databased of foods' environmental impact indicators: Poore & Nemecek, SHARP-ID and SU-EATABLE LIFE. Abbreviations: GHGE: Greenhouse-gas-emissions; WF: Water footprint.

analyses described were conducted in R software version 3.6.2 for MacOS (R Core Team, 2019). A p-value <0.05 was considered statistically significant.

3. Results

Table 2 describes all environmental impact indicators evaluated in the Portuguese population using the three databases for the total population and stratified by sex and age group. The average is not presented because some indicators showed a skewed distribution. Thus, the median and interquartile range were used to describe the indicators and are presented in Table 2.

Using foods' environmental impacts from Poore & Nemecek database resulted in daily individual median values of 6.17 kg Co_2eq, 13.45 m^2, 855 L, 26586 L, 39.7 g SO_2eq and 32.5 g PO_4^{3-}eq for GHGE, LU, F-WF, S-WF, AE and EE, respectively. The dietary environmental impacts slightly varied by sex and age group. Red and white meat were the food groups that mainly contributed to environmental footprints. Regarding water, however, fish and dairy (milk and cheese) were the top contributors.

The analysis of dietary environmental impact using SHARP-ID resulted in median GHGE and LU of 4.42 kg Co_2eq and 5.36 m^2 per day, respectively. Meat (red, white, and processed), fish and milk were the leading sources of GHGE, whereas meat (red, white, and processed), bread and milk were the core contributors to LU.

The daily median of GHGE was 5.64 kg Co_2eq when using the SU-EATABLE LIFE database, and the median value of WF was 5858 L. Red and white meat, bottled water, yoghurt, fish, and fresh vegetables were the key sources of dietary GHGE in the Portuguese population (Fig. 2). Regarding WF, the leading contributors were red and white meat, fresh fruit, water and yoghurt (Fig. 2).

The results for the common indicators GHGE and LU (WF is not directly comparable) show differences in the estimates depending on the database used. Regarding GHGE, Poore & Nemecek presented the highest median estimate, followed by SU-EATABLE LIFE and SHARP-ID. For LU, the median result obtained using Poore & Nemecek database is around 2.5 times higher than the estimated using SHARP-ID.

Despite these differences, the correlation coefficients estimated between indicators from the three databases are positive and moderate-to-high, even between different indicators from different datasets (Fig. 3). GHGE, LU, AE and EE from Poore & Nemecek and SHARP-ID all correlate strongly with each other $\rho \in [0.67–0.98]$, p-value<0.001. Indicators expressing water footprint from Poore & Nemecek (F-WF and S-WF) and SU-EATABLE LIFE (WF) are clustered in the graph and, with few exceptions, present lower correlations with the remaining indicators. GHGE from SU-EATABLE LIFE presents a strong correlation with WF from the same database ($\rho = 0.94$, p-value<0.001), whereas lower correlation coefficients with GHGE and other indicators from other databases.

Further information on the estimates and contributions by sex and age group for all databases used can be found in Table 2 and Tables S1–S11 (Supplementary Material).

4. Discussion

This study aimed to assess the impact of using different foods' LCA compilation databases in estimating the individuals dietary environmental impact using food consumption data from a nationally representative survey. The results from the current study show that using different databases of food environmental impacts, in combination with individual food consumption data, led to several differences. First we highlight the array of environmental indicators comprised in the databases and differences in the coverage of foods: Poore and Nemecek includes 6 indicators, GHGE, LU, AE, EE, S-WF and F-WF for 40 food groups; SHARP-ID has information for 2 indicators, GHGE and LU for 944 food items; and, finally, SU-EATABLE LIFE also encompasses 2 indicators, GHGE, WF, for 323 and 320 foods, respectively.

Then, as we anticipated due to methodological differences, heterogeneous sources and inputs for LCA data, and foods covered, the absolute estimates on individuals dietary environmental impact were heterogenous depending on the dataset used, even for the common indicators, with median results for GHGE of 6.17, 4.42 or 5.64 kg Co_2eq from Poore and Nemecek, SHARP-ID and SU-EATABLE LIFE, respectively and for LU of 13.45 or 5.36 m^2 from Poore and Nemecek, and SHARP-ID, respectively. We can argue on some possible specific reasons causing these discrepancies.

