Resources, Conservation & Recycling 160 (2020) 104908

Contents lists available at ScienceDirect

Resources, Conservation & Recycling

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

Full length article

Waste prevention, energy recovery or recycling - Directions for household food waste management in light of circular economy policy

Irmeline de Sadeleer[a,b,*], Helge Brattebø[a], Pieter Callewaert[b]

[a] Industrial Ecology Programme, The Norwegian University of Science and Technology, NTNU, Trondheim, Norway
[b] Ostfold Research, Stadion 4, 1671 Kråkerøy, Norway

ARTICLE INFO

Keywords:
Organic waste treatment
Food waste prevention
Recycling rates
Energy efficiency
Avoided greenhouse gas emissions

ABSTRACT

Waste amounts are growing with increasing wealth and population. To curb this trend and reduce adverse environmental impacts, food waste reduction has been sat on the political agenda, together with ambitious material recycling and greenhouse gas (GHG) emissions targets.

This study analyses the environmental benefits of two waste management systems for household organic food waste, namely recycling by anaerobic digestion (AD) and incineration. Recycling rates, energy efficiency and GHG emissions are reviewed to determine the environmental profile of the downstream systems. The avoided GHG emissions achieved by the respective waste management strategies are further compared with the ones achieved by food waste prevention strategies. The study combines a material flow analysis (MFA) assessing the downstream system with published life cycle analysis (LCA) results for the upstream system. The method was demonstrated as a proof-of-concept case study for the city of Trondheim, Norway.

It was found that the recycling of food waste with AD performs better in terms of recycling rates and GHG emissions than incineration, provided that diesel is substituted by biogas. However, the energy efficiency of the incineration process was found to be slightly higher than of the AD option. Nonetheless, relatively small reductions in food wastage (15% and 30%) resulted in large amounts of avoided emissions, outweighing the benefits of recycling strategies. For mitigating climate change, the prevention of food waste clearly stood out as the most effective strategy. Norwegian authorities should focus equally much on household food waste prevention than on optimising food waste management systems.

1. Introduction

The European Union's approach to waste management is currently based on two main pillars. On the one hand, the Waste Framework Directive (2008/98/EC, Article 4) favours waste prevention over reuse, followed by recycling, energy recovery and finally disposal (European Commission, 2008).

On the other hand, the Circular Economy package adopted by European Commission in 2015 advocates an economic system that leaves no waste to be landfilled and that keeps all material flows in the economy through reuse, redesign, material recovery or energy recovery (European European Commission, 2015). Two main elements are introduced: the landfill ban on specific waste fractions such as organic waste, and specific collection and recycling targets for the various waste fractions.

Several European cities have in the context of a circular economy recently implemented source sorting of household organic waste, as this fraction contains high energy and nutrient levels and has a high potential for recovery. Environmental and economic benefits have hence led European authorities to focus on organic waste recycling and to largely invest in biogas facilities, resulting in Europe now being the world's leading producer of biogas (Hamilton et al., 2015; Scarlat et al. 2018). Anaerobic digestion (AD) converts waste into biogas and digestate, which can be used to produce electricity, heat, fuel and soil amendment products (Bernstad and la Cour Jansen, 2011, 2012; Bernstad Saraiva Schott and Andersson, 2015; Khalid, Arshad, Anjum, Mahmood, and Dawson, 2011; Modahl et al., 2016; Scarlat, 2018). Previous studies have concluded that AD as waste management option results in net environmental benefits when compared to incineration, composting and landfilling (Bernstad and Andersson, 2015; Edwards, Othman, Crossin, and Burn, 2017; Evangelisti, Lettieri, Borello, and Clift, 2014; Khoo, Lim, and Tan, 2010; Raadal, Stensgård, Lyng, and Hanssen, 2016). In general, biogas-based energy systems release lower amounts of greenhouse gas (GHG) emissions than fossil-

* Corresponding author.
E-mail address: ids@ostfoldforskning.no (I. de Sadeleer).

https://doi.org/10.1016/j.resconrec.2020.104908
Received 13 June 2019; Received in revised form 21 April 2020; Accepted 23 April 2020

based energy systems, especially when biogas substitutes fuel in transportation (Niu et al., 2013; Lozanovski et al., 2014; Lyng et al., 2015). The environmental benefits, however, depend on technology choices, the substituted products, the impact categories analysed and the area under study (Bernstad and la Cour Jansen, 2011, 2012; Modahl et al., 2016).

Even though waste prevention is a top priority in the European waste policy, reducing food waste has only lately been recognized as a priority area both at an international, European and Norwegian level. Sustainable Development Goal (SDG) number 12.3 sat the topic on the agenda in 2015 by aiming at "halving per capita global food waste at the retail and consumer levels and reduce food losses along the production and supply chains, including post-harvest losses". In Europe, food waste reduction has become one of the priority areas in the Circular Economy package adopted in 2015. In Norway, an agreement between the government and the food industry was concluded in June 2017, aiming at reducing food waste by 50% by 2030 (Klima og miljødepartementet, 2017).

These resolutions are of importance as food waste is in fact a huge challenge. Approximately one third of the food produced worldwide is wasted throughout the supply chain, representing loss of resources consumed, such as water, land, energy and labour (FAO, 2013). 12% of the total Norwegian household food consumption is wasted (Stensgård and Hanssen, 2015), of which two thirds are avoidable food products (Bernstad and Andersson, 2015; Bjørnerud and Syversen, 2017; Syversen, Hanssen, and Bratland, 2018). Vanham, Bouraoui, Leip, Grizzetti, and Bidoglio (2015) estimated that as much as 80% the European food waste can be classified as avoidable. Food waste hence indirectly causes large environmental damages, in addition to the direct impacts of waste treatment at disposal, and therefore give rise to ethical, social and economic concerns. The prevention of food waste can remedy to several of these aspects (Eberle and Fels, 2016; Salhofer, Obersteiner, Schneider, and Lebersorger, 2008; UNEP, 2016), however, prevention measures have until now received far less attention than waste treatment and recovery measures.

Few previous studies have compared the environmental benefits of food waste prevention with the ones of various waste handling solutions for several indicators. This comparison has been partly covered by Bernstad and Andersson (2015), who concluded based on LCA methodology that food waste minimization strategies result in far greater benefits for global warming compared to both incineration and AD. This supports the conclusions presented by Matsuda, Yano, Hirai, and Sakai (2012). Further, Hamilton et al. (2015) concluded using MFA methodology that food waste minimization strategies result in greater energy saving potential compared to food waste recycling strategies. There is little literature on this topic, and since the Circular Economy package seems to focus mostly on recovery and recycling strategies, while the waste hierarchy and overall policy should give highest priority to prevention, this limited knowledge is seen as a problem.

