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Research article

Circular economy in waste management – Socio-economic effect of changes in waste management system structure

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A R T I C L E I N F O

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A B S T R A C T

Due to the fast development, the EU economy has grown over its own raw material production. To enable future economic development, the EU is trying to develop a sustainable and resource-efficient economy. This path is emphasized through the idea of „Closing the Loop“ which is integrated into EU legislation by a Circular Economy Package and emphasizes avoidance of waste production and its recovery. New waste management goals require significant changes in the waste management system structure which introduces new problems and one of them is an increase in the costs for the system users (citizens). To assess the impact of these changes, the time-dependent Life Cycle Assessment based waste, material, and energy flow tracking framework is adapted and used to calculate material and energy production which can be monetised. As waste management plants/facilities are built with public money, to provide public service, in economic calculations annual cash flow of the system is equalized with zero, taking into account all incomes (incomes from products like energy vectors, secondary materials, and compost) and expenses (like an investment and operating costs). From these calculations, variable (volatile, time-dependent) and average system gate fee (operating cost per tonne of input waste), which is charged to the system users, are calculated. A possible increase in system cost can cause the issue of social unacceptance, which decision-makers, elected by the citizens, want to avoid. Results show that energy recovery of waste generates higher income then material recovery while overall lower system costs, and lower sensitivity of the system cost, is observed in the material recovery based scenarios. The lowest system costs are calculated for the scenario which combines material and energy recovery and avoids investments in final disposal/recovery facilities by outsourcing this service. The main problem with outsourcing the final disposal/recovery stage is the uncertainty of the cost of such service. It is found that this kind of approach has not been previously used for the assessment of the socioeconomic sustainability of the whole waste management systems and represents a good tool for decision-makers.

1. Introduction

A large part of the EU industry's raw materials needs is covered by import (EC, 2018a). The European Commission (EC) launched the Raw Materials Initiative (EC, 2008) as a response to this problem, with a goal of ensuring a sustainable supply of raw materials, and European Innovation Partnership on Raw Materials (EC, 2013), with a goal to develop local raw materials industry. Path to severing the link between increasing energy and material consumption and the environmental impact from economic growth is outlined in the Roadmap to a Resource Efficient Europe (EC, 2011a) as a part of the Europe 2020 strategy in which the Circular Economy was proposed as the best concept to undertake an economy transformation (EC, 2010).

Next to metals, the EU is most dependent on fossil fuels import (EC, 2018c), where, in the last twenty years, the import of fossil energy carriers increased by 23% (EC, 2018d). In order to address these issues, the EU adopted the 2020 Climate and Energy Package (Council of the European Union, 2009) and 2030 Climate and Energy Framework (Council of the European Union, 2014), which is in line with the long-term goals set out in the Roadmap for moving to a competitive low-carbon economy in 2050 (EC, 2011b).

Next to problems of satisfying input, EU economic development as such is also unsustainable from the standpoint of output, i.e. the generation of large quantities of waste that do not fit into the established natural cycles. The total EU waste production in 2014 amounted to 2500 million tonnes, of which 53% was not reused, recovered or recycled

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(European Parliament, 2018a). The EU economy thus loses significant amounts of resources, i.e. potential secondary raw materials.

While such a large amount of generated waste represents a problem, it also provides the opportunity to facilitate previously identified problems. This is recognized by the EU, and the first steps towards a resource-efficient Europe were made through the Waste Disposal Directive (European Council, 1999) and Waste Framework Directive. Next steps were made by the EU action plan for the Circular Economy (European Parliament and Council, 2015), and Waste Package (European Parliament, 2018b) which include the changes in re-use and recycling goals for Municipal Solid Waste (MSW) (60% by 2030 and 65% by 2035), MSW disposal goals (max 10% by 2035) and recycling of packaging waste goals (70% by 2030.).

Although in the sustainability analysis, no aspect can be ignored, the life cycle approach has gained on importance in the waste management (WM) field because of the provisions of the Waste Framework Directive which stipulates that potential deviations from the WM hierarchy must be justified through considerations of impacts on the life cycle level. This is often achieved through a Life Cycle Assessment (LCA) which is a standardized scientific method for assessing the impact on the life cycle level. In addition, the EC emphasized the LCA as the "best framework for assessing the potential environmental impacts of products" (Commission of the European Communities, 2003). Therefore, a number of LCA approaches for the analysis of the Waste Management Systems (WMS) have been developed and used for a whole range of the analyses (Gentil et al., 2010): Montejo et al. (2013) used specialised WM LCA software to analyse potentials of MBT plants, so did Lima et al. (2018) for comparison of different WMS, while Zhou et al. (2018) used on-site data and non-specialised LCA software for environmental performance evolution of MSW management. Also, simpler assessments were conducted which are more suitable for decision making (Tomić and Schneider, 2017, 2018). There are few publications with combined environmental and economic assessments, as LCA is usually performed separately from the economic assessment, often with different scope, system boundaries, conditions/assumptions and analysis timeframe (Norris, 2001). Zhang et al. (2020) coupled LCA of blanket production from PET bottles and Franchetti (2013) analysed carbon emissions and the economy of Anaerobic Digestion (AD) of food waste using the LCA framework, but these analyses cover only a small part of the waste management/recovery system. On the other hand, LCA analysis entails monitoring of all mass and energy flows (internal and external), which can be used in economic analysis.

