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Environmental impacts of food waste: Learnings and challenges from a case study on UK

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ABSTRACT

Food waste, particularly when avoidable, incurs loss of resources and considerable environmental impacts due to the multiple processes involved in the life cycle. This study applies a bottom-up life cycle assessment method to quantify the environmental impacts of the avoidable food waste generated by four sectors of the food supply chain in United Kingdom, namely processing, wholesale and retail, food service, and households. The impacts were quantified for ten environmental impact categories, from Global Warming to Water Depletion, including indirect land use change impacts due to demand for land. The Global Warming impact of the avoidable food waste was quantified between 2000 and 3600 kg CO_2-eq. t^{-1}. The range reflected the different compositions of the waste in each sector. Prominent contributors to the impact, across all the environmental categories assessed, were land use changes and food production. Food preparation, for households and food service sectors, also provided an important contribution to the Global Warming impacts, while waste management partly mitigated the overall impacts by incurring significant savings when landfilling was replaced with anaerobic digestion and incineration. To further improve these results, it is recommended to focus future efforts on providing improved data regarding the breakdown of specific food products within the mixed waste, indirect land use change effects, and the share of food waste undergoing cooking. Learning from this and previous studies, we highlight the challenges related to modelling and methodological choices. Particularly, food production datasets should be chosen and used carefully, to avoid double counting and overestimation of the final impacts.

1. Introduction

About one third of the food produced globally is lost or wasted corresponding to an annual generation of roughly 1.3 billion tonne of food waste (Gustavsson et al., 2011). In Europe this figure is estimated to about 88 Mt corresponding to ca. 173 kg per capita (Stenmarck et al., 2016; data for EU28 as for 2012); in economic terms, this incurs a loss of 143 billion€ each year. Food waste is often distinguished between unavoidable and avoidable, the latter intended as the food (and eventually drinks) which at some point, prior to being thrown out, was edible (Quested and Johnson, 2009). The avoidable portion represents a waste of resources, as food demands land-use, energy, chemicals and materials in order to be produced and delivered to the different actors involved in the food supply chain. Such a loss of resources inevitably translates into considerable environmental impacts that ideally may be avoided by prevention or mitigated by enforcing best waste management practices.

A number of studies have assessed the impact of food waste using life cycle thinking approaches. Typically, there are two main methods to perform this assessment: applying top-down approaches, using for example input-output tables and related figures for the impacts, or bottom-up approaches, using more detailed products databases. Advantages and disadvantages of the two methods have been discussed elsewhere (Reutter et al., 2017). The same authors, applying environmentally-extended

Abbreviations: AC, acidification; AEN, aquatic eutrophication, nitrogen; AEP, aquatic eutrophication, phosphorous; ARD, abiotic resource depletion; dLUC, direct land use change; EF, ecological footprint; ET, Ecotoxicity; EU, Europe; EU28, Europe, 28 member states; FRD, fossil resource depletion; LCA, life cycle assessment; LUC, land use change; HTc, human toxicity, cancer; iLUC, indirect land use change; GW, global warming; MSW, municipal solid waste; OD, ozone depletion; PM, particulate matter; POF, photochemical ozone formation; RD, fossil resource depletion; SI, supporting information; TE, terrestrial eutrophication; UK, United Kingdom; WD, water depletion.

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input-output analysis, found that Australian food waste represents 9% of the total water use and 6% of greenhouse gas (GHG) emissions at the national level amounting to a total of 57,507 Gg CO_2-eq. annually. Applying a top-down approach and using global statistics from FAO (Food and Agriculture Organization of the United Nations), Kummu et al. (2012) concluded that food waste embeds ca. 23–24% of the total use of cropland, freshwater resource, and fertilizers for food production. Song et al. (2015) combined national statistics and surveys on consumption patterns with bottom-up life cycle inventories for food products to derive carbon, water and ecological footprints of household food waste in China, estimating an impact of 2500 kg CO_2-eq. t^{-1}. Other studies used instead a bottom-up LCA approach. For example, Bernstad and Andersson (2015), using a bottom-up LCA approach integrated with data derived from dedicated sampling campaigns, concluded that the impact of the avoidable food waste generated by Swedish households ranged between 800 and 1400 kg CO_2-eq. t^{-1}. Oldfield et al. (2016) quantified the impact of food waste in the Irish food supply chain to ca. 5600 kg CO_2-eq t^{-1}. Scholz et al. (2015) used a bottom-up LCA approach to quantify the average carbon footprint of the food waste generated by a supermarket chain in Sweden, estimating it to 1600 kg CO_2-eq. t^{-1}. A similar study was also performed by Brancoli et al. (2017) that quantified an impact of 2800–3100 kg CO_2-eq. t^{-1} depending on the waste management scenarios. Martinez-Sanchez et al. (2016), focusing on the indirect effects of prevention for the case of Denmark, estimated the impact of food waste from Danish households to ca. 1200 kg CO_2-eq. t^{-1}, again using bottom-up LCA. Chapagain and James (2011) calculated carbon and water footprint of the total and avoidable food waste generated by UK households, estimating that these correspond to 6% and 3% of the total water and C-footprint of the UK.

It should be noticed that all the above mentioned studies, except for Martinez-Sanchez et al. (2016) and Chapagain and James (2011), did not include a thororugh quantification of the environmental impacts associated with land use changes (LUCs) induced by the cultivation of food, later becoming waste. In LCA the LUC impacts are typically distinct into direct and indirect (dLUC/iLUC). While the first refers to a change in the use of the land, the second refers to the upstream consequences of demanding land regardless of the final use of it and reflects market-mediated effects occurring globally, beyond the border of the region under assessment (Schmidt et al., 2015). Accounting for these, when addressing biomass resources incurring a demand for land, is crucial to the LCA results as learned from the extensive literature and discussion on bioenergy/biofuels. This is particularly true for carbon-footprint results, typically worsen when iLUC impacts are included (e.g. Edwards et al., 2010; Hamelin et al., 2014; Searchinger, 2010, 2008; Tonini et al., 2016a, 2017; Wenzel et al., 2014). In addition to this, the majority of the studies only addressed one or a few impact categories (e.g. carbon and water footprint) and one waste generator (or sector of the supply chain), mainly households or wholesale/retail sectors as earlier mentioned. Further, no study, the authors are aware of, has so far attempted to address and identify the main source of uncertainties in the life cycle assessment of food waste, using state-of-the-art approaches. Last, when modelling land use changes and waste management system, methodological challenges related to possible double countings and other modelling issues arise. This may be due to the way life cycle datasets are provided, for example the emissions or processes included. No study, the authors are aware of, has so far attempted to identify and discuss these issues.

Keeping in mind these limitations and in the attempt to bridge the gap we find in the current status of the research, this study aims to: (i) quantify the environmental impacts of food waste generated by different sectors of the food supply chain, using UK as case study; (ii) identify the main contributors to the impacts within the supply chain; (iii) determine the main source of uncertainties and the need for further research efforts on data collection to improve the robustness of the results. In addition, based on the learnings from this and previous research, this study also attempts to highlight and discuss some of the main challenges arising when performing this type of studies. The focus is placed on the modelling of the waste composition, land use changes, waste management, and on the most important modelling parameters and scenario uncertainties.

2. Materials and method

2.1. Definitions

We followed the definitions given in the recent FUSIONS study (Östergren et al., 2014); accordingly, *food waste* is intended as the fraction of food and inedible parts of food, removed from the food supply chain to be recovered or disposed (including composted, crops ploughed in/not harvested, anaerobic digestion, bioenergy production, cogeneration, incineration, disposal to sewer, landfill or discarded to sea). This excludes (from being considered *food waste*) the fraction of food and inedible parts of food that is used for animal feeding or for production of biomaterials. Notice that *food waste* is different from *food losses* defined as un-harvested crops (left on-field), losses of livestock pre-slaughter (dead during breeding or dead during transport to slaughter) or losses of milk due to mastitis and cow sickness (Östergren et al., 2014). Similar definitions may be found in other studies (Gustavsson et al., 2011; Östergren et al., 2014; Stancu et al., 2015). Both *food losses* and *food waste* refer to food items intended for human consumption and include both avoidable and unavoidable waste. The *avoidable* food waste is here intended as the food (and eventually drinks) which at some point, prior to being thrown out, was edible conforming with Quested and Johnson (2009).

2.2. Scope and functional unit

The functional unit of the study is the life cycle (cradle-to-grave, i.e. from provision to waste handling) of one tonne of avoidable food waste generated by four individual sectors of the United Kingdom food supply chain, which are: (I) Processing, (II) Wholesale & Retail, (III) Food Service, and (IV) Households. From now onwards, this naming (with capitals) will be used to refer to each of these waste generators and to the associated scenario. Food waste at farming sector was not addressed due to lack of reliable data, as also stressed in WRAP (2017). The food waste generated at these four stages of the food supply chain differs both in terms of composition and also in terms of supply chain activities and waste management practices involved. The assessment encompasses the entire life cycle of the avoidable food waste from production of the food (then becoming waste) and associated land use changes, to distribution (production of the packaging, transport and store operations), eventual meal preparation, up to final waste treatment, recycling, and eventual disposal (including end-of-life of the packaging). It should be noted that, differently from other waste management LCA studies typically disregarding upstream activities prior to waste generation, all activities prior to generation of the waste were included in order to quantify the actual life cycle environmental impact of the avoidable food waste generated. The assessment was performed following the ISO standards for LCA (ISO, 2006a, 2006b). A consequential approach was applied (Weidema et al., 2009; Weidema, 2003). The geographic scope of the study is United Kingdom, i.e. the foreground inventory data for food waste composition, technologies, and the legislative context were as much as possible specific to UK conditions. Most data for waste and technologies referred to the period 2010–2015. Background life cycle impact assessment (LCIA) data were obtained from the ecoinvent v3.3 database, consequential system version (Wernet et al., 2016). Impacts associated with capital goods were included accordingly. The impact assessment was performed for ten selected environmental impact categories, namely: Global Warming (100-year time horizon; Forster et al., 2007), Terrestrial Acidification (Seppälä et al., 2006), Photochemical Ozone Formation and Particulate Matter (van Zelm et al., 2008), Aquatic Eutrophication Nitrogen and Phosphorous (Struijs et al., 2009), Human Toxicity cancer and Ecotoxicity (Rosenbaum et al., 2011), Fossil Resource Depletion (van Oers et al., 2002) and Water Depletion (Goedkoop et al., 2009). Notice that, with respect to Global Warming, the uptake/release of CO_2 biogenic from the food was assigned a characterization factor equal to 0, while the eventually sequestered CO_2 biogenic (within the 100-year time horizon considered) was assigned a factor equal to -1, following common practice for short-live biomass. The assessment was facilitated with the LCA-tool EASETECH (Clavreul et al., 2012).

2.3. Description of the scenarios and system boundary

We assessed four scenarios corresponding to the life cycle of the avoidable food waste generated by four sectors of the food supply chain (Processing, Wholesale & Retail, Food Service, and Households). Scenario I (Processing; Fig. 1a) included indirect land use changes, farming, processing, waste management, and transport (when applicable). Scenario II (Wholesale & Retails; Fig. 1b) included indirect land use changes, farming, processing, distribution (transport and store operations), and waste management. Scenarios III/IV (Food Service and Households; Fig. 1c and d) included indirect land use changes, farming, processing, distribution (transport and store operations), meal preparation, and waste management. It should be noticed that any food removed from the food supply chain and used for production of biomaterials/biochemicals and for animal feeding is not included in this assessment as it is not considered food waste in accordance with Östergren et al. (2014). The waste management, for each sector of the food supply chain, was modelled on the basis of the information provided in WRAP (2017) following current practices in UK; see Fig. 1. Since the report does not detail the partition of the biological treatment between anaerobic digestion and composting, this was further estimated to 50% on the basis of the information provided in NNFCC (2014); see calculations in Section 2 of the Supporting Information (SI; Table S18). Additionally, in the case of Households sector, it was assumed that all the avoidable food waste is sent to end-of-life treatment or to sewer (i.e. pet feeding was not considered; see Table S18, SI) to conform to the definition of food waste (Östergren et al., 2014; see Section 2.1 of this manuscript). With respect to the packaging waste generated along with the avoidable food waste, a share equal to 59.2% was sent for recycling, 4.9% incinerated, and 35.9% landfilled conformingly with current practice (DEFRA, 2016).

As common practice in waste LCAs, any co-products delivered by the waste management system, along with the primary service (treatment of the waste), were assumed to displace corresponding similar products in the market. These were identified in the marginal products (or technologies), i.e. those unconstrained products (or technologies) that would be affected by a change in demand. For electricity, the marginal mix for UK consisted of about 51% hard coal, 21% nuclear, 12% natural gas, and 15% hydro/wind/biogas power (Wernet et al., 2016; consequential system version, UK electricity). Using the same approach, the marginal heat was modelled as a mix of natural gas (82%), oil (15%), and coal (3%) using the average of 2011–2014 data series (IEA, 2017). Residual organics after biological treatment (i.e. compost and digestate) were assumed to displace urea, diammonium phosphate, and potassium chloride based on the content of N, P, and K conforming with previous investigations (Tonini et al., 2016b). While for P and K the substitution was assumed to be 100%, for N a substitution factor of 20% for compost and 40% for digestate was assumed conformingly with the figures provided by Boldrin et al. (2009) and Hansen et al. (2006), respectively. Similarly, recycled polyethylene, polypropylene, and paper material was assumed to displace corresponding virgin production. Bottom ash from incineration was assumed to be disposed of in mineral waste landfill following the approach of Manfredi and Christensen (2009). Fly ashes were assumed to be utilized for backfilling of old salt mines conformingly with the approach of Fruergaard et al. (2010). The inventory data are detailed in Section 2.4. The datasets used to model the marginal technologies/products may be found in Table 3.