This study aims at analysing the performance of these two respective strategies (transition to a circular economy and waste prevention) for the case of food waste based on three relevant indicators in the light of CE: recycling rates, energy efficiency and generated/avoided GHG emissions. For capturing these three indicators which are closely interlinked, we use a multi-layer MFA framework to model the waste management system. The upstream (production system) environmental impacts and downstream (waste management system) impacts are linked by mass balance principles and CO_2 calculations by coupling the MFA results with LCA literature for assessing the avoided emissions indicator.

The methodology is demonstrated by a proof-of-concept study for the city of Trondheim, representing a typical Norwegian city. The functional unit is based on a food waste composition analysis for this city. The conclusions drawn from this study can be applied to other European cities facing the same waste management situation.

2. Methodology

This study aims at analysing the performance of two respective strategies: the transition to a circular economy and waste prevention for the case of food waste based on three relevant indicators. For doing so, the downstream and upstream systems are modelled separately. The downstream model is developed using material flow analysis (MFA) methodology extended with energy and emission data for assessing recycling rates, energy efficiencies and emission levels for two different recycling systems. The upstream model calculates CO_2 emission from the food production system, using data from LCA studies in literature. Both models are tailored to fit the current food waste situation and the actual plans for the city of Trondheim, with the 2017 system as reference and alternative scenarios in 2020 and 2025 as comparisons. This methodology is a proof-of-concept. The full MFA model is presented in S.I.

Different definitions and terms are found in literature when it comes to food wastage. These should be defined precisely as they are used in this paper to avoid any confusion. Note that neither the definitions nor the scope of the study does not include packaging.

(1) *Food waste* is 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) (Östergren et al., 2014).
(2) *Avoidable food waste* refers to materials that could have been eaten, making no distinction between what is elsewhere called "possibly avoidable" or "preference loss" (e.g. peels, seeds). Moreover, food which has passed its by-use date is also considered as avoidable, as the consumer could have planned more effectively (Östergren et al., 2014).
(3) *Unavoidable food waste* refers to materials that could not have been eaten under normal circumstances, for instance bones and orange peels (Östergren et al., 2014).
(4) *Food waste prevention* are measures taken before a substance, material or product has become waste, that reduce: (a) the quantity of waste, including through the re-use of products or the extension of the life span of products; (b) the adverse impacts of the generated waste on the environment and human health; or (c) the content of harmful substances in materials and products (European Commission, 2008).

2.1. Case study description

As of 2017, Trondheim had ca 191 000 inhabitants (SSB, 2017), plus some 10–15 000 students with another formal home address, and is thereby the third largest city in Norway. A large share of the waste is today incinerated, with heat recovery feeding into a district heat network that serves 30% of the space heating demand of the city's buildings. This provides annually some 600 GWh heat supply of which ca 80% energy from waste and the remaining 20% from peak load energy sources (Brattebø and Reenaas, 2012; Lausselet et al., 2016; Varme, 2017). Currently, paper, plastic, glass, metal, and residual waste are the fractions sorted out from households. There are three main collection technologies: surface bins represent the bins on wheel that are placed in front of each household; underground receptacles represent containers usually placed at a central point in an urban area and serve multiple households; and vacuum systems that are either stationary or mobile. These currently collect 83%, 12%, and 5% of the household waste respectively. The two latter technologies are underground systems, which together aim at reaching a collection capacity of 50% by 2030. Hence, organic waste is currently not sorted out or treated independently but is sent to incineration in the residual waste fraction. The city administration, however, today investigates the possibilities for building a central sorting facility, including the use of near infrared technology, aimed at sorting out organic waste for biogas production and plastic waste for increased ratios of material recycling (Trondheim kommune, 2017).

2.2. Data acquisition

A composition analysis was conducted to estimate the avoidable food waste amounts contained in household waste in Trondheim. Waste samples of 400–500 kg from five different residential areas were collected, reflecting the social-demographic differences of the city. This was important as it has been shown that the food waste amounts differ with factors such as age, sex, wages and time consumption on food preparation (Stensgård, Prestrud, Jørgen, Og, and Callewaert, 2019). The residual waste was first divided into non-food waste and food waste. The food waste was subsequently categorized as avoidable and unavoidable, as recommended by Lebersorger and Schneider (2011) and Bernstad and Cànovas (2015). Eight avoidable fractions were distinguished: fruits and vegetables, bread and bakeries, fish, meat, dairy products, eggs, meal leftovers, and other usable products, as advocated by the Norwegian national handbook for composition analyses (2015). For three of the sampling areas, the meal leftover fraction was further classified into the categories of bread, fish and meat, and others in order to get an indication of the amounts of carbon-intensive products present in that specific fraction.

A composition vector was developed based on the average of the 5 areas, representing the share of unavoidable and avoidable and food waste fractions divided on the specific fractions (Table 1). Based on weight, the meal leftover fraction was found to be the most important fraction (28%) of avoidable food waste, followed by fruits and vegetables (25%) and bread and pastries (21%).

However, the uncertainties linked to the analysis are likely to be significant due to errors that occurred during the out-sorting process. As the results are based on the fraction weight, incorrect out-sorting of heavy products or the inclusion of packaging influence the results. Nonetheless, the results are comparable to the ones presented in the literature and therefore considered acceptable for the purpose of this study.

Bernstad and Casanovas (2015) present a graph compiling the available food waste fractions results across the literature. It is difficult to compare in detail the different studies, as the classification of the food fractions differ across the studies, affecting the percentage-based results. However, the fruits and vegetable share nearly always the largest, most often followed by bread and pastries and/or prepared food. In some studies, the categories “diary” and “others” were also significant. The study of Stensgård and Hanssen (2015) was not included in that overview, but the division of the categories and hence the results are comparable to this study. Their result present that the meal leftover fraction was the most important fraction (31%) of the avoidable food waste, followed by fruits and vegetables (27%) and bread (13%).

Understanding the composition of the food waste is a first step for proposing targeted and efficient reduction solutions.

Table 1
Composition vector of the reference scenario 2017.

Waste fractions	Waste composition kg/cap	%
Bread and pastries	8,61	14%
Fruits and vegetables	10,05	16%
Meat	3,48	6%
Fish	0,85	1%
Dairy	2,54	4%
Other usable products	3,87	6%
Eggs	0,16	0%
Meal leftovers	11,59	19%
Unavoidable food waste	20,07	33%
Total	61,21	100%

2.3. Downstream system

The system boundaries are two-fold: the upstream and the downstream system (Fig. 1). The first one, representing the food production system, is described in 2.4.1. The system boundaries of the downstream system include the municipal household waste system for managing organic waste. The system boundaries start with the collection of waste from the households. The waste is transported either directly to the incineration facility or to a central sorting facility. In the second option, the waste is, after further sorting in the central sorting facility, either directed to a biogas facility or to the incineration plant. The incineration process produces heat which is used for district heating purposes and ashes which are disposed of. The biological treatment produces biogas which is used as fuel, and digestate which is used as fertilizers as it recovers nitrogen and phosphorus.

2.3.1. System boundary description

.

2.3.2. Model description

.

