Country-specific life cycle inventories for human excretion of food products

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Abstract

Purpose A regionalized life cycle inventory model to account for human excretion of food products is presented, constituting an update of a previous model published by the author in 2008.

Methods The updated model provides country-specific estimates on toilet activities (use of toilet paper, tap water, soap) and wastewater/excreta management, the latter addressing not only wastewater collection and treatment as done in developed countries, but also decentralized options such as septic tanks, latrines, and open defecation. The model currently supports inventories for 92 countries, linked to the ecoinvent database. The model is tested here with three products (banana, breadcrumbs, cheese) in five countries (Bangladesh, Denmark, Niger, Israel, USA), where the results for human excretion, per kg food ingested, are compared to food production in terms of global warming and aquatic eutrophication.

Results and discussion The results show that besides a wide geographical variability, the environmental impacts of sanitation linked to food consumption have a higher magnitude than previously anticipated. In global warming, human excretion impacts can be of similar and even higher magnitude than food production, as far as products with a low carbon footprint are concerned, such as bananas. This relevance decreases for products with a higher carbon footprint such as cheese. Regarding aquatic eutrophication, the results suggest a potentially high relevance for human excretion in countries with poor sanitation, while the opposite holds true in countries with advanced treatment of wastewaters.

Conclusions We stress the importance of including human excretion in LCA studies of food products, especially when the goal is to identify life cycle hotspots or to assess dietary shifts.

Keywords Human excretion · Sanitation · Inventory model · Regionalization

1 Introduction

The end-of-life stage of food products inevitably leads to environmental impacts, in the form of emissions from human digestion and its derived excretion products. Following digestion, the human body not only releases CO_2 and small amounts of methane to the atmosphere (Bond et al. 1971), but it also releases liquid and solid excreta that needs management. In a developed-country context, excreta are typically managed through centralized sewers and wastewater treatment plants, while in developing countries human

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waste is often discharged with basic or insufficient treatment. According to UNICEF-WHO (2020), over half of the world's population, 4.2 billion people, use sanitation services that leave human waste untreated, threatening human and environmental health, while an estimated 673 million people have no toilets at all and practice open defecation.

The need to include human excretion of food products in life cycle assessment (LCA) was first mentioned by Andersson (2000), while simple quantitative approaches, mainly focusing on the fate of nutrients in food, were introduced by Ziegler et al. (2003) and Sonesson et al. (2004). It was not until 2008 that a full model was developed for this purpose by Muñoz et al. (2008). In this model, the nutritional composition of a specific food item or diet, defined as its content in water, carbohydrates, fat, protein, etc., is used to provide a mass balance for nutrients in the human body, direct emissions to the environment from the latter and an inventory for management of human excreta through toilet use and wastewater

treatment, using the wastewater treatment model from version 2 of the ecoinvent database (Doka 2007). Although this life cycle stage is commonly omitted in most food LCA studies, the model by Muñoz et al. (2008) has successfully been used in case studies assessing both specific food products, such as alcoholic drinks (Weidema et al. 2016), fish (Vázquez-Rowe et al. 2014), bananas (Roibás et al. 2016), beef (Ledgard et al. 2016), kiwi fruit (Mithraratne et al. 2008), and diets (Muñoz et al. 2010). Outside the field of LCA, the model has also been used to support material flow analyses (Krausmann et al. 2018) and input–output analyses (Merciai and Schmidt 2018).

A major shortcoming of the human excretion model by Muñoz et al. (2008) is the fact that as far as excreta management is concerned, it reflects the typical conditions in a developed country in terms of toiletries consumption, and most notably, wastewater management through treatment in a modern wastewater treatment plant (WWTP). As already mentioned in the first paragraph, insufficient access to basic sanitation is still today the daily reality for more than half of the world's population, making this model a poor representation of such conditions.

In this article, we present an update to the model by Muñoz et al. (2008), in which toilet use and the management of human excreta evolve from a static Western scenario to country-specific conditions, allowing us to reflect more accurately the range of environmental impacts from human excretion in different parts of the world with completely different sanitation realities.

2 Methods

Figure 1 shows a flow diagram of the activities included in the model.

2.1 Input data

The reference flow in the human excretion model is 1 kg food, as ingested. The necessary data to obtain country-specific LCIs for a given food item or diet include the following:

- Food/diet nutritional composition: as in Muñoz et al. (2008), the composition of the assessed food needs to be defined on a mass percentage basis, including water, carbohydrates, fat, protein, fibre, alcohol, organic acids (such as acetic acid in vinegar), and phosphorus. This is readily available information in nutrition databases, such as USDA's FoodData Central (USDA 2021) or the Danish Frida (DTU 2021).
- Defining a geographical scenario, referring to the country where the food is consumed. This choice automatically links in our new model to country-specific data on toilet use (see Sect. 2.3) and faecal water discharges (see

Sect. 2.4). Currently, the model stores such data for 92 countries.