2.4. Inventory data

2.4.1. Fractional composition of the avoidable food waste

The data on avoidable food waste for United Kingdom were taken from the information provided in WRAP (2016) for Processing and Wholesale & Retail sector, in WRAP (2013) for Food Service sector and in Quested and Johnson (2012) for Households. These reports provided data for avoidable food waste aggregated as food categories (e.g. meat, fruits and vegetables, bakery products, etc.). These were used to derive the food categories share. To derive the total annual generation (per aggregated food category), these shares were then multiplied by the (updated) total amount of avoidable food waste generated per sector, as reported in WRAP (2017). This information is thoroughly detailed in Tables S1–S9 of the SI. At this stage, some simplified assumptions needed to be taken to match the food waste categories reported in the reports from WRAP with the categories considered in this study (for details, see Table S1-S9, SI): (i) Meat & Meat Products, (ii) Milk & Dairy Products, (iii) Fish & Fish Products, (iv) Bakery & Dry Products, (v) Fruit & Vegetable, (vi) Drinks, and (vii) Other. At this point, in order to apply a bottom-up life cycle approach (and associated datasets), a further breakdown of these categories into the specific food products was necessary. To this purpose, the production and consumption patterns for United Kingdom were used on the basis of the statistics provided by UK Government (2017) for the period 2011–2014 (the average of the period was used). The share of each food product (e.g. chicken, beef, pork, white bread, tomato, etc.) in the mixed waste was calculated using the production (for Processing) and the consumption (for the remaining sectors) patterns. With respect to this, Table S10 (SI) details these production and consumption patterns, as retrieved and elaborated from UK Government (2017) for this study. An example of how the breakdown of food products was calculated is shown in the SI (Section 1.6 of the SI). Table S11 illustrates the final breakdown of the food products (shares over 100%), without considering the packaging materials. The amount and type of packaging material for each food product was then quantified on the basis of literature information (Table S12, SI). Once this was done, the information regarding the avoidable food waste fractional composition and the associated packaging was combined and presented in Table 1. This represents the input to the LCA (i.e. 1 t avoidable food waste plus associated packaging; see Fig. 1). Notice that for the category Drinks, all drink products were assumed as orange juice due to lack of waste composition data.

2.4.2. Chemical/physical/biochemical/nutritional properties of the food products

A database of the chemical/physical/biochemical/nutritional properties of the individual food products and packaging materials listed in Table 1 was established on the basis of the Danish food

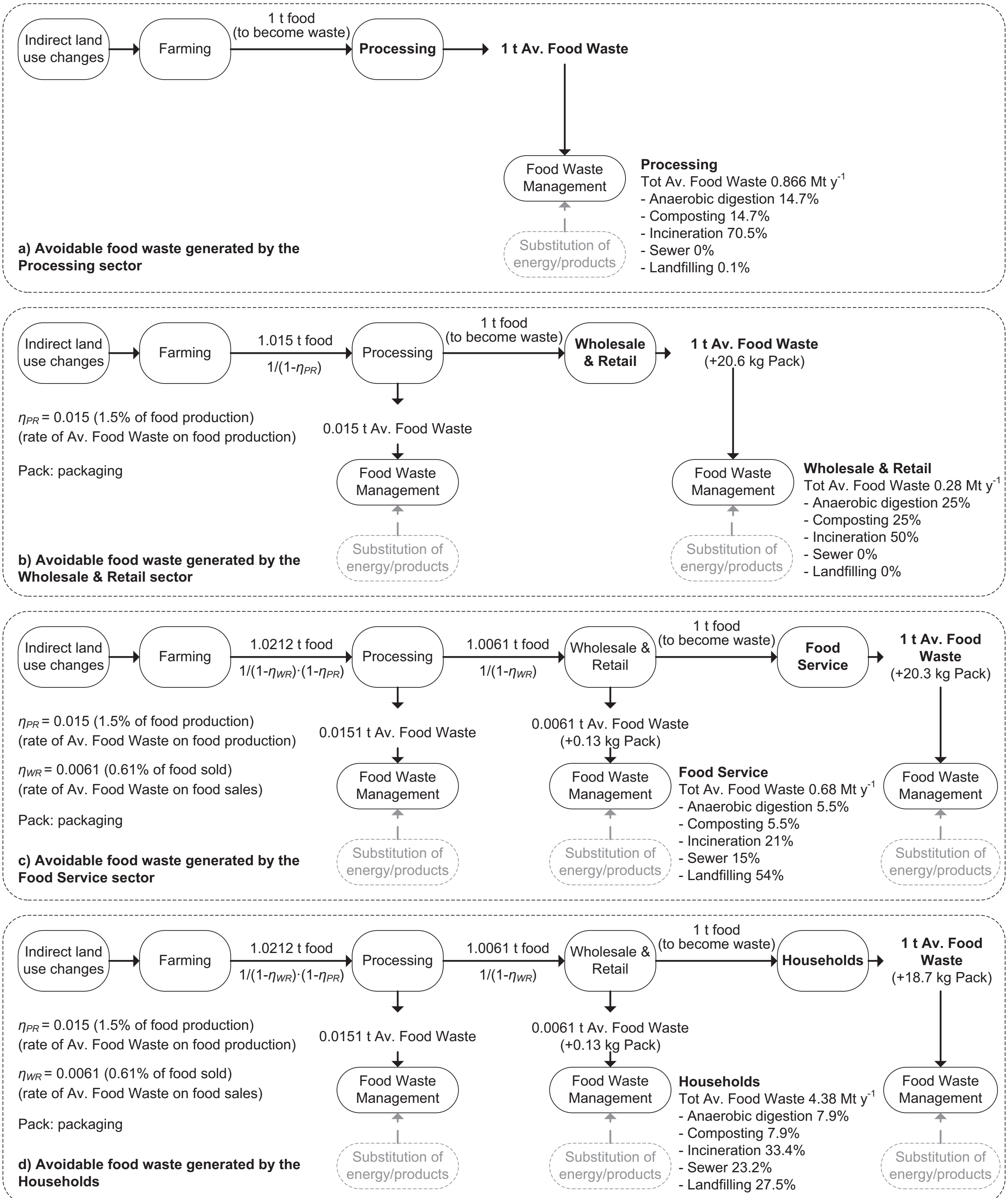

Fig. 1. System boundary for the assessment of the life cycle impacts of one tonne of avoidable food waste generated at: (a) Processing, (b) Wholesale & Retail, (c) Food Service, and (d) Households (the two latter share the same boundaries). Induced flows are illustrated with black-continuous lines. Avoided flows are illustrated with grey-dotted lines. Relevant information about the current food waste management practices at each sector is also provided on the basis of the information reported in WRAP (2017). With respect to the management of packaging waste: 59.2% was sent for recycling, 4.9% incinerated and 35.9% landfilled conformingly with current practice (DEFRA, 2016).

product database (DTU National Food Institute, 2017). The database provides more than 900 individual food products compositions. For the purpose of the study, average figures were calculated per type of food product reported in Table 1. For example, an average composition for cheese, pork, beef, milk, etc. was calculated. The theoretical methane potential was quantified using the Buswell formula (Symons and Buswell, 1933). The excel-file of the dataset elaborated for this study, containing a number of about

Table 1
Breakdown of the fractional composition (kg t^{-1}) of the avoidable food waste generated by Processing, Wholesale & Retail, Food Service, and Households sector. The breakdown of the associated packaging (kg packaging generated per one tonne of avoidable food waste) is also provided (except for Processing sector, for which no packaging was assumed).

Food category	Food product	Processing	Wholesale & Retail	Food Service	Households
Meat & Meat Products	Pork	43.2	12.1	20.2	14.2
	Beef	47.5	22.9	38.3	26.9
	Chicken	88.5	54.1	90.6	63.5
	Lamb/Mutton	16.2	8.0	13.4	9.4
Milk & Dairy Products	Milk	145.8	107.6	0.0	80.5
	Cheese	75.3	8.1	0.0	6.1
	Butter	5.0	2.9	0.0	2.2
	Yoghurt	4.8	14.4	0.0	10.7
Fish & Fish Products	Fish	24.0	7.4	10.9	9.2
Bakery & Dry Products	White bread Similar non-brown bread	51.5	85.7	66.4	48.1
	Brown, Whole meal, and Other bread	7.6	31.2	24.2	17.5
	Cakes, Biscuits, Other Bakery Products	12.2	60.8	47.1	34.1
	Cereals & Cereals Products	24.8	86.5	67.1	48.6
	Rice	8.7	20.6	16.0	11.6
	Flour	7.0	12.4	9.6	6.9
	Sugar	4.8	23.5	18.2	13.2
Fruit & Vegetable	Fresh oranges	0.0	9.4	14.3	10.1
	Other fresh citrus fruits	0.0	14.9	22.5	15.9
	Fresh apples	9.3	27.1	40.9	28.8
	Fresh pears	1.1	7.6	11.5	8.1
	Fresh grapes	0.0	10.4	15.7	11.1
	Fresh bananas	0.0	42.8	64.7	45.6
	Fresh melons	0.0	6.6	9.9	7.0
	Fresh cabbages	9.2	6.4	9.6	6.8
	Fresh cauliflower	3.7	11.3	17.1	12.1
	Leafy salads fresh	5.2	10.0	15.2	10.7
	Peas	3.9	4.1	6.3	4.4
	Beans	0.0	24.6	37.3	26.3
	Fresh carrots	28.7	19.7	29.8	21.0
	Fresh onions, leeks and shallots	14.1	21.8	33.0	23.2
	Tomatoes	3.7	27.4	41.4	29.2
	Potatoes (fresh and processed)	221.4	138.2	208.9	147.3
Drinks	Non-alcohol	32.7	34.4	0.0	106.0
	Alcohol	32.7	14.4	0.0	44.3
	Fruit juices	32.7	5.8	0.0	17.8
Other	Confectionery	34.6	4.8	0.0	21.8
Total Avoidable Food Waste		1000	1000	1000	1000
Packaging	Board packaging for liquids	0.0	4.1	0.0	3.6
	Polyethylene	0.0	5.4	9.1	6.4
	Polypropylene	0.0	6.3	7.6	5.5
	Kraft paper	0.0	4.8	3.7	3.3
Total Packaging		0.0	20.6	20.3	18.7

hundred food products, can be downloaded from the SI. The chemical composition of the packaging materials was instead taken from Riber et al. (2009).

2.4.3. Food production: farming and processing

For global processes involving food production, transport and associated refrigeration, we relied on state-of-the-art consequential datasets taken from ecoinvent v3.3 (Wernet et al., 2016). Global consequential datasets were available for fruit, vegetable, and dairy products. The list of the datasets used for each individual food product may be found in Table 2. For meat, data were available for animal farming activities, excluding further processing. Meat processing in slaughterhouses was then modelled conformingly with the information provided in the LCA Food database (2-0 LCA Consultants, 2007) using a consequential approach for any byproduct arising from the process consistently with the remaining datasets. On the basis of the information available, for pig, cows, and chicken a share equal to 83%, 60%, and 73% of the living animal weight becomes meat products sold to the market while the remaining mass (consisting of bones, animal bone meal, feather, feet, etc.) is sent to thermal treatment and credited with energy recovery, assuming that the residues are incinerated using the technology detailed in Section 2.4.6 of this manuscript. Fish production was also not available in Wernet et al. (2016) and thus modelled on the basis of the information provided by 2-0 LCA Consultants (2007) for trout (the inventory includes production, filleting, and freezing; see Table S13, SI) and pelagic fish, assuming cod fish (the inventory includes fishing, processing into fillet, and freezing; see Table S13, SI). The portion of the fish discarded during processing (17% and 56% for trout and cod fish, respectively) was assumed to be used for the production of fish meal, this credited with substitution of soymeal (thus soybean) on the basis of the respective proteins content. As soymeal production brings soy oil as coproduct, the reduced soy oil production was compensated with induced production of palm oil, following the soybean meal loop detailed by Dalgaard et al. (2008). The corresponding avoided land demand was also accounted for. Flour and bread production was also based on 2-0 LCA Consultants (2007), using a consequential perspective as for fish and meat products (Tables S14–S15, SI). This means that, the meals co-produced during flour production were assumed to substitute for animal feed. The corresponding avoided land demand was also accounted for. For drinks, an

Table 2
Summary of the life cycle inventory data used to model food production (farming and processing) and related source; land demanded is calculated accordingly. The original name of the dataset is displayed in italic. GLO: global market processes from ecoinvent v3.3, consequential system (Wernet et al., 2016).