While the economy is the main driver for solving material and energy-related problems of the EU economy, solutions for the waste-related problems are mainly directed to facilitate a solution for those two problems (next to ecological problems). Meeting the EU goals requires significant changes in the structure of the WMS which introduce new problems for the municipalities. Those problems are particularly emphasized in the newer EU member states which have problems with meeting even the older WM goals. One of the biggest problems is the social acceptance of these changes. WM is an activity that needs to be paid by system users (citizens). Consequently, changes in WMS costs represent an economic question that could accentuate or alleviate the problem of social acceptance by increasing or reducing waste management costs.

Due to the importance of the economic aspect, the techno-economic analysis of the MSW collection strategy was carried out by Franzese et al. (2009), but the performance of other parts of the WMS were not monitored. The results of a more detailed analysis of waste collection costs were presented by Syazwina et al. (2011), however, only for the actual system in Italy without looking at alternatives. There are also other papers that analyse the economy of WMS where Bertanza et al. (2018) analysed only a part of the system, while others considered only one technology (Tomić et al. (2017), Tan et al. (2015) and Tomić et al. (2016) Waste-to-Energy (WtE), Greco et al. (2015) collection and Luz et al. (2015) waste gasification). Unlike in previous analyses, Fei et al. (2018) conducted the complete economic analysis of the WMS in Japan but the emphasis has been put only on assessing the impact of household waste separation on the cost of existing WMS and alternative system configurations were not considered. Minoglou and Komilis (2013) estimated only the influence of the municipality's size on the total cost of WM. On the other hand, there are papers that have analysed different approaches and technologies for WM. Chifari et al. (2017) analysed various WM technologies and their capital (CAPEX) and operating (OPEX) costs, but the impact of the integration of these technologies on the overall WM was not assessed. The total economy of the WMS through nine different scenarios that cover different technologies has been analysed by Bel and Fageda (2010), but their analysis does not take into account the dimension of time.

When designing WM projects, future changes in the quantities and composition of waste should be taken into account. These changes are inevitable because of the previously identified legislative provisions, as well as socioeconomic and technological trends which are the result of social development. Analyses, and planning, on the principle of „business as usual", assume a constant increase in the quantity of MSW with uniform composition. This is linked to the Waste Kuznets Curve hypothesis (WKC) (Aleluia and Ferrão, 2017), which claims that waste generation correlates with economic growth to a certain point (threshold), after which the waste generation decreases. This threshold has been achieved in some households, i.e. in the provinces of Japan (Aleluia and Ferrão, 2017) and Italy (Khan et al., 2016), which show that solving the waste problem by building a new processing and recovery plants can become economically unsustainable, as the end of the continuous increase in the MSW generation is already evident. In the EU, although the absolute decoupling trend, of waste generation and economic growth, is not present, the link between waste generation and economic indicators is weaker than in the past, indicating a relative decoupling (Ichinose et al., 2015). In addition, policy decisions and development strategies also have an impact on the quantity and composition of waste. The introduction of new technologies and solutions within the municipal WMS, aimed at reducing waste production, reuse and recycling, also reduces the amount of waste to be managed (Mazzanti et al., 2008). These effects are particularly pronounced in the newer EU member states, which must rapidly implement changes to achieve the goals set out in the EU legislation. However, change cannot happen overnight (Mazzanti and Zoboli, 2008), and the economic sustainability of building new facilities can come under a question. Because of this, predicting future quantities is of most importance (Christensen et al., 2009). Also, changes should not cause an excessive increase in costs for citizens that would result in their disapproval of newly established systems. Such disapproval easily leads to problems such as non-participating in separate waste collection schemes and illegal waste disposal. Therefore, the system must be designed to limit the increase in costs during the projected economic period (system repayment period) and limit the sensitivity of systems economy to market changes.

In this paper, this socio-economic problem is analysed through a waste, material, and energy tracking approach built upon life cycle assessment (LCA) based framework where unit process (UPR) data are used for modelling of analysed technologies. Life cycle impact (LCI) UPR data present the process/technology on the principle of a black box, where all input and output, material, energy, and waste flows data are separately shown. The developed model is used to track input dependent mass and energy flows needed for calculation of material and energy production of the system which are used in subsequent economic calculations. Analysed technologies are further defined by input dependent functions defining its investment and operating costs. By using LCA based system modelling, holistic, system-wide socio-economic assessment is conducted, which puts light on relations among different technologies and WM systems that are in previous researches analysed independently of each other. In comparison with previous techno-economic analyses of WMS, where time-dependent changes were neglected or simply extrapolated from historical data, the time component is taken into account by considering socio-economic and legislation dependent changes on the quantity and composition of waste streams. In this research, the elaborated framework is used together with data on prognosed, time depended, changes in quantities and composition of waste, resulting in a holistic socio-economic analysis of the overall cost of the entire system in relation to a time scale, which represents a novel approach to these kinds of analysis. Reported results show WM cost per mass of mixed MSW, which at the same time represents the measure of social acceptability of proposed systems as the cost of the system always falls on the backs of its users, i.e. citizens. Through this research, authors are trying to answer the research question: What is the impact of changes in the structure of the municipal WMS, which are induced through legislation changes, on the socio-economic acceptability of such changes?

2. Methods

This study builds upon a previously published LCA based system modelling framework (Tomić and Schneider, 2017, 2018) which is developed for environmental sustainability assessment. From those publications LCA based waste, material, and energy tracking framework is adopted and used for modelling of new scenarios and expanded in a way to encompass the economics of WMS, which previously was not looked upon. The system boundaries include overall municipal WMS and all associated activities/processes leading to secondary raw material production, on the material side, and produced energy vectors, on the energy side. Post-production activities/processes have not been considered, and obtained products were valuated on corresponding markets. As far as the time frame is concerned, the analysis uses the annual input data, while the time component has been considered by conducting consecutive analyses. A discussed time frame of analysis is up to 3035, which is defined by a previously elaborated European legislative framework. The boundaries of specific systems are case dependent and explained in chapter 2.5.