2.2 Human excretion mass balance

Muñoz et al. (2008) developed a steady-state mass balance for food excretion, in which food constituents are classified into four input groups: water, degradable organic matter (fat, protein, carbohydrates, alcohol), non-degradable organic matter (fibre), and inorganics (phosporus, potassium, etc.). Each of these groups then contributes to three main output flows: respiration (containing mainly CO_2 and water), urine (containing mainly water, urea and inorganics), and faeces (containing water and organic matter). The contribution of each input group to the different output flows is based on average physiological data, such as for example how does water typically partition between respiration, urine, faeces, etc., and in the case of degradable organic matter, it is based on stoichiometric relationships.

The above-described mass balance calculations remain unchanged in this update. This assumes that the same partitioning of nutrients to air and excreta from ingested food is valid regardless of the geography, i.e. that an average citizen in Denmark, Ghana, or Thailand will excrete the same amount of CO_2, urine, and faeces after ingestion of the same quantity of the same food item. It is well known that in developing countries, solid excreta production per capita is higher than in developed countries (Rose et al. 2015); however, this can be attributed mainly to dietary differences, such as a higher fibre intake, rather than to differences in inherent nutrient absorption by different populations.

2.3 Toilet use

Activities related to toilet use are updated by providing country-specific estimates for consumption of the following items:

- Toilet paper.
- Tap water for toilet flushing and hand washing.
- Soap for handwashing.

Other toilet-use activities included by Muñoz et al. (2008) are disregarded, leaving them out of the inventory. This affects hand drying through the use of a towel or a hand drier and washing of the towel after a certain number of uses.

Data on per-capita-year toilet paper consumption was only found for 10 countries, in Statista (2018). An alternative source of data on this topic is the European Tissue Symposium (European Tissue 2013a, 2013b, 2013c, 2016);
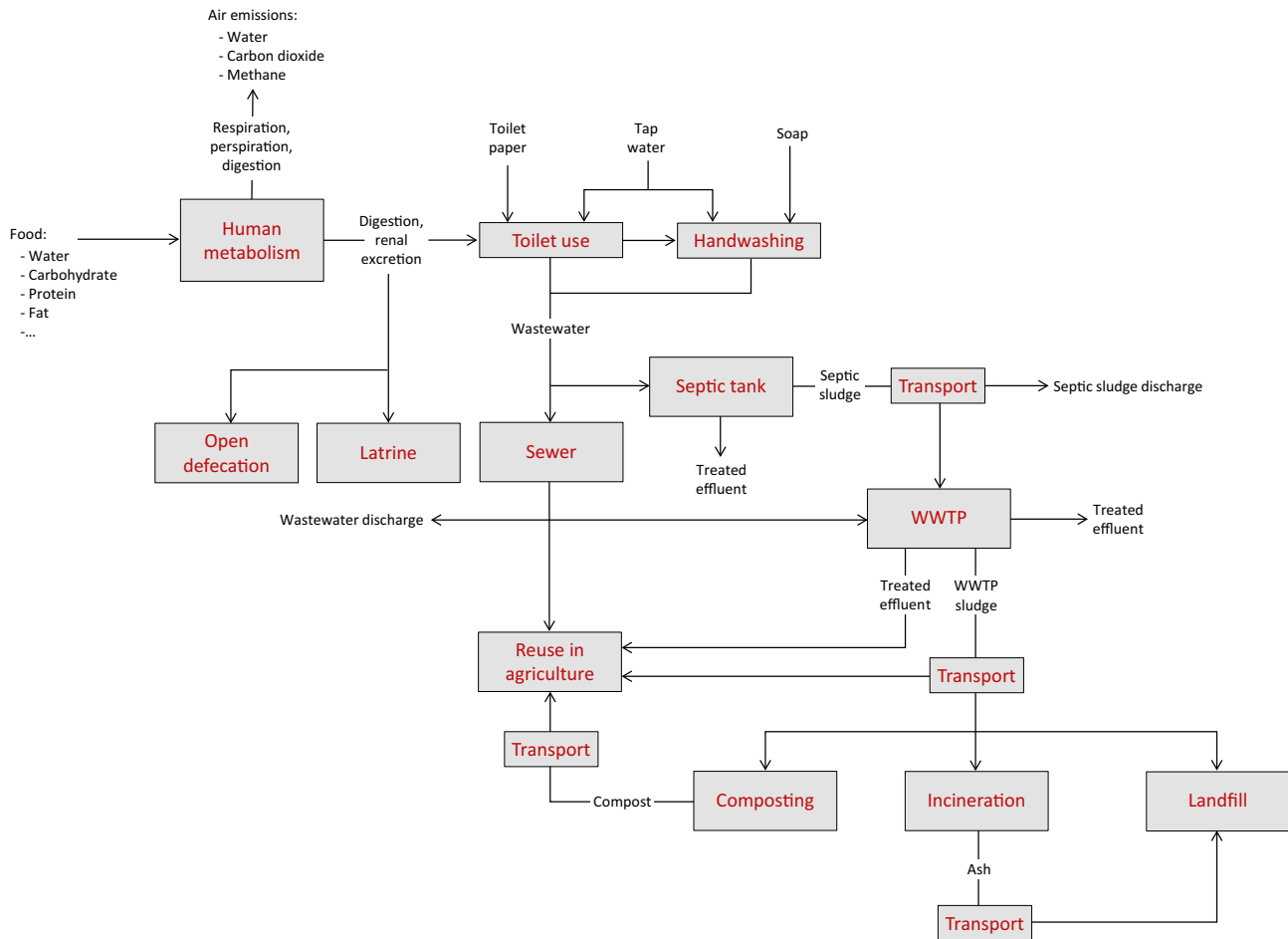
Fig. 1 shows a flow diagram of the activities included in the model