2.4. Upstream system

Following the MFA modelling principles of Brunner and Rechberger (2004), the model “A generic municipal solid waste management model” developed at NTNU (Callewaert, 2017) was adapted and applied to the organic waste system of the municipality of Trondheim (see S.I.).

The mass-balanced mathematical model analyses the resource and emission flows in the system, using three different system flow layers for this system definition. First, a material layer quantifies the annual flows of goods (on a waste fractions level) in the system. Second, an energy layer evaluates the associated flows of energy for each process in the system. Finally, an emission layer estimates GHG emissions (as CO_2-eq) from processes, transport and energy consumption. Due to the dependency between the layers, it is possible to examine how changes in the material flows, as a consequence of system changes over time, will influence the system-wide energy and emission performance.

The first layer calculates all material flows based on given waste flows quantities, on the composition vector and on known or assumed transfer coefficients for each process. A transfer coefficient in MFA theory determines how much of the sum of inflows to a given process is directed to a specific outflow direction. Transfer coefficients hence tell how efficient a process is in directing the waste throughflow in the desired downstream direction.

The calculated material flows are used to estimate the energy flows entering and leaving the system. The energy efficiency of the system is calculated by dividing the energy generated in incineration and biogas production with the feedstock energy from the waste and the consumption of energy from waste treatment processes and transport activities. This is used as an indicator for assessing the overall energy performance of the system.

The emission layer calculates the generated GHG emissions based on the results from the material and energy layer. The emissions included are caused by waste collection and transport, energy consumption during waste treatment processes and direct emissions from AD and incineration. Emissions caused by the life cycle of infrastructure are excluded. The emission factors are presented in S.I.

Additionally, avoided emissions are calculated based on the quantity of energy outputs calculated in the energy layer of the model. On the one hand, heat generated from the incineration process is assumed to replace electricity as heating source in households, thanks to district heating in Trondheim. On the other hand, biogas from the AD is assumed to substitute diesel in transport. For the substituted products, the amounts of energy generated are multiplied with the emission factors of

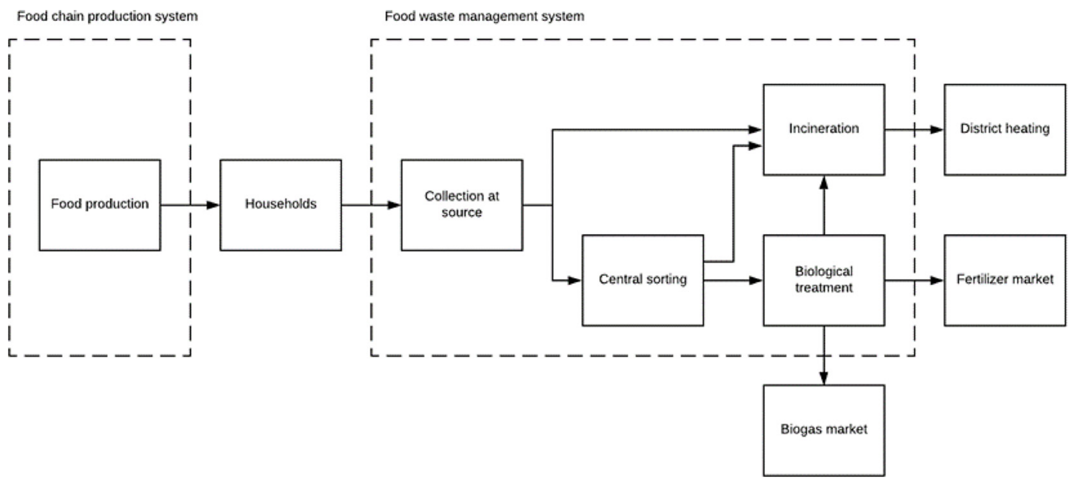

Fig. 1. Food chain system divided on upstream and downstream systems.

electricity (0,044 $kgCO_2eq/kWh$, ecoinvent 2) and diesel (0,273 $kgCO_2eq/kWh$, ecoinvent 2), representing avoided emissions and therefore with negative values.

2.4.1. System boundary description

Both the food supply chain and the waste management system are essential in the environmental assessment of food waste prevention (Bernstad and Cànovas, 2015). The upstream system, depicting the food supply chain, quantifies the embedded emissions of food commodities in a cradle-to-gate perspective.

The system boundaries of the upstream system follow the ones of Clune et al., 2017, Fig. 2.

At the farm, inputs from chemicals and fertilizers, fuel and energy inputs from irrigation and machinery for cultivation, harvesting and processing are included. In addition, transport and distribution to the regional distribution centre are part of the analysis. Outputs include emissions released from fertilized soils, plants and animals on the fields. The infrastructure, however, is not included.

It should be noted that the use phase which includes how consumers travel to shops, store and cook food is not included in the analysis. In fact, the aim of the study is to quantify the impacts of different political strategies which are out of reach for consumers. If a share of the avoidable food waste is properly prevented from being wasted at the household level, it can be assumed that the inflow of food commodities to the household is equally reduced. Consequently, it can be considered that the same amount of food commodities is avoided from being produced, and that the associated production-related emissions are

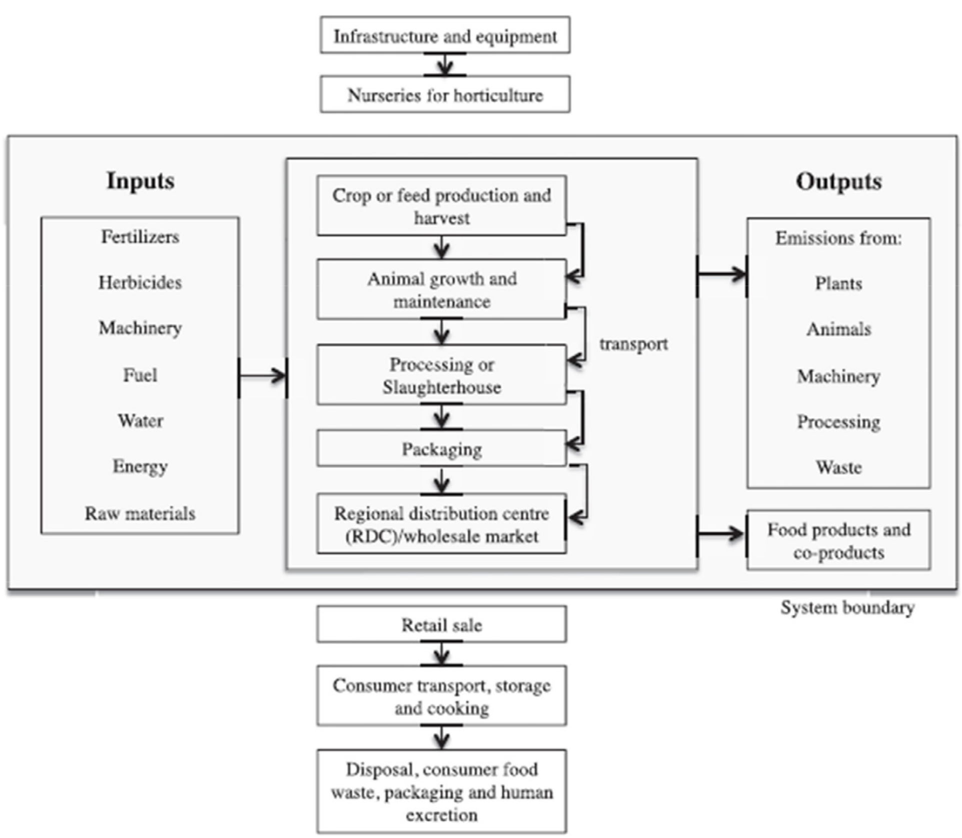

Fig. 2. System boundaries of the upstream system. Source: Clune et al., 2017.

avoided simultaneously. This analysis hence neglects the environmental impacts arising at the household level.