Food category	Food product	Life cycle inventory dataset and reference source	Land arable/pasture (m^2·year kg^{-1})
Meat & Meat Products	Pork	Data for farming: ecoinvent v3.3; *swine production, live weight; GLO* Data for processing: LCA Food Database; *Slaughtering and cutting of pigs*	4.9/0
	Beef	Data for farming: ecoinvent v3.3; *cattle production for slaughtering, live weight to generic market for red meat, live weight; GLO.* Data for processing: 2-0 LCA Consultants (2007); *Slaughtering of cattle*	4.45/4.42
	Chicken	Data for farming: ecoinvent v3.3; *chicken production; GLO* Data for processing: 2-0 LCA Consultants (2007); *Slaughtering of chicken*	2.17/0.02
	Lamb/Mutton	Data for farming: ecoinvent v3.3; *sheep production; GLO* Data for processing: 2-0 LCA Consultants (2007); *Slaughtering of cattle* (assumed as cattle)	2.87/19.75
Milk & Dairy Products	Milk	Data for farming and processing: ecoinvent v3.3; *cow milk production; GLO*	0.58/0.58
	Cheese	Data for farming and processing: ecoinvent v3.3; *cheese production, from cow milk, fresh, unripened; GLO*	3.96/3.93
	Butter	Data for farming and processing: ecoinvent v3.3; *production of butter; GLO*	−36.37/6.31
	Yoghurt	Data for farming and processing: ecoinvent v3.3; *yogurt production, from cow milk; GLO*	0.64/0.54
Fish & Fish Products	Fish	Data for farming and processing: 2-0 LCA Consultants (2007); Fresh water trout farming/Filleting of fishData for fishing and processing: 2-0 LCA Consultants (2007): *Demersal fish/Filleting of fish*	3.9/0
Bakery & Dry Products	White bread & Similar non-brown bread	Data for farming: ecoinvent v3.3; *market for wheat grain; GLO* Data for processing: 2-0 LCA Consultants (2007); *wheat bread, conventional, fresh*	2.17/0
	Brown, Whole meal, and Other bread	Data for farming: ecoinvent v3.3; *market for rye grain; GLO* Data for processing: 2-0 LCA Consultants (2007); *rye bread, conventional, fresh*	1.35/0
	Cakes, Biscuits, Other Bakery Products	Data for farming: ecoinvent v3.3; *wheat grain; GLO; wheat grain, feed; GLO* Data for processing: 2-0 LCA Consultants (2007); *Flour and oat flakes production*	3.1/0
	Cereals & Cereals Products	Data for farming: ecoinvent v3.3; *wheat production; GLO*	3.1/0
	Rice	Data for farming: ecoinvent v3.3; *rice production; GLO*	0.01/ 0
	Flour	Data for farming: ecoinvent v3.3; *flour production; GLO*	3.1/0
	Sugar	Data for farming: ecoinvent v3.3; *sugar; GLO*	0.67/0
Fruit & Vegetable	Fresh oranges	Data for farming: ecoinvent v3.3; *orange production; GLO*	0.22/0
	Other fresh citrus fruits	Data for farming: ecoinvent v3.3; *lemon production; GLO*	0.37/0
	Fresh apples	Data for farming: ecoinvent v3.3; *apple production; GLO*	0.36/0
	Fresh pears	Data for farming: ecoinvent v3.3; *pear production; GLO*	0.49/0
	Fresh grapes	Data for farming: ecoinvent v3.3; *grape production; GLO*	0.36/0
	Fresh bananas	Data for farming: ecoinvent v3.3; *banana production; GLO*	0.2/0
	Fresh melons	Data for farming: ecoinvent v3.3; *melon production; GLO*	0.09/0
	Fresh cabbages	Data for farming: ecoinvent v3.3; *cabbage production; GLO*	0.11/0
	Fresh cauliflower	Data for farming: ecoinvent v3.3; *cauliflower production; GLO*	0.17/0
	Leafy salads fresh	Data for farming: ecoinvent v3.3; *lettuce production; GLO*	3.44/0
	Peas	Data for farming: ecoinvent v3.3; *pea protein production; GLO*	3.09/0
	Beans	Data for farming: ecoinvent v3.3; *bean production; GLO*	3.16/0
	Fresh carrots	Data for farming: ecoinvent v3.3; *carrot production; GLO*	0.21/0
	Fresh onions, leeks and shallots	Data for farming: ecoinvent v3.3; *onion production; GLO*	0.21/0
	Tomatoes	Data for farming: ecoinvent v3.3; *tomato production; GLO*	0.23/0
	Potatoes (fresh and processed)	Data for farming: ecoinvent v3.3; *potato production; GLO*	0.41/0
Drinks	Non-alcohol	Assumed as Fruit juices	0.5/0
	Alcohol	Assumed as Fruit juices	0.5/0
	Fruit juices	Data for farming: ecoinvent v3.3; *orange; GLO* Data for processing: (Doublet et al., 2013)	0.5/0
Other	Confectionery	Data for farming: ecoinvent v3.3; *wheat grain; GLO; wheat grain, feed; GLO* Data for processing: 2-0 LCA Consultants (2007); *Flour and oat flakes production*	3.1/0
Packaging	Board packaging for liquids	Data for production and transport: ecoinvent v3.3; *market for liquid packaging board container; GLO*	0/0
	Polyethylene	Data for production and transport: ecoinvent v3.3; *market for polyethylene terephthalate, granulate, bottle grade; GLO*	0/0
	Polypropylene	Data for production and transport: ecoinvent v3.3; *market for polypropylene, granulate; GLO*	0/0
	Kraft paper	Data for production and transport: ecoinvent v3.3; *market for kraft paper, unbleached; GLO*	0/0

inventory for production of orange juice was established on the basis of information provided in Doublet et al. (2013). Based on this, 1 kg of juice requires an input of 2.29 kg wet weight oranges. The peel waste discarded during processing (ca. 46% of the wet mass of the orange used to produce the juice) was assumed to displace soymeal and maize on the basis of the digestible energy and protein content, this being the most common utilization of such residue (Doublet et al., 2013). Displacing soymeal induces a production of palm oil to compensate for the displaced soy oil, as earlier described. The corresponding avoided land demand was also accounted for (Table S16, SI). A detailed overview of the life cycle inventory elaborated for these processes can be found in the SI, Tables S13–S16.

2.4.4. Land use changes and related emissions

Following a consequential perspective, demanding or preventing one additional unit of food waste incurs a demand or prevention of corresponding land. As mentioned earlier, land use changes in LCA are typically distinct into direct and indirect (dLUC/iLUC). The impacts from dLUC, while important for more

Table 3
Summary of the life cycle inventory data used to model food waste management. LHV_{wb}: lower heating value (wet weight basis; i.e. as received); PE: polyethylene; PP: polypropylene; ww: wet weight.

Process or Technology	Brief description	Main inputs	Main outputs	Source
Collection	Collection truck 10 t; 10 km	Diesel: 0.09 L km^{-1}	–	Distance based on Eisted et al. (2009). Truck: JRC (2017)
Transport	Road truck 28–32 t, euroVI, highway. Digestate/compost: 20 km. Bottom ash: 100 km. Fly ash: 500 km. Packaging: 200 km	Diesel: 1.46 kg kg^{-1} km^{-1}	–	Generic distances assumed. Truck: JRC (2017)
Incineration	Grate-fired incinerator with wet flue gas cleaning and electricity recovery	Electricity: 86 kWh t^{-1} ww Fuel oil: 0.63 kg t^{-1} ww Chemicals (see Section 2.4.6)	Electricity: 25% (of the LHV_{wb}) to market. Bottom ash and fly ash. Wastewater to treatment	Riber et al. (2008) Danish Energy Agency (2012)
Anaerobic digestion	Termophilic digester, dry matter of the mixture ca. 10%. Gas engine for biogas utilization	Electricity: 49 kWh t^{-1} ww Diesel: 0.9 L t^{-1} ww Heat: 192 MJ t^{-1} ww	Electricity: 45% of collected biogas-energy. Digestate to use on-land. Fugitive CH_4: 2% of CH_4 generated	Boldrin et al. (2011). Gas engine: Nielsen et al. (2010) and Danish Energy Agency (2012)
Composting	Aerated tunnels provided with biofilters	Electricity: 53 kWh t^{-1} ww Diesel: 1 L t^{-1} ww	Compost to use on-land. Residues (ca. 5% of input) to incineration	Boldrin et al. (2011, 2009)
Use on-land	Spreading with tractor on sandy-loam soil	Diesel: 0.57 L t^{-1} ww	NH_3, NO_x, N_2O, NO_3, C sequestered (see Section 2.4.6). Nutrients NPK to plants	Yoshida et al. (2016)
Landfilling	Conventional landfill equipped with gas extraction/ recovery, bottom line, daily soil cover and final cover	Electricity: 8 kWh t^{-1} ww Diesel: 0.2 kg t^{-1} ww	Electricity: 45% of collected biogas-energy. Leachate, gas emissions (see Section 2.4.6)	Manfredi and Christensen (2009). Gas engine: Nielsen et al. (2010), Danish Energy Agency (2012)
Packaging recycling	Plastic: re-melting of PE and PP waste to granulate. Paper: board production from mixed paper and cardboard waste	Plastic: Electricity: 88 kWh t^{-1} ww. Fuel oil: 1.1 kg t^{-1} ww. Paper: Electricity: 1380 kWh t^{-1} ww. Fuel oil: 51 kg t^{-1} ww	Recycled PP and PE (granulate) to market. Recycled board.	Plastic: Swerec AB v3.3[α]. Paper: Skjern Papirfabrik v3.3[α]
Marginal electricity	Marginal electricity sources (UK market)	51% hard coal, 21% nuclear, 12% natural gas, and 15% hydro/wind	Electricity	ecoinvent v3.3 consequential, UK[β]
Marginal heat	Marginal heat sources (UK market)	82% natural gas, 15% coal, 3% fuel oil	Heat	Ecoinvent v3.3 consequential, UK[γ]
Marginal fertilizers	Marginal fertilizers (global market assumed)	–	Urea, diammonium phosphate, potassium chloride	Ecoinvent v3.3 consequential, GLO[δ]
Marginal materials	Marginal PE/PP/Paper (global market assumed)	–	PE, PP, Paper	Ecoinvent v3.3 consequential, GLO[ε]

[α] Unpublished data that may be found in the database of the LCA-tool EASETECH (Clavreul et al., 2012).
[β] Marginal electricity; low/high/medium voltage; UK.
[γ] Heat production, natural gas, at boiler fan burner low-NOx non-modulating <100 kW; Europe without Switzerland; heat production, heavy fuel oil, at industrial furnace 1 MW; Europe without Switzerland; heat production, at hard coal industrial furnace 1–10 MW; Europe without Switzerland.
[δ] Market for urea, as N; GLO; market for phosphate fertilizer, as P_2O_5; GLO; market for potassium chloride, as K_2O; GLO.
[ε] Market for polyethylene terephthalate, granulate, bottle grade; GLO; market for polypropylene, granulate; GLO; market for kraft paper, unbleached; GLO.

spatially-defined LCA studies focusing on the optimal use of a specific land, are nevertheless typically one or more orders of magnitude lower than iLUC, especially when considering annual crops (most food production is based on these; see Schmidt and Brandao, 2013; Tonini et al., 2012). Further, modelling dLUC requires knowing the specific location of cultivation and the alternative cropping/use of the same land. For these reasons, we did not consider dLUC in this study and only modelled iLUC impacts. Notice that a similar approach was taken in in other studies (Schmidt and Brandao, 2013; Tonini et al., 2016a; Vadenbo et al., 2017). To quantify iLUC it is necessary to calculate: (i) the amount of arable and pasture land demanded by a specific food product (e.g. as m^2·year) and (ii) the iLUC inventory of demanding arable/pasture land. Regarding the first, the land demanded by each food product composing the mixed waste detailed in Table 1 was calculated on the basis of the life cycle inventory used. The results can be found in Table 2. Once the total amount of arable or pasture land demanded was known, the iLUC impact was quantified by multiplying the (arable or pasture) land demanded with the iLUC inventory derived with dedicated causal-effect biophysical models (Tonini et al., 2016b for arable land; Schmidt et al., 2015 for pasture land). These iLUC inventories already account for the fact that the response to the demand is given by a combination of intensification of the production and expansion on nature. For a detailed description of the model, the readers should refer to the original publications.

2.4.5. Distribution: transportation, wholesale, and retail

For the transport from farming place (or processing facility) to wholesale and retails (i.e. final point of distribution to consumers) and for the associated refrigeration involved, we relied on state-of-the-art consequential datasets from ecoinvent v3.3. Datasets were available for most of the food products composing the mixed waste composition; for those products for which datasets were not available, simplified assumptions were taken; for example transport of Meat & Meat Products and Fish & Fish Products was assumed equal to that of cheese (global consequential market; Ecoinvent, 2017); transport of Bakery & Dry Products and of Other was assumed equal to that of wheat; transport of Drinks was assumed equal to that of milk (global consequential market; Ecoinvent, 2017). Table S17 (SI) provides a detailed overview of the datasets used.

The consumption of energy at wholesale and retails was based upon the information provided in the LCA Food database (2-0 LCA Consultants, 2007). This varied depending upon the type of food product, storing condition (cooling, freezing, ambient temperature, and duration), and size of the store (for retails only); the specific consumptions assumed for each individual food product are detailed in Table S17 (SI). Overall, for wholesale, electricity consumption ranged from 0.0078 (ambient conditions) to 0.61 (average of cooling and freezing conditions) kWh kg^{-1} day^{-1}, while heat equalled 0.0052 kWh kg^{-1} day^{-1}. The storage period varied between 1 and 3 days. For retails, electricity consumption ranged from 0.012 to 0.188 kWh kg^{-1}, while heat was between 0.0032 and 0.019 kWh kg^{-1}. For scenario III and IV (food waste generated by Households and Food Service), local transport of the food from the retail to the household (or food service activity) was assumed to be done using a passenger vehicle EuroIV driving an average distance of 5.7 km based on the findings of Future Foundation (2007) specifically for United Kingdom.

