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Quantifying the environmental impacts of a European citizen through a macro-economic approach, a focus on climate change and resource consumption

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ABSTRACT

As economies in the European Union are ultimately driven by the final consumption of citizens, policy makers need proper indicators to monitor the environmental impacts associated with this consumption. These indicators can be constructed using two different approaches, each having their strengths and limitations. The top-down approach is based on environmentally extended input–output analysis and quantifies the environmental impacts of product groups and services provided by industrial sectors. The bottom-up approach is based on Life Cycle Assessment and quantifies the environmental impacts of a selection of representative products. The bottom-up approach has already been used by the European Commission's Joint Research Centre to calculate the impacts of the final consumption per capita in the European Union in 2006. In this paper, we calculated these impacts through a top-down approach, using the Exiobase database. The covered household activities are food, consumer goods, mobility, shelter and services. The goal was to calculate all the impact categories recommended by the International Reference Life Cycle Data handbook, and compare both approaches. However, the categories ionizing radiation, toxicity and abiotic resource depletion could not be included, as some relevant emissions and resources are not available in Exiobase. To study more profoundly the impact on natural resources, we added the Cumulative Exergy Extraction From the Natural Environment to the impact assessment. When comparing both approaches, it can be concluded that there is a considerable shift in the results. This means that the information obtained by a top-down approach could supplement the information base for policy support.

1. Introduction

Today, sustainable development is one of the main challenges in many governmental policies, especially in that of the European Union (EU) (EC, 2009). To guide and monitor the transition towards a more sustainable society in terms of environmental performance, proper indicators are needed. The environmental impacts of an economic society are ultimately driven by consumption, either directly, as impact of the use phase of consumer products, or indirectly, as impact of the production and end-of-life phase of these products (Tukker and Jansen, 2006). Hence, the indicators should provide a clear view on the links between final consumption and environmental impacts (EC, 2012a).

In 2006, the European Commission's Joint Research Centre (JRC) published a report on the environmental impacts related to the final consumption in the EU25 (Tukker et al., 2006). This report, together with the corresponding article of Tukker and Jansen (2006), includes a review of the 11 most relevant studies about consumption in the EU over the last 5 years, identifying two different approaches: bottom-up and top-down.

The bottom-up approach is based on traditional Life Cycle Assessment (LCA), which involves detailed data collection on production processes to quantify the environmental impacts of a product system (Feng et al., 2011; Huysveld et al., 2013; Payen et al.,

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2015). The alternative for the LCA-approach is the top-down approach, referring to environmentally extended input–output analysis (EEIOA) (Hertwich, 2011; Steen-Olsen et al., 2012; Wiedmann et al., 2015). EEIOA links IO-tables, describing the transactions between industry sectors in an economy, with environmental extensions, describing the pollutants emitted and resources extracted by industry sectors per (monetary) unit output (Suh and Huppes, 2005).

The top-down approach will further be called 'IO-approach', and the bottom-up approach will further be called 'LCA-approach'. They are described more detailed in Materials and methods, Sections 2.2 and 2.3. Both approaches have their strengths and limitations. They complement each other: the strengths of the one are the limitations of the other, and vice versa.

The main strength of the LCA-approach is its high level of detail: it is possible to evaluate the contribution of specific products as such. Furthermore, the elementary flows (i.e. emissions and natural resources) in Life Cycle Inventory (LCI) databases are often very detailed, making it possible to calculate environmental impacts in an accurate way. However, the main advantage of the LCA-approach, i.e. its high level of detail, goes hand in hand with some limitations. The first limitation is the truncation or cut-off problem (Suh, 2002): as the product system under study represents only a part of the materials-product chain, a system boundary has to be drawn. However, because there is no information available on the flows outside this boundary, it cannot be ensured that these neglected flows are indeed negligible. Another limitation is that representative products have to be selected when the LCA-approach is used for analysing the final consumption, because it is not possible to include all the products due to the lack of readily available datasets. Furthermore, LCI datasets are often a mix of linked processes from different years and countries. This could have an influence on the impact results, since the upstream supply chain of a product may differ from country to country. These problems could improve when more complete LCI databases would become available in the future. Nevertheless, performing an LCA for new product systems remains a time-consuming task.

An IO-approach can overcome some of these limitations. First, there is no cut-off problem, since IO-databases cover entire material-product chains in an economy. For the same reason, it is no longer necessary to select representative products, when one wants to assess the impact of product groups, as all the products are already present as aggregated product groups in the IO-database. Another strength is that IO-databases are more consistent than LCA-databases: all datasets have the same reference year. A third strength is the availability of interregional data when multi-regional/world IO-databases are used, with a subdivision into various countries. These global databases enable the calculation of all indirect resource and emission flows embodied in imports and exports (Wiedmann et al., 2011). This is relevant for countries depending largely on import or export, as is the case for most member states of the EU. Of course, an IO-approach also has its limitations. The main limitation is the high level of aggregation: the product as such is not visible, and because many elementary flows are aggregated, some impact categories (e.g. ecotoxicity) cannot be calculated accurately. Another important limitation is the publication time of IO-databases. Often, there are several years between the publication of an IO-database and its base year (Suh, 2002). Nonetheless, once the database is established, calculations can be performed very quickly and easily.