however, consumption figures are aggregated including all uses of tissue paper, of which toilet paper is only one of them. Based on the data for 10 countries in Statista (2018), which are also included in the European Tissue figures, toilet paper constitutes 46 to 88% of all tissue paper consumed, depending on the country, with an arithmetic average at 60%. In the model, this average figure is adopted for all countries reported in the European Tissue data set. With this estimate, a total of 49 countries can be covered. For all other countries, per-capita-year consumption is estimated based on a correlation between toilet paper consumption in Statista (2018) vs. gross national income (GNI) per capita. The correlation is fitted to a power function as shown in Eq. 1:

$$TPC = 0.0147GNI^{0.6113} \qquad (1)$$

Where TPC is toilet paper consumption, in kg/person-year, and GNI is expressed in US dollar, obtained from the World Bank Open Data (World Bank 2021). TPC can then be expressed in kg/kg human excreta based on a global

average production of solid and liquid excreta of 1.65 kg/person/day (constituted by 0.15 kg faeces and 1.5 kg urine), as originally considered by Muñoz et al. (2008).

Tap water and soap consumption per kg human excreta in Muñoz et al. (2008) was set to 33 L water for toilet flushing, 4.5 L water for handwashing, and 10 g soap for handwashing. In the model update, these same values are applied, but depending on the population's access to different levels of sanitation, as follows:

- Populations with access to toilets are assumed to use tap water for flushing and handwashing, as well as soap.
- Populations with access to basic hygiene, defined by the WHO-UNICEF (2021) as availability of a handwashing facility on premises with soap and water, are assumed to use tap water and soap for handwashing, but no tap water for flushing.
- Populations without access to basic hygiene are assumed not to use tap water and soap at all.

A total of 100% of the population in advanced economies, as defined by the IMF (2020), is assumed in the model to have access to toilets. For the remaining countries, the percentage of population with access to toilets is taken as 100% minus the percentage of population with no access to sewer or an independent wastewater collection system such as septic tanks. Data on such access is obtained from the database in the WW LCI model v4 (see Sect. 2.4). Further, the percentage of population with access to basic hygiene ($\%_{access}$) is obtained, whenever available, from WHO-UNICEF (2019). Some countries, though, are not covered by this source. In such cases, $\%_{access}$ is obtained with Eq. 2, which correlates this access to the human development index (HDI), using a sigmoid curve:

$$\%_{access} = \frac{1}{1 + \left(\frac{HDI}{0.6206}\right)^{-7.4788}} \tag{2}$$

The function is obtained by regression of known data on $\%_{access}$ in WHO-UNICEF (2019) vs HDI data from UN (2020) for a total of 72 countries.

2.4 Management of human excreta

In this update, the WWTP model by Doka (2007) is no longer used. Inventories for the management of human excreta are instead obtained with the model WW LCI (Muñoz et al. 2017; Kalbar et al. 2018), developed to calculate life cycle inventories for chemical pollutants in urban wastewater discharges. WW LCI covers the entire supply chain for wastewater management: collection, centralized treatment at various levels (primary, secondary, tertiary), decentralized treatment (septic tanks), emissions from untreated discharges, as well as sludge treatment and disposal by means of optional anaerobic digestion, composting, reuse in agriculture, controlled and uncontrolled landfilling, incineration. Its recently released version 4 (Muñoz 2021) introduces additional processes of particular interest regarding management of human excreta in developing countries, such as treatment in waste stabilization ponds, decentralized management through latrines and open defecation. Besides life cycle inventory calculations, the model is equipped with a database containing wastewater management and sanitation statistics for 92 countries, representing 92% of the world's population.