2.4.2. Model description

Clune et al., 2017 performed a meta-analysis of 369 LCA studies published between 2000 and 2015, from which they created a GHG emission database for a large quantity of food products. In this study, the embedded GHG emissions of the different avoidable food waste fractions were calculated (Table 1) by aggregating the median GWP values of the appropriate food products presented by Clune et al., 2017. The emissions were then multiplied with the composition vector in percent, giving the amount of CO_2eq embedded in one kilo of avoidable food waste with the typical composition of the food wasted by households in Trondheim. This composition was used as reference composition.

It is unlikely, however, that all food fractions will be equally reduced by prevention measures. The change within the various fractions therefore follows the results of the ForMat project (Stensgård and Hanssen, 2015). Two composition analyses were conducted in an interval of 5 years in the Norwegian city of Fredrikstad and in the Hallingdal valley. During this period, food waste prevention measures were actively established. The difference in food wastage between the two analyses was concluded to be a consequence of these measures. The change in shares within the composition vector in the various projections is subject to high uncertainty, but is the only available data in a Norwegian waste prevention context. Table 2 presents the new composition vector effected by food waste prevention measures.

The production of the unavoidable food waste fraction is equal for all scenarios and projections and was therefore left out of this study, as suggested by Martinez-Sanchez, Tonini, Møller, and Astrup (2016). Following the same approach as Bernstad and Andersson (2015), only the avoidable food waste is prevented. For calculating the embedded emissions from the meal leftover fraction, its composition had to be estimated. During the composition analysis, the meal leftovers were classified in three categories: bread, fish and meat, and others. Based on the fraction weight, it was estimated that 15% of the meal leftover was constituted of bread and pastries, 15% of fish and meat, and 70% of a mixture of all other fractions, mainly others and fruits and vegetables. The amount of embedded GHG emissions of this fraction was estimated in accordance with this allotment.

The emissions embedded in one kilo of avoidable food waste with the reference composition amounted to 3.44 kg CO_2eq/kg avoidable wasted food. In comparison, the emissions released for producing one kilo of avoidable food waste with the prevention composition amounted to 3.88 kg CO_2eq/kg avoidable wasted food. These results are in line with the literature (Bernstad and Cànovas, 2015).The difference between these two values, i.e. 0.44 kg CO_2eq/kg, represents the change in upstream emissions if the waste composition is altered through prevention activities. It can be noted that the embedded emissions of one kilo avoidable food waste increase as the amounts of avoidable food waste is reduced, which is explained by a reduction of the low-carbon intensive product share (bread and pastries) but a stagnation in the high carbon-intensive product share (meat, fish, dairy products).

Table 2
Composition vector affected by food waste prevention measures.

Waste fractions	Waste composition kg/cap	%
Bread and pastries	2,67	7%
Fruits and vegetables	10,05	27%
Meat	3,24	9%
Fish	0,79	2%
Dairy	2,44	7%
Other usable products	2,79	8%
Eggs	0,16	0%
Meal leftovers	14,72	40%
Total	36,85	100%

2.5. Scenario development

This study compares 3 main scenarios for the years 2017, 2020 and 2025: a Reference scenario (RS), a Central sorting scenario (CS) and a Prevention scenario (PS).

Reference scenario (RS) - describes the current waste management solutions in Trondheim in 2017 and assumes these solutions are used towards 2025. Organic waste is collected together with the residual waste and sent to incineration for district heating. Projections for 2020 and 2025 account for increased population and thereby increased food waste amounts. The share of collection technologies is adjusted with time, with above-ground bins decreasing to 80% and 70% for the 2020 and 2025 projections respectively, and the underground receptacles and vacuum systems increasing to 14% and 20%, and to 6% and 10%, respectively. The collection technology influences the energy requirement of the collection process. The share of biodiesel used in transport is assumed to rise to 15% in 2020, and to 50% in 2025. The LHV of food waste is estimated at 2500 kJ/kg for fruits and vegetables, 9200 kJ/kg for fish and meat (Christensen, 2011) and 4150 kJ/kg for all other fractions (Hung and Solli, 2012). The amount of organic waste per inhabitant was calculated based on historic organic waste generation data from 2007, 2012 and 2015, which show a slight increase over the years. A linear regression was applied and lead to the following: 61.21 kg in 2017, 68 kg in 2020 and 71.7 kg in 2025, which were used as reference scenarios for the different years.

Central sorting scenario (CS) – examines the effects of a new central sorting facility separating organic waste and different plastic waste fractions with optical sorting and near-infrared technologies. Central sorting facilities are promoted as important technological tools for increasing collection and hence recycling rates. Variants of this technology are currently spreading as state-of-the-art waste management practice in Norway and is therefore of importance to examine more closely. Based on data from a similar facility at ROAF outside Oslo, the organic waste separation efficiency of the facility is set to 50%, which reflects the performance of the currently existing technologies (Callewaert, 2017). Half of the household food waste is thus directed to the incineration plant together with other waste fractions, while the successfully out-sorted second half is sent to AD for biogas production in Verdal, 95 km outside Trondheim. The methane yield of food waste is assumed to be 153 Nm3/t (Hung and Solli, 2012). According to the city's plans, the central sorting facility will not be in operation before 2025 and is therefore only modelled for this year. The collection technologies, the share of transport fuel and the LHV of the food waste fractions are equal to those assumptions used in the RS scenario.

Prevention scenario (PS) – investigates the consequences of prevention measures, which decrease the amounts of avoidable food waste in 2017, 2020 and 2025 with 10%, 15% and 30% respectively. The measures themselves are not defined, only the effects of reduced avoidable food waste amounts are analysed. These effects of prevention are applied also to the CS scenarios, in a combined PS + CS scenario. The collection technologies, the share of transport fuel and the LHV of the food waste fractions are equal to the assumptions used in the RS scenario. Like for the reference scenarios, the amount of organic waste per inhabitant for the prevention scenarios were calculated based on historical data on which a linear regression was applied. This lead to the following: 56.91 kg in 2017 including 10% reduction, 61.11 kg in 2020 including 15% reduction and 57.27 kg in 2025 including 30% reduction.