These limitations of the IO-approach are counterparts of the strengths of the LCA-approach, as described above. It is also possible to combine the strengths of both approaches through a hybrid analysis, as has been done in several studies (Hou et al., 2014; Pairotti et al., 2015; Weinzettel et al., 2014).

The insights gained in the review (Tukker et al., 2006; Tukker and Jansen, 2006) paved the way for further research, aiming to calculate the environmental impacts of an average EU citizen with one of these two approaches. In theory, both approaches should lead to the same result, provided that IO-data of very high level of disaggregation and LCA data for a vast number of processes are available, and that these data are geographically and temporally explicit.

In 2012, JRC published a new study in which they used the LCA-approach to calculate the environmental impacts per capita in the EU27 for the year 2006 (EC, 2012a). By the EU27, we understand its 27 member states since 2007. It must be mentioned that this study of JRC was a pilot project, and not intended to be directly used for policy support.

During the same period, a new IO-database was being developed in the Sixth and Seventh Framework Program of the European Commission on compiling and refining environmental and economic accounts (Tukker et al., 2009; Wood et al., 2015). This new database, Exiobase, addresses the issues mentioned in the review of 2006 (Tukker and Jansen, 2006). The main asset is that Exiobase is a multi-regional database, interlinking 43 countries with the rest of the world. This means all EU27 countries are included, and that flows embodied in imports can be taken into account. This global database is very suitable for the IO-approach.

Hence, the objective of this study is to assess the annual environmental impact of the consumption of an average EU27 citizen through an IO-approach, using Exiobase v.2. This database has 2007 as reference year, which is close to the reference year 2006 of the LCA-study performed by JRC (EC, 2012a). This way, the IO-results and the LCA-results are considerably comparable.

2. Materials and methods

2.1. Mathematical structure

EEIOA and LCA are based on the same mathematical structure, see eq. (1). This is well explained in the work of Heijungs and Suh (2002) and Huysman et al. (2014):

$$g = B \cdot A^{-1} \cdot f \tag{1}$$

Matrix B is the environmental matrix, describing the amounts of natural resources extracted or emissions released by the industrial system. Matrix A is the technology matrix, describing the transfer of flows within the industrial system. Vector f is the final demand vector, representing the output demanded from the industrial system. Finally, the inventory vector g of all direct and indirect resources extracted or emissions released for a certain final demand f can be obtained by eq. (1) (Suh and Huppes, 2005).

In EEIOA, matrix A is written as $I - Z$, with I the identity matrix and Z the direct coefficient requirement matrix or IO-table. IO-tables are mostly available in monetary units, as in the original work of Leontief (1936). For the construction of IO-tables, two methods of handling multi-output processes (allocation) are possible: the industry-technology assumption, equivalent to partitioning in LCA, and the product-technology assumption, equivalent to system expansion in LCA (Suh et al., 2010). Both methods can be used to construct an IO-table in an industry-by-industry form ($I \times I$) or a product-by-product form ($P \times P$). In an $I \times I$ form, each row and column represents an industry sector, and in a $P \times P$ form, each row and column represents an industry output, i.e. a product group or service. A more detailed description is given in the work of Rueda-Cantuche and ten Raa (2007). In this study, we used the $P \times P$ form, which is more in accordance with the LCA-approach, because the focus is on products instead of industry sectors (Suh, 2002).

Having obtained the inventory g, it is possible to calculate the corresponding environmental impacts. To calculate impact h_i for impact category i, inventory g has to be multiplied with characterization vector q_i, see eq. (2). This vector consists of characterisation factors, corresponding with the chosen impact characterisation method.

$$h_i = q_i \cdot g = \sum_{k=1}^{n} q_{i_k} \cdot g_i \tag{2}$$

2.2. The LCA-approach

In the study of JRC, five household activities were defined: food, consumer goods, mobility, shelter and services (EC, 2012a,b). For each activity, a selection of key product groups was made, and for each product group, representative products were selected from the International Reference Life Cycle Data System (ILCD) database (EC, 2010a,b), see. Subsequently, the environmental impacts of these representative products were calculated based on equation (2). More details about the impact assessment are given in Section 2.4.

Next, the impact of each representative product was multiplied with the consumption of an average EU citizen in the year 2006, e.g. (impact per cotton shirt) × (number of cotton shirts consumed per capita). If consumption data of different reference years was used, it would also be possible to see changes in consumption patterns over time. Theoretically, this result could be scaled up to account for the entire product group or activity. However, the methodological choice was made not to apply scaling in order to arrive at the index rather than the coverage of the total impact (EC, 2012b). This means that the impacts reflect only the representative products in the basket (Table 1).