In WW LCI, the composition of pollution in wastewater is defined at the level of individual pollutants, which are described in terms of chemical composition, physical–chemical properties, and degradability, among other parameters. In order to express human excreta as an input of chemical pollution in WW LCI, excreta are broken down into a set of eight individual components, namely:

- Water
- Urea (containing all nitrogen initially in proteins)
- Faeces
- Fibre
- Phosphate (containing all phosphorus initially in food)
- Sulfate
- Toilet paper
- Soap

Table 1 Components in wastewater/human excreta generated by human excretion and toilet use

Component	Chemical formula	Form in wastewater	Degradability in WWTP	Degradability in environment
Water	H_2O	n.a	n.a	n.a
Urea	CH_4ON_2	Dissolved	Yes[d]	Yes
Faeces	C_2H_4O[a]	Suspended	Yes[e]	Yes
Fibre	$C_6H_{10}O_5$[a]	Suspended	Yes[e]	Yes
Phosphate	PO_4	Dissolved	n.a	n.a
Sulfate	SO_4	Dissolved	n.a	n.a
Toilet paper	$C_6H_{10}O_5$[b]	Suspended	Yes[e]	Yes
Soap	$C_{18}H_{35}NaO_2$[c]	Emulsion[f]	Yes[e]	Yes

n.a. not applicable

[a]Same composition as in Muñoz et al. (2008)

[b]Assumed same as fibre (cellulose)

[c]Assumed as sodium stearate

[d]90% degraded in WWTPs with activated sludge

[e]60% settled and 30% degraded in WWTPs with activated sludge, 60% settled in septic tanks

[f]Treated as suspended matter

Table 1 shows key parameters used in WW LCI to characterize each of the above components. The chemical composition of some of these components such as water or urea is straightforward, but for others, an approximate composition is taken. This is the case for faeces, for which the approximate composition used here was already suggested in Muñoz et al. (2008), while for toilet paper, the composition can be fairly represented by cellulose fibres. Regarding soap, sodium stearate is used as an approximation for a surfactant in a typical soap bar. In the real world, though, liquid soaps and soap bars are constituted by formulations including not only surfactants, but also preservatives, dyes, and fragrances, among others. Thus, the approach used in the model can be seen as a rough approximation.

Regarding fate in WWTPs and in the environment, urea, faeces, fibre, toilet paper, and soap are assumed to be readily degradable. Urea is dissolved in wastewater and therefore can only be removed by WWTPs applying biological treatment, where 90% removal of urea is considered. Urea breaks down generating ammonium, which is present in the effluent of such WWTPs. Water and sulfate in WW LCI are assumed to be entirely discharged in the treated effluent of WWTPs, while for ammonium and phosphate, this depends on the treatment level. WWTPs with tertiary treatment in WW LCI include N and P removal by nitrification–denitrification and chemical precipitation, respectively. In such plants, N is partly emitted in the effluent as ammonium and part as nitrate, while part is lost to the atmosphere, mainly as N_2 gas. Regarding N_2O-N emissions, these are estimated in all WWTPs with biological treatment as 0.5% of the N input. Faeces, fibre, toilet paper and soap are modelled as pollutants in suspended form, where 60% is assumed to be partitioned to sludge in WWTPs with at least primary treatment and in septic tanks, while 30% and 10% is assumed to undergo degradation and effluent discharge, respectively, in WWTPs with biological treatment. Whenever organic matter reaches the environment via either treated/untreated effluent or sludge, WW LCI includes the emissions to air and water from degradation of this organic matter, such as carbon

dioxide, dinitrogen monoxide, methane and nitrate, based on chemical composition. In the particular case of methane, emissions are particularly affected by the discharge scenario, including such aspects as climate, or entry compartment to the environment (some methane is expected from aquatic environments, but none from soils receiving sludge). For further details on WW LCI v4, the reader is referred to the model documentation (Muñoz 2021).

2.5 Life cycle inventory results

As in Muñoz et al. (2008), the outcome of the model is an LCI with a reference flow of 1 kg ingested food item. The LCI includes inputs of toilet paper, tap water, soap, oxygen for human metabolism, direct emissions to air from respiration (CO_2, CH_4, water, heat losses), and activities linked with human excreta disposal (wastewater treatment, sludge disposal, etc.). The entire background system in this model links to ecoinvent data sets, and the obtained LCIs can be exported in comma-separated value (CSV) format to the software SimaPro (Pré 2021) for further impact assessment calculations.

2.6 Case study with example food products

The applicability and potential relevance of the developed model in the life cycle of foods are tested in this article by applying it to several common food items and comparing their environmental impact at the human excretion stage with their corresponding impact from cradle to gate. Three products are chosen for which inventory data are readily available in the ecoinvent database v3.6, namely banana, breadcrumbs, and cheese. Nutritional composition for these products is obtained from the Frida database (DTU 2021), see Table 2. The sensitivity of the human excretion model regarding different countries is tested by assessing consumption of these food items in Bangladesh, Denmark, Israel, Niger, and the USA. On the one hand, the choice of food products is meant to reflect from low to high food processing requirements, with bananas on the low end and cheese in the high end. Regarding countries, these are chosen to reflect different levels of economic development as well as different sanitation realities, as shown in Table 3.

