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Environmental impacts of different waste to food approaches

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All waste treatment options have environmental impacts. As waste to food is one of the many possible ways to valorise (or treat) food waste, environmental impacts of different waste to food processes need to be compared alongside other waste treatment methods. In addition, the environmental impact of the prevention of waste needs to also be compared to waste to food impacts. This chapter introduces the method of Life Cycle Assessment (LCA) to evaluate the environmental impacts of various production and treatment options. We highlight multiple methods to conduct environmental impact assessment, including a bottom up LCA, or a hybrid IO-LCA approach. We cover the drawbacks and limits of these different LCA methods. We highlight best practice waste to food environmental assessment case studies, including the REFRESH FORKLIFT toolkit. We intend for this chapter to be a broad introduction to this topics, empowering a decision maker or researcher to understand the processes, and limits of waste to food environmental impact assessments.

Key words

food prevention

food waste valorisation,

Input-Output,

Life Cycle Analysis,

waste hierarchy,

Waste to Food,

The environmental impacts of different waste to food approaches

Introduction

The disposal and treatment of food loss and waste causes the loss of organic resources and has a wide variety of environmental impacts due to the multiple processes involved in the products life cycle. However, much food waste still has economic or nutritive value, and by some definitions it could still be considered as a resource (Gedi et al., 2020; Thompson, 1979). Contemporary methods of food waste treatment, include incineration, composting, anaerobic digestion, animal feed, food rescue, the use of landfill and valorisation, each of these options lead to different environmental impacts.(Hoornweg et al., 2020; Salemdeeb et al., 2018, 2017a, 2017b; Schmidt Rivera et al., 2020)

Indeed, the valorisation of food waste into new food products is just one of the many methods available for processing food loss and waste and turning it into usable product (Burange et al., 2011). A broad definition of valorisation might include other waste treatment methods such as composting, anaerobic digestion or animal feed. However, in the context of this Waste-to-Food book we follow Gedi et al.'s narrower definition (Gedi et al., 2020) . Specifically, where technology and innovation are applied to create new methods for utilising available wasted resources (i.e. wasted food) to create *"new food, new by-products for food and feed formulations, new high value added food and feed components or other novel products from wasted food. This final option involves creating products for other sectors of the economy, where valorised materials could be utilised."*. This definition moves valorisation beyond the waste to energy paradigm, and into the space of waste to food.

To assess the environmental impacts of a product or a treatment method, we use a methodology called Life Cycle Assessment (LCA), which is a widely used systematic approach used to evaluating the environmental impacts of various production and treatment options. As a standardised method prescribed by the International Organisation for Standardization (BSI, 2006a, 2006b), LCA adopts a process-based modelling approach where a system is modelled using an inventory of processes representing inputs, outputs and potential environmental burdens of the system.

Assessing the environmental impacts of food waste valorisation options is complex due to the many possible (edible and non-edible) product and co-products that can be created from different food waste streams, as well as methodological issues like the goal and scope of the analysis (e.g. treatment of food waste vs production/recovery of enzymes from waste stream), functional units (e.g. mass of waste treated vs mass of enzymes produced)-each with their own environmental impacts, benefits and drawback (Östergren et al., 2018). However, as Waste to Food is a relatively new field of academic study, there are a limited number of studies on the environmental impacts of waste to food products; with not all possible combinations of food waste, process, and output product calculated (De Menna et al., 2020; Khoshnevisan et al., 2020; Lam et al., 2018; Östergren et al., 2018; Plazzotta et al., 2020; Torres-León et al., 2018).

In order for a decision maker to have full information, many LCA analyses need to be performed to compare potential avenues for food waste treatment, or different food waste valorisation options. This is because we need to compare the environmental, economic and social impacts of different valorisation options alongside other food waste treatment methods (including prevention, as well as other methods of producing comparison/comparative products).

LCA methods are used to understand the environmental impacts of waste to food options for a number of reasons: (1) compare the how different processes or technologies can produce more value-added products from a waste stream (e.g assessing green methods for pectin extraction from waste orange peels (Benassi et al., 2021)); (2) to compare different uses of a waste stream, (including alternative valorisation processes and waste treatment and prevention options , as well as other methods of producing comparison/comparative products (Brancoli et al., 2020; Davis et al., 2017; San Martin et al., 2016)); (3) compare and assess the environmental impacts of different valorisation process. This includes the identification of hotspots, and testing the influence of different conditions (geography, scale, etc) on the overall performance(De Clercq et al., 2019; Laso et al., 2016); and (4) to compare and assess the environmental impacts of producing a main product (e.g. anchovy fillets), considering the valorisation of waste generated as part of the production process and wider system boundaries (e.g., anchovy spines and heads into fish oil and meal, see (Laso et al., 2016)).

Due to the need to understand not only the environmental impacts of waste to food, but wider food waste treatment and prevention, this chapter will introduce LCA, describing how it links to the food waste hierarchy. We will then highlight the place of Waste to Food within the food waste hierarchy. Describe different approaches to life cycle analysis, highlighting their draw backs, and then provide case studies of current waste to food environmental impacts drawn from the available literature.