2.3. The IO-approach

In this study, we used the Exiobase version 2 database to perform the IO-approach. Currently, two versions of Exiobase are available: version 1 with reference year 2000 (Tukker et al., 2009) and version 2 with reference year 2007 (Wood et al., 2015). While version 1 interlinks 43 countries with one rest of the world (RoW) region, version 2 interlinks 43 countries with 5 RoW regions: Asia and the Pacific, America, Europe, Africa and the Middle East. Further on, version 1 considers 129 product groups per country in its $P \times P$ form, while version 2 considers 200 product groups per country in its $P \times P$ form. This increase in detail is situated in the waste treatment sectors. Exiobase version 3 is currently under development and will cover a time period of 19 years, from 1995 till 2014 (Wood et al., 2015).

It must be remarked that the available version of Exiobase is based on the industry technology assumption, which is not in accordance with recommendations in IO guidelines (Eurostat, 2008) as well as LCA standards (ISO, 2006). To be in compliance with standards, the IO-model should in fact be based on the product-technology assumption (Suh et al., 2010). However, this does not imply that the used version of Exiobase is not suitable, as the different approaches all have their benefits, i.e. viewpoints on this matter can be questioned.

The first step was the calculation of inventory g, corresponding with the final household demand vector f, using equation (1). However, before this could be done, the IO-table needed to be altered. In Exiobase v.2, capital investments are not part of the IO-table, but of the final demand, and hence the contribution of capital goods would not be included in the impact results. Since this would lead to serious underestimations, capital investments had to be integrated in the IO-table. The calculations were based on the FORWAST report (Schmidt et al., 2010) and can be found in the Supporting Information.

The second step was the calculation of the environmental impacts, using equation (2). In this step, inventory g was multiplied with a characterisation vector q, associated with a certain impact category. For example, if the impact category is Global Warming, the characterization factors are expressed in kg CO_2-equivalent per unit of each flow in inventory g, e.g. 25 kg CO_2 equivalents per kg methane. For an inventory flow of 10 kg methane, the total impact is 250 kg CO_2 equivalents. Finally, impact vector h was obtained, consisting of 9600 (200 × 48) rows which represent the product groups (and services) in each country.

These product groups had to be classified into the five household activities defined by JRC (EC, 2012b): food, consumer goods, shelter, mobility, services. To do so, we used the FORWAST report (Schmidt, 2010). This report classifies the product groups in Exiobase into ten activities instead of five: clothing, communication, education, health care, housing, hygiene, leisure, meals, security and social care. For example, the product group 'dairy products' is classified into the activity 'meals'. These ten activities were distributed over the five main household activities as described in Table 2 and the Supporting Information.

2.4. Impact assessment

In the LCA-approach of JRC, the impact assessment follows the recommendations of the ILCD handbook (EC, 2010a,b), covering the impact categories Global Warming, Particulate Matter, Photochemical Ozone Formation, Acidification, Eutrophication Terrestrial, Eutrophication Freshwater, Eutrophication Marine, Human Toxicity cancer, Human Toxicity non-cancer, Ecotoxicity, Ozone Depletion, Ionizing Radiation, Water Depletion, Land Use and Resource Depletion.

The goal was to cover the same impact categories in the IO-approach. However, this was not possible for each impact category. The impact category Ionizing Radiation could not be included because none of the required elementary flows are covered in Exiobase. For the impact categories Human Toxicity and Ecotoxicity, elementary flows are insufficiently available to make, in our opinion, an adequate assessment of the impact category. The

Table 1
Composition of the basket-of-products in the LCA-approach (EC, 2012b).

Household activity	Product groups	Representative products
Food	Meat and seafood	Beef, pork, poultry
	Dairy products and eggs	Milk, butter, cheese
	Crop based products	Sugar, oils and fats
	Vegetables	Potatoes
	Fruits including tomatoes	Apples, oranges
	(Non)alcoholic beverages	Coffee, beer
Consumer goods	Clothing	Shoes, cotton shirt
	White goods	(Dish)washer, refrigerator
	Electronics	Laptop
Mobility	Private transport	Mid-class car
	Public transport	Train, bus, plane
Shelter	Single/two-family house	Single house
	Multi-family houses	Multi-family house
	High-rise buildings	High-rise building
Services	Bars and restaurants	(Omitted from study)
	Leisure activities	(Omitted from study)
	Education	(Omitted from study)
	Tourism	(Omitted from study)

Table 2
Distribution of activities over the five household activities.

Main activities	Distribution of activities in FORWAST report
Food	▪ Food from activity meals
Consumer goods	▪ Activity leisure ▪ Goods from activity meals (e.g. tableware) ▪ Energy and water use from activity meals ▪ Goods from activity hygiene (e.g. soap)
Mobility	▪ Activity communication, except for 'radio, television and communication equipment'
Shelter	▪ Activity housing ▪ Energy and water from activity hygiene (e.g. showering)
Services	▪ Activities education, security, health care and social care ▪ Services from other activities, except for public transport

elementary flows in Exiobase represent only 0.2–0.4% of the total list of elementary flows in the toxicity impact methods. But also in the LCA-approach, a remark was made concerning the toxicity impacts: their precision is relatively low compared to other impact methods.