First, the different structure of the dataset implied using different approaches to merge the foods' environmental footprints with the food consumption data due to the diverse structures of the databases. We used the FoodEx2 code as the linkage key for SHARP-ID and SU-EATABLE LIFE. However, not all foods had an exact match (EM), and for a share of foods reported in the survey, a value based on the hierarchical proximity (HP) was attributed (SHARP-ID: 62% EM and 38% HP; SU-EATABLE LIFE: 30% EM and 70% HP for GHGE; 43% EM and 57% HP for WF). For Poore & Nemecek, the FoodEx2 method was not feasible, and we matched the data by food group similarity. In this database, information on environmental footprints for highly processed foods such as cookies, pastries, and soft drinks was unavailable. Thus, we excluded these foods in this analysis, likely underestimating the environmental impacts. However, we estimated that the foods excluded from this assessment represented only around 20% of the total energy intake in the sample. A recent study has proposed an algorithm to apply Poore & Nemecek's data to a vast set of processed products from the UK and Irish markets, overcoming this limitation (Clark et al., 2022). Yet, it was not possible to replicate the proposed methodology in our study due to a lack of product formulation data.

Table 2
Distribution of dietary environmental impact indicators (Median and Interquartile Range) in the Portuguese population (IAN-AF, 2015–2016), using different data sources: Poore & Nemecek, SHARP-ID and SU-EATABLE LIFE.

	Poore & Nemecek	SHARP-ID	SU-EATABLE LIFE
	Median (P25–P75)		
Greenhouse-gas emissions (kg CO_2eq)			
Total Population	6.17 (4.46–8.41)	4.42 (3.44–5.64)	5.64 (4.26–7.36)
Age group			
Children (<10 years)	5.03 (3.32–7.15)	4.20 (3.29–5.33)	4.33 (3.06–5.83)
Adolescents (10–17 years)	6.68 (4.9–8.99)	4.84 (3.83–6.1)	6.06 (4.71–7.74)
Adults (18–64 years)	6.48 (4.74–8.75)	4.62 (3.63–5.85)	6.07 (4.69–7.79)
Elderly (≥65 years)	5.28 (3.83–7.18)	3.62 (2.85–4.59)	4.44 (3.4–5.75)
Sex			
Female	5.00 (3.74–6.60)	3.76 (3.00–4.69)	4.74 (3.7–6.01)
Male	7.68 (5.63–10.32)	5.29 (4.16–6.66)	6.83 (5.18–8.82)
Land Use (m^2)			
Total Population	13.45 (9.22–19.41)	5.36 (4.02–7.11)	–
Age group			
Children (<10 years)	12.87 (8.34–19.01)	4.51 (3.2–6.13)	–
Adolescents (10–17 years)	15.06 (10.48–21.48)	6.36 (4.91–8.24)	–
Adults (18–64 years)	14.04 (9.73–20.1)	5.68 (4.34–7.43)	–
Elderly (≥65 years)	10.97 (7.56–15.8)	4.18 (3.22–5.43)	–
Sex			
Female	10.9 (7.69–15.29)	4.48 (3.47–5.77)	–
Male	16.85 (11.54–24.32)	6.48 (4.86–8.53)	–
Water footprint (L) [a]			
Total Population	–	–	5858 (4501–7544)
Age group			
Children (<10 years)	–	–	4488 (3238–5958)
Adolescents (10–17 years)	–	–	6153 (4832–7799)
Adults (18–64 years)	–	–	6258 (4898–7955)
Elderly (≥65 years)	–	–	4816 (3747–6158)
Sex			
Female	–	–	4963 (3945–6189)
Male	–	–	7044 (5428–9003)
Freshwater Withdrawals (L) [a]			
Total Population	855 (674–1056)	–	–
Age group			
Children (<10 years)	727 (530–934)	–	–
Adolescents (10–17 years)	876 (697–1076)	–	–
Adults (18–64 years)	878 (699–1079)	–	–
Elderly (≥65 years)	805 (634–997)	–	–
Sex			
Female	742 (596–904)	–	–
Male	990 (797–1200)	–	–
Scarcity-Weighted Freshwater Withdrawals (L) [a]			
Total Population	26586 (21311–32397)	–	–
Age group			
Children (<10 years)	22786 (16966–28799)	–	–
Adolescents (10–17 years)	27202 (21992–32970)	–	–
Adults (18–64 years)	27274 (22047–33057)	–	–
Elderly (≥65 years)	25107 (20106–30670)	–	–
Sex			
Female	23250 (19035–27819)	–	–
Male	30640 (24975–36768)	–	–
Acidifying Emissions (g SO_2eq)			
Total Population	39.7 (30.0–51.4)	–	–
Age group			
Children (<10 years)	32.7 (22.7–44.1)	–	–
Adolescents (10–17 years)	41.8 (32.12–53.4)	–	–
Adults (18–64 years)	41.8 (32.1–53.5)	–	–
Elderly (≥65 years)	33.7 (25.3–43.9)	–	–
Sex			
Female	32.5 (25.3–40.9)	–	–
Male	49.1 (37.8–62.2)	–	–
Eutrophying Emissions (g $PO_4^{3-}eq$)			
Total Population	34.0 (24.9–45.2)	–	–
Age group			
Children (<10 years)	26.5 (17.6–37.1)	–	–
Adolescents (10–17 years)	34.4 (25.5–45.6)	–	–
Adults (18–64 years)	35.5 (26.3–46.8)	–	–
Elderly (≥65 years)	31.1 (22.8–41.4)	–	–
Sex			
Female	28.1 (21.0–36.8)	–	–
Male	41.6 (31.0–54.3)	–	–