We also highlight that a good open access supplement to this chapter is the REFRESH projects' FORKLIFT (FOod side flow Recovery LIFe cycle Tool) valorisation spreadsheet tool, and accompanying documentation (REFRESH, 2020). As the FORKLIFT spreadsheet learning tool indicates generic life cycle greenhouse gas emissions and costs for using 6 key examples of unpreventable food processing co-products, by-products or wastes (apple pomace, pig blood, brewers spent grains, tomato pomace, whey permeate, and oilseed press cake). This can be worked through to show off some of the concepts discussed within this chapter.

The food waste hierarchy and LCA of waste to food.

To aid the selection of food waste disposal technologies, the EU provides guidelines on which disposal technologies are preferable (ReFood, 2013). This so-called food waste hierarchy stipulates that governments should prioritise efforts (in order of most to least preferable) to (1) reduce food waste, (2) redistribute it (e.g., to the homeless or recycle it as animal feed), (3) compost, (4) recover energy through anaerobic digestion, and finally, (v) landfill the remainder. Table 1 shows how different geographies and legal frameworks have slightly different versions of the food waste hierarchy.

A key research gap exists with regards to quantifying the environmental impacts of the food waste hierarchy (Bernstad Saraiva Schott et al., 2016; Van Ewijk and Stegemann, 2016). Food waste technologies (including waste to food valorisation) included in the food waste hierarchy are site and technology dependent (Laurent et al., 2014a). Therefore, without a solid and rigorous understanding of all stages of the food waste hierarchy on a local level, it would be an arduous task to identify best practices and maximise net environmental benefits. In order to demonstrate the environmental benefit of the food waste hierarchy, it is necessary to develop scientifically based tools and holistically evaluate all its stages in a consistent manner.

Quantifying the overall environmental impacts of each component of the food waste hierarchy is a crucial step in achieving a more comprehensive understanding of the issue at stake and developing effective strategies to reduce its environmental burden (Teigiserova et al., 2020). Although current literature is abundant with studies that have attempted to evaluate the environmental impacts of different components of the food waste hierarchy, the results reported in these studies fail to deliver a holistic evaluation of the environmental impacts of all components of the food waste hierarchy with coherence. This key limitation can be attributed to two factors: firstly, the majority of studies in the literature suffer from a high level of uncertainty and inconsistency in overall results (Bernstad and la Cour Jansen, 2012; Cleary, 2009); secondly, the selective approach taken in the majority of reviewed studies which focuses solely on specific components of the food waste hierarchy (usually downstream treatment options). These studies tend to overlook other options such as food waste redistribution, waste to food valorisation, and prevention due to the complex nature of models required in order to be able to quantify their environmental impacts.

Table 1 Food waste hierarchies in EU, UK, and USA

Priority	Waste framework directive	WRAP food and material hierarchy	Food waste pyramid	Food recovery hierarchy
	EU (EC, 2008)	UK (Downing et al., 2015)	UK (ReFood, 2013)	USA (USEPA, 2015)
Highest	Prevention	Prevention	Reduce	Source reduction
	Preparation for reuse	Optimisation	Feed people in need	Feed people in need
	Recycling	Recycling	Feed livestock	Feed livestock
	Recovery	Recovery	Compost and renewable energy via AD	Industrial use compost
Lowest	Disposal	Disposal	Incineration or landfill	Incineration or landfill

Life Cycle Assessment (LCA) is a systematic approach to quantifying and evaluating inputs, outputs, and potential environmental burdens of a product, a process, or an activity throughout its life cycle "cradle to grave", including all major stages (e.g. raw material extraction, manufacturing, use, and disposal) (BSI, 2006a). LCA consists of four different phases that are iterative and inter-connected as shown in Figure 1. Typically, the first phase is to define the goal and scope of the study, which is then followed by an inventory analysis. It can optionally include an impact assessment phase before finishing with an interpretation of the analysis results.

Figure 1 The general methodological framework for life cycle assessment, adapted from BSI (2006).

LCA has been increasingly used to evaluate the environmental impacts of food waste management options (Boldrin et al., 2011; Martínez-Blanco et al., 2010). Nevertheless, a review study by (Bernstad and la Cour Jansen, 2011) shows that the environmental impact of waste management options varies largely in literature. These significant variations could be attributed to several methodological and technical limitations associated with the use of LCA, despite being an ISO standardised and wide-spread approach. The lack of consistency amongst existing studies was also reported by Laurent et al. (2014a,b) who analysed 222 studies under a broader umbrella of global waste management and found that the lack of consistency plays a key role in influencing results reported in the literature. A summary of these limitations as well as advantages of LCA are presented in Table 2. These limitations have primarily led to the failure of LCA to draw comprehensive system boundaries around studied projects. Decisions on inclusion or exclusion of processes are usually arbitrary and often based on professional judgment rather than scientific reasoning. Therefore, cut-off criteria in system boundaries could potentially lead to system incompleteness, alternatively referred to as truncation error (Suh et al., 2004). Other system-specific shortcomings associated with the use of LCA to model food waste treatment options are provided in the following sections.

Table 2 Advantages and disadvantages of life cycle assessment.