For the impact category Abiotic Resource Depletion, the most dominant elementary flows are aggregated in one group, making a comprehensive impact assessment impossible. These are the flows for metals with a high depletion risk, which have high characterization factors compared to more common metals, minerals and fossil fuels. Germanium for example has a characterization factor of 19,500 kg Sb eq./kg, which is very high compared to the characterization factor of iron (1.66E−06 kg Sb eq./kg) or natural gas (3.73E−07 kg Sb eq./kg). However, in Exiobase, these dominant metal flows (40 in total) are aggregated in only one group ('other metals'). Hence, the impact of Abiotic Resource Depletion cannot be assessed properly in Exiobase.

In summary, the ILCD impact categories included in the IO-approach are: Global Warming, Particulate Matter, Photochemical Ozone Formation, Acidification, Eutrophication Terrestrial, Eutrophication Freshwater, Eutrophication Marine, Ozone Depletion, Water Depletion and Land Use.

The most widely known impact category in this list is of course Global Warming, characterizing the effect of emitted greenhouse gases. Therefore, the results of Global Warming are discussed thoroughly in Section 3.1. The results of the other impact categories are included in the Supporting Information and discussed shortly in Section 3.2.

From a historical perspective, most of these ILCD impact categories are focussed on impacts related to emissions. Only Land Use, Water Depletion and Abiotic Resource Depletion are considered regarding natural resource consumption. But due to the reasons mentioned above, Abiotic Resource Depletion could not be properly assessed in the IO-approach. This means our impact assessment would lack a good analysis of natural resource consumption.

Therefore, we selected an additional, more comprehensive, characterisation method to assess the impact of natural resource consumption: CEENE or the Cumulative Exergy Extraction from the Natural Environment (Dewulf et al., 2007; Swart et al., 2015). The CEENE method is recommended by Liao et al. (2012) as the most appropriate thermodynamics-based method for characterising resource consumption since it covers all resource types, including land occupation. The CEENE results are discussed in Section 3.1.

The CEENE-method is based on the exergy concept, which stands for the maximal amount of work that can be retrieved from a resource when bringing it into equilibrium with the natural environment (Dewulf et al., 2008). The result is usually expressed in megajoules of exergy (MJ_{ex}). To make the results easier to interpret, they can be converted into exergy-based tonnes of oil-equivalent. In analogy to energy-based tonnes of oil-equivalent, 1 exergy-based ton of oil-equivalents is defined as the amount of exergy contained in one tonne of crude oil: 45.7 MJ_{ex}. CEENE considers a very wide range of natural resource: fossil fuels, nuclear resources, metal ores, minerals, water resources, abiotic renewable resources, atmospheric resources and land resources. Land resources can basically be accounted for in two ways: by the content of the biomass harvested or by the area and time needed to produce the biomass (land occupation). To avoid double counting, one way of accounting has to be chosen. In the first version of CEENE (CEENE v.2007), all resources are accounted for by land occupation, using the photosynthetic solar exergy as a proxy (Dewulf et al., 2007). In the second version of CEENE (CEENE v.2013), resources from natural systems are accounted for by the exergy content of the biomass, e.g. wood from rain forests, while resources from human-made systems are accounted for by land occupation, e.g. agricultural crops (Alvarenga et al., 2013), using the deprived potential natural net primary production (NPP) as a proxy. The latter made it possible to establish spatially differentiated characterisation factors for land occupation. CEENE v.2007 and v.2013 have been operationalized to LCA with the Ecoinvent database (Alvarenga et al., 2013; Dewulf et al., 2007), while CEENE v.2013 has been operationalized to EEIOA with the Exiobase database v.1 (Huysman et al., 2014).

For this study, CEENE v.2013 had to be coupled with Exiobase v.2. This means that for each elementary flow in the inventory, a characterisation factor (here an exergy value) was determined. For example, the characterization factor of the elementary flow 'brown coal' is 10.3 MJ of exergy per kg. This coupling, together with a complete list of updated characterisation factors, is described in the Supporting Information.

3. Results and discussion

3.1. Global warming and resource consumption

The impact results of the IO-approach and the LCA-approach are presented in Fig. 1. Fig. 1a shows the results for the impact in terms of Global Warming, while Fig. 1b shows the results for the impact in terms of Resource Consumption (CEENE). Fig. 1c presents again the results for Resource Consumption, but this time in a resource-contribution profile, illustrating how much each natural resource type contributes to the total impact. However, in the LCA-approach, land resources are not included due to lack of characterisation factors in the ILCD methods (EC, 2012b). To allow a better comparison between the IO-results and LCA-results, the IO-results are presented twice: once including land resources and once excluding land resources. From Fig. 1c, it can be noticed that land resources are only highly significant (75%) for the activity food, in which they