2.1. Technology modelling

To model the analysed systems, energy and material (and waste) inputs and outputs of each technology are tracked. Most of the used technology data are adapted from Ecoinvent LCI UPR datasets (Ecoinvent Centre, 2016), while the Mechanical and Biological Treatment (MBT) facility and gasification facility are modelled on the basis of literature data for facility for gasification of MSW for a production of electricity (MBT (McDougall et al., 2001), and gasification (Ducharme, 2010)).

Mainly European UPR LCI data were used for modelling of the system and are adapted to suit the local waste composition and the resulting Lower Heating Value (LHV), biogas and landfill gas production, as well as the share of methane. Waste sorting technologies data are adapted so that sorted waste material streams correspond to the local waste composition. LHV and quantities (shares) of mixed MSW, sludge from the AD, a high calorific fraction of waste from MBT (Refuse Derived Fuel (RDF)) and residual wastes were used to calculate the overall input streams to thermal treatment plants. These calculations were done on the basis of each analysed time period, as described in chapter 2.2. Quantities of individual waste streams were calculated from the actual and prognosed waste collection data and technical data for each particular technology.

As composition of some individual waste fractions were not given in reports, they were modelled on the basis of the statistical information on metal packaging (Croasdell et al., 2015), for the proportion of aluminium and steel in metal waste, and on the basis of the UPR data for plastic waste sorting technologies, for quantities of individual plastic materials in plastic waste.

As the construction of a new WtE plant is considered, its UPR set is modelled on basis of a modern (existing) cogeneration WtE plant in Sønderborg with a total efficiency of energy transformation of 90.5% (cogeneration ratio of 0.22) (Energinet, 2012), and used, with input waste characteristics and quantities, to calculate plants energy production.

Through Ecoinvent LCI data, the landfill is modelled as a regulated landfill with a landfill gas collection system that is subsequently flared off. Since in this analysis collected landfill gas is used for further energy transformations, the existing LCI UPR data set is adapted according to US Environmental Protection Agency (USEPA) (AP-42) (USEPA, 2019) compilation of air pollution factors related to MSW landfills, which contains a mathematical model for estimating landfill emissions and emission reduction factors in accordance with an emission control system. Using these data, values related to flaring off the landfill gas were deducted from the values reported in the data set and the value of the generated landfill gas is added to the LCI UPR dataset.

2.2. Amounts, composition and energy potential of waste

The common method for predicting the MSW amounts takes into account population growth and the long-term changes in the amount of generated waste per capita, which are simply extrapolated. This simple and quick approach neglects social and economic influences on the amount of generated MSW (Karavezyris, 2001). As consideration of socioeconomic indicators enables a significant reduction in prediction error (Armstrong, 2001), an LCA-IWM prediction model (Boer et al., 2005) was used for predicting changes in the MSW amounts and composition. By setting the prediction boundary conditions, as defined by the European Directive on Landfilling and the Waste Framework Directive for the period up to 2020 and the Waste Package for the period up to 2035, the predicted quantities also encompass the influence of EU legislation.

Composition and LHV of waste are interconnected by chemical composition as expressed by the Mendeliev equation (Sharma, 2006). The chemical compositions of individual fractions were obtained from Magrinhos and Semiaos (2008) research while LHV of corresponding waste streams were calculated from the share of individual waste fractions. LHV of the residual sludge from the AD process is taken from the Ecoinvent LCI dataset for the technology of thermal treatment of digestate and accounts to 2.42 MJ/kg.

Biomethane is generated by two technologies: a landfill gas collection system and an AD plant. Landfill gas yield was modelled using the LandGEM model with an assumed biogas collection period of 30 years. In this model, the total generation of biomethane and its time distribution are determined by decomposition rate (k) and potential for methane generation (L_o). The decomposition rate was modelled for mixed MSW and therefore its value was 0.04, which correlates to generating 70% of the total possible amount of biogas within 30 years in regions with more than 630 mm precipitation per year (Frischknecht et al., 2007). The potential for methane generation L_o is modelled on the basis of the share of each type of bio-waste in mixed MSW and calculated specifically for each analysed system and time step of the analysis. In the analysis of the existing situation, it was not possible to neglect historical waste, therefore to calculate the landfill emissions, it was assumed that the same amount of waste was disposed of during the previous period. The amount of the generated biogas in the AD plant is calculated using the same mathematical model, but without the time distribution of biogas generation and overall produced biogas is generated in the same year in which the biowaste was processed.

2.3. Socioeconomic analysis

In this section analysed technologies are additionally defined through economic functions. Economic data were adapted from the literature as CAPEX data, dependent on the annual capacity of the considered technology, and OPEX data, dependent on the amount of processed waste. CAPEX and OPEX data for modelling financial side of

WM technologies are not easy to obtain and vary significantly from case to case. This is due to (micro) location economic specificity, differences in used technology, differences in the definition of capital/operational costs for equipment suppliers/plant operators, etc. Because of it, given economic functions were modelled based on several different MSW facilities (by using real (reported) data or (in some cases) estimates), by which mean data for not specific (generic) plants are modelled by given functions. Functions for waste separation and disposal technologies are presented in Table 1.

As plants for sorting of separately collected waste fractions do not have high investment cost and therefore can be built as distributed facilities, economic data for these plants are expressed as a function of the total cost (c_{tot}) - Equation (11)

$$ctot = 51,515x^{0.73}[€] \tag{11}$$

where x represents the input stream in tons per day.

