Waste Management 117 (2020) 136–145

Contents lists available at ScienceDirect

Waste Management

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

Environmental impacts of waste management and valorisation pathways for surplus bread in Sweden

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ARTICLE INFO

Article history:
Received 12 May 2020
Revised 2 July 2020
Accepted 21 July 2020

Keywords:
Food waste
Bread
LCA
Waste hierarchy
Waste treatment
Prevention

ABSTRACT

Bread waste represents a significant part of food waste in Sweden. At the same time, the return system established between bakeries and retailers enables a flow of bread waste that is not contaminated with other food waste products. This provides an opportunity for alternative valorisation and waste management options, in addition to the most common municipal waste treatment, namely anaerobic digestion and incineration. An attributional life cycle assessment of the management of 1 kg of surplus bread was conducted to assess the relative environmental impacts of alternative and existing waste management options. Eighteen impact categories were assessed using the ReCiPe methodology. The different management options that were investigated for the surplus bread are donation, use as animal feed, beer production, ethanol production, anaerobic digestion, and incineration. These results are also compared to reducing the production of bread by the amount of surplus bread (reduction at the source). The results support a waste hierarchy where reduction at the source has the highest environmental savings, followed by use of surplus bread as animal feed, donation, for beer production and for ethanol production. Anaerobic digestion and incineration offer the lowest environmental savings, particularly in a low-impact energy system. The results suggests that Sweden can make use of the established return system to implement environmentally preferred options for the management of surplus bread.

1. Introduction

Food waste leads to a loss of the resources that are invested in the supply chain, such as water, energy, fertilizers, and land, which are used in the production, transport and storage of food products. These materials and processes have different environmental impacts, such as global warming, acidification, and eutrophication. Therefore, the loss of food results not only in the loss of the product itself, but also of all of the resources used in the supply chain. This problem has been recognized by the United Nations in the Sustainable Development Goals (United Nations, 2015). Specifically, the goal "sustainable consumption and production patterns" has a sub-target 12.3 that aims to halve food waste in the consumer and retail levels and to reduce food loss in the production and supply chains.

Bread waste is a large part of the global food waste. The Waste and Resources Action Programme (WRAP) has estimated that bread waste is 10% of all food waste generated in the United Kingdom (WRAP et al., 2011). Brancoli et al. (2019) have estimated that 80,410 tons of bread is wasted in Sweden each year, which is the equivalent of 8.1 kg per capita/year.

Besides the large quantities of bread waste, the distribution scheme for bread in Sweden and other countries such as Norway (Stensgård and Hanssen, 2016), Austria (Lebersorger and Schneider, 2014), the Netherlands (Weegels, 2010) and Germany (Brosowski et al., 2016), makes it an interesting product since it is not mixed with other food waste and can therefore be managed separately to other waste streams. The bread distribution is conducted within a full service scheme that involves a take-back agreement (TBA) between retailer and supplier. This means that the bakeries are responsible for ordering the bread that is supplied to the supermarkets, which includes the forecasting, placement, and removal of products from the supermarket shelves. More importantly, the bakeries are also financially responsible for the unsold products and its collection and treatment, thereby operating in a reverse supply chain (Eriksson et al., 2017; Ghosh and Eriksson, 2019). This reverse logistics enables a clean flow of bread, i.e. not mixed with other food fractions, and provides opportunities for different waste treatment pathways such as ethanol or animal feed production, which are not always viable for a mixed waste fraction. These pathways are an alternative to incineration or

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https://doi.org/10.1016/j.wasman.2020.07.043
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anaerobic digestion, which are the most common treatment methods for mixed food waste in Sweden (Avfall Sverige, 2017).

In order to comply with legal frameworks and environmental quality objectives set by governments, waste management systems have progressively become more complex (Brancoli and Bolton, 2019; Manfredi et al., 2011). The European Union have established the EU Waste Framework Directive (European Commission, 2008), which sets basic concepts and defines a priority for different waste management alternatives. According to this hierarchy, prevention of waste is the preferred option, followed by reuse, material recycling, energy recovery and disposal as the least preferred alternative. Nevertheless, the hierarchy does not always provide the best waste management alternative due to differences in local conditions such as energy supply mix and the waste treatment technology. The life cycle assessment methodology (LCA) is often used to support deviations from the waste hierarchy in a robust scientific way. LCA uses a holistic approach, which includes the whole life cycle of the material or process under study, thereby reducing the risk of shifting environmental impacts from one part of the waste management system to another (Hauschild and Barlaz, 2010; Manfredi et al., 2011).

Although previous studies have assessed the environmental impacts of different food waste management options (e.g. Albizzati et al., 2019; Khoo et al., 2010; Saer et al., 2013; Xu et al., 2015), few have studied systems dedicated to bread waste. Vandermeersch et al. (2014) studied the production of animal feed from bread waste and concluded that the valorisation of bread waste into animal feed was environmentally preferable in comparison with anaerobic digestion. Eriksson et al. (2015) assessed the environmental impacts of six different waste management options for different food fractions, and concluded that among the products studied, bread had the highest potential to reduce greenhouse gas emissions.

This study builds on the existing literature and contributes further by systematically assessing, using LCA, the environmental impacts associated with different options for managing bread surplus in Sweden. The goal of the LCA is to compare the following options: source reduction, donation, animal feed production, ethanol production, beer production, anaerobic digestion and incineration. Although the exact amounts sent to each treatment is unknown, the alternatives included in this study are the ones which are already implemented in Sweden or that would be possible to implement (Fazer, 2016; Polarbröd, 2016; Pågen, 2018). Currently, bread surplus is also used for yeast production in Sweden (Pågen, 2018), however, it was not included in this study due to lack of data. The environmental savings offered by these waste management schemes are also compared to reducing the production of bread by the amount of surplus bread. The relative environmental savings offered by the different waste management options and their comparison with waste prevention is compared to the waste hierarchy (European Commission, 2008).