In parallel to the above analysis, the three mentioned products are also assessed with the model by Muñoz et al. (2008), in order to assess differences between the two models. This is done by linking the inventories obtained from the Excel file by Muñoz et al. (2008) to ecoinvent v3.6 data sets. In this case, only one LCI per food product is obtained. In the appendix, we provide the LCIs for all products and countries.

Table 2 Nutritional composition, in g/100 g, for banana, breadcrumbs, and cheese (DTU 2021)

Nutritional composition	Banana	Cheese	Breadcrumbs
Water	75.3	49.7	10
Protein	1.1	28.9	11.6
Fat	0.2	17.7	1.5
Carbohydrates	21	1	69.8
Fibre	1.6	0	6.9
Phosphorus	0.026	0.44	0.1

Table 3 Sanitation scenarios for faecal discharges according to the WW LCI v4 database

Country	Sewer to discharge	Sewer to WWTP			Septic tank	Latrine	Open defecation	Wastewater reuse in agriculture
		Primary	Secondary	Tertiary				
Bangladesh			5.2%		13.5%	81.2%	0.1%	
Denmark			2%	89%	9%			
Israel	2.2%	6.5%	37.6%	52.7%	0.9%	0.1%		98%
Niger					8.6%	20.1%	71.3%	
USA	5.5%	6.4%	28.5%	40.6%	19%			6%

All ecoinvent v3.6 data sets in this case study refer to the consequential system model in the database. Production of bananas and cheese reflects the global market mix in the database, after setting to zero subsequent transport of the product. For breadcrumbs, the data in ecoinvent reflect production in Western Europe. Regarding impact assessment, this is carried out at mid-point level for two impact categories, namely global warming and aquatic eutrophication. The former is assessed with the global warming potential for a time horizon of 100 years (GWP-100) according to the fifth IPCC's assessment report (Myhre et al. 2013), where CO_2 from biogenic sources is assumed to be neutral and the GWP-100 of methane is adjusted as explained in Muñoz and Schmidt (2016): 27.75 and 30.5 kg CO_2-eq/kg methane from biogenic and fossil sources, respectively. Aquatic eutrophication is assessed by means of the CML method (Guinee et al. 2001). We acknowledge that this method can be considered outdated; however, it is chosen in this case study in order to simplify the interpretation of results, given that it provides a single indicator that includes the contribution to eutrophication from emissions of chemical oxygen demand, nitrogen, and phosphorus. The human excretion model is in any case compatible with more recent methods such as those for marine eutrophication by Cosme and Hauschild (2017) and for freshwater eutrophication by Payen et al. (2021).

3 Results and discussion

In Table 4 we show, as an example, the LCI for human excretion of the three products in Denmark, per kg food ingested. It must be highlighted that this table does not include the inventory for managing the wastewater/human excreta produced. Complete LCIs are shown for all food products and countries in the appendix.

Figures 2 and 3 shows the results for human excretion of bananas, breadcrumbs, and cheese in the five chosen countries and two environmental indicators. The results are broken down into different contributing activities, as follows:

- Toilet paper: includes toilet paper production.
- Soap: includes soap production.
- Water supply: includes the production and supply of tap water.
- WW and sludge: includes all activities related to wastewater and sludge management (infrastructure, energy use, etc.), except direct emissions specified below.
- Methane: includes direct emissions of methane from wastewater and sludge management.
- N2O: includes direct emissions of dinitrogen monoxide from wastewater and sludge management.
- CO2: includes direct emissions of carbon dioxide from wastewater and sludge management (CO_2 from biogenic sources is neutral as mentioned in Sect. 2.6).
- Air emissions: includes direct emissions of ammonia and nitrogen oxides from wastewater and sludge management.
- Water emissions: includes direct emissions to water from treated and untreated effluents, including leachate from landfills.

Table 4 Inventory for human excretion of banana, breadcrumbs, and cheese in Denmark