Advantages	Disadvantages
• A process-based assessment method that quantifies potential environmental impacts associated with a product or product system through its life cycle. • A standardised methodology with wide availability of methodological and procedural guidance to aid the application of this method (BSI, 2006b; JRC, 2012) • the availability of various peer-reviewed LCI databases (e.g., ecoinvent and the European life	• A labour and time intensive method. • Entails a high level of uncertainty associated with the aggregation of results into different impact categories, particularly toxicity-related impacts (Finnveden et al., 2000; Pennington, 2001; Reap et al., 2008b). • Life cycle inventory data for specific processes is scarce. Previous studies tend to use data of similar processes to model technology-specific processes and/or scale-up/simulate

cycle database and the World Food LCA database). • This method has been adopted widely to develop policies and make strategic decisions (Finnveden and Lindfors, 1998; Finnveden et al., 2005) • LCA allows for comparisons between different products and product systems (Bernstad and la Cour Jansen, 2012) • The availability of several software applications and tools that make it easier to conduct a LCA analysis	pilot/laboratory data of new processes or technologies. • Generally, LCA models of waste management quantify environmental impacts per kg or tonne of waste managed. This implies that a) environmental burdens of the total quantities of waste generated are not affected by changes to management measures investigated (Ekvall et al., 2007). Moreover, when assessing valorization routes (waste to food, waste to energy), the environmental burden of the waste streams itself are considered zero, which does not promote or account for the reduction/prevention of food waste and does not offer reliable information for decision making processes (e.g. capital investment) (Slorach et al., 2020) • Involves a drastic simplification of a complex real system (Ekvall et al., 2007) • The difficulty to translate LCA results into clear recommendations that could be adopted by policy makers. This is due to the high level of assumptions associated with it.

Assessment of food loss and waste management using LCA

LCA relies heavily on both data and software applications. Reviews of LCA papers of 1) wider food waste treatment options (Salemdeeb et al., 2018, 2017a; Slorach et al., 2020), and 2) waste to food valorisation (Caldeira et al., 2020) both indicate that reviewed studies can not only differ in ranking but also in absolute values for the same methods of waste treatment of valorisation. For example, the global warming potential of incineration of food waste varies from +250 to -350 kg CO_2-eq. per tonne of food waste (Salemdeeb et al., 2018). The large range in reported results amongst technologies make it hard to settle on a generic conclusion that can be broadly applied. Therefore, a comparative conclusion between food waste treatment and valorisation methods cannot be made considering these large differences. A similar issue in ranking alternatives was identified between AD and composting, as well as between incineration and landfill in a meta-review study (Morris et al., 2013).

These disparities in reported results can be attributed to various factors and specificities associated with each LCA study (e.g., the adopted assessment methodology, data input and assumptions made as part of the analysis) but not due to the actual environmental impacts of processing food waste (Bernstad Saraiva Schott et al., 2016). A similar reasoning has been discussed by (Laurent et al., 2014a), who state that the highest contribution in results are due to the background system which is location and technology dependent. Overall, decisive factors that lead to these discrepancies in literature are:

1. Truncation error and differences in system boundary;

2. The inclusion/exclusion of capital goods; and

3. Technological choices (e.g., process efficiency or substituted products).

The first factor consists in the difficulty of drawing a comprehensive system boundary around the studied project, which ensures full inclusion of all key processes. The system boundary can be defined as the "unit processes that are part of a product system" (BSI, 2006a). In an actual food waste management facility, there are essentially numerous processes that need to be included such as operation stage, reject reprocessing and maintenance activities. Decisions on inclusion or exclusion of these processes are usually arbitrary and are often based on professional judgment rather than scientific reasoning. Therefore, truncation (cut-off) criteria in system boundaries could potentially lead to system incompleteness, alternatively referred to as truncation error (Suh et al., 2004). For example, (Eriksson et al., 2015) excludes the environmental benefit of the use of compost as a substitute for synthetic fertiliser. These decisions to exclude these processes undermine potential benefits of these options. The magnitude of this incompleteness varies according to the type of each system, but it can reach up to 50% (Lenzen, 2000; Suh et al., 2004). Moreover, results of studies by (Bullard et al., 1978), (Lenzen, 2000), and (Miller and Blair, 2009) suggest that even extensive process-based inventories for complex systems do not achieve sufficient system completeness. It would be ideal if truncation error could be reduced without the need to depend on the labour-intensive and time-consuming LCA approach. Although the issue has been identified in numerous waste-management-related studies (Cleary, 2009; Laurent et al., 2014a), there has been little discussion surrounding possible methods to tackle it.

The second factor that could lead to substantial underestimation of the total environmental burdens is the decision whether to include environmental impacts associated with "capital goods", a term commonly used to refer to goods, services and energy inputs required at the construction stage of a food waste downstream treatment facility. Generally, excluding the environmental impacts of capital goods is a common trend in most waste-management studies; Laurent et al., (2014a) conducted a review study that found that 88% of waste-management-LCA studies exclude capital goods. Even more worrying is that 26% of the studies claim that capital goods have insignificant ecological impacts (Laurent et al., 2014a, 2014b).