2. Materials and methods

2.1. Definitions

The FUSION Project (Östergren et al., 2014) defines food waste according to its disposal or valorisation route. Food waste is defined as "any 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, co-generation, incineration, disposal to sewer, landfill or discarded to sea). The FUSIONS framework does not classify products that are converted to animal feed or bio-based materials as food waste. Products used in biochemical processing are not defined as food waste either. While acknowledging this definition, in this study all bread which is produced for human consumption and that is not consumed by humans is referred to as waste, regardless of its final disposal or valorisation route. The only discrepancy with the FUSIONS definition is when the bread is used to produce animal feed. We justify this classification by the fact that no bread is baked for animal feed, and that animal feed can be produced directly from wheat, without the need of the processing required for bread production. The use of wheat as animal feed is common, and although most wheat is grown for human consumption (with low quality and surplus wheat being redirected to produce feed), it is also grown for feed (Blair, 2018; Inra et al., 2016; Lalman and Highfill, 2011). Bread that is donated is, of course, not defined as waste. This study uses "surplus" as a general term to define both bread that is wasted or donated.

2.2. Goal and scope definition

The functional unit of the study is the management – or prevention – of 1 kg of surplus bread in Sweden. The production and disposal of packaging is not included because it is assumed similar in all treatment and valorisation scenarios assessed. As mentioned above, the surplus bread can be reduced at the source, donated, or sent to different waste management pathways, namely animal feed production, ethanol production, or beer production (Fig. 1). The return bread flow, described in Fig. 1, is the flow that is currently the most suitable for all the scenarios assessed in this study, since it is collected separately. Nevertheless, the remaining flows in the supplier-retailer interface could also be treated via the scenarios described in this study if the proper agreements between the actors are reached. These flows are in-store, which are products normally branded by the retailer, and are not submitted to take-back agreements, and bake-off, which are products baked from pre-made dough in supermarket stores or supplied by a bakery in a nearby store. The residual flow comprises the flows outside the supplier-retailer interface, which are not under take-back agreements and are usually disposed together with the other organic waste fractions and unfit for the scenarios in which segregated bread is necessary.

The system boundaries includes receiving the surplus bread and the further valorisation, waste management, or donation. The geographic scope is Sweden. The temporal scope is the waste management and valorisation pathways that are currently used, or possible to be used, in Sweden. The ecoinvent database version 3.5 was used for the background processes. The characterization methods are based on ReCiPe 2016 v1.1 midpoint and endpoint method, Hierarchist version (Huijbregts et al., 2016). The impact categories are presented in the Supplementary material in Table SM.1.

This study has the goal to compare the prevention of baking surplus bread and the different technologies for treatment and valorisation of surplus bread, and can be classified as an account study in the International Reference Life Cycle Data System (JRC, 2010). Therefore, this study uses an attributional approach and, consequently, average data for the upstream and downstream processes are used, including the product substitution.

2.3. Inventory of waste management and valorisation pathways

As shown in Fig. 1, seven scenarios were investigated in this study: source reduction, donation, animal feed production, ethanol production, beer production, anaerobic digestion, and incineration. The multi-functionality of the scenarios, i.e. the co-products and services, were treated using system expansion, so that the management options are credited by accounting for the environmental impacts of the substituted co-products (using the market average).

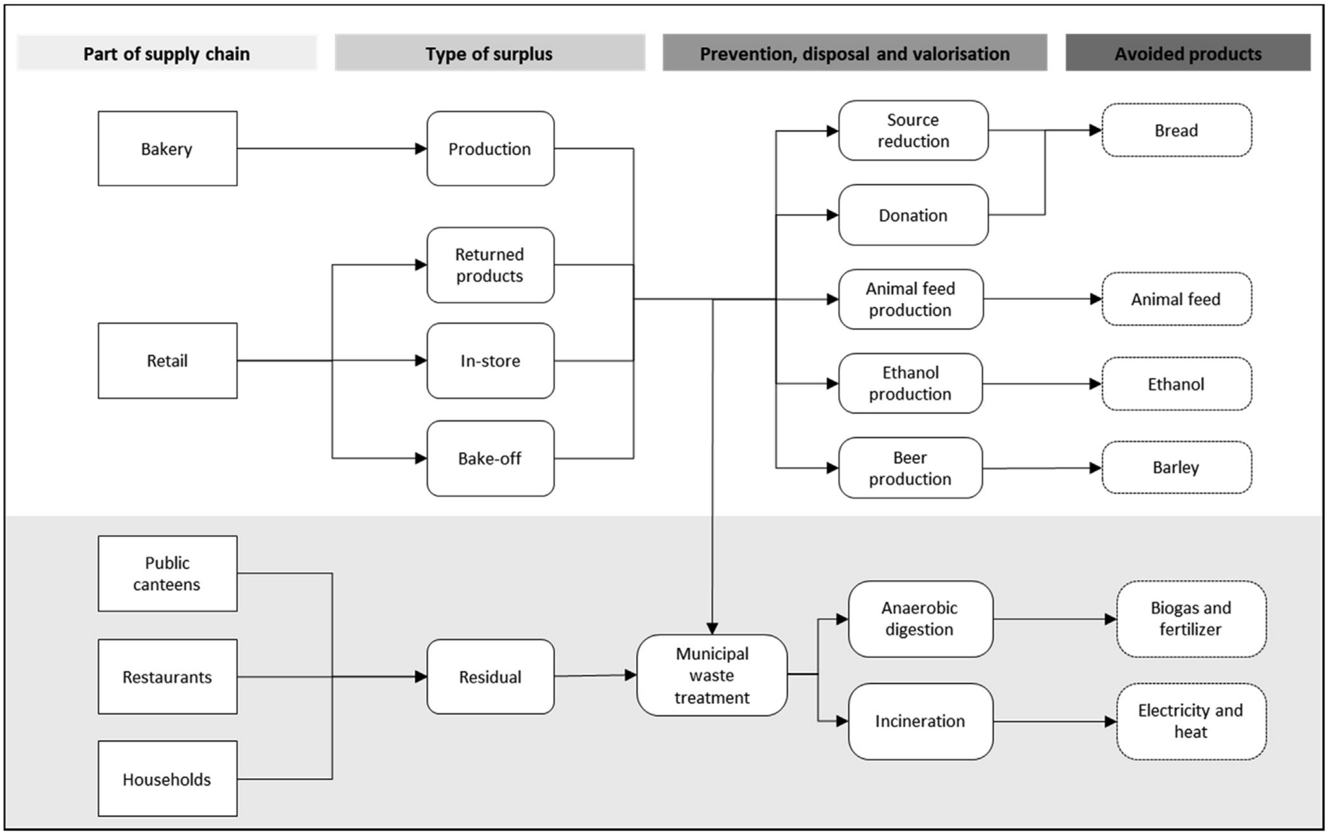

Fig. 1. Bread surplus flows, waste treatment and valorisation scenarios, and avoided products. The area highlighted in grey shows common flows of mixed food waste, which are not feasible for the management pathways highlighted in the white area.