The economy of the AD facility is modelled on the basis of available economic (Minoglou and Komilis, 2013; Franckx et al., 2010) and data on subsequent transformations of the obtained biogas (Karellas et al., 2010). Cost functions for the AD plant in the dependence of the capacity are given by equations (12) and (13). The landfills, whose economy is described in the equations in Table 1, also include landfill gas collection systems, where collected gas is transformed into other energy vectors. The economy of a biogas cogeneration (CHP) engine is modelled based on Karellas et al. (2010) research. The CHP engine represents a significant investment, which is confirmed by a fact that it increases the investment by 33.62%, compared to the AD plant without it. According to the same source, maintenance and operating costs are estimated at 5% of the investment cost. Regarding the costs of biogas upgrade to biomethane, they are modelled on the basis of data for different technologies (Warren, 2012), while CAPEX and OPEX functions are modelled by regression analysis - Figs. 1 and 2.

The cost of the Compressed Natural Gas (CNG) station is defined by real technologies data (Mitchell, 2015). The capital cost function dependent on a plant's capacity is defined by regression analysis - equation (24). Maintenance and operation costs are estimated at 5% for plants with the capacity of up to 130 Nm^3/h of CNG, with the cost increment of 0.16% per Nm^3 of CNG after that value.

Economic data for WtE plants were calculated based on an extensive overview of available data, based on which a set of economic data for existing plants is constructed and regression analysis was carried out to obtain plants CAPEX function in a relation to its size – Fig. 3.

Operating costs are estimated on the basis of literature data (Eunomia, 2009; Lončar et al., 2009) to 4% of investment cost for the fixed part and 19.1 €/t of processed waste for the variable part. These costs do not include labour costs, which is calculated based on the average monthly gross salary and number of employees (Lončar et al., 2009; EC, 2006).

Regarding gasification plant, its economic functions are also modelled through regression analysis from available data on MSW gasification plants where produced syngas is internally used for the production of electricity. Fig. 4 presents a regression analysis for estimating CAPEX function, and Fig. 5 for OPEX function.

An overview of the obtained economic functions is given in Table 2.

These functions define the system's expenditures while incomes are generated through energy and secondary raw materials sales, as well as waste management fees paid by system users.

In the legislation framework of a number of countries, for defining a value of secondary raw materials from waste, data from specialist trade publisher portals like EUWID and Letsrecycle are used. In this research, Letsrecycle material prices data are used because of their accessibility – Table 3.

Such a system is commonly funded by the citizens through a municipal utility company. As this system provides a public service and is not designed to make a profit, the socioeconomic model tracks the minimum required cost of the system that has to be covered by the system users (MSW management fee (T)) in order to equalize the annual cash flow with zero – Equation (28).

$$\sum_{i=1}^{n} Cc_i + \sum_{i=1}^{n} Co_i - \sum_{j=1}^{m} (M_j c_j) - \sum_{k=1}^{o} (E_k c_k) - T = 0 \tag{28}$$

C_c – total annual cost caused by the capital cost of the technology *i*.
C_o – total annual operating and maintenance cost of the technology *i*.
M – the waste material stream entering a secondary material market *j*.
E – produced energy flow by technology *k*.
T – MSW management fee paid by its users
c_j – purchase price of secondary material *j*
c_k – the purchase price of the energy vector produced by the technology *k*.

Reducing the total cost of the system results in a reduction in the MSW management fee which is paid by the citizens. MSW fee is paid by the amount of produced mixed waste which stimulates citizens to participate in the operation of the WMS through a system of separate collection of MSW.

2.4. Definition of case studies

The previously described methodology was extended by locally dependent variables. The analysed case is an example of the city with over 790 thousand inhabitants (~1200 inhabitants/km^2) and 99% of the population is participating in the MSW collection system (EC, 2015a, 2015b). The city has a low level of primary waste separation (13.5%) and represents new EU members with municipal WMS which still does meet the objectives of the EU legislation.

Changes in the WMS are modelled through six possible scenarios and three time frames. The considered technologies cover mostly used technologies for the treatment of MSW in the EU and gasification which is a new technology in this sector. The possibility of thermal treatment of residual waste in cement plants is also considered.

The generated waste data in the time frame 2020 are adopted from MSW report (Požgaj et al., 2019) and composition and morphological characteristics analysis report (Domanovac et al., 2018). As analysed city still does not meet EU legislation goals for 2020, during forecast, it is assumed that it will meet them with five years delay (2025), while the time frame to 2035 assumes achieving all the objectives of existing EU legislation. Projected amounts of MSW are shown in Table 4, the composition of mixed MSW is presented in Table 5, and waste energy potential in Table 6.

A large reduction in the mixed waste by 2025, compared with Today's quantities (Table 4), is due to an increase in primary waste separation (to meet legislation goals). Total waste production in the period from 2025 to 2035 is on the same level as MSW generation growth compensates for the impact of a further increase in primary separation efficiency.

The LHV (Table 6) was calculated from chemical composition of

Table 1
Economical functions of separation and disposal facilities.