Only a few studies have examined the impact of capital goods. (Otoma et al., 1997) conducted the first study that highlighted the significant impact of capital goods in evaluating the environmental performance of waste management infrastructure. Otoma's work has encouraged more researchers to consider capital goods, estimating that environmental burdens associated with capital goods could cover up to 10% of the total life cycle impacts (Finnveden et al., 2005; McDougall et al., 2001). These estimates are supported by more generic findings from (Frischknecht et al., 2007), who showed that capital goods contribute substantially to abiotic resource depletion, climate change, and ecosystem damage.

Brogaard and Christensen, (2016) comprehensively quantified the environmental impacts of capital goods in waste management systems (Brogaard and Christensen, 2016). The methodology of the study is based on a conventional process-based LCA model and uses life cycle inventories of capital goods which were compiled in previous projects (Brogaard et al., 2015; L K Brogaard et al., 2013; Line K Brogaard et al., 2013). Reported results vary greatly depending on the technology and the impact category. For example, variations reported as follow: 1–17% for Global Warming, 0.05–99% for Freshwater Eutrophication, 10–92% for Human Toxicity-Cancer Effect and 1-31% for the Depletion of Abiotic Resources – Fossil. The most important contributions by capital goods were made by the high use of steel and concrete.

The third factor to be considered in modelling food waste downstream management options is the wide variety in technical specifications of modelled technologies. These variations could lead to differences in allocation and substitution rates or the type of marginal/average data used to model affected by the induced change to the system (i.e., marginal process) (Laurent et al., 2014a, 2014b).

Food waste prevention compared to Waste to food

There have been several endeavours to quantify the environmental benefits of food waste prevention (De Laurentiis et al., 2020). Unsurprisingly, an overarching consensus exists on the overwhelming environmental benefits of food waste prevention. (Bernstad Saraiva Schott and Andersson, 2015) have shown that food waste prevention yields greater benefits for GWP compared to modern downstream alternatives for food waste treatment such as anaerobic digestion and composting, with estimated benefits ranging from 1800-3300 kg CO_2eq. per tonne of food waste prevented. A similar study conducted in Germany estimated that 1200-3300 kg CO_2eq. could be saved as a result of the avoidance of wasting unconsumed food (Gruber et al., 2014).

In spite of the overall consensus on the substantial environmental benefits associated with food waste prevention activities, reviewed studies report environmental benefits that vary dramatically; as shown in Figure 2. Bernstad Saraiva Schott and Cánovas (Bernstad Saraiva Schott and Cánovas, 2015) concluded that these variations are largely explained by differences in system boundary delimitations and assumptions related to reduced food production.

Most recently Slorach et al (Slorach et al., 2020) calculated that a 20% or greater waste prevention was more efficient at reducing global warming impacts than any standard waste treatment method (AD prioritised), and so recommended a mix of both treatment and prevention. Comparisons of prevention of food waste with Waste to Food, are harder to find, with some studies (Davis et al., 2017; De Menna et al., 2020; Vandermeersch et al., 2014) providing guidance and frameworks for this type of assessment.

Figure 2 A comparison of the different estimates of GHG savings from avoiding (preventing) one tonne of food waste. The error bars illustrate the ranges reported in each study.

There are limitations of the current food waste prevention LCAs. Firstly, all of the reviewed studies were conducted using a process-based life cycle assessment approach and, therefore, inherit the widely discussed limitations of LCA such as system boundary, data inconsistency, study-specific scenarios and variations in the definition of the functional unit (see below). For instance, the majority of the studies ignore the fact that only a portion of food waste can be avoided while the remaining fraction, ranging between 30-40% of total weight, could be unavoidable). This would consequently lead to an overestimation of benefits associated with food waste prevention ((Bernstad Saraiva Schott and Andersson, 2015). Another example stems from the fact that food is an outcome of a series of complex and inter-connected activities such as agriculture, processing, manufacturing, storage, distribution and retails. Thus, modelling such a complex system leads to arbitrary decisions on whether to include or exclude processes and subsequent inconsistencies in the level of completeness.

The second key limitation that leads to these substantial discrepancies in reported results is the scarcity of data, particularly data used in the evaluation of the avoided upstream production impacts. Generally, reviewed studies use generic food production life-cycle inventory and secondary data sources to model avoided food production. For example, the (Gentil et al., 2011) study, based on a Danish context, is modelled on a UK-based study by Ventour (2008) while (Bernstad Saraiva Schott and Andersson, 2015) use 20 different sources to extract LCI data related to food production activities. None of the studies listed above, investigates or reports the level of uncertainty associated with data used to model the avoided upstream production impacts.

There is clear evidence that emissions associated with food products vary not only by geographical location but also by production method and the season of harvesting (in case of vegetables and fruit products) (Canals et al., 2008a; Poore and Nemecek, 2018). This could be attributed to different farming and production techniques that are greatly affected by local conditions (e.g., energy sources, weather conditions and feed sources). Additionally, variations in environmental impacts of the same product could be observed across different time spans for the same.

Another level of uncertainty is added as a result of the type of food products (whether organic or conventionally grown foods). Apart from (Bernstad Saraiva Schott and Andersson, 2015), none of the reviewed studies differentiate between the specific characteristics of respective farming methods or take into consideration differences between organic and conventional food products. The literature review is abundant with studies that quantitatively confirm the high level of variation in environmental impacts associated whilst comparing organic and conventionally-grown food products; their differences vary between -81 to +130% for fruits and vegetables, and -41% to +45% for arable crops (Meier et al., 2015).