Fig. 2 shows the system boundaries and the main activities for the waste management and valorisation scenarios considered in this study, including the avoided products in the system expansion.

2.3.1. Donation and source reduction

Donation systems are often complex and dynamic. This study uses a scenario to represent a hypothetical donation system, which is a simplification of the large variety of activities that take place in Sweden. The modelling includes transportation between the retail and the donation centre (40 km) and from the later to the households (5 km), based on Bergström et al. (2020). The bread losses before and at the donation centre was assumed to be 20%, similar to Bergström et al. (2020). The bread losses in households was estimated to be 32%, since this is this figure reported by WRAP for UK households (WRAP et al., 2011). This loss rate is representative for average consumers, and it might be different when considering donation recipients. The loss rate can be affect for example by the relative lower shelf life of donated products and by differences in the behaviour of people who purchase the bread or received it via donation.

The wasted bread was assumed to be treated by the average waste technologies used in Sweden for the organic waste fraction, described in Fig. 1 and Fig. 2b as "Municipal waste treatment". The "Municipal waste treatment" is composed by anaerobic digestion (40%) and incineration (60%), which are the most common methods for household waste treatment in Sweden (Avfall Sverige, 2017).

The modelling of source reduction is used in this study as a benchmark, since surplus bread should, in principle, not be manufactured. The maximum level of prevention is modelled, i.e. avoiding the production of one kg of bread.

2.3.2. Animal feed production

Feed production was modelled using a pig feed recipe from Sirtori et al. (2007), where some ingredients from the original recipe were replaced by bread waste. The recipes are described in Table SM.4 in the Supplementary Material. The recipe described in Sirtori et al. (2007), ensured that the feed with bread waste (bread feed) was equivalent to the original recipe (original feed) in terms of the conversion rate and chemical composition parameters such as gross energy, protein content, dry matter and crude fibre. Thus, it was ensured that the growth performance of the pigs was not affected by the substituted ingredients. Consequently, the total intake for each type of feed is different and was modelled accordingly. Sirtori et al. (2007) have reported higher average daily gain and back fat thickness for the animals fed with the bread feed. System expansion was used accounting for the fact that the bread feed avoids the production of the original feed.

The LCI includes the ingredients that are necessary for the production of the bread feed and the downstream processes. Ingredients with a contribution lower than 5% in mass were not considered. The valorisation of bread waste for feed production starts with the receiving of the bread, unpacking, crushing, mixing with the other ingredients and packing (Fig. 2c).

2.3.3. Ethanol production

The ethanol production was based on a typical Swedish ethanol plant modelled by Brancoli et al. (2017a). Data for the inventory was obtained by lab scale experiments (Ferreira et al., 2015; Nair et al., 2015), a pilot scale unit and computer simulation (Rajendran et al., 2016) done in Aspen Plus® (v.8.4) (Aspentech: Burlington, MA, USA).