[a] In the Poore & Nemecek database, water footprint is given by two separate indicators. Freshwater withdrawals (F-WF) refers to the direct calculation inventory items: irrigation withdrawals; irrigation withdrawals embedded in feed; drinking water for livestock; water for aquaculture ponds; and processing water. The authors assumed that all irrigation water is evapotranspired or embedded in the product, and none is returned to the watershed through percolation to estimate Scarcity-Weighted Freshwater Withdrawals (S-WF), which may be an overestimation. Because this methodological approach does not apply to the SU-EATABLE LIFE database, we presented these indicators separately.

Differences in the LCA compiled data used in the databases are also possible reasons for the differences found in the results. Food LCA data is subject to considerable variability that may be reflected in these different databases. Different management practices, soil types and climates, the timescale of the study affecting seasonality, distinct transportation modes and distances and subsequent processing, retailing and consumption activities, diverse storage time, packaging and food preparation are some aspects that contribute to the LCA data variability and possible differences between the databases (Notarnicola et al., 2017).

Notwithstanding the absolute differences perceived, the analysis of food contributors showed many similarities across all databases for the mutual indicators, with animal-based products, particularly meat and dairy, ranking at the top of the highest contributors. An exception was found for the indicators associated with water footprint, evaluated using Poore & Nemecek and SU-EATABLE LIFE, where both absolute footprint values and contributions present pronounced disparities, suggesting profound methodological differences between databases concerning this indicator. The correlation analysis corroborates these results and suggests that indicators show significant positive moderate-to-high correlations (especially GHGE, LU, AE and EE), independently of the databased considered, whereas water presents slightly lower