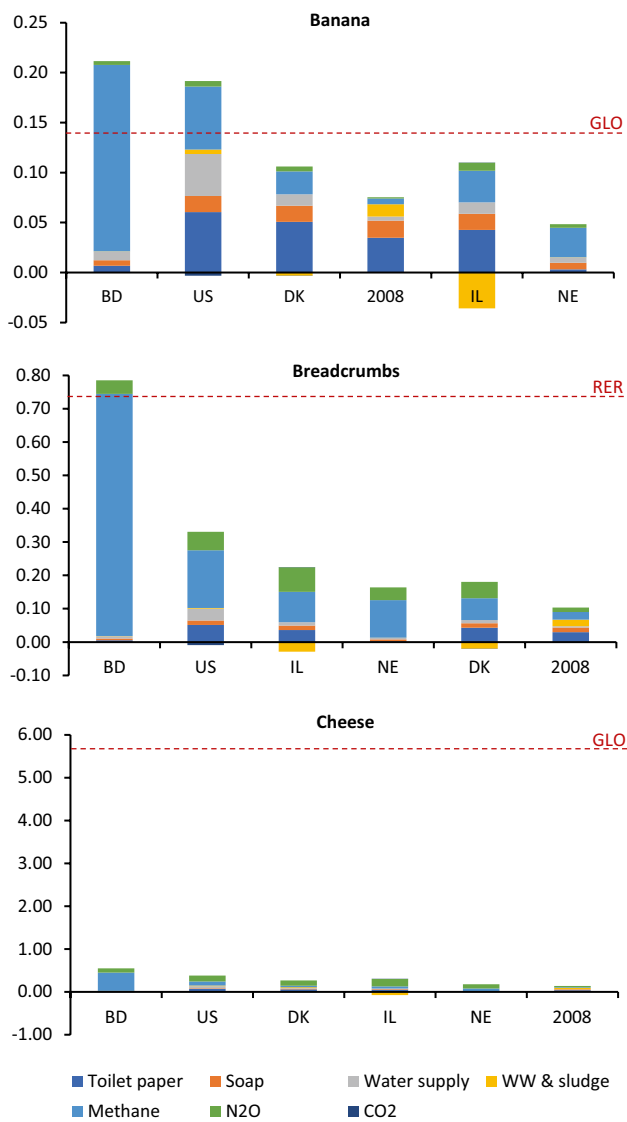|	Banana	Breadcrumbs	Cheese
Inputs from nature:			
Oxygen (g)	237	908	845
Inputs from technosphere:			
Food ingested (g)	1000	1000	1000
Toilet paper (g)	10	9	12
Tap water (g)	22,332	18,888	24,670
Soap (g)	6	5	7
Outputs to nature (respiration):			
Carbon dioxide, biogenic (g)	325	1,212	885
Methane, biogenic (g)	0.1	0.4	0.3
Water (g)	315	196	285
Outputs to technosphere:			
Wastewater containing human excreta (g)	22,938	19,402	25,348

Fig. 2 Results for global warming associated to human excretion, in kg CO$_2$-eq/kg banana, breadcrumbs, and cheese ingested in Bangladesh (BD), Denmark (DK), Israel (IL), Niger (NE), United States (US), and the 2008 version of this model by Muñoz et al. (2008). Dashed lines show the global warming results for banana as global (GLO), breadcrumbs in Western Europe (RER), and cheese as global (GLO)

3.1 Global warming

Figure 2 shows the results for human excretion of bananas, breadcrumbs, and cheese in the five chosen countries regarding global warming, expressed in kg CO$_2$-eq/kg food ingested. The figure also shows with dashed flat lines the global warming impact per kg of these products, from cradle to gate, according to the consequential version of ecoinvent v3.6. Figure 2 also shows the corresponding

results for human excretion according to the model by Muñoz et al. (2008), labelled as '2008'. All these results are also shown in the appendix in tabular form.

In the results for all three products in Fig. 3, we can observe first of all the variation in the total CO$_2$-eq figures. The ratio highest/lowest (excluding the '2008' results) is 3.1 for cheese, 4.4 for bananas, and 4.9 for bread, therefore substantiating the hypothesis that different sanitation realities lead to substantially different environmental impacts. For all products, Bangladesh consistently appears as the country with the highest emissions, and this is due to methane emissions originating in latrines and septic tanks, the main
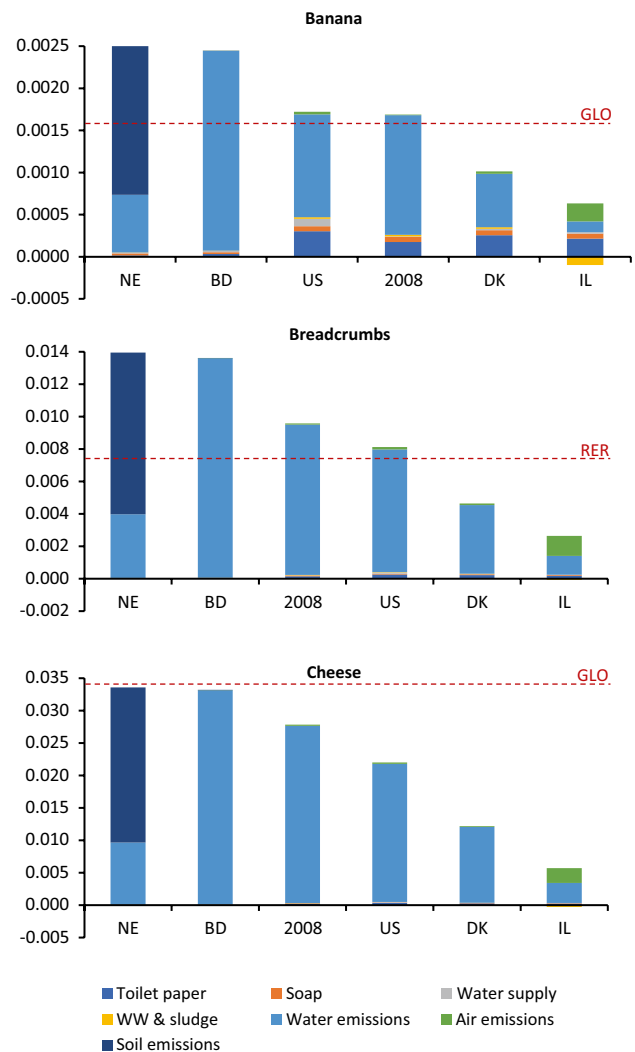