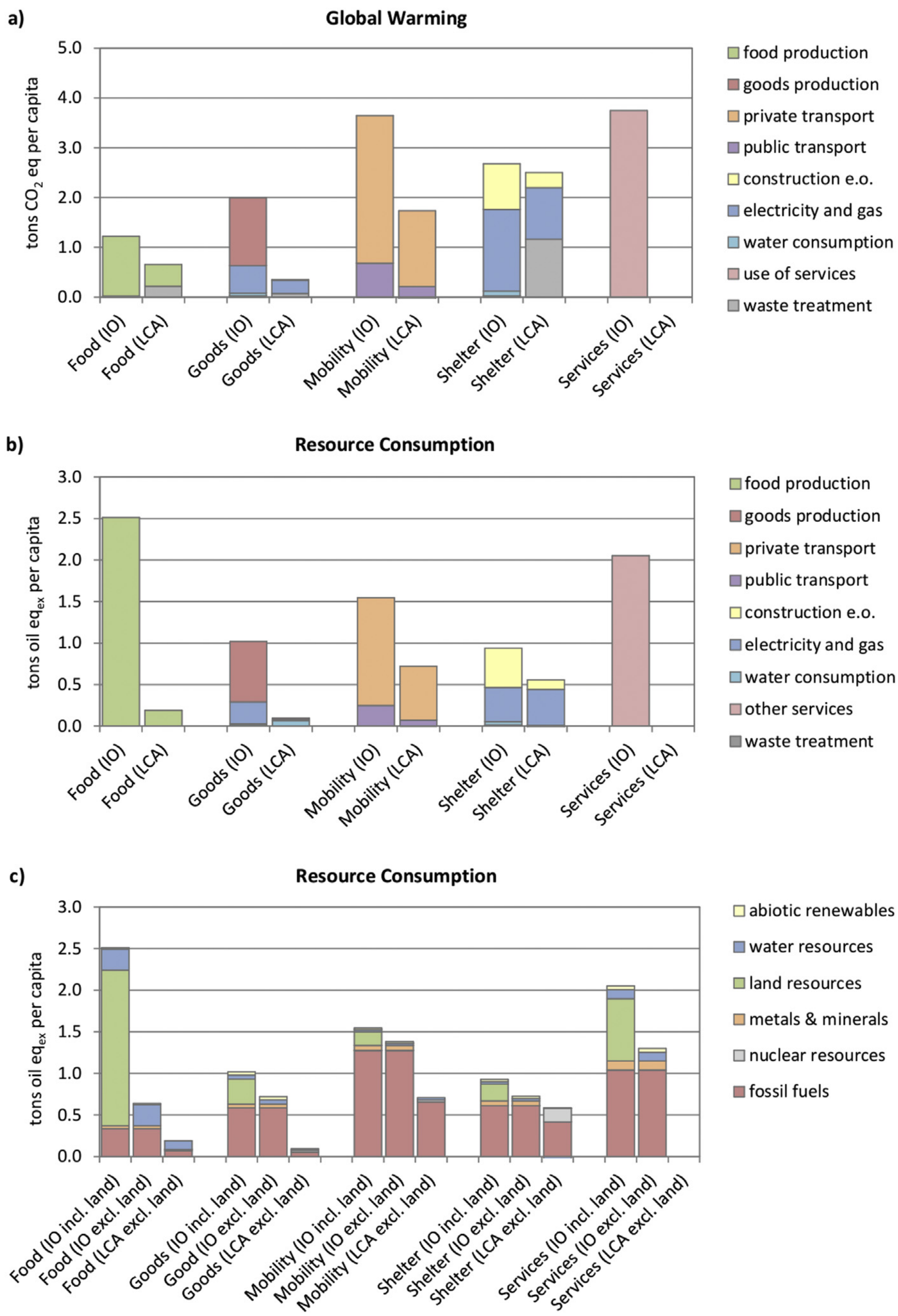

Fig. 1. IO-results and LCA-results of the consumption of an EU citizen in the year 2007. Impacts are expressed in terms of (a) Global Warming, in tons of CO_2 equivalents per capita (tons CO_2 eq) and (b, c) Resource Consumption, in exergy-based tons of oil-equivalents per capita (tons oil eq_{ex}). In (b) and (c), the IO-results are presented inclusive and exclusive land resources. Goods = Consumer Goods. To be in accordance with the study of JRC (EC, 2012b), the activity services include all services except public transportation, which is part of the activity mobility.

are needed for agriculture and livestock. There is also a small contribution of land resources to the activity consumer goods. This is mainly due to extracted wood, e.g. for the production of paper and wooden furniture.

These figures illustrate that the IO-results are higher than the LCA-results, for both Global Warming and Resource Consumption. The first reason for this difference in results is of course the difference in methodology: EEIOA versus LCA. As mentioned in the introduction, EEIOA does not have to deal with the mentioned cut-off problem. Therefore, IO-results are generally higher than LCA-results. The second reason is the basket-of-products selection for each activity. The IO-approach gives higher impact results, because it considers a much broader product range for each activity than the LCA-approach. This is mainly because no upscaling was applied in the LCA-study (EC, 2012b), meaning that the impacts reflect the representative products only, see Section 2.2.