2.4.6. Food waste management technologies and processes

For all the scenarios, regardless of the final treatment, food waste collection from the generators was assumed performed with a collection truck consuming 0.09 L diesel per km per tonne of food waste collected; the collection route was assumed equal to 10 km based on typical figures from literature (Eisted et al., 2009). Anaerobic digestion was modelled as state-of-the-art wet termophilic technology with a methane yield equal to 70% of the theoretical methane potential (Angelidaki and Batstone, 2010), which was quantified for each individual food product on the basis of the biochemical composition as earlier mentioned (see excel-file available in the SI). Electricity consumption for operations was set to 49 kWh t^{-1} ww food waste treated while heat consumption was calculated as the energy required to heat up the food waste from 8 °C (annual average temperature in UK) to 55 °C (termophilic conditions) assuming water content of the mixture in the fermenter equal to 90%. The produced biogas was assumed to be used in natural gas engines for electricity generation with gross efficiency equal to 45% on the energy content of the biogas-input (Danish Energy Agency, 2012). Fugitive emissions were assumed to equal 2% of the methane generated. The residual digestate (dry matter content 5%) was assumed to be transported 20 km and applied on land with tractors having fuel consumption of 0.57 L diesel t^{-1} digestate applied. Emissions of nitrogen following application were modelled conformingly with the findings of Yoshida et al. (2016) for temperate sandy loam soil: N_2O-N (to air; 2.8% of N applied), NH_3-N (to air; 7.5% of N applied), NO_3-N (to water; 35.3% of N applied). The carbon sequestration, within a 100 year time-horizon, equalled 13.2% of the C applied with the digestate. The composting technology was modelled as aerated tunnels provided with biofilters for exhaust gas cleaning conformingly with the plant described in Boldrin et al. (2011). The degradation of each material fraction was modelled as a percent of the volatile solids content in the incoming waste; this corresponded to ca. 70% volatile solids degradation for animal and vegetable food waste. The electricity consumption equalled 53 kWh t^{-1} food waste. The main emissions were CH_4, N_2O, and NH_3. Conformingly with Boldrin et al. (2011, 2009), the fugitive CH_4 emissions were set to 0.2% of the degraded C, and N_2O emissions to 1.4% of the degraded N. About 98.5% of the degraded N was in the form of NH_3 of which 99% was oxidized in biofilters. The rejects from the process were assumed incinerated. The produced compost (65% dry matter content), similarly to the digestate, was assumed to be transported 20 km and applied on land with tractors having fuel consumption of 0.57 L diesel t^{-1} compost applied. Emissions of nitrogen following application were modelled conformingly with Yoshida et al. (2016) for temperate sandy loam soil: N_2O-N (to air; 4.5% of N applied), NH_3-N (to air; 1.6% of N applied), NO_3-N (to water; 21.8% of N applied). The carbon sequestration, within a 100 year time-horizon, equalled 11.3% of the C applied with the compost.

The incineration plant was modelled as a grate-fired incinerator equipped with wet flue gas cleaning, non-selective catalytic reduction of NO_x, Hg and dioxin removal by activated carbon. Net electricity efficiency was assumed equal to 25% relative to the lower heating value, wet basis, of the waste input conformingly with expected performances for state-of-the-art waste-to-energy technologies in the period 2015–2030 (Danish Energy Agency, 2012). The materials and resources consumption for operations and flue-gas cleaning were: 0.63 L oil t^{-1} waste as auxiliary fuel, 0.02 kg NaOH t^{-1} waste, 0.34 kg $CaOH_2$ t^{-1} waste, 1.53 kg NH_3 t^{-1} waste, 1 kg activated carbon t^{-1} waste and 5.7 kg $CaCO_3$ t^{-1} waste for flue-gas cleaning. Following the approach of Riber et al. (2008), air emissions were divided into either process-specific emissions (emissions independent of waste composition but proportional to the amount of waste incinerated) or waste-specific emissions (determined by output transfer coefficients, i.e. for heavy metals). The emission of NO_x was 0.85 g t^{-1} MSW while SO_2 emission equalled 0.029 kg t^{-1} MSW assuming a degree of desulphurisation higher than 98.5%. Selected air emissions for relevant heavy metals were (as % of input): 0.2% (As), 0.1% (Cd), 0.01% (Cr), 0.0018% (Cu), 0.004% (Mn), 0.125% (Ni) and 0.015% (Pb) conformingly with available operational data from existing incinerators.

The landfill technology was modelled as an up-to-date conventional landfill with energy recovery and engineered measures to prevent emissions of gas and leachate to the environment. These included bottom liner, leachate collection system and leachate treatment prior to discharge of treated leachate to surface water bodies, and top soil cover, gas collection system and flaring. The provision of energy and materials to site and on-site operations were included. Data were adapted from Manfredi and Christensen (2009); the inputs to the landfill were electricity (8 kWh t^{-1} waste), diesel oil for vehicles operating on site (0.2 kg t^{-1} waste), soil and clay (in total, 0.26 t t^{-1} waste), polyethylene (membranes; 0.23 kg t^{-1} waste), polyvinylchloride pipes and steel (0.01 and 0.14 kg t^{-1} waste). The landfill had an average filling depth of 10 m and a bulk waste density of 1 t m^{-3} waste. Gas generation was modelled according to the tiered approach suggested by IPCC using decay values for temperate regions (Eggleston et al., 2006). On this basis, decay factors for food and cartons/cardboard materials equalled 0.137 y^{-1} and 0.019 y^{-1}, respectively. Gas collection was modelled according to four time periods, adapting the approach described in previous publications (Manfredi and Christensen, 2009): during the years 0–5 after waste disposal, 35% of the gas was assumed to be collected (daily top cover assumed), then 65% (years 5–15; temporary top cover assumed), then 75% (years 15–55; final top cover assumed). During the remaining 45 years, no gas was assumed to be collected (closure). The uncollected gas fraction received partial oxidation in the top soil cover. In respect to this, adapting the approach of Manfredi and Christensen (2009), oxidation efficiencies were specified for each gas constituent for the four time periods. With respect to methane, the efficiency of oxidation to CO_2 equalled 10% with daily top cover, 20% with intermediate top cover, and 36% with final top cover. All the collected gas during 100 years was assumed combusted in gas engines for electricity production. Likewise for anaerobic digestion, the electricity recovery efficiency of the gas engine equalled 45% of the energy content of the biogas-input (Danish Energy Agency, 2012). Regarding leachate, the net infiltration through the top cover was assumed on average equal to 300 mm y^{-1} during 100y. Following the approach of Manfredi and Christensen (2009), four periods were considered to take into account the variation of the concentration of the different chemicals in the leachate. With respect to leachate collection, capture efficiencies were set to 99.9% during the first 80 years of the landfill life and 87% afterwards. Uncollected leachate was assumed to reach the groundwater. The collected leachate was treated in a wastewater treatment plant for purification, modelled conformingly with the plant detailed in Yoshida et al. (2015). The treated leachate was assumed discharged to surface water bodies. The same wastewater treatment technology was used to model treatment of food waste discharged in the sewer. Recycling of polyethylene and polypropylene into the respective granulates was modelled using data from a Swedish plastic recycling plant (Swerec AB, 2017), while recycling of paper was modelled according with a Danish plant (Skjern papirfabrik, 2017). Additional information (e.g. transportation distances) may be found in the summary of the life cycle inventory data used to model waste management technologies and processes (Table 3).

2.5. Uncertainty analysis

To address parameter uncertainty, the state-of-the-art approach described in Bisinella et al. (2016) was followed. This includes perturbation analysis to identify the most sensitive parameters and uncertainty propagation to derive overall uncertainties. The perturbation analysis was performed following the "one-at-a-time" approach (Bisinella et al., 2016). According to this method, each parameter is changed one-at-a-time keeping the other ones fixed to evaluate which parameters are the most sensitive in the scenario. The parameters considered in the model were varied by 10%. The list of all the parameters perturbed may be found in Table S19 (SI). Once the perturbation analysis was done, the uncertainty of each individual parameter used in the model was quantified analytically, conformingly with the method described in Bisinella et al. (2016), on the basis of the uncertainty range assumed for each individual parameter used in the model. The total uncertainty of the characterized results quantified for the baseline, for each environmental impact category, was then obtained as the sum of the contributions of the individual parameters uncertainties.

The uncertainty range was based, whenever possible, on the information available from literature sources. This was the case of waste treatment technologies efficiency (e.g. energy recovery), energy consumption for cooking and cooling of food (for Households and Food Service), and substitution efficiency of mineral fertilizers. For energy consumption of wholesale and retails, it was assumed that this varied between the consumption needed for refrigeration and for freezing; this, however, only applied to selected food products, i.e. meat, fish and dairy. The remaining was assumed to be stored at ambient temperature. When information was not available from literature sources, best-guess assumptions were taken. For example, for transportation of food products but also of waste and treatment residues, it was conservatively assumed a variation of 200% around the value assumed in the baseline. For the management of the packaging, it was assumed that the share of packaging recycled varied between 59.2% (current performance) and 80% (overall 'recovery target' on the basis of the EU packaging directive). The share not recycled was assumed to be incinerated or landfilled maintaining the current proportion between the two options. For the remaining parameters, a variation of 20% around the value taken for the baseline was assumed. This was the case of the land demanded for the individual food products and of the food production impact. For the food waste management, the share of food waste sent to the individual treatments was also assumed to have an uncertainty range of 20%. For simplicity, each path was assumed as 'independent variable'. A variation of each of these pathways was assumed to affect only the share sent to landfilling, this being assumed as the alternative option (and thus correlated). The share of the iLUC response from intensification and expansion was assumed to span from 100% expansion to 100% intensification. For a detailed overview of the ranges adopted in the study refer to Table S19 (SI).

In addition to the parameter uncertainty analysis, a sensitivity analysis was performed on the food waste fractional composition breakdown used in the baseline (see Table 1) in order to evaluate the importance of this on the results. For each individual environmental impact category, the extreme variation of the result (min-max) was quantified by varying the specific food products composing the mixed food waste, at the same time maintaining the fulfilment of the functional unit (1 t of avoidable food waste). This exercise aims at illustrating the importance of the specific food products composing the mixture on the final LCA results.

3. Results

The LCA results are illustrated in Fig. 2 as characterized impacts per tonne of avoidable food waste, wet weight basis. The breakdown of the contributors to the impact is also displayed. These are grouped as: (i) Land Use Change (including expansion and intensification), (ii) Food Production (including farming and processing), (iii) Packaging Provision (including production and transport to the place where it is used for packaging of the food), (iv) Distribution (including packed food products transport from

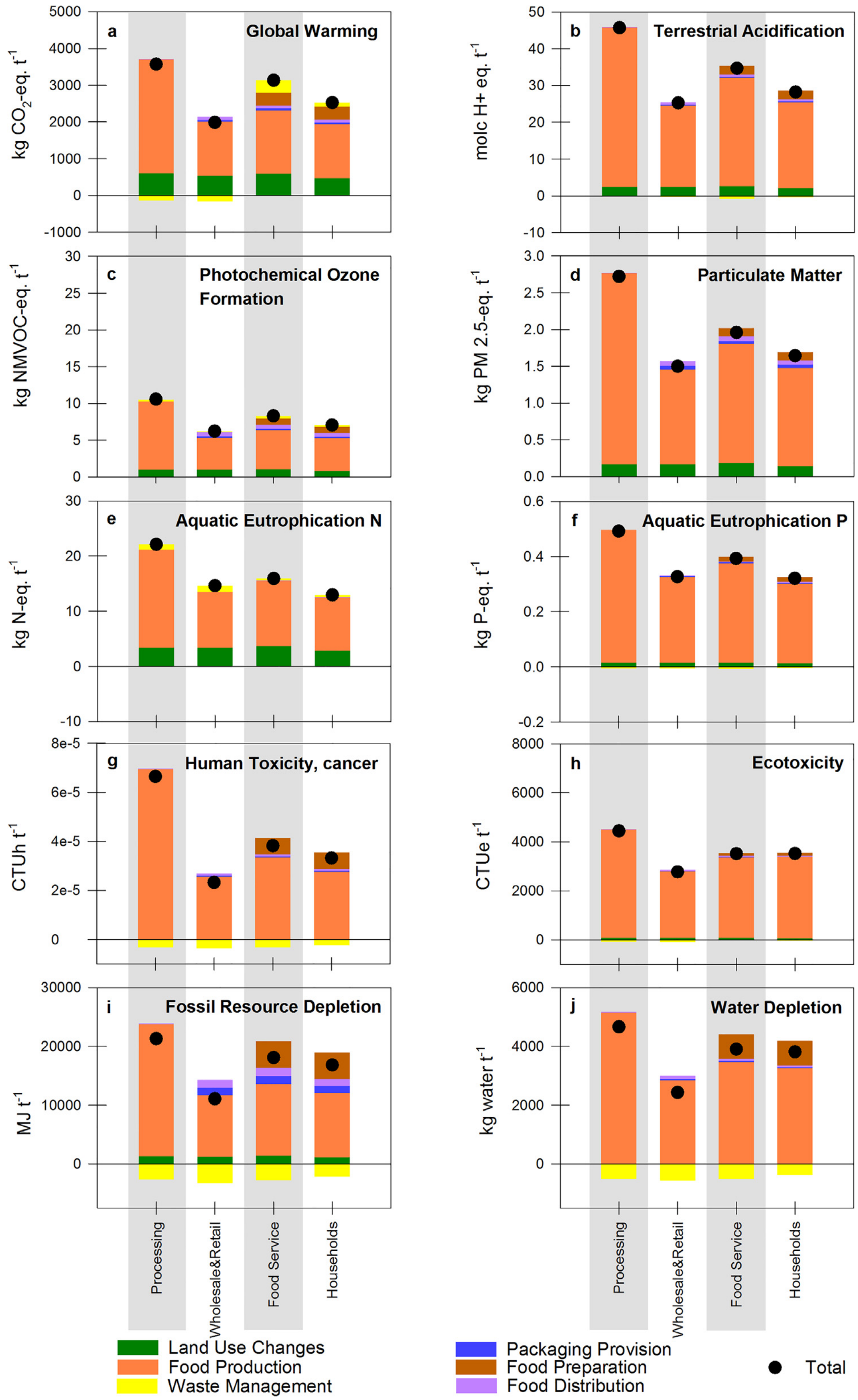

Fig. 2. Characterized LCA results for the environmental categories addressed in this study: food waste impacts during the entire life cycle, from production (including iLUC) to final waste management and disposal. The functional unit is one tonne of avoidable food waste (wet weight basis) with the composition given in Table 1.