Fig. 3 Results for aquatic eutrophication associated to human excretion, in kg PO$_4$-eq/kg banana, breadcrumbs, and cheese ingested in Bangladesh (BD), Denmark (DK), Israel (IL), Niger (NE), United States (US), and the 2008 version of this model by Muñoz et al. (2008). Dashed lines show the eutrophication results for banana as global (GLO), breadcrumbs in Western Europe (RER) and Cheese as global (GLO)

sanitation options in this country. For completely different reasons, USA appears as the second country with the highest emissions in all products. This is due to higher inputs of toilet materials (paper, soap, water) and to emissions of methane from septic tanks, used by around 1/5th of the population. The lowest emissions for bananas and cheese correspond to Niger. This is explained, on the one hand, to relatively low toilet inputs associated to a low percentage of population with access to basic hygiene. On the other hand, the majority of the population practices open defecation, which involves no management of excreta and also low methane emissions. It is also worth noticing the case of Israel, where a substantial credit is obtained, associated to substituted seawater desalination thanks to wastewater reuse. In WW LCI v4, countries with high water scarcity, high income (and access to the sea), as is the case for Israel, freshwater from natural bodies is considered a constrained resource, whereby an increase in demand will induce supply from reverse osmosis desalination, considered as the marginal technology. Denmark also presents a credit, although of smaller magnitude, in this case associated to energy recovery from sludge incineration, which in WW LCI v4 is assumed to substitute natural gas. Finally, results for the 2008 model consistently fall in the lower end of CO_2-eq emissions in all three products, showing that a generic European scenario, as described by Muñoz et al. (2008) is in many cases likely to underestimate the real impacts of human excretion, as we see in all three products but especially in breadcrumbs and cheese.

Concerning the relevance of human excretion compared to food production in terms of global warming, we can see in Fig. 2 that this relevance tends to decrease as the impact of food production increases. In bananas, the product with the lowest global warming impact per kg among the three assessed, we can see that human excretion can be as relevant as food production. When a product with a higher carbon footprint, such as breadcrumbs, is considered, human excretion is only expected to become a hotspot in countries where sanitation induces high methane emissions, such as Bangladesh in our case study. Finally, when cheese is considered, human excretion seems unlikely to appear as a hotspot in the life cycle. We argue that this low relevance can be extrapolated to food products with a high carbon footprint, such as meat, but not for dairy products such as milk, yoghurt, or even eggs, which, although from animal origin, present a substantially lower carbon footprint (see, as an example, global warming impacts for these products consumed in Denmark, according to the Big Climate Database v1 (Concito 2021)).

3.2 Aquatic eutrophication

Figure 3 shows a set of graphs equivalent to those in Fig. 2, this time displaying the results for aquatic eutrophication. As

in global warming, the results show a considerable variation in the absolute PO_4-eq emissions, even bigger in this case. The ratio highest/lowest (excluding the '2008' results) is 4.7 for bananas, 5.4 for breadcrumbs, and 6.2 for cheese. On the one hand, the countries with the highest emissions appear to be Niger and Bangladesh, with approximately the same results in all three products. The main difference between the two is that while in Bangladesh, the main contribution comes from water emissions (mainly N and P) and in Niger, the main contribution (71% of the total score) is associated to soil emissions. This is due to the fact that 71% of the population, as seen in Table 3, practices open defecation, where WW LCI v4 labels emissions in human excreta (urine, faeces, etc.) as emissions to soil. Emissions of N and P to soil could be expected to have a lower eutrophication impact than the same flows emitted to water, see for example in the ReCiPe impact assessment method (Huijbregts et al. 2016). While we agree with this, the issue falls in the domain of life cycle impact assessment (LCIA), where such aspects can be tackled by using more sophisticated LCIA methods than CML. On the other hand, Denmark and specially Israel both appear as the countries with the lowest emissions. The two countries count with WWTPs treating most wastewater up to a tertiary level, which in WW LCI v4 is modelled as removal of N and P and sand filtration. The lower impact achieved in Israel is caused by the fact that virtually all its treated wastewater is reused in agriculture. This diverts emissions of N and P in the treated effluent from rivers and sea to soil; however, agricultural soil in WW LCI v4 is treated as part of the technosphere, where N and P in effluents discharged to agricultural fields are treated as intermediate flows subject to emission calculations similar to those conducted for fertilisers. It can be seen in Fig. 3 that as a consequence of wastewater reuse, substantial ammonia emissions are expected, labelled in the graphs as 'Air emissions'.