Technology	CAPEX [€]	OPEX [€/t]	Equation
Closed composting [a]	$c = 2,000x^{0.8}$	$o = 2,000x^{-0.5}$	(1, 2)
Open composting [a]	$c = 4,000x^{0.7}$	$o = 7,000x^{-0.6}$	(3, 4)
Landfill (x > 60,000) [a]	$c = 3,500x^{0.7}$	$o = 150x^{-0.3}$	(5, 6)
Landfill (x < 60,000) [a]	$c = 6,000x^{0.6}$	$o = 100x^{-0.3}$	(7, 8)
MBT [b]	$c = 23,844x^{-0.404}$	$o = 3,353.1x^{-0.404}$	(9, 10)

c – investment cost [€], o – operation and maintenance costs [€/t], x – input waste stream [t/year].
[a] **Minoglou and Komilis (2013).**
[b] **McDougall et al. (2001).**

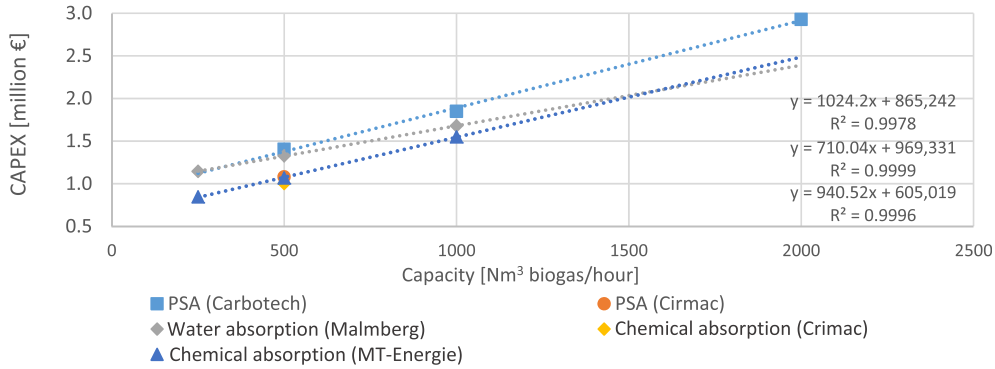

Fig. 1. Regression analysis of CAPEX functions for biogas purification technologies.

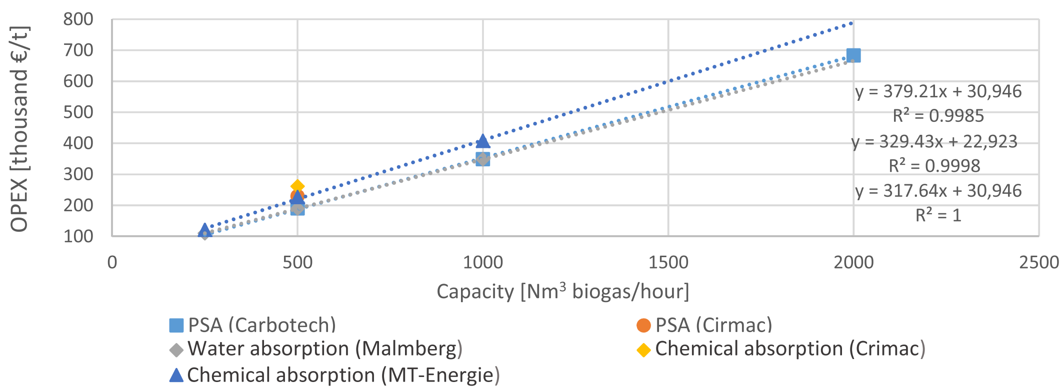

Fig. 2. Regression analysis of CAPEX functions for biogas purification technologies.

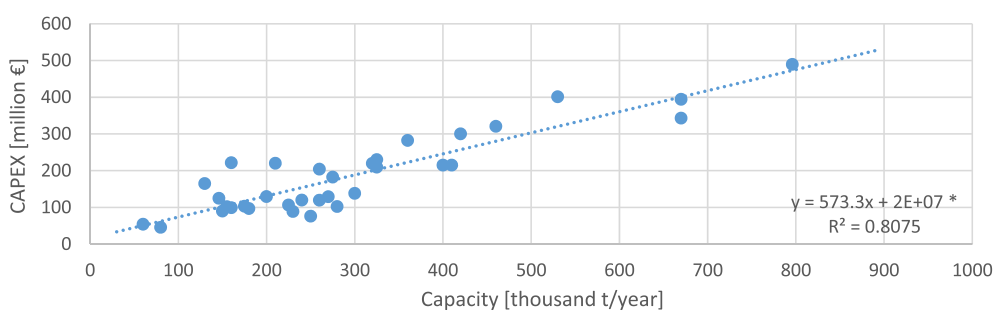

Fig. 3. Regression analysis of investment costs of cogeneration WtE plants. Based on data gathered from: Wood et al., 2013; A2A Group, 2015; Friotherm, 2015; Granatstein, 2000; UK Government, 2013a; UK Government, 2013b; ARUP, 2012; Green Investment Bank, 2013; Waste Management World, 2014; Kalogirou, 2010; Kalogirou, 2011; State of Green, 2011; Energia, 2015; Eew, 2013; Eew, 2014; Kalogirou, 2013; SACE, 2011; HZ INOVA, 2014; EPEM, 2014.