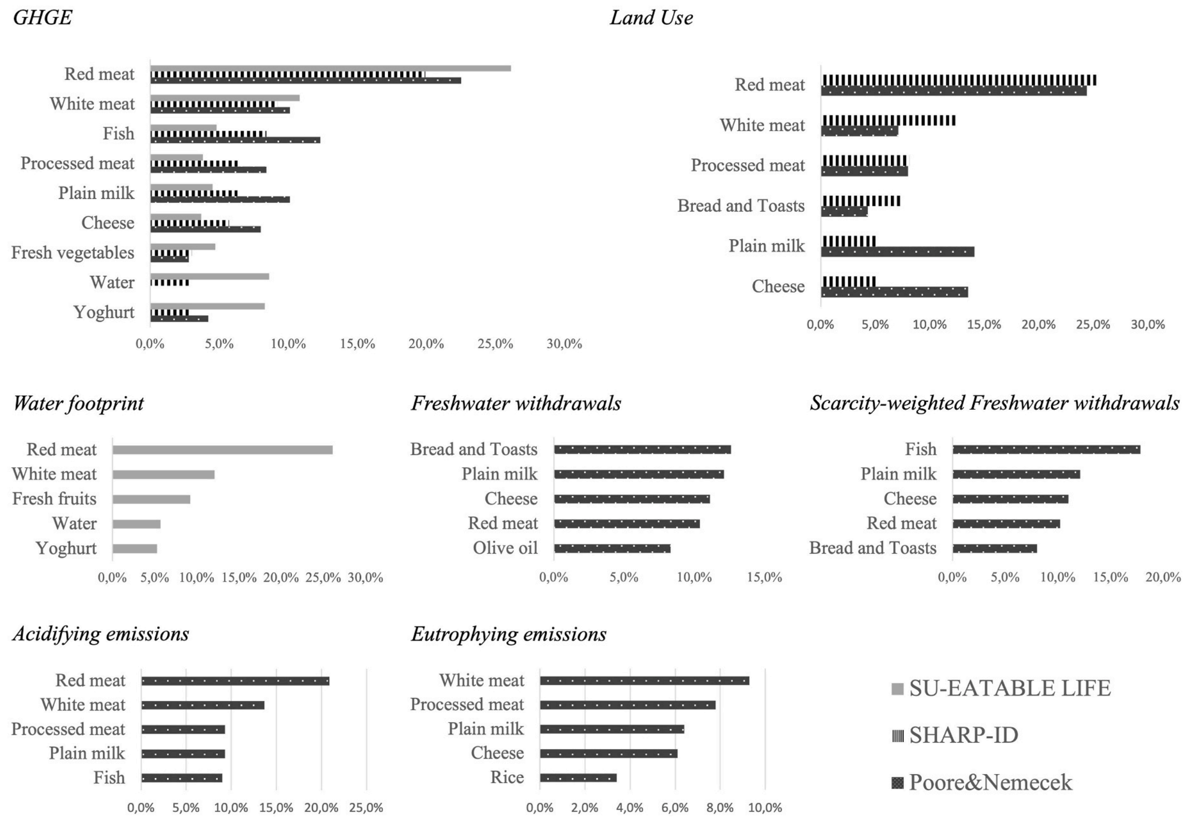

Fig. 2. Contribution (%) of food groups to dietary environmental impact indicators, in the Portuguese population (IAN-AF, 2015–2016), using different data sources: Poore & Nemecek, SHARP-ID and SU-EATABLE LIFE.

correlations. In SU-EATABLE LIFE, however, GHGE and WF have a very high correlation, which was unexpected. Previous studies have shown differences in these indicators because foods with lower carbon footprint, such as fruits, vegetables and pulses, can have higher water footprints due to higher irrigation needs (Harris et al., 2020; Springmann et al., 2018a, 2018b). The correlations between GHGE from SU-EATABLE LIFE and GHGE (or other indicators) from the remaining databases were lower than the other correlations observed, suggesting that the validity of this dataset to estimate GHGE may be poorer. However, as we do not have a gold standard, conclusions regarding this topic are hampered.

Acknowledging the impact of using different data sources is essential to guide the process of determining the procedures for harmonized dietary environmental impact assessment which, in its turn, is crucial for future planning and monitoring of sustainability policies and strategies. Examples of strategies to reduce dietary environmental effects include targeting overconsumption by promoting an energy intake reduction, which has been linked to higher environmental impact or focusing on a qualitative change, supporting the substitution of food items with a higher environmental footprint (e.g., meat and dairy) with more sustainable and healthier alternatives (Mertens et al., 2019b; Vellinga et al., 2019; Vieux et al., 2012). In previous research, dietary shifts were among the most effective strategies for reducing the food system's environmental impact (Bryngelsson et al., 2016; Garnett, 2011; Meier and Christen, 2013; Notarnicola et al., 2017; Perignon et al., 2017; Stehfest et al., 2009). Promoting sustainable food consumption and facilitating the change to a healthy, sustainable diet are goals of the European Commission in its Farm to Fork Strategy (European Commission, 2020). In the view of a Target, Measured and Act approach, proposed originally to tackle Sustainable Developmental Goal 12.3 (Food Waste) (Champions 12.3, 2020), but easily applicable to this topic, and considering the global dietary transition to tackle climate change and ensure the environmental boundaries as the *target*, *measuring* the environmental impact of dietary choices and patterns in a harmonized and standardized way, ideally at the individual level poses as crucial to *act*, defining appropriate and more specific policies and strategies to reach the target. Measuring the environmental impact with harmonized data and indicators is also relevant for monitoring the implementation and the impact of policies that tackle sustainability issues at the global level to ensure comparability (Sustainable Development Solutions Network, 2015).