Significant variations associated with the type of food products could be attributed to several factors. The first factor is the lack of reliable data sources as some of the studies used data taken from a small number of farms (<10) and then generalized their findings across different food categories. This practise is questionable and so are the results as representative of the farming system. Another aspect is the difference of yield in outputs: the production yield of organic-grown food products varies largely and therefore this might lead to huge variations. Finally, the majority of these studies adopt a conventional LCA method in the assessment process and therefore inherit LCA limitations (such as, truncation error, allocation issues), could play a significant role in these observed variations.

The second challenge in modelling food waste prevention is the dependency of the food basket on imports and the impact of international trade in a wider context. Therefore, and in order to achieve more accurate results, there is a need to account for the source of food product imports.

The source of food production (whether nationally or internationally) introduces a high level of uncertainty into the analysis of environmental impacts. Although domestic food products are usually assumed to have less environmental burdens than imported products (this may seem obvious at first glance due to the international transport of food), this is not the case as energy and emissions intensities of both production and international transport vary largely across countries and products (Canals et al., 2008b). Therefore, a need exists to adopt a disaggregated approach when modelling food products (and waste to food products) to account for both the type and the source.

The final factor that results in substantial variation in estimated benefits from preventing food waste concerns the question of the inclusion of the rebound effect: the avoidance of food waste in households leads to increased effective income and additional expenditure on alternative products and services (Binswanger, 2001; Brookes, 1990; Khazzoom, 1980). As this additional expenditure generates additional GHG emissions, the environmental benefits of minimizing food waste can be partially or completely offset. If the economic savings were to be spent on carbon-intensive goods or services (e.g. air travel or domestic heating), it is even plausible for food waste prevention to create higher environmental burdens than disposing of food waste via other waste management alternatives (Martinez-Sanchez et al., 2016).

The rebound effect associated with food waste prevention has not been addressed in literature except rarely – indeed no study calculates waste to food products with a rebound effect. A few studies looked at the impact of the rebound effect in a similar context (i.e., sustainable consumption and changing diets see (Alfredsson, 2004) and (Lenzen and Dey, 2002).

A handful studies evaluated the environmental impacts of food waste prevention, including the rebound effect, and compared it with other food waste disposal options (i.e., incineration, co-digestion of food waste with manure and animal feed). (Martinez-Sanchez et al., 2016)) show that although prevention provides the highest environmental gains, it could also incur the highest environmental burden if the monetary savings from unpurchased food commodities were re-spent on GHG-intensive goods and services. These results are even larger than those discussed previously (Alfredsson, 2004; Druckman and Jackson, 2008). Salemdeeb et al. (2017) estimated that including the rebound effect could reduce overall GHGs emissions attributed to waste prevention in the UK by up to 60% (Salemdeeb et al., 2017a).

To conclude, this section has shown that conventional approaches to investigating environmental benefits associated with food waste prevention are insufficient in the context of behavioural and systemic effects, as well as a globalized world. The food and waste to food industries are inter-connected and supply chains are currently fragmented across countries and products often transit through multiple countries before arriving to final markets. The multi-faceted nature of the food system requires a wider approach to capture activities that would be time-consuming and resource intensive to be covered using a conventional process-based life cycle assessment approach. In order to quantify the environmental benefits associated with food waste prevention, it is necessary to include the following factors:

1. The adoption of high-resolution models for a better representation of the food production mix taking into consideration the global food supply chain; and
2. The inclusion of the rebound effect in reducing environmental benefits of food waste prevention.

Environmental assessment: towards a holistic approach

LCA has been extensively applied to evaluate the environmental impact of waste disposal options (Cleary, 2010; Laurent et al., 2014a). This is primarily due to three reasons: (1) the technique's flexibility in modelling and evaluating case-specific options; (2) the availability of pre-defined life cycle inventory datasets for the majority of waste disposal processes that could be easily modified to model specific waste management scenarios; and (3) the variety of environmental impact categories that could be investigated in order to provide a comprehensive environmental evaluation.

While acknowledging the analytical strengths of LCA, the method stills suffer from several limitations such as truncation error, system expansion and the exclusion of capital goods, as previously highlighted. In addition, the literature has demonstrated how LCA is proven to be an inadequate tool when quantifying the overall environmental impacts of food waste upstream options, in particular food waste prevention. It fails to take into consideration the multi-faceted, and dynamic nature of food waste prevention (in particular modelling the environmental impacts of food imports and the inclusion of the rebound effect). The majority of studies reviewed did not deal with these issues and those which did attempt to do so reported results with a high level of uncertainty. Therefore, a necessity exists on the importance of the introduction of a comprehensive and systematic approach to investigate the environmental impacts of the food waste hierarchy.