production site to final consumers as well as the operations in wholesale and retail), (v) Food Preparation (including refrigeration at Households/Food Service and eventual cooking), (vi) Waste Management (including collection, treatment, credits for energy/products recovery, transport and final disposal of the treatment residues). From now onwards, this naming (with capitals) will be used to refer specifically to each of these contributors. Values above zero represent environmental burdens; those below zero represent environmental savings. The final (net) impact, per each individual category, is the sum of burdens and savings, and it is

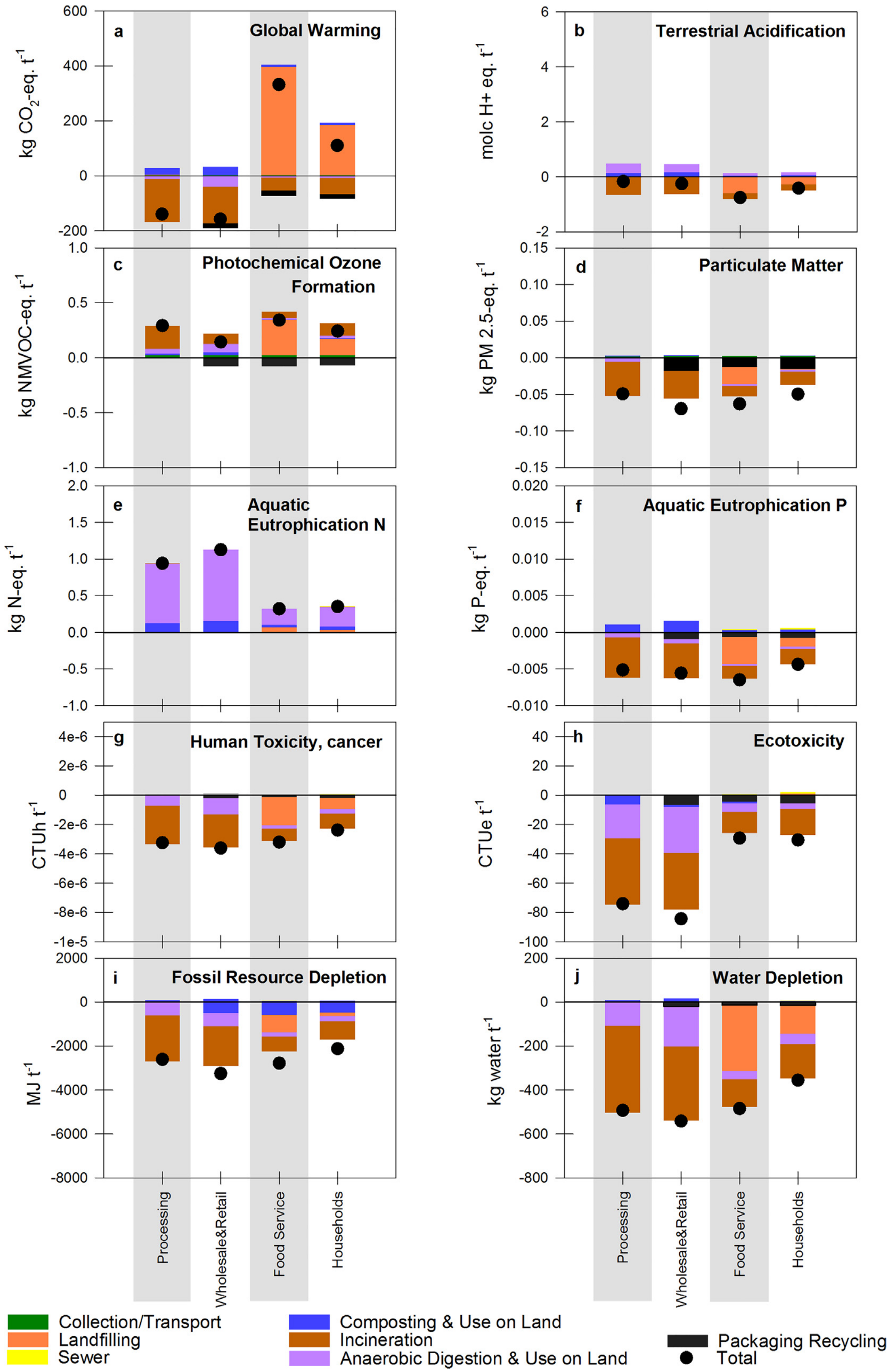

Fig. 3. Characterized LCA results for the environmental categories addressed in this study: focus on Waste Management impacts. The functional unit is one tonne of avoidable food waste (wet weight basis) with the composition given in Table 1.

illustrated with a circular indicator in Fig. 2. Fig. 3 focuses on the impact of Waste Management, with breakdown of the activities involved. For completeness and transparency, Table S20 (SI) provides the numerical results with the contributor's breakdown

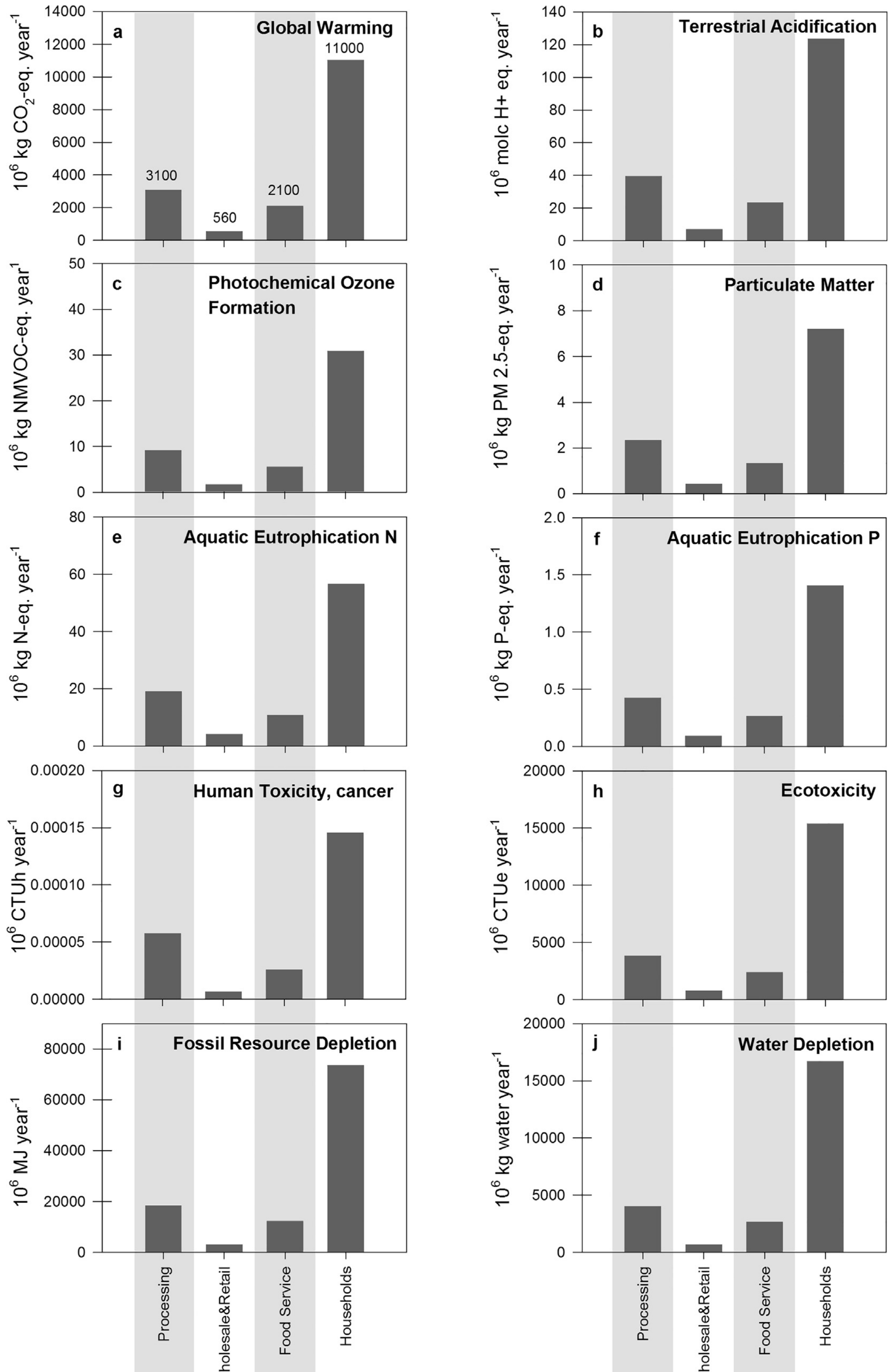

Fig. 4. Annual total environmental impacts (reported as characterized LCA results) per each sector addressed in this study. The annual amount of avoidable food waste generated at each sector is (wet weight basis): 0.866 Mt at Processing, 0.28 Mt at Wholesale & Retail, 0.68 Mt at Food Service, and 4.38 Mt at Households (see also Fig. 1).

expressed as percentage of the burdens and of the savings. Finally, Fig. 4 displays the annual total environmental impacts per each sector.

3.1. Global Warming

The Global Warming impact ranged from 2000 kg CO_2-eq. t^{-1} for Wholesale & Retail to 3600 kg CO_2-eq. t^{-1} for Processing sector (Fig. 2a). For Food Service and Households, the impact equalled 3100 and 2500 kg CO_2-eq. t^{-1}, respectively. Food Production was the main contributor to the impact, followed by Land Use Change. The burden from Food Production was primarily connected to the content of meat and dairy products in the mixed food waste, these having the largest CO_2 footprint among the considered food products. Indeed, across the four sectors addressed in the study, Processing was the one with the larger share of meat and dairy products, this finally incurring a higher impact. The same reasoning applies to Land Use Change impacts, also significantly affected by the share of meat and dairy products in the mix. Food Preparation was a significant contributor to the overall burden of Food Service and Households (12–14% of the burden; Table S20, SI). The contribution from Distribution, albeit not negligible, was nevertheless small compared with the others, ranging from 12 (for Processing) to 90 (for Households) kg CO_2-eq. t^{-1}, i.e. 0.3–4.2% of the burden. The contribution from Packaging Provision was the smallest, about 45–50 kg CO_2-eq. t^{-1}, i.e. ca. 2% of the burden. Waste Management contributed with savings in the sectors Processing and Wholesale & Retail (−139 and −157 kg CO_2-eq. t^{-1}), as illustrated in Fig. 3a. This was because the food waste from these sectors was mainly managed through anaerobic digestion or incineration, incurring substantial environmental savings because of energy recovery and substitution of alternative means of production. In the remaining sectors, instead, the prevalence of landfilling as waste handling technique induced burdens on Global Warming (110–332 kg CO_2-eq. t^{-1}). While energy recovery is also applied in landfill (as in anaerobic digestion), the efficiency of gas extraction and recovery is significantly decreased compared to controlled conditions in dedicated fermenters. It derives that the related savings from displacement of the alternative energy sources are not sufficient to compensate for the burdens derived, mainly, from methane leakages.

3.2. Acidification, photochemical ozone formation and particulate matter

The impact on Acidification, Photochemical Ozone Formation, and Particulate Matter (Fig. 2b–d) generally followed a similar trend to that of Global Warming both with respect to the ranking of the sectors and to the impact contributions. Again, the principal contributor was Food Production, which share on the impact ranged between 66% in the category Photochemical Ozone Formation and 95% in Acidification (Table S20, SI). In the category Photochemical Ozone Formation, the contribution from Distribution, Food Preparation, and Waste Management was rather significant compared to Acidification and Particulate Matter where the impact was mainly due to the sole Food Production (79–95%) and Land Use Change (5–11%). The reason for this was the emission of NO_x and volatile organic compounds during food products transportation (part of Distribution), cooking (part of Food Preparation), and biogas combustion in landfill and anaerobic digestion plants (part of Waste Management). Waste Management, overall, incurred savings, albeit negligible, in Acidification and Particulate Matter across all sectors. These occurred because the savings derived from energy recovery (at incinerators, landfills, and anaerobic digestion plants) with related substitution of alternative forms of production exceeded the burdens derived from on-land application of digestate after anaerobic digestion and composting operations (i.e. an advantageous balance of NO_x, NH_3, SO_x, and particulate emissions; Fig. 3b–d).

3.3. Aquatic eutrophication nitrogen and phosphorous

Expectedly, the impact on Aquatic Eutrophication, both in relation to nitrogen and phosphorus, was driven by Food Production owing to the nutrients leaching following on-land application of mineral fertilizers during farming operations (Fig. 2e and f). Per unit of avoidable waste, the sector Processing showed the highest footprint due to the higher share of meat and dairy in the mixed waste composition, similarly to the results for the other categories. In Aquatic Eutrophication Nitrogen, Land Use Change and Waste Management were also important contributors to the impact. For the former, because of nitrogen leaching following increased use of mineral nitrogen fertilizers, this being part of the market response to the demand for arable land; for the second, because of nitrogen leaching following application on-land of the digestate after anaerobic digestion of food waste. Accordingly, the impact of Waste Management was more evident in the sectors Processing and Wholesale & Retail because of the increased application of anaerobic digestion as waste handling technique (Fig. 3e and f). The impact of incineration is generally negligible on these categories owing to the strict NO_x emission control in modern plants (Astrup et al., 2015; Turconi et al., 2011). That of landfilling is also typically limited, under the assumption that a proper system for leachate collection and treatment is operated (Manfredi and Christensen, 2009). Notice that Waste Management induced savings on Aquatic Eutrophication Phosphorous, owing to the fact that the credits from energy recovery and mineral P-fertilizer substitution were larger than the burden from P-leaching, typically negligible when applying digestate and compost on-land compared to N-leaching as discussed in previous studies (Hansen et al., 2006).