2.6. Sensitivity analysis

A sensitivity analysis is used for assessing the robustness of certain parameters, and thereby their influence on the system variables. Input variables and assumptions are deliberately changed one at a time to analyse how they affect the outcome of the modelling. The changes in results are measured through the sensitivity ratio (SR) which is the fraction of relative change in the results (R) over the relative change in the input parameter (P) *(*Sandberg et al., 2017).

$$SR_p = \frac{\Delta R/R0}{\Delta P/P0}$$

The sensitivity analysis was only performed on the main parameters of the MFA system, influencing the three layers. The analysis was performed for the CS scenario of 2025 as this would allow a comparison of the parameters influencing both the AD and the incineration processes.

3. Results

The results are three-fold according to the three assessed indicators: material recycling, energy efficiency, and emission levels.

Regarding recycling rates, there is a common understanding in the EU that these must be increased in a circular economy. In addition to producing biogas, a biogas facility also creates biorest which recycles nitrogen and phosphorus. The analysed biogas facility uses a dewatering system, which leads to the nitrogen leaving the biorest stream. Only phosphorus is then recycled as fertilizer. However, the biogas facility under study has done tests regarding the use of liquid biorest, which would allow a recovery of the nitrogen in addition to the recovery of phosphorus (Ecopro, 2012).

Regardless of the amounts of nitrogen and phosphorus recycled, the European Commission defines in the waste legislation that all inputs to the AD facility are considered "*material recycled if the digestate is used as fertilizer in agriculture*" (European Commission, 2011). This means that the scenarios using AD obtain increased recycling rates compared to the RS, as long as the digestate is used as fertilizers.

As a result, CS and PS + CS scenarios reach 50% material recycling for the food waste fraction on the account that half of the waste is treated with AD. In comparison, the RS scenario obtains no material recycling as the total waste amounts are incinerated.

The net energy generation for the three scenarios at all points in time are presented in Fig. 3, together with the disaggregated consumption and generation factors. The net generation (yellow bars) is the result of the energy generated as biogas and district heating (green bars) minus the energy consumption in transport and processing (blue bars).

Given increased waste amounts, the energy consumption and generation are slightly increased over time. Not surprisingly, the prevention scenario at all times displays lower efficiencies compared to the RS scenarios because reduced waste amounts lead to reductions in consumed and recovered energy.

For all scenarios except the CS scenario, the process energy consumption is largely dominant over the transport energy consumption. It is higher in the CS scenario because of the long transport distances to the biogas facility.

Even if AD is capable of recovering slightly higher energy amounts than incineration (CS and PS + CS scenarios compared to the RS and PS in 2025), the net energy generation is slightly decreased due to the long transport distances.

The amounts of generated and avoided emissions are presented in Fig. 4 for the three scenarios at all points in time. The bars on the upper side of the graph present the amounts of generated emissions, whereas the bars on the lower side represent the avoided emissions.

The PS 2025 scenario and the PS + CS scenario clearly demonstrates the largest amounts of avoided emissions across all scenarios. The prevention of food waste (green bars) has undoubtedly the highest impact as climate mitigation strategy.

The substitution of diesel with biogas (dark blue bars) also leads to avoided emissions, as does the substitution of electricity with district heating (light blue bars) although to a lesser extent. As diesel has a much higher emission factor than electricity, its substitution highly increases the amounts of avoided emissions. Nonetheless, both substitution options result in far less avoided emissions than the prevention of food waste. The benefits of fertilizer substitution with digestate was neglected.

The avoided emissions outweigh the generated emissions in all scenarios, except in the current PS scenario. This latter is explained by the fact that the composition of the food waste arising with the influence of prevention measures include more carbon-intensive products. The prevention activities in the current RS are hence resulting in higher levels of GHG emissions, as the change in the share of fractions outweighs the benefits of 10% reduction in waste amounts.

The amounts of generated emissions are higher in the CS and PS + CS scenarios due to the increase of transport related emissions (yellow bars) compared to the RS and the PS scenarios. Only small emissions are released by the incineration process (orange bars) and the recycling process (red bars).

To analyse the robustness of the results, a sensitivity analysis was conducted for the most important parameters (Table 3). The food waste separation efficiency of the central sorting facility was analysed in terms of how it influences the system recycling and energy efficiencies. As expected, an efficient food waste out-sorting in the central sorting facility is crucial for improving the system recycling efficiency. However, it turns out to only slightly influence (reduce) the system energy efficiency. The energy efficiency is in fact much more influenced by changes in the methane yield and the LHV of "other food waste fractions". Regarding emission levels, the emission factor for diesel used in transport was found to be very sensitive, as most of the waste truck fleet is fuelled on diesel.

4. Discussion

This chapter first discusses the results and assumptions used in the study. Second, the limitations of the methodology are reviewed.

The system boundaries in this study exclude the household level, with storage in refrigerators and food preparation. As the meal leftover fraction stands for the largest share of avoidable food waste, its prevention would also influence the amount of energy consumed. Further, the study did not account for the rebound effect. As households spend less on food when food waste is prevented, the environmental impacts might be reallocated with spending on other products. This aspect should be taken into account for a holistic environmental policy development.

Comparing avoided emissions from improved waste management systems with the ones obtained from food waste prevention offer insights in the environmental potential of upstream versus downstream climate change mitigation strategies. In that regard, prevention strategies clearly result in larger benefits than recycling strategies. 30% reduction in avoidable food waste gave more than 5 times larger benefits than what was obtained with improved recycling strategies in the CS scenario. It must be noted that these conclusions are based on the assumption that a reduction in food waste leads to a reduction of food production. Avoided emission from the food production process was hence the determining factor for the overall benefits of food waste prevention, as observed also by Bernstad and Andersson (2015); Gentil, Gallo, and Christensen (2011) and Matsuda et al. (2012). There is however a risk that the amount of food waste prevented at the household level will arise higher up in the food chain, e.g. at the retail or production level. Such a shift in waste production will hence not prevent any GHG emissions – it is then necessary to have good waste management recycling systems in place, and therefrom avoid emissions through substituting carbon intensive products.