In Fig. 3, we can also see that the eutrophication results for the generic model by Muñoz et al. (2008), fall approximately in the middle of the range of scores by the different countries. Therefore, this model underestimates impacts for countries with poor sanitation, while it overestimates them in the opposite situation, in spite of this model representing a typical European scenario. A likely reason for Muñoz et al. (2008) overestimating impacts of countries like Denmark and Israel is the fact that the WWTP model by Doka (2007), representing the Swiss average plant in the early 2000s, presents a low N removal (25% reduction in total-N with respect to the influent), while modern plants applying nitrification denitrification typically achieve N removal levels above 75% (Von Sperling 2007).

Regarding relevance compared to food production in terms of aquatic eutrophication, the results prove that human excretion is a potential hotspot in the life cycle of any of the three assessed products. This is in agreement

with previous studies on broccoli (Muñoz et al. 2008), fish (Vázquez-Rowe et al. 2014), and the average Spanish diet (Muñoz et al. 2010). As an example, human excretion of bananas in Niger and Bangladesh involves a clearly higher impact than banana production. The same can be said for breadcrumbs produced in Western Europe. Even for cheese, the product with the highest eutrophication impact among the ones assessed, human excretion in Niger and Bangladesh shows the same magnitude as cheese production.

3.3 Discussion

The model presented in this article presents an advance when it comes to modelling a complete life cycle inventory of food products. It builds upon the model by Muñoz et al. (2008), which is now 13 years old. In this update, we still rely on its valid excretion mass balances, describing the fate and biochemical transformations of food in the human body as a function of a simple set of food descriptors (food composition), while we change what we consider to be outdated, namely the model for human excreta management. Further, the country-specificity of the life cycle inventories is achieved without burdening the user with additional needs on input data, thanks to the toilet use approximations presented in Sect. 2.3, plus the database on human excreta management incorporated in the WW LCI model v4.

As in Muñoz et al. (2008), we advocate for inclusion of human excretion as a life cycle stage in LCA studies. Our small test with a few products and countries has shown that human excretion can be a hotspot, and that the environmental impact of this life cycle stage shows a substantial geographical variation. While previous studies like that by Muñoz et al. (2010) on the average Spanish diet found that only eutrophication is likely to be of concern when it comes to human excretion, we have shown in this study that results for global warming, the 'quintessential' impact category in LCA studies, can be heavily influenced by human excretion, depending, as shown, on the product and on the geography. Thus, studies trying to understand life cycle hotspots, as well as those comparing dietary shifts, should include human excretion. Nevertheless, as pointed out in Muñoz et al. (2008), there may be situations where human excretion can safely be left out, for example studies assessing different production methods for the same product. However, some authors seem to disagree, and the reality is that, to date, most LCA studies systematically omit this life cycle stage. As an example, Svanes and Aronsson (2013), stated that: '…the human body is a part of nature and when the food is ingested the energy and materials are

returned to nature'. First, we do not see why humans should be seen any differently from any other species in the animal kingdom when it comes to mass and energy balances. LCA studies of beef and milk production systematically address the metabolism of cows, which leads to production of methane emissions as well as manure. Obviously, cows produce milk and beef, while humans do not produce, but consume, milk and beef. Yet, LCA is a tool to assess not only production, but also consumption choices. Second, although it is true that energy and materials are returned to nature, this is very often not a direct transfer, as it involves the building and operation of complex sanitation systems (sewers, WWTPs, landfills, etc.). All these activities fall within the technosphere and as such need to be included in the inventory analysis. Even in extreme cases when such infrastructure is lacking, as is the case for open defecation, our first argument still applies, i.e. emissions to nature must be registered in an inventory analysis, regardless of whether they originate in humans or cows, and the effects of such emissions, whenever possible, must be captured in the impact assessment phase.

4 Conclusions

We have presented a regionalized inventory model to account for human excretion of food products. The results of applying this model in a case study assessing three products (banana, breadcrumbs, cheese) in five countries (Bangladesh, Denmark, Israel, Niger, USA) suggest that besides a wide geographical variability, the environmental impacts of sanitation linked to food consumption have a higher magnitude than previously anticipated, at least for the two environmental impact categories assessed, namely global warming and aquatic eutrophication. In global warming, human excretion impacts can be of higher magnitude than food production, as far as products with a low carbon footprint are concerned, such as bananas in our case study. This relevance decreases for products with a higher carbon footprint such as cheese. Regarding aquatic eutrophication, the results suggest a potentially high relevance for human excretion in countries with poor sanitation, while the opposite holds true in countries with advanced treatment of wastewaters. As in Muñoz et al. (2008), we again stress the importance of including human excretion in LCA studies of food products, especially when the goal is to identify life cycle hotspots or to assess dietary shifts.

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