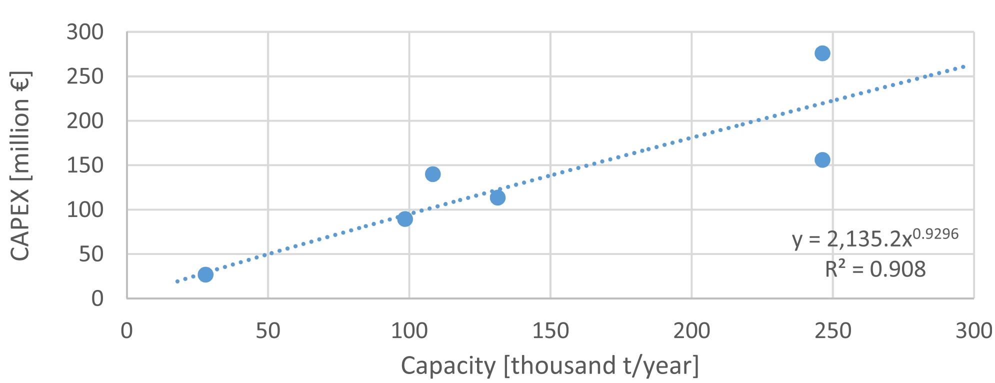

Fig. 4. CAPEX function for gasification of MSW technology with electricity production.

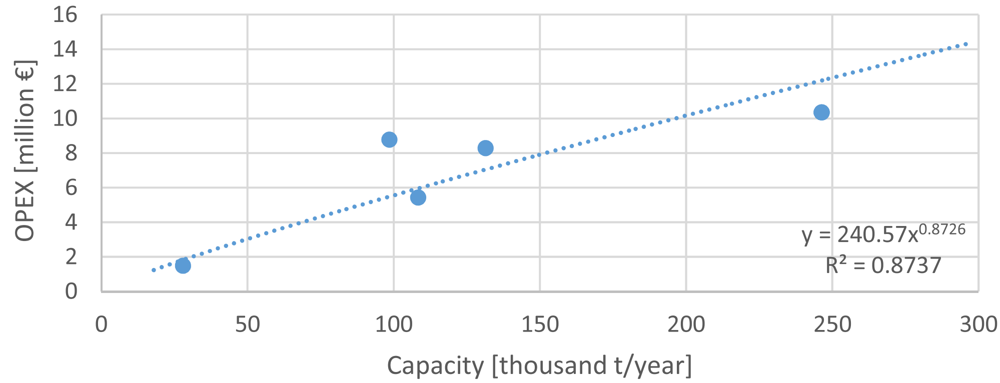

Fig. 5. OPEX function for gasification of MSW technology with electricity production.

Table 2
Economic functions for energy recovery and energy transformation plants.

Technology	CAPEX [€]	OPEX	Eq.
AD	$c = 26,194x^{0.6}$	$o = 12,723x^{-0.6}$ [€/t]	(12, 13)
WtE	$c = 573 - 3x + 2\cdot 10^7$	$o = 51.6x$ [€][a]	(14, 15)
Biogas cogeneration plant	$c = 415.97y^{0.6}$	$o = 90,544y^{-0.6}$ [€/m^2]	(16, 17)
Biogas upgrade plant:			
- Water absorption	$c = 710.04y + 969,331$	$o = 317.64y + 30,946$ [€]	(18, 19)
- Chemical adsorption	$c = 940.52y + 605,019$	$o = 379.21y + 30,946$ [€]	(20, 21)
- PSA	$c = 1,024.2y + 865,242$	$o = 329.43y + 22,923$ [€]	(22, 23)
CNG plant	$c = 15,243z + 243,028$	$o = -3.249z^2 + 167.6z + 19,442$ [€]	(24, 25)
Gasification plant	$c = 2,135.2x^{0.9296}$	$o = 240.57x^{0.8726}$ [€]	(26, 27)

c – investment cost, o – operation and maintenance cost, x – input waste [t/year], y – input biogas flow [m^3/h], z – output flow of CNG [m^3/h].
[a] For the plant in full operation.

Table 3
Market prices of secondary raw materials (January 2018) (Letsrecycle, 2018).

Material	Price [€/t]
Glass	10.10
Paper	39.28
Compost	26.97
Steel	131.86
Aluminium	1133.44
PE	32.54
PET	50.79

waste and waste fractions using the Mendeliev equation, while the biomethane production from the AD plant was calculated using literature data on characteristics of AD substrates (Al Seadi et al., 2008) and shares of the MSW biowaste fractions – Equation (29). Based on mixed waste composition, L_0 for the municipal landfill was also calculated (Equation (30)) and the time distribution of the production is taken into account using the LandGEM model.

$$L_{0,AD} = \left(\frac{P_{bp,k}}{S_{m,k}} \cdot \frac{x_k}{x_k + x_v}\right) + \left(\frac{P_{bp,v}}{S_{m,v}} \cdot \frac{x_v}{x_k + x_v}\right) [\mathrm{m}^3/\mathrm{t}] \quad (29)$$

$$L_{0,odl} = L_{0,AD} \cdot x_{bio} \quad [\mathrm{m}^3/\mathrm{t}] \quad (30)$$

$L_{0,AD}$ – the potential for biomethane generation through the AD of biowaste [m3/t].
$P_{bp,k}$ – biogas yield for kitchen waste [m^3/t].
$P_{bp,v}$ – biogas yield for garden waste [m^3/t].
$S_{m,k}$ – methane content in biogas, kitchen waste [%]
$S_{m,v}$ – methane content in biogas, garden waste [%]
x_k – the share of kitchen waste [%]
x_v – the share of garden waste [%]
$L_{0,odl}$ – the potential for generating biomethane through landfill gas [m^3/t].
x_{bio} – the biowaste share in the waste that is disposed of at the landfill (%)

2.5. Defining scenarios

To give an answer to the research question, a scenario-based analysis was carried out. The analysed scenarios consist of the same technologies for material recovery. Regarding waste separation technologies, scenarios differ in the presence of MBT facilities for secondary separation of mixed waste. Energy recovery technologies include MSW landfill with the landfill gas collection system, incineration of mixed MSW in the cogeneration WtE plant, thermal treatment of residual waste in cement kilns, thermal treatment of RDF from MBO plants in the cogeneration thermal treatment plant, AD of bio-waste and gasification of MSW. Furthermore, three variants have been analysed for possible subsequent transformation and use of biogas and landfill gas - cogeneration of heat and electricity in a biogas engine, upgrading (purification) of biogas to biomethane and injection of the produced biomethane to the gas network and the transformation of the produced biomethane into the CNG for the use in motor vehicles.