Combining prevention measures and a switch to AD would, nonetheless, offer optimal solutions for food waste management based on

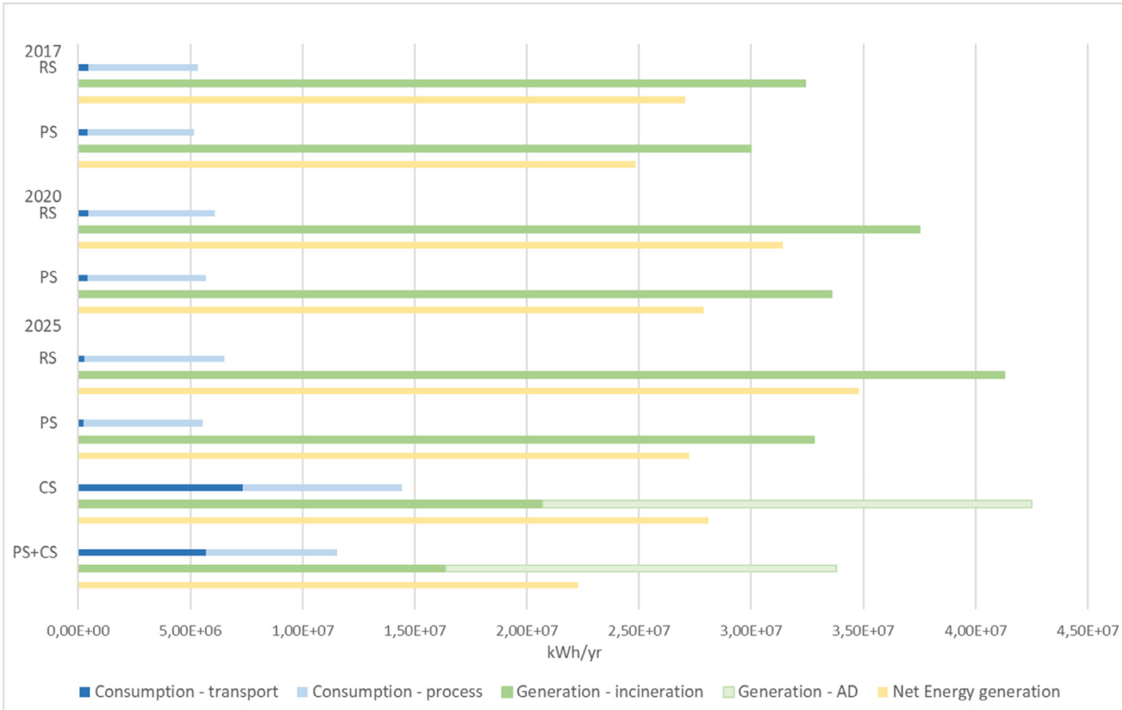

Fig. 3. Energy consumption, generation and net energy consumption for all scenarios.

the analysed indicators. It must be noted that the AD process depends on food waste as feedstock. Investing in a biogas facility will create a market for the food waste and might therefore not incentivize the prevention and reduction of food waste at the household level.

Further, analysing the differences between the RS 2025 and CS 2025 scenarios, excluding the upstream prevention results, allows for a comparison of the performance of the household food waste management systems for the three assessed indicators.

First, in accordance with the definition of the EU (European Commission, 2011), AD is the only treatment option resulting in material recycling. The sensitivity analysis disclosed that the effectiveness of the central sorting facility is a crucial parameter, highly affecting the recycling rate. Optionally, organic waste can be collected in separate bins and directly transported to a biogas facility, avoiding

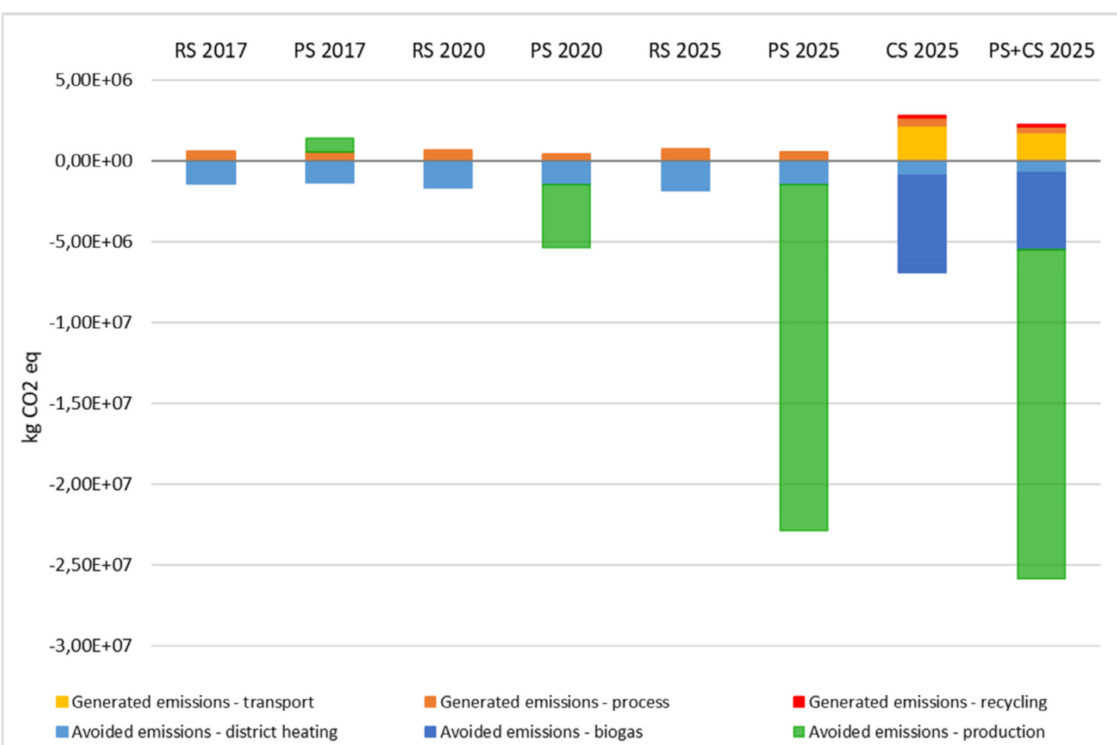

Fig. 4. Total emissions generated and avoided in kg CO_2 eq for all scenarios.

Table 3
Sensitivity analysis.

Parameters	Recycling efficiency SR	Energy efficiency SR	Emissions intensity SR
tkm X12		0,015	
Separation efficiency CS facility	0,997	−0,055	
Methane yield		0,763	
LHV meat		0,091	
LHV all other fractions		0,552	
Process emissions - El			0,027
Process emissions - Diesel			0,002
Process emissions - Heat			0,004
Process emissions - Oil			0,018
Transport emissions - Diesel			0,227
Emission factor for electricity			0,094

the diversion through a central sorting facility. Based on the experiences from ROAF, this would reduce the contamination of the other waste fractions, especially paper and plastics, allowing for overall higher recycling rates (Callewaert, 2017; Unander, 2017).

Second, it is beneficial to recover the feedstock energy present in food waste, as the generated energy amounts largely outweigh consumed energy across all scenarios. Even though the AD process generates slightly more energy than the incineration process, the CS scenario requires higher energy amounts because of the longer driving distance to the biogas facility, causing the total energy efficiency to decrease. The LHV, and especially the methane yield, were found to be quite sensitive parameters. The latter was assumed to be slightly overestimated (Hung and Solli, 2012), which might have given too high energy amounts generated for the AD process. However, the biogas facility in the case study operates with co-digestion: a feedstock mix consisting of sewage sludge, organic household waste and fish sludge. This mix delivers higher amounts of biogas than if only organic waste was used as input (Edwards et al., 2017). Therefore, the methane yield in use is higher than if only food waste would have been digested. In addition, it should be taken into account that the energy recovery from incineration can easily be connected to a heat or electricity grid. In comparison, biogas and fertilizers from digestate are not necessarily convenient to use without any infrastructural changes and due to premature markets or policy constraints. This might lead to the results of the study being more theoretical than practically implementable.