More importantly, the globalised nature of the food production system coupled with the fact that ours is a planet of finite resources make it crucial to address the issue of food waste comprehensively. The discussion above has shown how different stages of the food waste hierarchy are inextricably intertwined as well as linked to other systems outside the waste management sector. For example, the food-waste-to-animal-feed option is directly linked to the livestock industry. The era of specific-case research is over as a new era has begun that requires the food waste challenge to be considered within the global food supply chain and indirect interactions with other industries. The process of using existing LCA methods is time consuming and expensive. These limitations, coupled with the multi-faceted nature of food waste, make the environmental evaluation of food waste prevention practices an arduous task. This has created the need for a holistic approach to the environmental evaluation of food systems to be identified as a new global research agenda by the United Nations Food and Agriculture Organization and the World Health Organisation (Haddad et al., 2016).

In order to address the research gap identified in the above literature, a top-down economic-based input-output method can be integrated into a conventional process-based LCA model using an attributional approach to develop a hybrid technique integrating a consequential approach. This technique was introduced to overcome the limitations of conventional process-based LCA (in particular truncation error), another type of environmental impact assessment can be used. This other method is a top-down economic-based input-output method, and can be used independent or as a hybrid method with LCA.

Input-output analysis

Input-output based LCA (IO) is a top-down approach developed by Leontief to describe the complex interdependencies of industries within an economy. IO table, shown in Figure 3, has been widely used to evaluate the environmental performance of various real-world applications (Murray and Joy, 2010). The roots of environmental applications can be traced back to 1968 (Daly, 1968; Isard et al., 1968). Leontief also provided a key methodological extension to cover environmental problems (Leontief, 1970). Leontief's concept is based on linking the environmental impact of IO sectors using an environmental 'satellite' at the bottom of the input-output table (IOT). Environmental satellites augment the IOT with additional rows or columns to reflect pollution generation and abatement activities (Miller and Blair, 2009). They can display noneconomic flows to and from intermediate sectors (in mixed units) such as pollutant emissions, energy generation or use, waste, water, or even social indicators such as employment levels or accidents per sector.

IO tables, illustrated in Figure 3, are primarily derived from Supply-Use table within a national economy. IO table is a matrix of numbers with rows and columns labelled as sectors of the economy. IO table consists of the following elements;

Intermediate sectors Z: the sectors of the economy that produce goods and services- are listed on both axes allowing for intersection of each sector with all economy's sectors (including the same sector). Z_{ij} indicates the monetary value of outputs produced by industry I and purchased by industry j.

Primary inputs V: elements of production that are exogenous to the framework of the economic production.

Final demand F: is the consumption of goods and services in their final form by exogenous elements (e.g. households, government, exports etc.).

Figure 3 A schematic diagram of an IO table, adapted from Lenzen and Reynolds, (2014).

National accounting states that the total output of a sector (x) equals the total expenditure of the same sector (x'), which in turn means that

$$x^{'} = V + T = T + F = x \qquad (1)$$

In order to calculate the proportion required by each industry output i for the production of one unit of industry j, each cell of T (Z_{ij}) is divided by the corresponding sector of x. Thus we create the direct requirements matrix, alternatively called production matrix, $A = Z\hat{x}^{-1}$where the '∧' over a vector denotes a diagonal matrix out of vector x, such that x_i is located at $\hat{x}_{ii}$and $\hat{x}_{ij} = 0$ whenever $i \neq j$.

X is the total output of industry I, which is the sum of the total industry's outputs consumed by other industries and final demand F

$$x = Ag + F \qquad (2)$$

Solving this equation for x

$$x = (I - A)^{-1}F \qquad (3)$$

Where I is an identity matrix. $(I - A)^{-1}$ is widely known as the Leontief inverse matrix. Assuming a fixed relationship between a sector's input and output, in other words, input coefficients are scale insensitive. The total output of an industry x required by an arbitrary final demand for industry output y is calculated by:

$$X = (I - A)^{-1}y \qquad (4)$$

In an operation similar to the direct requirements matrix, a link between an economic activity and environmental health can be created – the amount of pollution emitted per (monetary) unit of output of sector i, $g_i = {}^{e_i}/_{x_i}$. Wiedmann (2013) refers to this as the Direct Intensity Multiplier (DIM), or the direct 'pollution' coefficient. Rearranging this term gives the emission from sector i, $e_i = g_i x_i$. Total emissions in matrix notation can be written as $E = GX$.

Substituting in the total requirements matrix from Equation (4), the total Intensity multiplier, or the total 'pollution' coefficient can be found, $E = G(I - A)^{-1}$. This gives the total pollution generated by one unit of final demand for products from sector i.

Numerous factors have made IO analysis a prominent analytical tool to evaluate the environmental impacts of any system in a consistent and holistic manner. The process of conducting an IO analysis is straightforward due to the abundance of freely available software applications such as EIOLCA (Hendrickson et al., 2006), and CMLCA (Heijungs and Suh, 2002). It is also an efficient, inexpensive, less data-intensive and less labour-intensive approach (Lenzen, 2000). More importantly, IO takes into account the whole system (upstream processes) without the need for system cut-offs, thus avoiding the boundary issues and truncation errors (Lenzen, 2000; Tukker et al., 2006).

Despite its capability of capturing upstream environmental impacts, several limitations of this technique need to be acknowledged. The level of aggregation is one of the key concerns raised by various researchers (Suh, 2003; Tukker et al., 2006). Highly aggregated tables may lead to significant errors (Lenzen, 2000). (Lave et al., 1995) echoed these observations and acknowledged that aggregated tables tend to combine different commodities into one sector level. Thus, average multipliers will be used to represent all these commodities under the same sector.