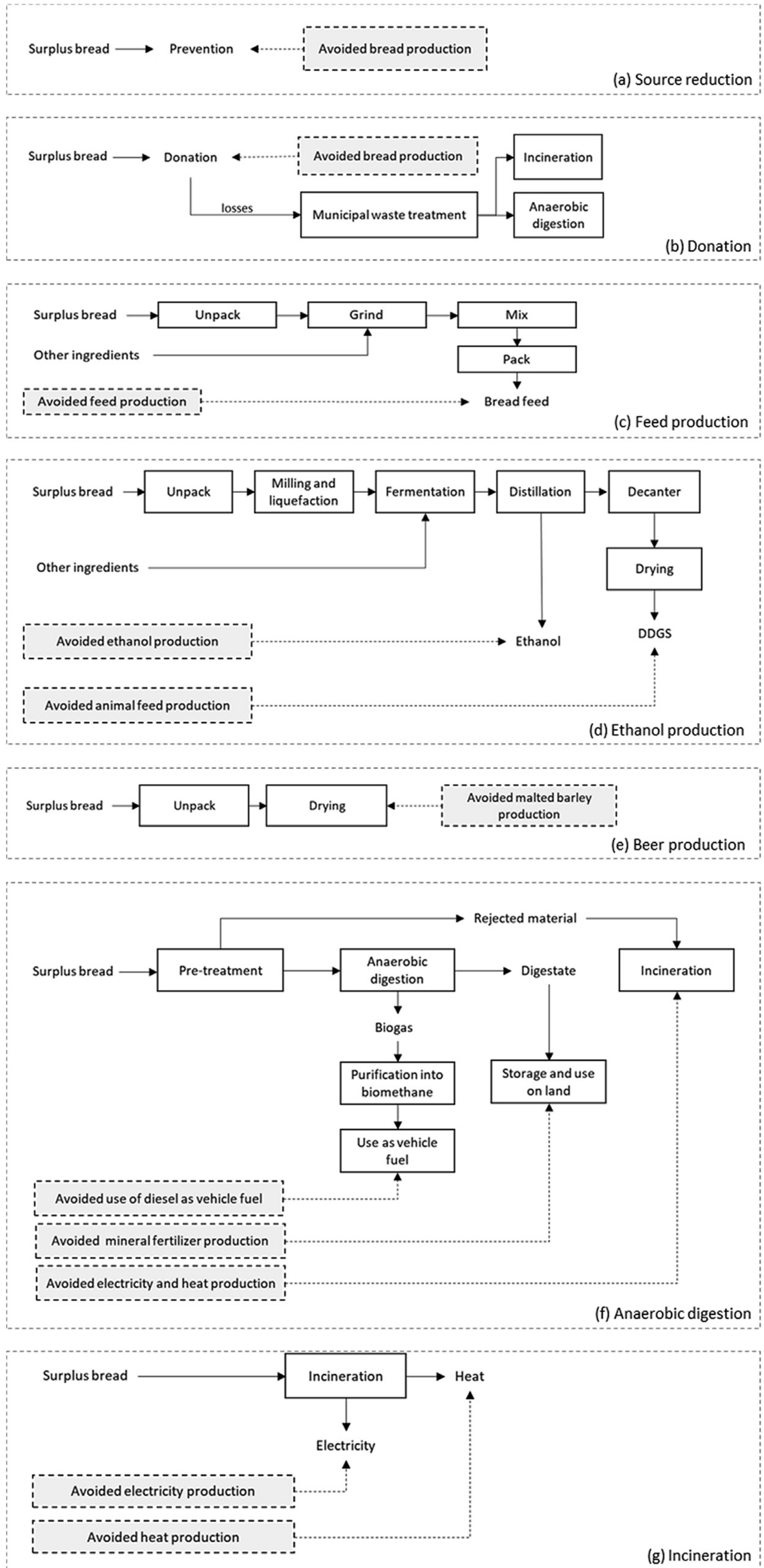

Fig. 2. System boundaries for prevention and the valorisation and waste management scenarios considered in this study. The dashed lines and dashed boxes represent the substitution of products in the market. (a) Shows the source reduction of bread. (b) The donation of the surplus bread. (c) The use of surplus bread as animal feed. (d) The valorisation of bread via ethanol production. (e) The production of beer. (f) Shows the anaerobic digestion process, and (g) the incineration process.

The ethanol production yield was based on data from Ebrahimi et al. (2008), where the separate hydrolysis and fermentation of waste wheat bread was studied. They obtained overall ethanol yield of 350 g per kg of bread. This value is similar to that obtained by Pietrzak and Kawa-Rygielska (2014), who reported a yield of 354.4 g of ethanol per kg of bread waste.

The production of ethanol has dried distillers grains with solubles (DDGS) as the main co-product, which is used as animal feed. The amount of replacement feed products was calculated using system expansion according to its energy and protein content. It was considered that the by-product (DDGS) was used as cattle feed and the digestible energy for ruminants was used (Inra et al., 2016). The digestible crude protein (dCP) was determined from the crude protein content by subtracting the unavailable protein content (Eq. (1)) as done in (Tonini et al., 2016). It was assumed that DDGS replaced average market for energy and protein feed, as described in ecoinvent database (Wernet et al., 2016). The biochemical parameters used for the calculations are presented in Table 1.

$$dCP\ \left(\mathrm{kg\ kg^{-1}\ DM}\right) = \frac{(0.93 * CP * 100 - 3)}{100} \tag{1}$$

where DM is dry matter and CP is crude protein.

2.3.4. Beer production

Beer is produced primarily from a cereal grain, hops and water. Barley is the most common cereal that is used. Recently, small breweries have started to use surplus bread in their recipes, substituting part of the malted barley, originally used as a source of sugar for fermentation (Almeida et al., 2018; Connolly, 2019). Two recipes were analysed in this study, one from Almeida et al. (2018) and another from a brewery in the United Kingdom (UK) which made their recipe public (Toast Ale, 2020). In both recipes, 25–28% of the original malt was substituted with dried bread. The remaining ingredients were assumed the same as those used in the production of the standard beer (Almeida et al., 2018).

It was assumed that the energy consumption for the drying of bread is similar to the drying of the malted barley. Therefore, for simplification and assuming that the beer produced with bread surplus will avoid the production with malted barley, the only process that was modelled was the substitution of barley by bread. The changes in weight of barley when malted is 0.807 kg malt/kg barley based on Briggs (1998).

2.3.5. Anaerobic digestion

The anaerobic digestion was based on the inventory from a generic plant treating organic waste and manure that was sorted at the source. The characteristics and the inventory analysis of the plant is shown in Table SM.2 in the Supplementary Material. The biogas potential of bread was based on Dubrovskis and Plume (2017) and calculated as 0.6 N m^3 kg^{-1} VS. The yield was set to 70%, which means that the biogas generated was 0.42 N m^3 kg^{-1}. Based on Dubrovskis and Plume (2017), the biogas composition is 48% methane and 52% carbon dioxide. It was assumed that 2% of methane is emitted as gas leakage from the digester (Andreasi Bassi et al., 2017; Møller et al., 2011). Pre-treatment is necessary due to the presence of the bread packaging. Based on studies done by Brancoli et al. (2017b), it was assumed that 44% of the bread waste is lost in the pre-treatment process. The losses in pre-treatment can vary between different plants depending on the technology employed. According to Bernstad et al. (2013), 2–45% of incoming wet waste is lost as refuse and Møller et al. (2011) indicate losses of 41% in wet weight or 55% in relation to volatile solids. The material that is rejected in the pre-treatment process is sent to incineration. The model for incineration is similar to the model described in Section 2.3.6, with the difference that it takes into account the increase in the water content of the bread after the pre-treatment.

The biogas that was produced was upgraded to methane in order to be used as vehicle fuel. This is the most common use of biogas in Sweden (Larsson et al., 2016). The details on the use of upgraded biogas and digestate are described in the Supplementary Material.

Table 1
Biochemical composition for relevant parameters of the by-products and replacement products (Inra et al., 2016).

Biochemical parameter	Unit	Dried distillers grains with solubles
Crude protein	% DM	37.3
Digestible crude protein	% DM	34.7
Gross energy	MJ/kg DM	20.5
Digestible energy[a]	MJ/kg DM	16.1

[a] Ruminant digestible energy.

2.3.6. Incineration

The waste-to-energy process was adapted from the ecoinvent process "*treatment of biowaste, municipal incineration with fly ash extraction*" which represents the incineration of a mixture of garden and food waste (Wernet et al., 2016). For simplification, the emissions to air, water and land used the model for mixed organic waste, based on the waste composition and transfer coefficients.

The energy production was modelled according to the calorific value of bread. The higher heating value (HHV) for bread was calculated as 14.84 MJ/kg, which is derived from the correlation shown in Eq. 2 (Eboh et al., 2016). The elemental compostion of bread was based on Tonini et al. (2018). The lower heating value (LHV) was calculated as 13.44 MJ/kg dry matter, and 10.35 MJ/kg as received.