Third, when comparing generated emissions with avoided emissions, it is clear that the avoided emissions outweigh the generated ones in all scenarios. Nonetheless, the net benefit of the CS scenario is 5 times greater than of the RS scenario. From an emission perspective, it is hence beneficial to treat food waste by AD rather than by incineration, even though the generated emissions are larger in the CS scenario. It can be concluded that substituting diesel is more advantageous than substituting electricity. It must be noted that the Norwegian electricity mix was applied, influencing the results by its low carbon-intensity. This assumption influences the difference between the scenario results more than it would if the Nordic or European electricity mix had been applied. In addition, the sensitivity analysis disclosed that both the emission factors for diesel and electricity were influential, especially the latter one, which might also contribute to overestimate the low emissions of the CS scenario. Additionally, the avoided emissions of the CS scenario would have been increased if the substation of fertilizers with digestate had been included.

The emission factor for diesel is rather influential on overall emission results. This can explain the high values of the transport process in the CS scenario, where driving distance is decisive. An option for reducing the consumed energy amounts would be to either have a nearer location of the AD facility, to fuel the trucks entirely on biogas or another type of carbon neutral fuel, or to transport the waste by train.

The methodology in use has clear limitations. The presented MFA model is more appropriate for modelling complex waste management systems with several waste fractions. The downstream indicators give relatively straightforward results for the analysed scenarios, but the links between them are not always obvious. The MFA system consistently allows analysing these trade-offs, but is not used to its full potential when applied to this simplified system.

In addition, the analysis would have been strengthened by a cost-benefit analysis. However, the literature shows that using food waste for biogas production is socioeconomically profitable compared to incineration (NIRAS, 2013; Randby, 2016). Therefore, Norwegian authorities have proposed a regulation on the sorting of food waste from households (Miljødirektoratet, 2018). In this regard, a cost-benefit analysis comparing different treatment methods of food waste would not have political influence, as the question has already been debated upon. In the European context, the same argumentation yields: because the Circular Economy Package requires higher amounts of recycled materials which can only obtained for food waste with biogas production, the cost would not have a real influence. A cost-benefit of prevention measures compared to recycling measures would however been of interest and is suggested as further research.

In the author's eyes, the most interesting result is the comparison of the avoided emissions obtained by the upstream and various downstream strategies. It can be argued that an LCA would have been a more robust and appropriate methodology for analysing this question. The aim of this study was however to analyse different indicators for the downstream system in the context of a transition to a circular economy; and compare the energy and emission performance with the potential upstream energy and emission savings. For this aim, we view the presented methodology as robust.

One should be cautious in applying the actual values presented in the results chapter. Due to the uncertainties introduced with the composition analysis, this study only aims at ranking the performance of the various strategies and reveal critical parameters that influence the overall performance level.

5. Conclusion

The environmental benefits of household food waste prevention were compared to the benefits from various waste management strategies in regard to recycling rates, energy efficiency and emission efficiency, using MFA methodology combined with published LCA results. The method was demonstrated as a proof-of-concept case study for the city of Trondheim, Norway. In a reference scenario, food waste is treated together with residual waste and sent to incineration. A central sorting facility is introduced in a central sorting scenario, aiming at outsorting parts of the household food waste for use as feedstock in biogas production. A food waste prevention scenario was also tested, considering the effects of a reduction of 10%, 15% and 30% avoidable food waste in 2017, 2020 and 2025, respectively.

The most effective food waste management strategy seems to be a combination of prevention and recycling strategies. On the one hand, focus should primarily be on prevention strategies for mitigating climate change. The developed scenarios only considered a small reduction in avoidable food wastage, but these had significant benefits in terms of future CO_2 emissions. On the other hand, emphasis should be placed on the use of AD for biogas production as the future waste recycling option. This waste management system would mitigate resource depletion, as it highly increases the recycling rates, and would lower the emissions compared to the current incineration process in use. One should be cautious in applying the actual values presented in the results chapter. Due to the uncertainties introduced with the composition analysis, this study only aims at ranking the performance of the various strategies and reveal critical parameters that influence the overall performance level.

Even if prevention measures have been identified, their effects and environmental benefits are considered difficult to quantify (Salhofer et al., 2008) and are therefore seldom examined (Gentil et al., 2011). Further research on the topic is needed to successfully reduce the amounts of avoidable food waste, especially at the household level.

References

Bernstad, A., la Cour Jansen, J., 2011. A life cycle approach to the management of household food waste - A Swedish full-scale case study. Waste Manag. 31, 1879–1896. https://doi.org/10.1016/j.wasman.2011.02.026.

Bernstad, A., la Cour Jansen, J., 2012. Review of comparative LCAs of food waste management systems – Current status and potential improvements. Waste Manag. 32, 2439–2455. https://doi.org/10.1016/j.wasman.2012.07.023.

Bernstad, A., Andersson, T., 2015. Food waste minimization from a life-cycle perspective. J. Environ. Manage. 147, 219–226. https://doi.org/10.1016/j.jenvman.2014.07.048.

Bernstad, A., Cànovas, A., 2015. Current practice, challenges and potential methodological improvements in environmental evaluations of food waste prevention - A discussion paper. Resour. Conserv. Recycl. 101, 132–142. https://doi.org/10.1016/j.resconrec.2015.05.004.

Bjørnerud, S., & Syversen, F. (2017). Rapport plukkanalyse Trondheim 2017.

Brattebø, H., Reenaas, M., 2012. Comparing CO2 and NOX emissions from a district heating system with mass-burn waste incineration versus likely alternative solutions – City of Trondheim, 1986–2009. Resour. Conserv. Recycl. 60, 147–158. https://doi.org/10.1016/J.RESCONREC.2011.11.001.

Brunner, P., Rechberger, H., 2004. Practical Handbook of Material Flow Analysis. Lewis Publishers, Boca Raton.

Callewaert, P. (2017). Analysing the sustainability performance and critical improvement factors of urban municipal waste systems (Master's Thesis).

Christensen, T., 2011. Solid Waste Technology & Management, Wiley. ed. Blackwell Publishing Ltd, Chichester, United Kingdom.

Clune, S., Crossin, E., Verghese, K., 2017. Systematic review of greenhouse gas emissions for different fresh food categories. J. Clean. Prod. 140 (2), 766–783. https://doi.org/10.1016/j.jclepro.2016.04.082.

Eberle, U., Fels, J., 2016. Environmental impacts of German food consumption and food losses. Int. J. Life Cycle Assess. 21, 759–772. https://doi.org/10.1007/s11367-015-0983-7.