Scenario Existing System describes the current state of the local WMS – Fig. 6. Figs. 6–13 show analysed scenarios and their internal waste streams (with black lines) as well as products (energy and material) (dotted lines). On these Figures, only waste flows classified as municipal waste are shown. Biogas production is marked with grey arrow shape because the biogas is not a final energy vector and it needs to be further transformed before final consumption.

To make presentation clearer and more understandable, material recovery chain, which is the same in all scenarios, is in scenario diagrams shown as aggregated chain Recyclebels – Collection system – Sorting of recyclables, with its interactions with energy recovery chain. The overall material recovery chain is shown in Fig. 7, where the MBT plant and its interactions are shown in grey color as it is not a part of all scenarios.

Previous diagrams show that scenario Existing system is based on waste disposal on the regulated landfill and that material recovery goals are fulfilled only through the primary waste separation. Energy recovery is present only in the form of landfill gas collection, where biogas can be transformed into electrical and thermal energy (which can be used in

Table 4
Current and forecast data on the quantities of collected MSW.

	2020[b]		2025		2035	
	Quantity [t]	Share [%]	Quantity [t]	Share [%]	Quantity [t]	Share [%]
Glass waste	2305.05	0.91%	5500	1.27%	12,900.00	2.17%
Metal waste	220.01	0.09%	3400.00	0.78%	7600.00	1.28%
- Fe waste [52]	23.28	0.01%	359.80	0.08%	804.25	0.14%
- Al waste [52]	196.73	0.08%	3040.20	0.70%	6795.75	1.14%
Paper waste	10,071.11	3.99%	42,900.00	9.90%	68,800.00	11.57%
Plastic waste	3539.56	1.40%	26,500.00	6.12%	63,600.00	10.69%
- PE [a]	2743.84	1.09%	20,542.64	4.74%	49,302.33	8.29%
- PET [a]	795.72	0.32%	5957.36	1.37%	14,297.67	2.40%
Biowaste	7846.51	3.11%	104,963.78	24.22%	132,700.00	22.31%
- Green/Garden waste	1582.73	0.63%	95,031.50	21.93%	121,805.44	20.48%
- Food/Kitchen waste	6263.78	2.48%	9932.28	2.29%	10,920.09	1.84%
Mixed municipal waste	216,631.28	85.89%	115,200.00	26.58%	105,300.00	17.70%

[a] Calculation based on UPR data for plastic sorting technology.
[b] Based on Požgaj et al. (2019)..

Table 5
Current and forecasted data on the composition of mixed MSW.

	2020[a]		2025		2035	
	Share [%]	Quantity [t]	Share [%]	Quantity [t]	Share [%]	Quantity [t]
Paper waste	16.10	63,031.12	18.59%	19,927.20	26.96%	27,439.20
Plastics waste	18.10	61,403.01	34.45%	38,229.90	29.46%	35,596.80
Metal waste	2.60	2558.46	4.55%	1624.50	3.61%	1359.60
Glass waste	3.10	8373.14	7.15%	5523.30	5.98%	5067.60
Garden waste	6.47	11,861.95	1.67%	2490.90	1.33%	3090.00
Biowaste	33.63	61,635.60	8.78%	13,104.30	7.29%	16,933.20
Other waste	20.00	23,723.89	24.81%	27,508.20	25.37%	34,237.20

[a] Based on Domanovac et al., 2018

Table 6
Energy characteristics.

	2020	2025	2035
Mixed waste LHV [MJ/kg]	8.93	12.25	12.16
Biomethane potential (landfill) [m^3/t][b]	76.29	48.92	47.96
Biomethane potential (AD) [m^3/t][a]	92.92	113.38	113.82

[a] For biogenic fraction.
[b] For 30-year collection of landfill gas.

district heating systems), upgraded (to biomethane) and injected into the gas grid and/or subsequently compressed to the form of CNG for use in vehicles – Fig. 8.

Scenario Mass Burn is based on the local municipal WM plan (Mužinić et al., 2007) (Fig. 9) where mixed waste is used in the WtE plant for energy production, which also separates secondary metals from the produced ash and returns them to the material recovery chain. Burnable residual wastes, generated from technological processes in the analysed system, are also diverted to thermal treatment. Treatments of wastes that cannot be classified as municipal are internalized within each waste-producing technology – expenses associated with the external technologies for its processing (such as inert material dumps, hazardous waste treatment or underground disposal of certain waste categories).

Scenario Material Recovery avoids energy recovery and puts an emphasis on material recovery. It adds the MBT facility, so it does not rely solely on primary waste separation to increase the recycling rate – Fig. 10. Separated RDF fraction is thermally treated in an external plant (cement kiln) as a way of avoiding landfilling, which is in line with European legislation. Cement kiln is separated from the rest of the systems with a dotted line as it is a part of the RDF market. Residual waste of a significant LHV from other technologies in the MSW treatment chain can be also treated in the cement kiln. A separated biological fraction from mixed waste is bio-stabilised and appropriately landfilled, as it is too contaminated to be placed on the market.

Scenario Energy Recovery integrates all energy recovery technologies which are recognized through local legislation and illustrates the potential of energy recovery - Fig. 11.