The net thermal efficiencies for heat and electricity based on the LHV was modelled as 59% and 19% respectively, which is in accordance with the base-line plant in Eboh et al. (2019). The net energy production was thereby estimated as 1.94 MJ/kg of electricity and 6.10 MJ/kg of thermal energy. The produced heat and electricity was assumed to substitute the Swedish mix for electricity and heat, described in Section 2.3.7.

2.3.7. Energy modelling

The results of an LCA are usually strongly dependent on both the energy used and avoided in each process. Electricity was modelled as the average mix of technologies used to produce and distribute the electricity in Sweden according to the ecoinvent database (Wernet et al., 2016).

The modelling of the mixed technology for heat is based on the heat statistics published by the European Commission (2019), where heat production is divided into six categories, namely solid fossil fuels, oil and petroleum products, natural gas and manufactured gas, renewables and biofuels, wastes non-renewable, and other. Each fuel category was further divided into specific fuels, as shown in Table 2. When the specific fuel was not identified from the European Commission (2019) statistics, data from Statistics Sweden (2019) was used. This is similar to the approach described by Andreasi Bassi et al. (2017). The detailed description of the modelling of the energy systems are described in Section 2.4 of the Supplementary Material.

Fuels that contributed less than 5% to its fuel category were not considered. Table 2 shows the gross heat production by major fuel groups and the specific fuel composition of the major fuel groups in 2013. This data is based on the energy statistical country datasheets from Eurostat (European Commission, 2019) combined with the ecoinvent processes (Wernet et al., 2016).

Table 2
Correspondence between the fuel in the Eurostat data (European Commission, 2019) and the processes imported from ecoinvent (Wernet et al., 2016).

Fuel	Process from ecoinvent	Average mix heat
Solid fossil fuels		
Hard coal	Heat and power co-generation, hard coal (SE)	3.9%
Oil and petroleum products	Heat and power co-generation, oil (SE)	1.3%
Natural gas and manufactured gas		
of which natural gas	Heat and power co-generation, natural gas, conventional power plant, 100 MW electrical (SE)	1.8%
others	Treatment of blast furnace gas, in power plant (SE)	3.1%
Renewables and biofuels		
Solid biofuels and renewable wastes	Heat and power co-generation, wood chips, 6667 kW, state-of-the-art 2014 (SE)	71.2%
Ambient heat	Air-water heat pump 10 kW heat production (Europe without Switzerland	6.1%
Wastes non-RES	Heat, for reuse in municipal waste incineration only market for (SE)	12.7%

2.3.8. Transportation

The different valorisation or waste management schemes studied in this work require different infrastructures, e.g., plants for manufacturing ethanol or beer and donation centres. The availability of these infrastructures are different for the various valorisation or waste management options. For instance, the infrastructures for anaerobic digestion and incineration are present in all regions in Sweden, while the number of ethanol plants and animal feed producers are lower and located in specific regions. For this reason, the transportation distances for anaerobic digestion and incineration are expected to be lower than those for ethanol or feed production.

Transportation of the surplus bread to the valorisation or waste management site was not included in this study. Instead, a threshold for maximum distances that bread could be transported to a certain valorisation or waste management before the environmental benefits of the specific valorisation or waste management option would be lost was calculated.

3. Sensitivity analysis

A sensitivity analysis was performed based on a scenario analysis in order to investigate the influence of assumptions and sensitive input parameters to some of the results (Clavreul et al., 2012).

Different products may be substituted by the surplus bread when using it to produce animal feed. It is possible that the bread substitutes single products, such as barley or soymeal, or that it substitutes different ingredients in a feed recipe. Therefore, a sensitivity analysis was performed using scenarios where different feed products were substituted. The substitution was based on the digestible crude protein and digestible energy, similar to what was done for the DDGS in ethanol production. One of the scenarios used an ecoinvent average market for protein and energy feed (market for feed ecoinvent) (Wernet et al., 2016) and the second considers barley as the energy feed and soymeal as the protein feed. Moreover, two recipes with different replacement rates of bread from Kumar et al. (2014) was also assessed. Due to the functional unit of this study, namely 1 kg of surplus bread, and since the percentage of the replacement by bread in each recipe varied, the amount of feed produced per kg of bread also varied. For instance, in Kumar et al. (2014), between 25% and 50% of the ingredients were substitute by bread resulting in the production of 4.0 kg and 2.0 kg of feed per functional unit respectively. The detailed descriptions of the scenarios are available in Section 2.2 of the Supplementary Material.

The sensitivity analysis for anaerobic digestion used a scenario where the biogas was used in stationary engines for electricity and heat production. The emissions were based on Nielsen et al. (2010). In this scenario, it is not necessary to upgrading the biogas.

For energy production systems, namely anaerobic digestion and incineration, the electricity and heat consumed and produced have a significant impact on the results. Therefore, two scenarios were modelled for the source of electricity and heat for the foreground processes - one with a clean source and another with a dirty source. The energy sources were chosen from the Swedish energy mix. The dirty technology that was chosen was hard coal for heat and electricity. The clean energy was modelled with wind for electricity production and wood chips for heat. A sensitivity analysis for ethanol production included a scenario which assumed that the ethanol substituted petrol as vehicle fuel and the DDGS replaced soybean meal and barley.

4. Results

The results from the LCA are the potential environmental impacts, and not a precise prediction of absolute values for each impact category.

Fig. 3 shows the characterization results for each impact category. The environmental impact for each valorisation or waste management option is shown in relation to the best result in each impact category, which is arbitrarily given an impact of −100%.

The trend seen by the results in the eighteen impact categories supported the waste hierarchy: source reduction of bread waste is the preferred option followed by feed production, donation, beer production and ethanol production. There is no clear preference between these four latter valorisation pathways. The worst waste management options, with the exception of four impact categories (ionizing radiation, marine eutrophication, land use and water consumption), are anaerobic digestion and incineration, which are the most common waste management schemes in Sweden. Source reduction has the highest environmental savings in sixteen impact categories (all except ionizing radiation and fossil resource scarcity).