Ecopro (2012). Utprøving av flytende biogjødsel. Retrieved from 1.03.2020 https://ecopro.no/teknologi/pioner-pa-omrader/.

Edwards, J., Othman, M., Crossin, E., Burn, S., 2017. Anaerobic co-digestion of municipal food waste and sewage sludge: a comparative life cycle assessment in the context of a waste service provision. Bioresour. Technol. 223, 237–249. https://doi.org/10.1016/j.biortech.2016.10.044.

European Commission, 2008. Directive 2008/98/EC. Off. J. Eur. Union 312, 3.

European Commission (2011). Commission Decision of 18 November 2011 establishing rules and calculation methods for verifying compliance with the targets set in Article 11(2) of Directive 2008/98/EC of the European Parliament and of the Council (notified under document C(2011) 8165)OJ L 310, 25.11.2011, p. 11–16.

European Commission (2015). Closing the loop: commission adopts ambitious new Circular Economy Package to boost competitiveness, create jobs and generate sustainable growth. Press release. https://ec.europa.eu/commission/presscorner/detail/en/IP_15_6203.

Evangelisti, S., Lettieri, P., Borello, D., Clift, R., 2014. Life cycle assessment of energy from waste via anaerobic digestion: a UK case study. Waste manag 34 (1), 226–237. https://doi.org/10.1016/j.wasman.2013.09.013.

FAO (2013). Food wastage footprint: impacts on natural resources - Summary report.

Gentil, E.C., Gallo, D., Christensen, T.H., 2011. Environmental evaluation of municipal waste prevention. Waste Manag. 31, 2371–2379. https://doi.org/10.1016/j.wasman.2011.07.030.

Hung, C., Solli, C., 2012. Biogas from municiapal organic waste environmental holy grail? Energy Procedia 20, 11–19. https://doi.org/10.1016/j.egypro.2012.03.003.

Khoo, H.H., Lim, T.Z., Tan, R., 2010. Food waste conversion options in Singapore: environmental impacts based on an LCA perspective. Sci. Total Environ. 408 (6), 1367–1373. https://doi.org/10.1016/j.scitotenv.2009.10.072.

Khalid, A., Arshad, M., Anjum, M., Mahmood, T., Dawson, L., 2011. The anaerobic digestion of solid organic waste. Waste Manag. 31, 1737–1744. https://doi.org/10.1016/j.wasman.2011.03.021.

Lausselet, C., Cherubini, F., Del, G., Serrano, A., Becidan, M., Strømman, A.H., 2016. Life-cycle assessment of a Waste-to-Energy plant in central Norway: current situation and effects of changes in waste fraction composition. Waste Manag. 58, 191–201. https://doi.org/10.1016/j.wasman.2016.09.014.

Lebersorger, S., Schneider, F., 2011. Discussion on the methodology for determining food waste in household waste composition studies. Waste Manag. 31, 1924–1933. https://doi.org/10.1016/j.wasman.2011.05.023.

Martinez-Sanchez, V., Tonini, D., Møller, F., Astrup, T.F., 2016. Life-cycle costing of food waste management in Denmark: importance of indirect effects. Environ. Sci. Technol. 50, 4513–4523. https://doi.org/10.1021/acs.est.5b03536.

Matsuda, T., Yano, J., Hirai, Y., Sakai, S., 2012. Life-cycle greenhouse gas inventory analysis of household waste management and food waste reduction activities in Kyoto, Japan. Int. J. Life Cycle Assess. 17, 743–752. https://doi.org/10.1007/s11367-012-0400-4.

Miljødirektoratet, 2018. Utsortering og materialgjenvinning av biologisk avfall og plastavfall. M-1114. https://www.miljodirektoratet.no/publikasjoner/2018/oktober-2018/utsortering-og-materialgjenvinning-av-biologisk-avfall-og-plastavfall/.

Modahl, I., Lyng, K.-.A., Stensgård, A., Saxegård, S., Hanssen, O.J., .Møller, H., Arnøy, S., Morken, J., Briseid, T., & Sørby, I. (2016). Biogassproduksjon fra matavfall og møkk fra ku, gris og fjørfe. Status 2016 (fase IV) for miljønytte for den noske biogassmodellen BioValueChain. 1623.

NIRAS (2013). Kortlægning af indsamlings- og forbehandlingsmetoder for organisk affald. Projekt nr. 210105. https://danskaffaldsforening.dk/sites/danskaffaldsforening.dk/files/media/document/kortlaegning_af_indsamlings-_og_forbehandlingsmetoder_for_organisk_affald.pdf.

Östergren, K., Gustavsson, J., Hansen, J., Møller, H., Research, O., Anderson, G., Bellettato, C., Canali, M., Falasconi, L., Gaiani, S., Vittuari, M., Salhofer, S., Linzner, R., & Caspar, B. (2014). FUSIONS Definitional Framework for Food Waste.

Raadal, H.L., .Stensgård, A.E., .Lyng, K.A., .& Hanssen, O.J. (.2016). Vurdering av virkemidler for økt utsortering av våtorganisk avffall og plastemballasje. OR.01.16.

Salhofer, S., Obersteiner, G., Schneider, F., Lebersorger, S., 2008. Potentials for the prevention of municipal solid waste. Waste Manag 28, 245–259. https://doi.org/10.1016/j.wasman.2007.02.026.

Stensgård, A.E., Hanssen, O.J. (2015). Matsvinn i Norge 2010-2015. Sluttrapport fra ForMat-prosjektet. OR.17.16.

Stensgård, A.E., .Prestrud, K., Jørgen, O., Og, H., & Callewaert, P. (2018). Matsvinn i Norge, Rapportering av nøkkeltall 2015-2017. OR.28.18.

Stensgård, A.E., .Prestrud, K., Hanssen, O.J. & Callewaert, P. (2019). Matsvinn i Norge, Rapportering av nøkkeltall 2015-2018. OR.32.19.

Syversen, F., Hanssen, O.J., .& Bratland, H. (2018). Nasjonal beregning av mengde matsvinn på forbrukerleddet. 1/2018.

Trondheim kommune (2017). Planprogram - Avfallsplan for Trondheim kommune 2018-2030.

Unander, S.M.O. (2017). Analysing the sustainability performance and critical improvement factors of urban municipal waste systems (Master's Thesis).

UNEP (2016) Food Systems and Natural Resources. A Report of the Working Group on Food Systems of the International Resource Panel. Westhoek, H, Ingram J., Van Berkum, S., Özay, L., and Hajer M. Job Number: DTI/1982/PA ISBN: 978-92-807-3560-4.

Vanham, D., Bouraoui, F., Leip, A., Grizzetti, B., Bidoglio, G., 2015. Lost water and nitrogen resources due to EU consumer food waste. Environ. Res. Lett. 10, 084008. https://doi.org/10.1088/1748-9326/10/8/084008.

Varme, S. (2017). https://www.statkraftvarme.no/om-statkraftvarme/vare-anlegg/norge/trondheim/ (accessed 9.13.18).