3.4. Human toxicity cancer and ecotoxicity

The toxicity impacts reflected the trend seen for the other categories, where higher shares of meat and dairy in the mixed waste induced worse performances; this is the case of the waste generated at the Processing sector. The impact on Human Toxicity (Fig. 2g) was mainly associated to Food Production due to the release of metals to water and soil following use of chemicals and fertilizers during farming. Food Preparation, because of the energy expenses for cooking, had a non-negligible contribution, equalling about 17–19% of the burden. Waste Management contributed with savings thanks to energy recovery and related substitution of alternative means of production (Fig. 3g). This applied to all sectors, but was the highest in the sectors Processing and Wholesale & Retail because of the better energy recovery efficiency, as here most of the food waste was treated through anaerobic digestion or incineration. The impact on Ecotoxicity (Fig. 2h) was almost totally governed by Food Production, mainly because of the use of herbicides and pesticides during farming. The burden was then higher whenever the share of meat and dairy products in the mixed food waste was larger, these having the higher footprint in terms of Ecotoxicity. This was the case of the Processing sector, due to the mixed waste composition considered.

3.5. Fossil resource and water depletion

The trend of the results for the two resource categories (Fig. 2i, j) was similar and the ranking between sectors performances comparable to that of the other categories. Expectedly, the results for Fossil Resource Depletion mirrored those of Global Warming, ranging from 11,000 MJ t^{-1} (Wholesale & Retail) to 21,000 MJ t^{-1}

(Processing). Regarding depletion of fossil resources, while Food Production again brought the larger burden, Food Preparation and Waste Management also were significant contributors to the impact. The first with a burden owing to the energy expenditures for cooking, and the second with environmental savings because of the energy recovery at incinerators, biogas plants, and landfills with related substitution of alternative means of production, partly including fossil sources. Non-negligible was also the burden derived from Distribution and Packaging Provision, both involving consumption of fossil resources. The impact on Water Depletion ranged from 2400 kg water t^{-1} for Wholesale & Retail to 4700 kg water t^{-1} for Processing. The higher content of meat and dairy products in the mixed waste was the main reason for the worse impact in the Processing sector. Likewise for Fossil Resource Depletion, Food Preparation and Waste Management (for the latter, refer to Fig. 3i, j) also contributed with significant burdens and savings, owing to energy consumption and recovery, respectively.

3.6. Annual total environmental impacts

Considering the total amount of avoidable food waste generated annually, the Households was outstandingly the sector of the supply chain with the highest environmental impact among all, across all the ten environmental categories assessed, as illustrated in Fig. 4. This is because of the notably larger amount of food waste generated by the households compared with the remaining sectors of the supply chain (i.e. 4.38 Mt y^{-1}; see Fig. 1). The annual carbon footprint of the Households sector equalled ca. 11 Mt CO_2-eq., while the annual carbon footprint of the entire post-farm supply chain amounted to ca. 16.8 Mt CO_2-eq., this figure accounting for all the direct and indirect effects (iLUC and credits from waste management) of generating avoidable food waste. For comparative purposes, it should be noticed that this figure corresponds to about 3.4% of the annual total carbon footprint of the United Kingdom for the year 2015, quantified to 496 Mt CO_2-eq. (UK Government, 2017).

3.7. Parameter and scenario uncertainty

Fig. 5 illustrates the most important model parameters with respect to their relative contribution to the overall uncertainty. Only parameters with a relative contribution above 3% are shown. All in all, food production (the sum of all the contributions from the individual food products), iLUC (following variation of the intensification/expansion ratio), share of food waste cooked, and energy consumption for cooking were the most important contributors to the overall uncertainty across the ten environmental categories investigated. The parameters relating specifically to food preparation (share of food waste cooked and energy consumption for cooking) only applied to the sectors Food Service and Households, as it was assumed that no cooking occurred in the two remaining sectors. Because of this, in the sectors Processing and Wholesale & Retail, most of the uncertainty was attributed to Food production impact and iLUC. These results highlight that the parameters related to distribution (food transport distances, cooling consumption, etc.) and to waste management technologies (e.g. electricity recovery at incinerator or anaerobic digestion plant, biogas yield, digester leakage, waste collection and transportation distances, etc.) were less important on the final uncertainty of this case study. This was true for all the assessed environmental impact categories, with the exception of Human Toxicity where also transportation (including waste collection and transport) and other parameters uncertainty (e.g. fertilizer substitution) appeared to be important. Regarding the waste treatment technologies, it should be noted that the low importance of parameters such as the energy recovery efficiency is also the consequence of the relatively cleaner energy system assumed in the first place for United Kingdom. Dirtier emission factors, e.g. in terms of GHG emissions of electricity production, may change this result, as the contribution of the waste-to-energy recovery on the environmental impact becomes larger. Other parameters, which importance on the overall uncertainty was found to be minor, were the efficiency of the food supply chain (i.e. food wasted over total food output from each sector) and the share of food waste sent to the alternative final treatments. The overall uncertainty for the impact categories addressed was quantified in the range of 5–25% of the baseline result, except for the category Human Toxicity cancer, for which the uncertainty was significantly larger, up to about 100% of the baseline value (Fig. 6).

Fig. 6 also illustrates min-max characterized results obtained after running the model with extreme variations in the initial food waste composition (i.e. that used in the baseline, reported in Table 1), in terms of specific food products composing the mixture. This means, for example, having 100% of the Meat & Meat Products category composed of chicken (or other products), or 100% of the Milk & Dairy Products category composed of butter (or other products), etc. As highlighted by Fig. 6, these changes incurred much larger variations in the LCA results than those caused by the parameter uncertainty. In extreme cases, some of the results may even be negative owing to the fact that some food products have environmental credits because of the co-products generated following the consequential approach. This is the case of butter in Ecoinvent v3.3, where some impacts are negative due to the savings derived from the substitution of buttermilk and skimmed milk. This exercise, though illustrating extreme variations in the results following extreme variations in the food waste mixture, nevertheless highlights that knowing the specific food products composing the mixed food waste is very important to derive robust results.

4. Discussion

4.1. Comparison with the results from previous studies

This section compares the results of this study with the relevant literature. The comparison and discussion focus particularly on bottom-up studies dealing with mixed food waste (i.e. not on single food products) and Global Warming, as most of the studies available have assessed this impact category only. Table 4 summarises the main differences across the studies. The Global Warming results of this study are generally in the higher end of the range found in literature for bottom-up studies (Table 4). The main reason for this is the inclusion of indirect land use change impacts, which in this study contribute to the Global Warming impact with about 470–600 kg CO_2-eq. t^{-1} depending on the mixture of food products composing the waste, thus on the sector considered (for details, refer to Table S20, SI). This impact is not included in the other LCA studies on food waste, with the exception of Martinez-Sanchez et al. (2016) and Chapagain and James (2011). Martinez-Sanchez et al. (2016) calculated the C-footprint of the avoidable portion of household food waste in a region of Denmark to 1200 kg CO_2-eq. t^{-1}. This figure is much lower than that in this study because Martinez-Sanchez et al. (2016) did not include food preparation and store operations. Further, most importantly, as opposite to this study, the waste management in Martinez-Sanchez et al. (2016) contributed with larger savings (ca. −400 kg CO_2-eq. t^{-1}) as waste was assumed to be incinerated with electricity and heat recovery and these then credited by substitution of coal-based electricity and natural gas-based heat, respectively. The difference in waste management techniques (incineration with electricity and heat recovery in DK versus mostly landfilling in UK for the case of households food waste)

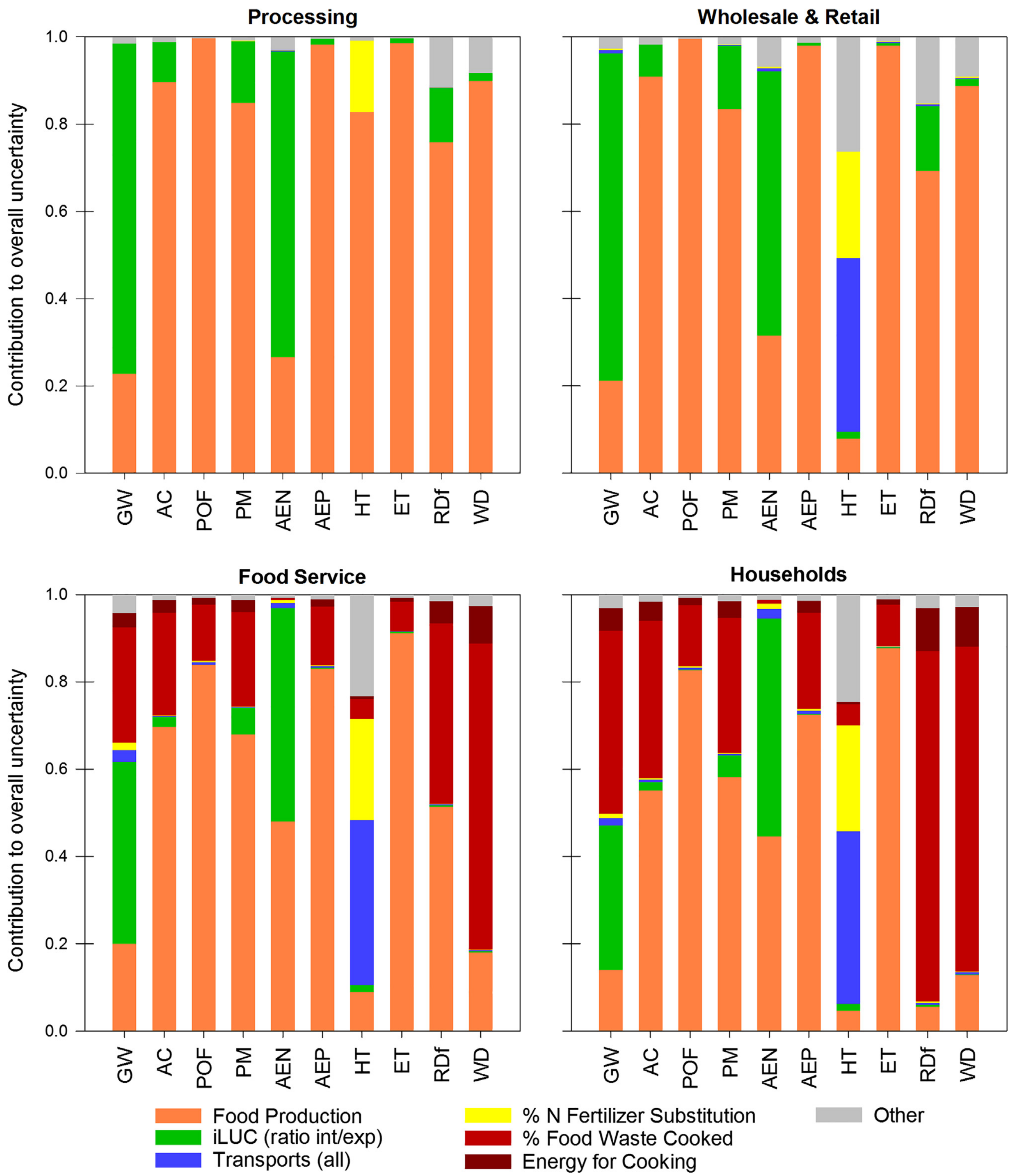

Fig. 5. Uncertainty contributions from the parameters used in the LCA model for each environmental impact category (i.e. relative importance on the overall parameter uncertainty).

and in the assumption for the substituted electricity source (coal in DK versus a much cleaner mix in UK) finally determine a significant delta (ca. 570 kg CO_2) between the two case studies. When applying the scenario assumptions of this study to Martinez-Sanchez et al. (2016) the Global Warming raises to ca. 2100 kg CO_2-eq. t^{-1}, closer to the result of this study. The remaining difference is then explained by the different waste composition and datasets used to model food production.

Bernstad and Andersson (2015) quantified the Global Warming of household food waste in Sweden between 800 and 1400 kg CO_2-eq. t^{-1}. The range reflects a sort of min-max range of the results due to variations in the datasets and modelling assumptions. This figure is lower than this study because: (i) indirect land use change and storage/refrigeration (at the household) impacts were not included, (ii) waste management contributed with substantial GHG savings for the same reasons as in Martinez-Sanchez et al.