The waste management and valorisation pathways yield environmental savings in most of the impact categories. The environmental savings from source reduction come in its majority from the avoided production of wheat, similarly to donation and ethanol production. Moreover, source reduction has a relative higher environmental performance in comparison with the other scenarios because it does not have any environmental impact associated with downstream processes, such as transportation, or waste treatment and valorisation pathways. In terms of global warming (GW), preventing the production of surplus bread avoids the emissions of −0.66 kg CO_2 eq per kg of bread. Donation is similar to source reduction, although transportation and losses were included in this scenario. The majority of the environmental impacts during ethanol production come from crops used as feedstock, and the environmental savings determined in the present study are obtained when these crops, modelled as wheat, are substituted by bread waste. This resulted in a net savings of −0.56 kg CO_2 eq per kg of bread valorised as ethanol. The environmental savings from feed production come from the substitution of ingredients, such as maize and barley, by surplus bread and result in an avoided burden of −0.53 kg CO_2 eq per kg bread. The savings observed from beer production, e.g. −0.46 kg CO_2 eq in the GW category, are a result of the substitution of malted barley by surplus bread.

The relatively poor performance of anaerobic digestion is due partially to the amount of material that is rejected in the pretreatment. For the GW category, it resulted in a savings of −0.02 kg CO_2 eq per kg of bread. Anaerobic digestion has the high-

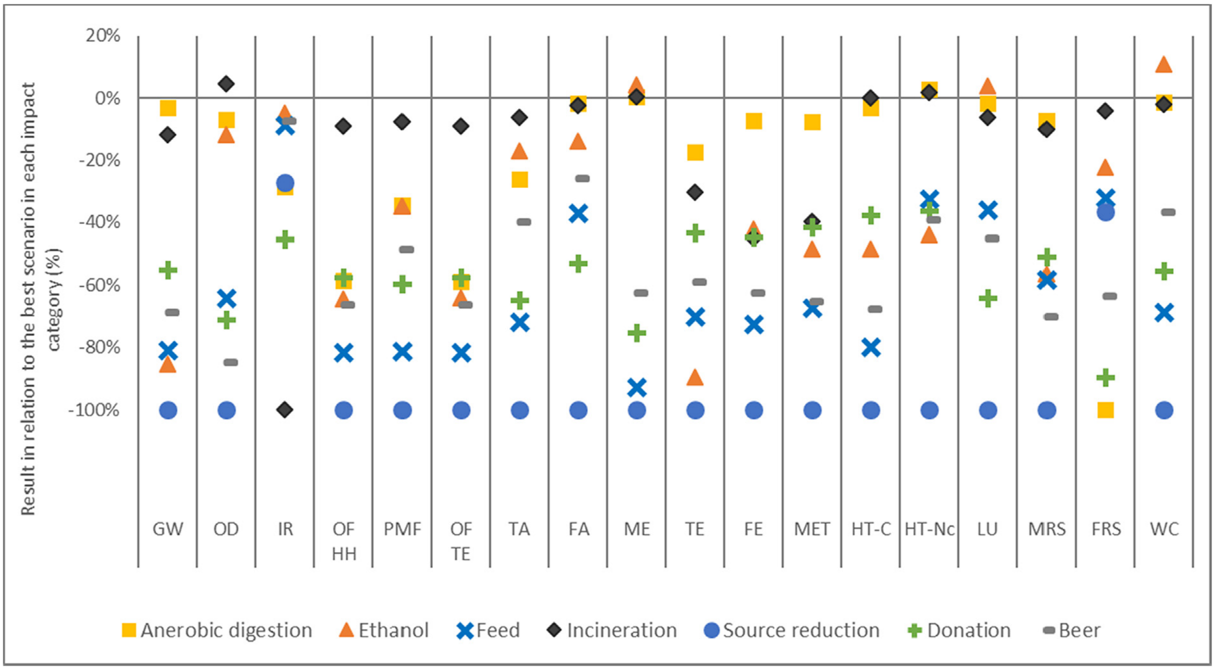

Fig. 3. Characterized results per kg of bread for the 18 impact categories analysed in relation to the best scenario in each category: Global warming (GW), Stratospheric ozone depletion (OD), Ionizing radiation (IR), Ozone formation, Human health (OF, HH), Fine particulate matter formation (PMF), Ozone formation, Terrestrial ecosystems (OF, TE), Terrestrial acidification (TA), Freshwater eutrophication (FA), Marine eutrophication (ME), Terrestrial ecotoxicity (TE), Freshwater ecotoxicity (FE), Marine ecotoxicity (MET), Human carcinogenic toxicity (HT-C), Human non-carcinogenic toxicity (HT-Nc), Land use (LU), Mineral resource scarcity (MRS), Fossil resource scarcity (FRS), Water consumption (WC).

est environmental savings in the fossil resource scarcity impact category due to the substitution of diesel by the upgraded biogas produced.

Electricity and heat production in Sweden have relatively low environmental impacts, which results in lower credits for the substituted electricity and heat studied here. Therefore, the low credits from substituted energy and the relatively low heating value of bread yields a small environmental savings for incineration (e.g. -0.08 kg CO_2 eq per kg of bread in the GW category). The detailed results for the characterization step in Fig. 3 is described in Section 3.1 of the Supplementary Material.

The valorisation and management pathways for surplus bread can be divided into three clusters with regards to their environmental savings. Source reduction of surplus bread has the highest performance in the majority of the impact categories. The next cluster includes donation as well as ethanol, beer and feed production. After source reduction, these pathways offer the highest environmental savings, although the relative value of the savings varies between the different impact categories. The cluster with the lowest environmental savings includes incineration and anaerobic digestion, which had the smallest savings in more than 80% of the impact categories.