For the activity food, the IO-results are presented more detailed in Fig. 2a and b. When considering for example Global Warming, the largest impact contribution comes from the group 'other products' (39%, a.o. rice, cereals, bread, pasta, preserved food), followed by meat (24%), vegetables and fruits (12%), dairy products (11%), fish products (6%), beverages (6%) and sugar, fats, vegetable oils (2%). These results can be compared with those presented by Schmidt and Merciai (2014), who calculated the impact on Global Warming of the world food consumption in 2007. To do so, they used a hybrid version of Exiobase v.2, integrating the economic IO-model with mass flows analysis. In their results, the largest impact contribution comes from meat (40%), followed by dairy products (19%), other products (17%), vegetables and fruits (10%), beverages (7%), fish products (5%) and sugar, fat and vegetable oils (2%). The main reason for this shift in contribution is that diets from all over the world are included, instead of only from the EU27. Furthermore, Schmidt and Merciai (2014) includes the contribution from indirect land use changes, and also, their work is based on a hybrid form of Exiobase v.2, while our results are calculated with the economic model.

For the activity consumer goods, the IO-results are presented more detailed in Fig. 2c and d. For both Global Warming and Resource Consumption, the largest impact contribution comes from wearing apparel (e.g. clothing, shoes), followed by furniture and other manufactured goods (e.g. toys, sport goods).

Furthermore, for the activity mobility, the IO-approach considers a broader range of private and public transport modes. For example, public transport includes not only trains, buses and planes (see), but also sea transportation, inland water transportation and other land transportation services, e.g. taxis.

Regarding the activity shelter, a remark needs to be made concerning the Global Warming impact. For 'construction' (which also includes refurbishment and maintenance of the house), and 'electricity and gas', the IO-results are higher than the LCA-results, which can be explained by two reasons mentioned above. However, for 'waste treatment' (e.g. water from flushing the toilet, cleaning), the LCA-result is higher than the IO-result. Hence, the impact of wastewater treatment may be underestimated in the IO-