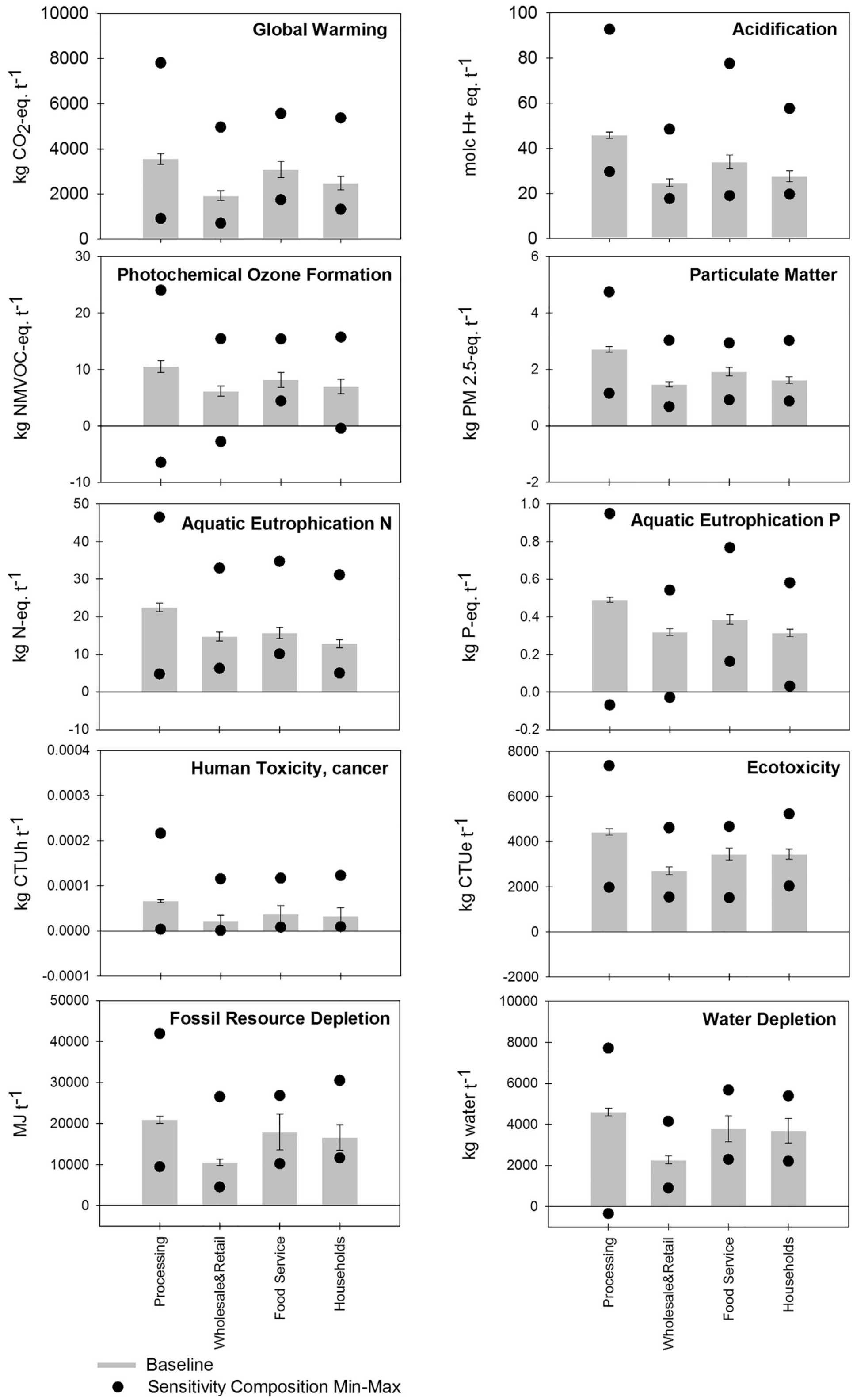

Fig. 6. Results of the parameter uncertainty propagation and of the sensitivity analysis on the food waste composition. The histogram shows the mean result values (baseline). The error bars represent the full extent of the parameters uncertainty around the mean result value. The uncertainty was obtained with the analytical procedure illustrated in Bisinella et al. (2016). The round indicators illustrate the extreme min-max variation of the LCA results obtained after changing the food products fractional composition. The functional unit is one tonne of avoidable food waste (wet weight basis) with the composition given in Table 1.

(2016). When applying the assumptions of this study to Bernstad and Andersson (2015), the Global Warming raises above 2000 kg CO_2-eq. t^{-1}, again closer to the result of this study. The residual difference with the footprint calculated in this study is a consequence of differences in waste composition and datasets used to model food production. It should also be noticed that the share of food

Table 4

Comparison of different impact assessment studies on (mixed) food waste. The different assumptions and methods used are underlined; AC: Acidification; AEN: Aquatic Eutrophication Nitrogen; AEP: Aquatic Eutrophication Phosphorous; ARD: Abiotic Resource Depletion: EF: Ecological Footprint; ET: Ecotoxicity; GW: Global Warming (kg CO_2-eq. t^{-1}): HTc: Human Toxicity cancer; OD: Ozone Depletion; PM; Particulate Matter; POF: Photochemical Ozone Formation.

Study	GW	Sector	Functional unit	LCA Approach	Modelling/boundaries (compared to this study)	Environmental categories assessed	Fractional Food Waste composition	Database for food production	Waste treatment assumed (energy substituted)
Martinez-Sanchez et al. (2016)	1200[α]	Households (Denmark)	Avoidable Food Waste	Bottom-up	Not included: packaging, store operations, food preparation	GW, AEN, PM, WD	Based on sampling campaign on a similar region	Mostly ecoinvent v3.1 (consequential)	Incineration with electricity/heat (coal/natural gas)
Bernstad and Andersson (2015)	800–1400	Households (Sweden)	Avoidable Food Waste	Bottom-up	Not included: iLUC, store operations	GW	Based on sampling campaign	Mix of different studies	Incineration with electricity/heat (coal/natural gas)
Chapagain and James (2011)	3800[β]	Households (UK)	Avoidable Food Waste	Bottom-up	Not included: packaging, store operations; iLUC reported separately (expansion only).	GW, WD	Modelled	Mix of different studies	62% landfilling (not specified) 8% incineration (not specified) 8% composting (not specified) 22% sewer (not specified)
Brancoli et al. (2017)	2800–3100[γ]	Wholesale & Retail (Sweden)	Avoidable Food Waste	Bottom-up	Not included: iLUC	GW, AC, AEN, ARD, OD, PM, TE	Based on sampling campaign	Blonk consultants	Different management scenarios (average Sweden)
Scholz et al. (2015)	1600	Wholesale & Retail (Sweden)	Avoidable Food Waste	Bottom-up	Not included: iLUC, packaging, store operations	GW	Based on sampling campaign	Mix of different studies	Incineration with energy recovery (average Sweden)
Oldfield et al. (2016)	5500–6100[δ]	Entire food supply chain (Ireland)	Food Waste	Bottom-up	Not included: iLUC, packaging, food preparation	GW	Modelled (assumed equal to UK)	Mix of different studies	Incineration with energy recovery (not specified)
Monier et al. (2010)	1700–2070 (1900)	Manufacture Households Others (EU)	Food Waste	Bottom-up	Not included: iLUC	GW	Modelled	Average figures were used for 7 food categories	EU average (not specified)
FAO (2013)	2100[ε]	Entire food supply chain (Global)	Food Wastage (loss + waste)	Bottom-up	Not included: iLUC	GW, WD, Biodiversity	Modelled	Not specified	Landfilling (not specified)
Song et al. (2015)	3600	Households (China)	Food Waste	Hybrid	Not included: iLUC, waste management	GW, WD, EF	Modelled	Mix of different studies	Composting (none)
Reutter et al. (2017)	17,860[ζ]	Entire food supply chain (Australia)	Avoidable Food Waste	Top-down	–	GW	Not addressed	Input-Output tables	Not addressed (not specified)
This study	2000–3600	Processing Wholesale & Retail Food Service Households (UK)	Avoidable Food Waste	Bottom-up	–	GW, AC, POF, PM, AEN, AEP, HTc, ET, FRD, WD	Modelled	Mostly ecoinvent v3.3 (consequential)	Based on the current UK situation (see Fig. 1)

[α, γ, δ] The range includes the credits from waste management; all ranges have been recalculated after personal communication with the authors.
[β] This figure does not include iLUC impacts; the authors also provide separately an impact for iLUC, assuming that all land is derived through expansion (deforestation) only.
[ε] Recalculated dividing the total annual impact provided in the study (3.3 Gt CO_2-eq. y^{-1}) by the food wastage amount given in the same study (1.6 Gt y^{-1}).
[ζ] Recalculated dividing the total annual impact provided in the study (57,504 Gg CO_2-eq. y^{-1}) by the food waste generation in Australia (3,219,404 t y^{-1}).

waste undergoing cooking was 27% in Bernstad and Andersson (2015) against 10% in this study. Yet, the energy consumption for cooking assumed in Bernstad and Andersson (2015) was lower than this study, finally leading to a comparable impact from food preparation. Chapagain and James (2011) quantified the C-footprint of the avoidable food waste generated by the UK households to 3800 kg CO_2-eq t^{-1} (this does not include iLUC). The figure is sensibly higher than this study because of the notably increased farming and processing (i.e. food production), transport, and waste management impacts in Chapagain and James (2011). The reason for this difference is the inventory data used in their study, aggregated from different publications dated 1999–2009, and presenting significantly higher impacts than the consequential datasets used in this study. This is likely a consequence of cut-offs and allocation rules used to handle co-products as well as of obsolete data. Further, it appears that credits from co-products (e.g. in waste management) were not considered in the study, incurring higher impact estimates.

For the wholesale/retail sector, Scholz et al. (2015) quantified the average C-footprint of a supermarket chain in Sweden to 1600 kg CO_2-eq t^{-1}. The Global Warming impact quantified in this study is higher (ca. 2000 kg CO_2-eq t^{-1}) because of the inclusion of iLUC impacts (contributing with additional ca. 540 kg CO_2-eq t^{-1}, see Table S20); without such contribution, the results of the two studies would be comparable, except for some differences due to the different waste composition and datasets used to model food production impacts. In the study of Brancoli et al. (2017) the current Global Warming impact of food waste generated in a Swedish supermarket was quantified between 2800 and 3100 kg CO_2-eq t^{-1}. This, albeit not including iLUC related to food production, is much higher compared to Scholz et al. (2015) that also focused on supermarkets. The main reason for this is the choice of datasets used due to model food production (taken from Blonk, 2015) which provide higher impacts for most food products (e.g. meat, dairy and bread) compared to the background datasets used by Scholz et al. (2015) and in this study. A main reason for this is the cut-offs and allocation rules used to handle co-products.

Oldfield et al. (2016) quantified the average Global Warming impact of the food waste in the entire Irish food supply chain to ca. 5500–6100 kg CO_2-eq t^{-1} depending upon the waste management technology considered. This figure, albeit not including iLUC and food preparation impacts, is higher than this and the other earlier-mentioned studies. The reason for this is the significantly higher C-footprint figures (compared to, for example, Ecoinvent v3.3 data) used by the authors to model the food production impact on Global Warming, e.g. for meat products (23.7 vs. 5–8 kg CO_2-eq. kg^{-1} in this study, depending on the composition mixture) and vegetables (2.1 vs. 0.8–2 kg CO_2-eq. kg^{-1} in this study, depending on the composition mixture). This is similar to what seen earlier for Chapagain and James (2011). Monier et al. (2010) quantified the average C-footprint of the EU food waste to 1900 kg CO_2-eq t^{-1}, this ranging from 1700 at manufacturing (farming and processing) to 2070 kg CO_2-eq t^{-1} at households. The score is lower than this study because LUC effects were not included in the analysis. Further, the background datasets used to model the C-footprint were roughly based on previous studies and not up-to-date as also stressed by the authors. A comparable figure is reported in FAO (2013) for the global World food supply chain, again not including LUC impacts. With respect to top-down and hybrid studies, Song et al. (2015) estimated the C-footprint of household food waste in China to 2500 kg CO_2-eq t^{-1}. This, while comparable to the result of this study in terms of magnitude, nevertheless did not include iLUC, food preparation, and waste management burdens and/or credits making the comparison between the studies very hard. Including these aspects would likely change the overall C-footprint. Reutter et al. (2017), using input-output

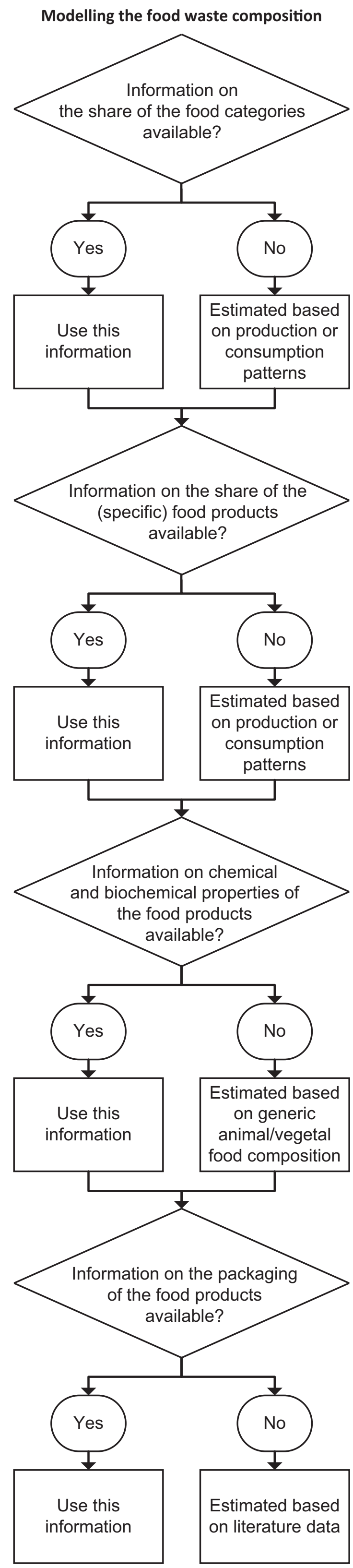

Fig. 7. Framework approach to model the food waste fractional composition in bottom-up LCA studies depending on data availability.

analysis and including land use change emissions, estimated a significantly higher C-footprint figure for the case of Australia, compared to this and other studies. This is likely due to the different approaches taken and is somehow expected as lower impacts are typically seen when applying bottom-up compared to top-down LCA approaches as exemplified and largely discussed in previous research (among the others: Lenzen and Dey, 2000; Majeau-Bettez et al., 2011).