The results from the different impact categories cannot be compared since they have different units. It is also not easy to ascertain the relative importance of the categories, and hence it is difficult to conclude which valorisation or management pathway offered the most environmental savings (across all impact categories). Although the use of weighting is controversial due to the underlying judgments, the method was used here to simplify and to facilitate analysis of the results. Fig. 4 shows the weighted results, where the results from the different impact categories are aggregated into a single score based on ReCiPe weighting sets (Goedkoop et al., 2009). The weighted results supports the hierarchy of source reduction as the best practice, followed by the cluster containing feed production, donation, beer production and ethanol

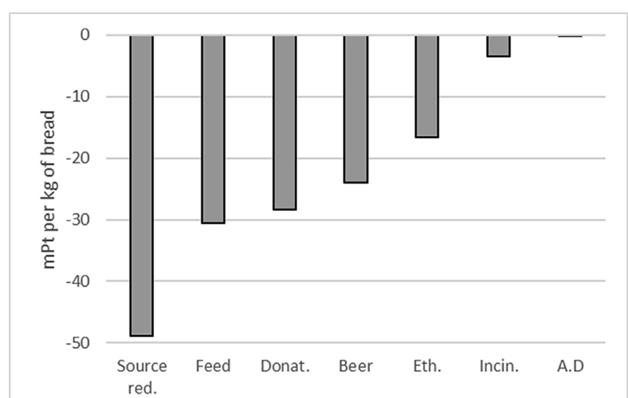

Fig. 4. Weighted results in milli points (mPt) per kg of bread for the scenarios analysed: source reduction (Source red.), feed production (Feed), donation (Donat.), beer production (Beer), ethanol production (Eth.), incineration (Incin.) and anaerobic digestion (A.D).

production and the final cluster of anaerobic digestion and incineration that offer the lowest environmental savings.

Besides the scenarios of prevention, namely source reduction and donation, it is possible to aggregate the results for the management of surplus bread in two groups based on the availability of infrastructure and the environmental savings offered by the scenario. The first group contains ethanol, beer and feed production, which have larger environmental savings than anaerobic digestion and incineration but at the same time, having limited available infrastructure in Sweden, and consequently might require long transportation distances. The second group includes anaerobic digestion and incineration, which have widely available local infrastructure but comparatively lower environmental savings.

Thus, assuming that the current infrastructure persists, it is important to define the threshold on the maximum transport distance that would still allow the management options in the first group to be superior. The calculation was done by comparing the two closest management scenarios in each group, i.e. incineration from the second group and ethanol production from the first group, using the weighted values (see Fig. 4). The calculations show that the surplus bread can be transported an additional 730 km, including return journeys, to the ethanol facilities before this valorisation pathway is no longer preferred to incineration. Of course, the threshold distances are longer for the other valorisation pathways and when compared to anaerobic digestion.

4.1. Sensitivity analysis

The results for the scenarios analysed in the sensitivity analyses are summarized in Fig. 5, and extend the results for the base scenarios, which are also show in Fig. 4. The characterization results are described in the Section 3.2 of the Supplementary Material. This is an attributional study and, as pointed out by Andreasi Bassi et al. (2017), the sensitivity analysis results should be used to better understand the current situation and not to assess potential impacts of future choices.

The highest percentage variation in the results was when changing the type of energy that was substituted during anaerobic digestion and incineration. When dirty energy is substituted by energy recovery from the system, there are larger environmental benefits, and *vice versa* for clean energy. The change in the energy mix from clean to dirty affects the results significantly. This indicates that the actual substitution that takes place in the market is critical for assessing the environmental benefits of using anaerobic digestion or incineration for bread waste treatment. Feed and ethanol production were less sensitive to the scenarios that were assessed. For feed production, the market average described in the ecoinvent database (Wernet et al., 2016) resulted in the lowest benefit comparing to the other feed scenarios assessed.

5. Discussion

Bread waste arises in different parts of the supply chain (Fig. 1). These flows are more or less suitable for the different waste management or valorisation practices. For example, returned bread, which is not contaminated with other food waste fractions, is a flow that is suitable for the majority of the waste management and valorisation options. In contrast, household waste, which is a mixed flow of different organic fractions, is suitable for less alternatives and is treated as a common municipal waste fraction. The most common methods for treating mixed food waste in Sweden are anaerobic digestion and incineration (Avfall Sverige, 2017), and these are the options studied in this work (Fig. 1).

The results from this study indicate that the return system, which is already implemented in Sweden, can be used to explore segregated waste management and valorisation pathways for surplus bread. These include ethanol and feed production, which have higher environmental savings compared to the municipal waste treatment methods. For example, the weighted results in Fig. 4 show that producing animal feed impacts the environment 8 and 160 times less than incineration and anaerobic digestion, respectively.

Surplus bread has been historically used as feed, and bread is characterized as a low-risk product provided it does not contain products from animal origin. Therefore, it is essential to ensure proper waste separation to avoid contamination with animal by-products. The main concerns with the use of surplus bread as animal feed are the moisture content and the nutrient variability. The latter risk can be minimized by introducing bread waste as part of a feed recipe (Inra et al., 2016). Besides the environmental benefits, selling the surplus bread as feed or as an ingredient to feed producers can provide an additional revenue stream to the bakeries and retailers. Moreover, it can reduce the costs associated with the waste treatment. Barriers to using surplus bread as animal feed