The second issue is the age of the data and the fact that IO tables usually take years to be compiled. For instance, the UK IO analytical tables for the year 2010 were published in 2013 (ONS, 2011). Although some studies underestimate changes as a result of data's age, attention should be paid to considering the variety of factors which might affect results (e.g. inflation, and import assumptions). (Suh, 2004) has also raised these concerns about the IO data especially when this data is used to assess new or rapidly developing sectors. Thirdly, IOA databases entail a high level of uncertainties, which could be due to sampling error, data age, and allocation/aggregation errors. Finally, environmental satellite data, linked to IO models, relies upon a national inventory of environmental emission, which is usually not complete (ONS, 2013).

IO-based hybrid LCA

IO-based hybrid LCA (hereafter referred to as Hybrid-LCA), firstly introduced in 1970s, was developed to overcome the incompleteness of LCA systems as well as aggregation issues associated with IO analysis (Bullard et al., 1978; Lenzen, 2000). Hybrid LCA integrates more reliable data into a comprehensive input-output model to capture all embedded impacts within a system. Hybrid-LCA has been considered as "state-of–the-art" in evaluating the environmental impacts of any system (Wiedmann et al., 2011). Numerous studies applied hybrid-LCA in real-world applications (Joshi, 1999; Lenzen and Munksgaard, 2002; Suh, 2004; Wiedmann et al., 2011).

There are three different types of hybrid-LCA; tiered hybrid analysis (Bullard et al., 1978), input-output based hybrid LCA (Joshi, 1999), and integrated hybrid analysis (Suh and Huppes, 2005; Suh, 2004). In the tiered hybrid analysis, direct requirements (i.e., major lower-order upstream requirement and downstream requirements) are quantified using process analysis, while the remaining higher-order upstream processes are covered by input-output analysis (Bullard et al., 1978). In the IO-based hybrid analysis, input-output sectors are disaggregated to allow for an analysis of a specific product or service (Joshi, 1999). When inputs that are required to produce the product or service are known and direct environmental burdens from the production process are available, then the total environmental impact of the product of interest can be estimated by adding the environmental burden from the input processes and the direct environmental burden. In the integrated hybrid analysis, a consistent mathematical framework is developed, with the process-based system represented by a technology matrix by physical units and the input-output system represented by monetary units. Among the three methods, the tiered hybrid method is the most straightforward, but it also bears the issue of double counting (Hou, 2014). The integrated approach is the most comprehensive; however, it is more complicated in both matrix construction and data collection (Suh, 2004).

IO-based hybrid LCA has rarely been used in waste management studies although it has been recommended in several studies (Finnveden et al., 2009; Rodríguez-Alloza et al., 2015). Possible explanations may include: (1) existing environmental models for the environmental evaluation of waste disposal options are completely process- based; 2) IO-based hybrid LCA has been primarily developed for product systems rather than service systems; and (3) the lack of monetary data that is required to carry out the analysis using IO models (i.e., a detailed breakdown of the financial cost for processing waste).

Environmentally extended multi-regional IO analysis

In order to address the limitations of the environmental evaluation of food waste prevention considering the global food supply chain, an environmental Multi-Regional Input-Output approach (MRIO) is considered in this study. It is a geographically extended version of the conventional single region IO-based LCA model. It also allows the tracing of the production of a "typical product" of economic sectors, quantifying the contributions to the value of the product from different economic sectors in various countries. This technique has been developed in order to address the complexity of the global supply chain and its ecological impacts – it is estimated that 5.3 Gt of CO_2 emissions are embodied in international trade flows (Peters and Hertwich, 2008). Three major MRIO databases exist: WIOD (Dietzenbacher et al., 2013), EORA (Lenzen et al., 2012a; 2012b) and EXIOBASE (Tukker et al., 2009; 2013).

The MRIO model has been used to evaluate the environmental impacts of similar lifestyle and consumption patterns upon global ecosystems (see e.g., Kastner et al., 2011 and Ivanova et al., 2015). Druckman (2011) used a MRIO-based model to investigate the impact of eliminating food waste by adopting a sustainable consumption approach (i.e., buying less food products). However, scenarios considered in this study overlook other stages of the food waste hierarchy and focus mainly on capturing environmental benefits associated with upstream food production. More importantly, the adopted MRIO model fails to provide a detailed quantification of emissions due to the high level of data aggregation: (1) the applied model includes only 13 regions, and (2) all food products are aggregated into a single sector. Therefore, reported results fails to further investigate the environmental impacts associated with different food products (for example, rice from India or coffee beans from Brazil). This is a key point that needs to be addressed taking into consideration the consensus existing on the significant contribution of imports on GHG emissions (Su et al., 2010).

A more detailed UK-specific MRIO study was carried out by (Minx et al., 2008) using structural path analysis in a generalized MRIO model covering 57 sectors and 81 world regions to identify GHG emission hotspots in the international supply chain of meat products consumed in the UK. The analysis results estimate that meat products account for more than half of the UK's food related carbon footprint. Results also suggest that CO_2 is the predominant burden (compared to other GHG emissions) in the global supply chain of meat products. This could be attributed to the large energy input required in this sector across all stages of production and distribution, in particular refrigeration (Foster et al., 2006).