4.2. Methodological learnings and challenges

This section discusses the main methodological learnings and challenges on the basis of the experience from this and previous studies on the subject. The focus is on the following issues: (i) modelling the food waste composition, (ii) modelling land use changes, (iii) avoiding double counting, and (iv) data uncertainties.

4.2.1. Modelling the food waste composition

In order to define the composition of the waste generated by each sector of the food supply chain, the decision framework outlined in Fig. 7 may be followed. Based on the experience from this and other studies with similar scope (Table 4), the information available from reports and statistics may typically have two very distinct levels of detail: food categories and/or food products. The food categories (e.g. meat, dairy, fruit and vegetables, etc.) comprise a number of food products. Food products consist of a further breakdown of these categories. Using the food categories resolution level may be enough when applying top-down life cycle approaches, i.e. using national input-output tables. When applying bottom-up life cycle approaches, the knowledge of the specific food products composing the mixed waste is necessary, as the LCI datasets are typically available per type of specific product (e.g. chicken). Ideally, if information on the specific food products composing the mixed food waste is available, this should be used. If this is not available, the compositional data on the specific food products could be derived departing from the food categories on the basis of the production (for Processing sector) and of the consumption patterns (for Wholesale and Retail, Food Service, and Households sector) for the region under assessment, using top-down national statistics. The assumption behind this is that the shares of products wasted are proportional to the production or consumption pattern. This approach was applied and exemplified in the present study for the UK case, as the breakdown of the specific food products was not available.

Once the breakdown of the food products is obtained, it is also necessary to detail the specific chemical/physical/biochemical properties of these food products in order to perform mass/substance/energy balances because the performance of the waste management technologies is affected by the biochemical and physical composition of the feedstock (e.g. biogas production or heating value depends upon biochemical and physical properties). These properties could be based upon detailed food datasets, as in this study, or roughly estimated based on generic vegetable and animal food waste chemical composition data as provided in Riber et al. (2009) or in ECN (2013). This latter pathway was applied, for example, in the study of Martinez-Sanchez et al. (2016) and Brancoli et al. (2017). In the present study, we provide a ready-to-use datasets of chemical, physical, biochemical, and nutritional composition for a number of typical food products elaborated on the basis of the Danish food product database (DTU National Food Institute, 2017). This can be downloaded from the Supporting Information of this article and used in any LCA-tool that enables the user to define a matrix of material biochemical and physical composition (e.g. Clavreul et al., 2012).

Yet, knowing the composition of the food does not provide the full picture of the mixed waste composition as food waste often comes along with packaging, e.g. in retails, food service or households. In order to know the amount and type of packaging, primary data should be used when available as illustrated in Fig. 7, e.g. from waste characterization analyses based on sampling campaigns. This was applied in the study of Bernstad and Andersson (2015), for example. Alternatively, the type (i.e. cartons or polyethylene) and the amount of packaging for each food product can be estimated using generic figures provided in literature. This pathway was applied in this and other studies, e.g. Brancoli et al. (2017), as providing a detailed insight into the packaging material composition was beyond the scope. However, improved knowledge of the composition of the packaging generated along with the food waste is important for studies assessing and comparing the impact of individual packaging materials, for example in the endeavour of optimizing product design and end-of-life.

4.2.2. Modelling land use changes

The inclusion of these, particularly iLUC, is crucial to the LCA results, as shown in this study and learned from the extensive literature on bioenergy (e.g. Hamelin et al., 2014; Searchinger, 2010, 2008; Edwards et al., 2010; Tonini et al., 2017, 2016a, 2016b; Wenzel et al., 2014). From a methodological perspective, the impacts from iLUC may be quantified using the figures provided with dedicated biophysical/deterministic models or with economic models. The first are based on linear elaboration of global statistics regarding crop production (e.g. FAO, 2017), deforestation (e.g. FAO, 2010) and fertilizers consumption (e.g. IFA, 2017). They already account for the fact that the response to an additional demand for a food product consists of a combination of expansion into natural vegetation and intensification of current production practices. This ratio is typically calculated on the basis of historical data, although future series may also be applied using the same maths. For example, Tonini et al. (2016b) calculated that the response to a demand for one additional hectare of arable land consists of 25% contribution from expansion and 75% contribution from intensification, using global agricultural statistics for the period 2000–2010 (dry mass basis). The iLUC inventory was then derived as a sum of the effects related to expansion (accounting for the biomes subject to deforestation) and to intensification (accounting for increased yields and use of mineral NPK fertilizers). An alternative approach is to simulate market-mediated effects of increasing food waste (thus production) using economic equilibrium models, in order to identify the areas affected by deforestation/conversion to cropland. This should then be combined with information on the carbon stocks for vegetation and soil in the regions subject to agriculture expansion. This type of analyses has been done for selected crops, mainly in the context of biofuel mandates (e.g. Edwards et al., 2010; Kloeverpris et al., 2010; Valin et al., 2015) or to estimate land use effects of changes in diet, for example in Tukker et al. (2009). While there is no agreement in the LCA community on the choice of the method to derive iLUC impacts, it should be borne in mind that the use of economic models may contrast with the basic principle of "full elasticity of supply" in consequential LCA, as thoroughly discussed in other publications (Schmidt et al., 2015). One of the main issues related to the use of these models is the fact that they account for short-term food price variations, assuming consequent changes in the consumption (e.g. decrease in food demand). Accounting for these fluctuations may be relevant in the context of policy scenario analyses to simulate short-term effects, but it is in contrast with the principle of the full elasticity of supply of consequential LCA as stressed in Weidema et al. (2009). Regardless of the method used, the case study presented in this research illustrates the crucial importance of including iLUC contributions when assessing food waste environmental impacts.

4.2.3. Avoiding double counting

When assessing food waste impacts, it is necessary to pay attention to the following potential double counting: (i) LUC emissions and (ii) waste management emissions. Regarding the first, some LCI datasets already contains LUC CO_2 emissions in the inventory for crop or, more in general, animal/food farming (e.g. swine, cow, fruit, etc.). In ecoinvent v3.3 (consequential system; Wernet et al., 2016), for example, LUC CO_2 emissions (for LUC, only CO_2 emissions are indeed considered) are quantified according to the approach described in the related report (Moreno Ruiz et al., 2014). This approach considers the historical land use changes occurring in the market where the crop/food is produced, which is a geographically-defined market, e.g. soybean produced in Spain, or in Brazil, etc. These changes include local transformations, i.e. direct land use changes, occurring due to the displacement of certain crops to make space for the one under assessment (e.g. for soybean), within the geographic region under question. Indirect effects occurring elsewhere, outside the country borders, are not included as well as intensification effects (i.e. no iLUC). In other words, this represents a sort of crop-specific and country-specific LUC effect (accounting for expansion-only), which only accounts for what has historically happened in relation to the crop under assessment in the country under question. To avoid double counting with iLUC, the approach followed in this study was to neglect these emissions and apply instead only iLUC inventories reflecting the upstream impact of demanding land, regardless of the use of this land (i.e. the type of crop involved and where). Such approach has been detailed earlier (Schmidt et al., 2015; Tonini et al., 2016b).

With respect to waste management, it should be borne in mind that generating avoidable food waste at one stage of the food supply chain means that the efficiency of the food supply chain is less than 100%, i.e. the next sector in the chain should take this efficiency factor into account as earlier illustrated in Fig. 1. Ecoinvent v3.3, for example, accounts for 12% losses for fruit and vegetable between production and consumers, i.e. during distribution, this encompassing transport and wholesale/retails, based on the global average figure provided in the FAO report (Gustavsson et al., 2013). If, such as in this study, the wholesale/retail sector is part of the boundary and the associated waste management is known and modelled, the default distribution loss (e.g. 12%) given by the LCI dataset should be cancelled out when modelling the impact of the following sector in the chain (e.g. Households or Food Service) to avoid double counting, and the actual efficiency of the system ("food wasted over food produced or sold", for example at Wholesale & Retail) should be used on the basis of the actual information available. Accordingly, if the waste management is known (i.e. partitioning of the food waste among incineration, landfilling, anaerobic digestion, composting, sewer, etc.), the associated environmental impact should be applied when modelling the food waste scenario of a sector placed later in the food supply chain (refer to Fig. 1).

4.2.4. Data uncertainty

On the basis of this and previous studies, the scenario uncertainties causing major variations in the results are: the choice of the datasets used to model food production and the composition of the food waste, in terms of specific food products composing the mix. Regarding the first, we discussed earlier the importance of this choice on the final magnitude of the results. The higher impacts of some studies are explained by the background datasets used to model food production. The importance of the second was highlighted in Fig. 6: this, albeit assuming extreme variations of the food waste composition, nevertheless illustrates the importance of obtaining solid information on the food products composing the mixed waste. Concerning parameter uncertainty, the results of this study highlighted that: (i) ratio intensification/expansion for iLUC, (ii) variation of food production impact, (iii) share of food waste cooked, and (iv) energy consumption for cooking represent the most important contributors to the overall parameter uncertainty. The uncertainty related to the iLUC ratio could be decreased by narrowing the range of variation (here conservatively assumed 0–100%), e.g. by elaborating different historical time series and coming up with a more defined range of variation. The second could be both seen as a parameter or a scenario uncertainty, depending on whether the user assesses the related uncertainty by assuming a range of variation and propagating the error, as in this study, or with a sensitivity analysis, e.g. by using a totally different dataset to model food production. No study, the authors are aware of, thoroughly assessed the uncertainty of food production in one way or the other. It should also be borne in mind that up-to-date consequential datasets for food processing (e.g. slaughtering, filleting, juice extraction, etc.) are not yet widely available. In this study, for example, we mostly relied on processing datasets adapted from 2-0 LCA consultants (2007), as ecoinvent v3.3 only provides data up to the farming stage for the majority of the food products. This is therefore an aspect that may be improved in future studies. The share of food waste undergoing cooking and, to a minor extent, the associated energy expenses are also very important in relation to the overall uncertainty. The information about the portion of food waste cooked may be obtained with observations based on sampling campaigns as in Bernstad and Andersson (2015) or by relying on estimates as in this study (Quested and Johnson, 2012). Yet, specific information and literature on this is generally scarce; this represents therefore an area where further research is needed to improve the quality of the data used in the assessment. Same goes for the energy spent for cooking, for which a large variation exists in the available literature data.

4.3. Hotspots for environmental improvement and policy

The results of the study highlight that, environmentally, food production and land use changes are the most important contributors to the impact. Prevention and minimization strategies incurring reduction in food and land demand, therefore, are likely to outcompete any other solution offered by end-of-pipe waste management technologies. This is in agreement with most of the previous studies on the subject, e.g. Bernstad and Andersson (2015), Gentil et al. (2011), Oldfield et al. (2016), Scholz et al. (2015), and Song et al. (2015). Yet, these findings are true only under the assumption that monetary savings from unpurchased (prevented) food are spent for activities or goods having lower environmental impacts than the prevented food waste as pointed out in Martinez-Sanchez et al. (2016). These so-called indirect effects may reduce or ultimately cancel out the environmental savings of prevention, thus highlighting the crucial importance of how tools such as subsidies, taxes, or education campaigns should be used in order to stir consumer's behaviour and avoid shifting the environmental impacts to other sectors. In order to achieve substantial environmental savings, prevention measures should particularly focus on the households, this being the stage of the supply chain having the highest environmental impact owing to the large amount of avoidable food waste generated. This was highlighted in this research for the case of UK, but also in many other studies, among the others Östergren et al. (2014) and Chapagain and James (2011). Besides prevention, other handling strategies such as redistribution and use for animal feeding may as well achieve substantial environmental savings, as they also partly induce savings of food production and corresponding land, similarly to prevention. Examples of this may be found in recent studies from Brancoli et al. (2017) and Eriksson et al. (2015, 2016) for the case of supermarket food waste, although the iLUC savings from avoiding feed production were only accounted for in Brancoli et al. (2017). In this context, when assessing scenarios of redistribution or use for animal feeding, both incurring land use change effects, including the related iLUC impacts appears desirable in order to capture the full environmental consequences. Finally, the results of this study also emphasize the importance of phasing out landfilling and instead promoting strategies for energy and nutrients recovery from food waste, primarily anaerobic digestion. The benefits of this are evident from the savings incurred by waste management when landfilling is not the prominent option, as highlighted in this study for the case of the processing and wholesale/retail sectors. Such results are aligned with previous literature. In particular, a recent and comprehensive analysis of the benefits of anaerobic digestion strategies for UK may be found in Styles et al. (2016).

5. Conclusion

The impacts of avoidable food waste were quantified for ten environmental categories for the case of UK. The C-footprint ranged from 2000 to 3600 kg CO_2-eq. t^{-1}, depending upon the food waste composition. This figure is generally in the higher end of the results found in previous studies because of the inclusion of indirect land use changes. Food production and indirect land use changes were highlighted as the largest contributors to the environmental burdens from food waste. Food preparation was also found to be a significant contributor to the environmental impacts, while waste management partly mitigated the overall impacts by incurring significant savings when landfilling was replaced with anaerobic digestion and incineration. The results emphasize the importance of prevention and minimization strategies to achieve substantial environmental improvements, as these are mainly connected to decreasing production and demand for land. To further reduce the uncertainty of the results, it is recommended to focus the effort on providing improved data regarding the breakdown of the food products composing the mixed waste, iLUC effects, food production datasets, and the share of food waste undergoing cooking.

Disclaimer

The views expressed in the article are the sole responsibility of the authors and in no way represent the view of the European Commission and its services.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at https://doi.org/10.1016/j.wasman.2018.03.032.

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