The results from this study clearly demonstrate that differences in the dietary environmental impact may simply be due to differences in the LCA data used. Thus, when establishing comparisons between different settings or different strategies/policies a common framework would ensure that comparisons reflect real differences in dietary patterns as well as an unbiased monitoring on the effectiveness of such policies. For instance, to compare the Portuguese dietary environmental impact estimated from our study with other settings is necessary to consider the data used in such studies. The results using SHARP-ID (GHGE – 4.4 kgCO2eq; LU – 5.4 m2) are comparable to the Mertens et al. study results reporting the dietary environmental impact of four European countries, which also used the SHARP-ID (Mertens et al., 2019b). Our estimates for GHGE and LU were lower than all estimates presented in Mertens study (*GHGE* – Denmark: 5.2 kgCO_2eq, Czechia: 5.4 kgCO_2eq, Italy: 5.1 kgCO_2eq, and France: 5.9 kgCO_2eq; *LU* – Denmark: 6.7 m^2, Czechia: 7.1 m^2, Italy: 6.6 m^2, and France: 7.3 m^2), suggesting that Portuguese dietary patterns have lower environmental impact concerning these two indicators.

Nevertheless, using a common foods' environmental impact database to assess dietary environmental footprint implies an additional limitation. These databases present standardized values of the indicators covered, based on the literature, that are not country-specific. Thus, specificities of national food systems that may affect environmental

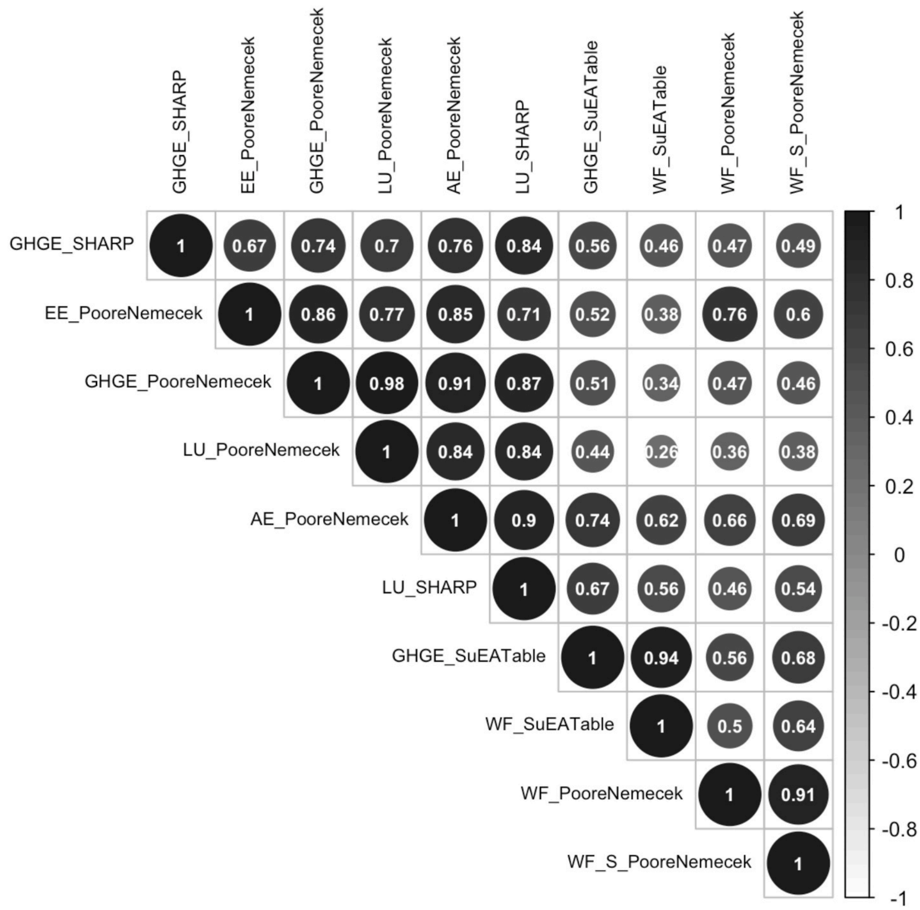

Fig. 3. Correlation coefficients between all indicators from the three databases: Poore & Nemecek, SHARP-ID and SU-EATABLE LIFE. All estimates present p-value <0.05.

impacts were not considered (Garnett, 2011). Nonetheless, these databases are relevant sources of standardized LCA data, especially for countries that don't have comprehensive LCA research covering most food products consumed. For instance, in the specific case of Portugal, according to a previous scientific review, only a few agri-food LCA studies are available, and those do not have a systematic and country-scale approach compromising its regional accuracy and comparability (Morais et al., 2016). Furthermore, using these databases to study dietary environmental footprints in different countries, such as the previous example, provides estimates that adequately reflect the diverse effects of distinctive food consumption patterns on several environmental impact indicators, despite uncertainties (Mertens et al., 2019b).