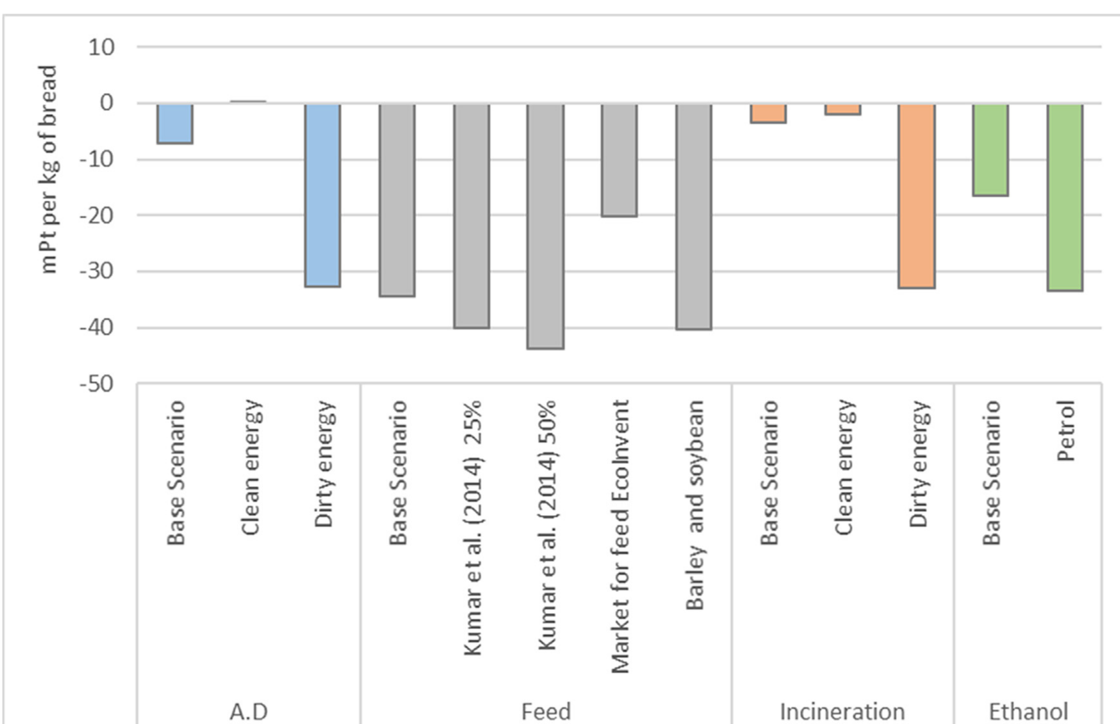

Fig. 5. Weighted results in milli points (mPt) per kg of bread for the scenarios assessed in the sensitivity analyses: anaerobic digestion (A.D), animal feed production (Feed), incineration, and ethanol production (Ethanol).

are identifying suitable selling channels for the products and complying with legislative requirements, particularly when the core business of bakeries and retailers are not the production of animal feed. The regulations are aimed to control the quality and safety of the feed produced, in order to protect animal health (Parfitt et al., 2016). It includes, among other things, registering the seller as a Feed Business Operator with their local authority, ensuring proper control of chemical or physical contamination with appropriate facilities and processes, and ensuring traceability processes that cover production and distribution.

A policy brief from the European Union (Hirschnitz-Garbers and Gosens, 2015) states that waste-based production of bioethanol can help mitigate environmental impacts and the competition between energy and food crops. The former claim is in agreement with the findings of this study. The document also states that one of the major challenges is the organisational effort and logistics for obtaining the waste. Bread waste is a good feedstock since the logistics of returns are already in place. Brancoli et al. (2019) estimated that 40,240 tons of bread is wasted each year by bakeries and retailers in Sweden. This is a waste flow that already has a logistics system in place and can thereby easily be used for bioethanol production. The potential of ethanol production from bread waste is estimated to be 12,000 tons per year, corresponding to approximately 8% of the Swedish annual production (SEA, 2019).

Beer production is an alternative to increase the diversity of infrastructure for bread valorisation pathways and consequently to decrease the transportation distances required. Moreover, microbrewery business models, such as the Toast Ale in the UK (Toast Ale, 2020), is a growing sector that requires relatively lower investments in infrastructure compared to the other scenarios. Donations are important from the environmental perspective and in the reduction of food waste, yet its relevance is demonstrated through its social implication on food security. However, due to the high amount of bread surplus in Sweden (Brancoli et al., 2019), it is unlikely that donations would be sufficient to solve the issue. Therefore, it is necessary to develop strategies on how to avoid surplus bread to be produced in the first place. Source reduction can be achieved, as pointed out by (Brancoli et al., 2019), through changes in the distribution system for bread, as products that are sold under take-back agreements have higher loss rate in comparison with products that are not governed by such agreements. However, such changes might require policy changes to encourage the actors involved to implement such actions (Brancoli et al., 2019). Anaerobic digestion and incineration have the advantage of being well-established technologies in Sweden. These processes can stabilize and convert different types of organic products to valuable products such as biogas and fertilizer from anaerobic digestion, and electricity and heat from incineration. However, as highlighted in this study, the uncontaminated flow of bread waste should be used at higher levels in the waste hierarchy due to the increased environmental benefits.

As shown above, differences in the availability of infrastructure for the waste management scenarios studied here might influence the environmental savings. Transportation distances must be taken into account when using the results from this study for a specific case. The results presented here estimate that surplus bread can be transported 730 km further to plants that produce animal feed, ethanol or beer that to plants for anaerobic digestion or incineration. If longer transportation distances are required then these options lose their benefits. It is important to assess specific cases individually. The transportation distances can be even more critical when selecting between the option to produce feed, beer or ethanol. A limitation of this study is the exclusion of the packaging material in the analysis. However, it has no influence in the treatment and valorisation scenarios assessed, since the production and disposal of the material is the same for all scenarios. The exception is source reduction, which would benefit from the avoided production of packaging. A previous study (Brancoli et al., 2017b) shows that packaging contributes around 10% in the results of the impact assessment in the life cycle of bread, depending on the impact category assessed. Nevertheless, this would not change the conclusions of this study regarding the ranking of the scenarios.

The results presented here can be compared with previous studies. For example, Eriksson et al. (2015) have done an LCA, limited to the global warming potential impact category, where different options for managing bread waste were compared. They found that incineration has larger environmental savings than donation, which in turn, has larger savings than anaerobic digestion. In contrast, the present study found greater environmental benefits for donation than for incineration and anaerobic digestion. This is also true when only considering the GW impact category, as in Eriksson et al. (2015). This difference can be explained by the choice of substituted products for anaerobic digestion and incineration. For instance, the incineration scenario in Eriksson et al. (2015) assumed that the energy produced by bread waste substituted fossil peat. As expected, these results are similar to those obtained here when the surplus bread substitutes dirty energy (see the sensitivity analysis discussed with respect to Fig. 5). The results of this study are also in agreement with Brancoli et al. (2017b), who concluded that using bread waste as animal feed has higher environmental benefits in comparison with anaerobic digestion.

6. Conclusion

The results from the life cycle assessment conducted here indicate a clear hierarchy for the valorisation and management of surplus bread. Source reduction, donation, or production of ethanol, beer or feed are favoured over anaerobic digestion and incineration. These two least preferred methods are currently used for treating municipal food waste in Sweden. Shifting from the current waste management pathways to the more environmentally friendly schemes can lead to environmental savings of 0.56 kg CO_2 eq. kg^{-1} surplus bread in the global warming potential category.

The current distribution system for bread in Sweden, where surplus bread is not mixed with other food waste fractions, facilitates implementation of the more environmentally friendly valorisation options studied here. Hence, two of the largest barriers for implementing these valorisation options – organisation effort and proper logistics – have already been overcome. Although the results presented in this study are valid for Sweden, they are also applicable to other countries, observing the discrepancies described in the sensitivity analysis.

Declaration of Competing Interest

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

Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.wasman.2020.07.043.

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