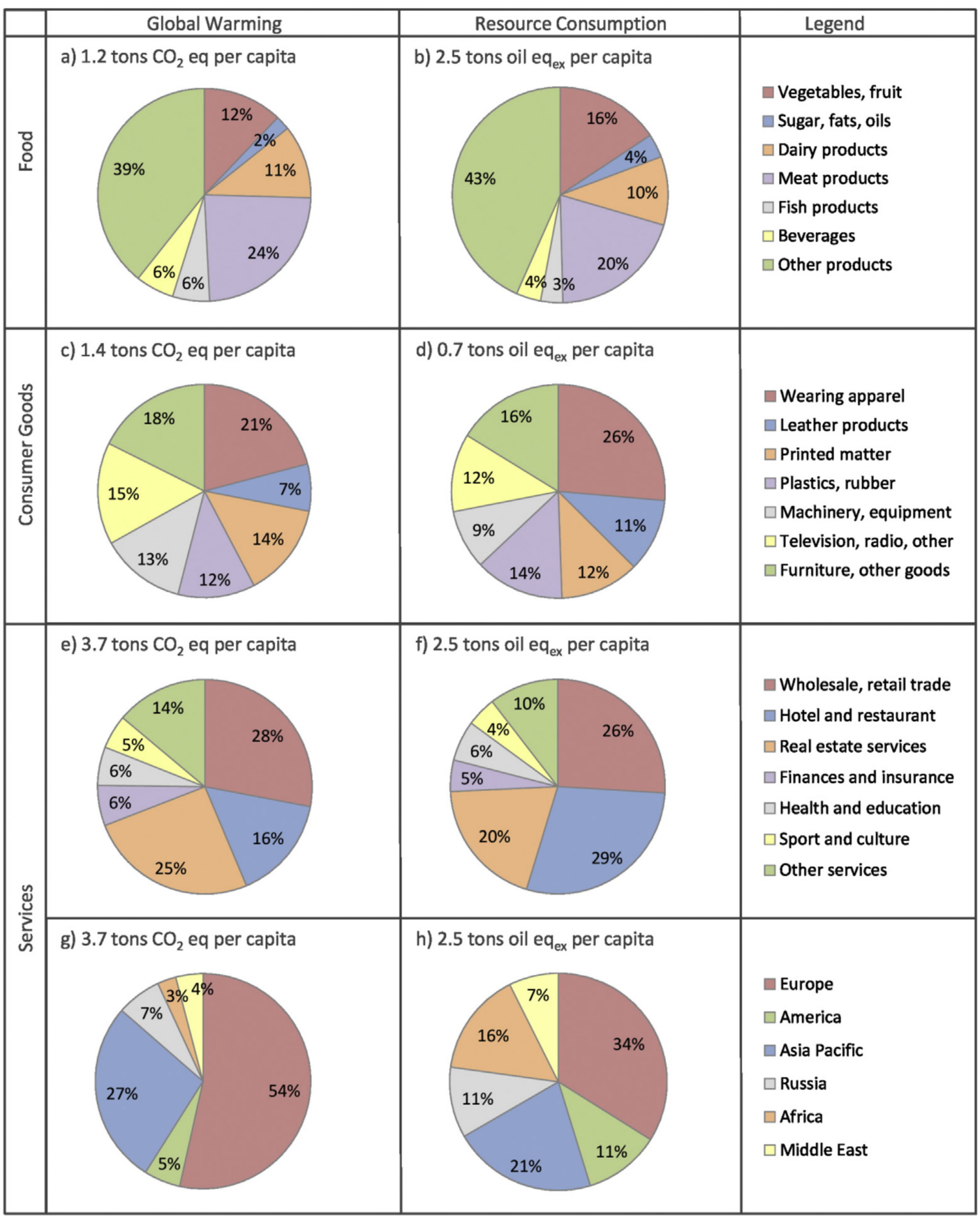

Fig. 2. IO-results of the consumption of an EU citizen in the year 2007. Contribution to total impact in terms of Global Warming, in tons of CO_2 equivalents per capita (tons CO_2 eq), or Resource Consumption, in exergy-based tons of oil-equivalents per capita (tons oil eq_{ex}), is presented per product group (a, b, c, d, e, f) and per geographical location (g,h).

approach or overestimated in the LCA-approach. When making a comparison with other studies (Tukker et al., 2006; Weidema et al., 2005), it seems that an overestimation in the LCA-results is more likely.

For the activity services, no LCA-results were available in the study of JRC due to lack of data (EC, 2012b). Hence, a comparison with the IO-results is not possible. These IO-results are presented more detailed in Fig. 2e and f. Fig. 2e shows that the largest contribution to Global Warming comes from wholesale and retail trade services (28%), followed by real estate services (25%) and hotel and restaurant services (16%). Fig. 2f illustrates that the largest contribution to Resource Consumption comes from hotel and restaurant services (29%), followed by wholesale and retail trade (26%) and real estate services (20%). Both wholesale/retail trade services and real estate services involve a lot of transport, which results in a high consumption of fossil fuels (more than 50%) and thus a high Global Warming impact. For Resource Consumption, hotel and restaurant services have an even higher impact contribution due to their high consumption of land resources needed for food production. If the alternative approach would be used (see Section 2.4), these high-impact services would be distributed over the other household activities. For example, real estate services would be assigned to the activity shelter, while hotel and restaurant services would be assigned to the activity food. As a consequence, the global warming impact would increase for the activities food (+43%) and shelter (+35%), while it would decrease for the activity services (−40%). Similar, the resource consumption impact would increase for the activities food (+21%) and shelter (+42%) and decrease for the activity services (−47%).

The impact contributions of the IO-results are also presented by their geographical location in Fig. 2g and h. This way of presenting is unique for our IO-approach, since it is based on a world input–output database. Fig. 2g indicates that the largest impact contribution to Global Warming is situated in Europe (54%), since the services are also consumed in the EU. However, Fig. 2h shows that the impact contribution to Resource Consumption is more globally spread: 34% of resources is extracted in Europe, 11% in America, 21% in Asia-Pacific, 11% in Russia, 16% in Africa and the 7% in the Middle-East. This is intuitively logical, since extraction of natural resources like metals, minerals and fossil fuels, is mostly situated outside of Europe.

As mentioned in the beginning of the paragraph, the main reason for the difference between the IO-results and LCA-results is the higher completeness of the IO-approach (no cut-offs, broader range of products, services are included). Moreover, there is an additional reason for the difference between both results: the reference year and country of origin of the datasets. In the IO-approach, data is available for 48 different countries, all having 2007 as reference year. To calculate the impact of a European citizen, we used the data of the EU27 countries. In the LCA-approach, LCI-data are multiplied with macro-economic consumption data of the EU27. These macro-economic consumption data have 2006 as reference year, but the LCI-data are based on different years and different countries of origin. For example, the dataset for potatoes is associated with production in Germany in 2010, while the dataset for apples is associated with production in China in 2009. However, we expect the influence of this additional reason to be much lower.

Of course, the IO-results also have their limitations: they are determined by the way product groups are selected and classified into five household activities, as explained in Section 2.3. In the classification chosen for this study, all the services are assigned to the activity services (except for public transportation), to allow a good comparison with the LCA-approach of JRC. Alternatively, it would also be possible to apply the classification of the FORWAST report (Schmidt, 2010). For example, they assign 'real estate services' to housing (shelter), and 'hotel and restaurant services' to meals (food). Using this alternative approach, the final result will be somewhat different.

3.2. Other impact categories

As mentioned in Section 2.4, the results for the ILCD impact categories Particulate Matter, Photochemical Ozone Formation, Freshwater Eutrophication, Marine Eutrophication, Ozone Depletion, Water Depletion and Land Use are given in the Supporting Information.

When comparing the IO-results to the LCA-results, an overall observation is the notable shift in the ranking of activities. Consider for example Particulate Matter. In the LCA-results, the hotspot activity is shelter, followed by food, mobility and consumer goods. In the IO-results on the other hand, the hotspot activity is mobility, followed by services, consumer goods, shelter and food. The main reason is again the higher completeness of the IO-results (no cut-offs, broader range of products, services are included), see Section 3.1. This means that the influence of using a different approach should not be underestimated.