(Wiedmann, 2009) also confirmed the effectiveness of using a MRIO approach by investigating different applications. The author used a MRIO model to estimate the ecological footprint of energy input embedded in UK trade in 2002 and compare the results with a conventional method (i.e., the product land use matrix, a method that is used in the national footprint accounts compiled by the Global Footprint Network). Results reveal large differences and hardly any correlation between the two methods. These could be attributed to the failure of the product land use matrix method to include trade in services (especially transport services) and upstream impacts of energy goods (fossil fuels) as well as the use of inappropriate embodied energy factors in this method. These studies confirm that international MRIO models could play a crucial rule in the future to estimate the overall environmental impacts of imports and exports of nations with the possibility of tracking their origin via inter-industry linkages, international supply chains and multi-national trade flows.

In spite of the analytical power of the MRIO model (as demonstrated above), this approach is subject to certain limitations. Firstly, it inherits all uncertainties specific to input output-based models, discussed previously (e.g., which include uncertainties in source (survey) data, allocation and multipliers (Lenzen, 2001, Hawkins et al., 2007, Weber, 2008). Aggregation is the second key limitation, in particular when a high-impacting sector is combined with a low impacting sector to form an aggregated and an average impacting multiplier. Considering the MRIO model by Druckman (2011), meat products, vegetables, bakery and dairy manufacturing processes are all aggregated into a single sector (i.e., food and non-alcoholic drink). Lenzen et al. (2004) examined the effect of sector aggregation as well as international trade on Danish carbon multipliers and trade balances. Whilst the inclusion of Danish exports led only to minor corrections, the explicit modelling of Danish imports, as well as sector disaggregation was concluded to be important for overall accuracy. Literature is abundant with other studies that confirm Lenzen’s conclusions by reporting a high level of uncertainty due to data aggregation (Weber, 2008; Weber and Matthews, 2007; Kanemoto and Tonooka 2009).

Despite the substantial uncertainties associated with MRIO models, their analytical strength would provide a better approach to capturing the environmental benefits of food waste prevention across the global supply chain. More importantly, a MRIO model that is based on a highly disaggregated database (e.g., EORA, EXIOBASE or WIOD), would enable us to track emissions across products and countries. Currently no study has used EIO, environmental

MIOR or a hybrid EIO-LCA to examine waste to food environmental impacts. However, this is a natural next step for research in this area to take.

Examples of waste to food environmental assessment

As highlighted by Calderia et al (2020) there has been a growth in recent literature using LCA to evaluate the environmental performance of different waste to food processes. The majority of studies consider one input item and then calculate and compare multiple output edible and nonedible products. Single items that have LCA investigations include bread (Lam et al., 2018) brewer's spent grain and barley straw (Garcia-Garcia and Rahimifard, 2019; González-García et al., 2018), chicken eggshell (Chung et al., 2019), chicory grounds (Vauchel et al., 2018), citrus and carrot waste for pectin (Garcia-Garcia et al., 2019), fish canning (Monteiro et al., 2018), fruit and vegetable (Plazzotta et al., 2020), olive mill waste (Frascari et al., 2019), rice straw (Belaud et al., 2019), shellfish (Iribarren et al., 2010), sugar cane processes by-products (Moya et al., 2013), and whey by-products (Summers et al., 2015),

We also highlight that outputs from the REFRESH project (Östergren et al., 2018; REFRESH, 2020; Scherhaufer et al., 2020) provide 6 key examples of unpreventable food processing co-products, by-products or wastes (apple pomace, pigs blood, brewers spent grains, tomato pomace, whey permeate, and oilseed press cake). We again suggest readers examine these case studies for clear applications of possible waste to food environmental impact assessments.

Summary and recommendations to practitioners

This chapter provided a review of previous attempts that have been made to quantify the environmental impacts of waste to food. Although the LCA of waste to food processes are still a 'young' field, the evidence base is growing; and good practices have been established (as seen in the REFRESH case studies). Indeed, we are excited to watch the field of waste to food environmental impact assessment grow over the coming years.

Practitioners can adopt either a bottom up LCA, or a hybrid IO-LCA approach to quantify the environmental impacts of waste to food. However, practitioners need to understand the drawbacks and limits to the methods, and how comparable results are between studies. The main take away points for future practitioners to note are as follows:

- Quantifying the environmental impacts of food waste treatment and waste to food suffers from several methodological limitations that lead to high level of uncertainty such as truncation error and the exclusion of capital goods. This may mean separate studies are not directly comparable.

- Evidence exists for the potential environmental benefits that could be achieved if waste to food was used in conjunction with food waste prevention. However, there is little no evidence to quantify this for each specific use-case. All future waste to food LCA projects need to compare each side product option with other downstream food waste treatment options.

- Previous studies fail to provide a comprehensive analysis of embodied environmental impacts of food waste prevention when compared to waste to food (or other treatment methods); a holistic approach has yet to be developed by taking into consideration the global food supply chain and the rebound effects.

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