Comparing our results with previous research supports that standardized measurement of dietary environmental impact may improve the reliability of the identification of specific high-impact groups and high-contributing foods across countries, allowing for better-targeted approaches. Thus, in a pan-European context, where there is a Common Agricultural Policy that strengthens the intra-EU food trade (Piglowski, 2021), and given the absolute differences and lack of standardization and shortage of covered food products among the currently available data sources, our results highlight the need for official guidelines and data that can ease the assessment of dietary environmental impact, improving knowledge in this field. As most environmental impact indicators seem strongly correlated for facilitating purposes, selecting the most divergent indicators (e.g., GHGE and WF) may be advocated, or a single composite environmental indicator may be proposed, as suggested in a previous study (Clark et al., 2022). Higher awareness of the actual environmental impact of the population diet is desired for regulators to guide their policies, for consumers wanting to make informed and more sustainable choices, for the food industry and retailers that are increasingly setting carbon-neutrality goals and incorporating ecolabels in the packaging of food products (Clark et al., 2022).

In the current study, we proposed to assess the individual dietary impact through multiple publicly available foods' environmental impact indicators databases applied to a common food consumption database from a representative sample of the Portuguese population. Given that LCA data can be highly heterogeneous, and these databases differ regarding indicators covered, foods included, and data sources consulted for compilation, we expected to observe differences in the absolute estimates. The methodological framework proposed in this study allowed us to estimate the magnitude of such differences, simultaneously evaluating the correlation amid indicators within and between databases to address the relative differences. In conclusion, our results corroborated that the data on the environmental impacts of food items affect the assessment of the diet's environmental impact as different databases of environmental impacts led to diverse estimates for common indicators when linked to the same food consumption data. Nonetheless, most indicators were highly correlated, independently of the database used. The choice of the database should be aligned with the study objectives, namely, in terms of indicators to assess (i.e., choose the database that covers the indicators of interest) or comparison to previous studies (i.e., select the same database to be comparable). Furthermore, we argue that an official harmonized and consolidated European database of food's environmental impacts, covering a range of the most divergent indicators, along with expert-defined guidelines for dietary environmental impact assessment, would be valuable for research in sustainable diets as well as for establishing and monitoring specific policies and actions by multiple stakeholders that promote dietary transitions.

Funding

The IAN-AF 2015–2016 received funding from the EEA Grants Program, Public Health Initiatives (grant number PT06-000088SI3). The EEA Grant Program had no role in this article's design, analysis or writing. This work was also supported by Fundação para a Ciência e Tecnologia (FCT) through the projects UIDB/04750/2020 and LA/P/0064/2020, and the Doctoral Grant SFRH/BD/146078/2019 (CC).

CRediT authorship contribution statement

Catarina Carvalho: Conceptualization, Methodology, Formal analysis, Writing – original draft. **Daniela Correia:** Methodology, Formal analysis, Data curation. **Sofia Almeida Costa:** Conceptualization, Writing – review & editing. **Carla Lopes:** Conceptualization, Supervision, Project administration, Writing – review & editing. **Duarte Torres:** Conceptualization, Supervision, Project administration, Writing – review & editing.

Declaration of competing interest

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

Data availability

Data will be made available on request.

Acknowledgements

The National Food, Nutrition, and Physical Activity Survey of the Portuguese General Population (IAN-AF 2015–2016) had institutional support from the General Directorate of Health (DGS), the Regional Health Administration Departments, the Central Administration of the Health System (ACSS) and the European Food Safety Authority (CFT/EFSA/DCM/2012/01-C03). The authors acknowledge the IAN-AF consortium, all these institutions and persons involved in all phases of the IAN-AF 2015–2016, and the participants.

The authors also acknowledge the SYSTEMIC project, a knowledge hub on Nutrition and Food Security, in collaboration with JPI-HDHL (Healthy Diet for a Healthy Life), FACCE-JPI (Joint Programming Initiative for Agriculture, Climate Change, and Food Security), JPI-OCEANS (Joint Programming Initiative Healthy and Productive Seas and Oceans).

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jclepro.2023.137973.

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