Specific for Acidification and Terrestrial Eutrophication, some remarks need to be made. In the LCA-results, the hotspot activity is food, followed by shelter, mobility and goods. The large impact of food is mainly caused by ammonia emissions from meat and dairy production. The impacts of shelter and mobility are caused by NO_x and SO_x emissions from cars and energy use. In the IO-results, the hotspot activities are mobility and services, followed by shelter, consumer goods and food. The low impact of food appears to be caused by a underestimation of ammonia emissions from meat and dairy products in Exiobase. Meat and dairy products contribute only 0.3% (Acidification) and 0.6% (Terrestrial Eutrophication) to the total impact of the activity food, compared to 83% and 86% in the LCA-results. A comparison with the study of Weidema et al. (2008), which reports that meat and dairy products contribute 25% (Acidification) and 30% (Terrestrial Eutrophication) to the total impact of EU27 consumption in the year 2000, indicates that ammonia emissions are indeed underestimated in the Exiobase database.

4. Conclusions

In this article, we calculated the environmental impacts of the consumption of an average EU citizen through a top-down IO-approach. The results were compared with an earlier study of JRC, in which a bottom-up LCA-approach was used. Yet, as mentioned in the introduction, the study of JRC was a pilot project, meaning that revision and improvement is ongoing. This must be taken into account when interpreting the results.

The goal was to obtain IO-results for all the ILCD recommended impact categories, as done in the LCA-approach. However, it was not possible to make an adequate, comprehensive impact assessment regarding (1) Ionizing Radiation, because none of the required elementary flows are present in Exiobase; (2) toxicity impacts, as the elementary flows in Exiobase represent less than 0.5% of required flows; and (3) Abiotic Resource Depletion, since the most dominant elementary flows are aggregated in one group.

In the article, the focus is on Global Warming, as this is the most widely applied impact category. The results of other impact categories are provided in the Supporting Information. However, since Abiotic Resource Depletion could not be included, the study would lack a good assessment of natural resource extraction. Therefore, the CEENE impact method was used, which accounts for a wide range of natural resources: fossil fuels, nuclear resources, metals, minerals, abiotic renewables, water and land resources.

The main difference between our study and other studies that apply the IO-approach, is the impact assessment. For example, the similar studies mentioned in the introduction do not perform an impact assessment based on ILCD handbook. Mostly, they stay at inventory level by calculating the material footprint (in kg metals, minerals, fossil fuels and biomass), the water footprint (in m^3 blue water) or the ecological footprint (in global hectares). The only actual impact is the carbon footprint (in kg CO_2-eq.), which is assumed to be equal to Global Warming. Moreover, our study is the first to apply the CEENE method in an IO-approach, making it possible to account for the extraction of natural resources in a consistent way. Further, our paper includes renewing figures on the geographical distribution of environmental impacts, illustrating from which countries resources are extracted, or in which countries pollutants are emitted. Such figures provide more information about socio-economic consequences, as the burden of EU consumption is often shifted to other countries.

When comparing the IO-results with the LCA-results, one of the main observations is that there are large shifts in the ranking of the consumption activities, caused by the higher completeness of the IO-approach (no cut-offs, inclusion of a broader range of products, inclusion of services). Consider for example Global Warming: in the LCA-results, the hotspot activity is shelter, followed by mobility, food and goods. In the IO-results, the hotspot activities are mobility and services, followed by shelter, goods and food. This means that the influence of using a different approach on the final results is large and should not be underestimated, which may be relevant input to policy support. They use the results to monitor the environmental impacts associated with the consumption behaviour of EU citizens, but also to develop policies that will reduce this environmental impact. For example, when looking at the IO-results for Global Warming, the two hotspot activities are mobility and services. Their high impact is mainly caused by the consumption of fossil fuels, as can be derived from the Resource Consumption results. For mobility, this is primarily due to private transport. Also for services, this is mostly due to transport (e.g. commercial transport related to wholesale and retail trade services). This example illustrates that transport is still a major sustainability issue that policies need to monitor, e.g. by improving public transportation. If for example, hypothetically, 50% of the private transport is replaced by train transport, the Global Warming impact will decrease almost 30%.

Overall, we can conclude that the IO-approach (based on Exiobase) is very suitable for this type of studies. The main limitation is that not every ILCD impact category can be calculated. However, there are also precision issues with toxicity impacts in the LCA-approach, and the CEENE method can be used as a substitute for Abiotic Resource Depletion. Nonetheless, the CEENE method cannot assess the actual depletion of metals. To be able to calculate the impact of metal depletion with IO-databases, the elementary metal flows and associated mining sectors need to be disaggregated, which is a difficult issue that will probably not be solved in the near future. A possible preliminary solution to include the impact of metal depletion is a hybrid analysis.

Associated content

Supporting Information with additional information on the countries and regions in Exiobase (section A), inclusion of capital investments (section B), selection of the basket-of-products (section C), CEENE impact assessment method (section D) and ILCD recommended impact assessment methods (section E).

Notes and disclaimers

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Appendix A. Supplementary material

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.jclepro.2016.02.098.

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