Unlike the previous one, in this scenario, the MBT plant does not

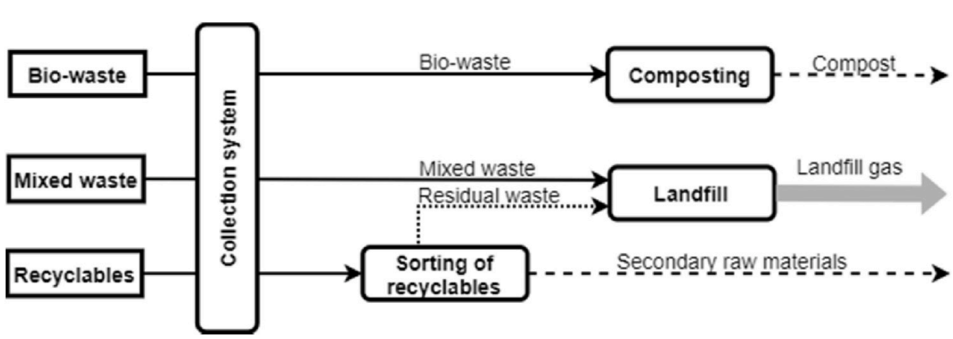

Fig. 6. Existing system.

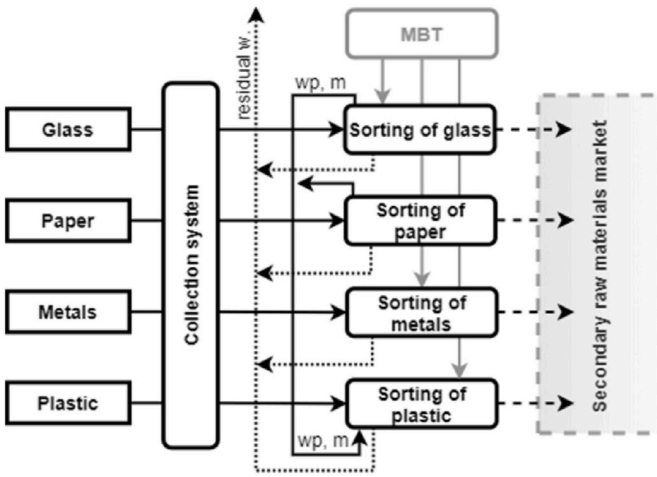

Fig. 7. Material recovery chain.

separate the plastic fraction which increases the amount and LHV of RDF and, thus, the systems energy recovery rate. Also, the separated biological fraction is energy recovered in the AD plant, and produced biogas is subsequently transformed as shown in Fig. 8.

Scenario Material Recovery with AD is based on the Material Recovery scenario and exchanges the composting facility with AD facility (Fig. 12) to solve the local problem of the small market for compost and removes the need for bio-stabilisation of secondary separated biowaste. It boosts energy recovery at the expense of material and represents a middle ground between material and energy recovery.

Scenario Gasification is showing possible integration of technology which is not common in the field of energy recovery of MSW. This scenario represents a relatively simple system that is similar to the Mass Burn scenario with the difference of separation of recyclable materials from waste streams before thermochemical treatment – Fig. 13. By doing this, material recovery is boosted by separating materials that cannot be gasified and everything else is used for syngas production which is subsequently used for electricity generation.

3. Results and discussion

Based on the described scenarios, the calculations were made

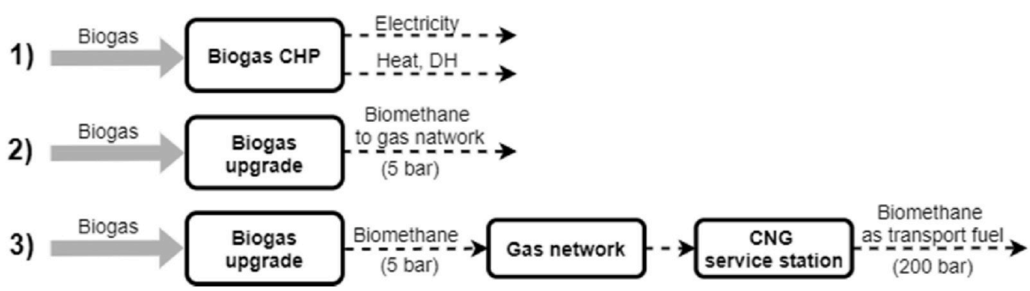

Fig. 8. Analysed biogas use variants.

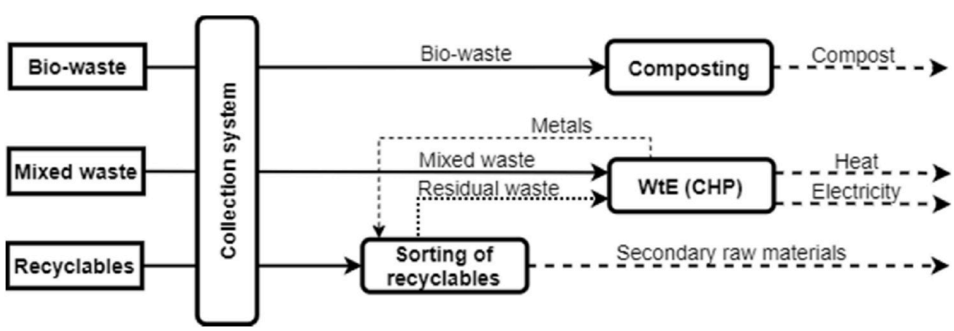

Fig. 9. Mass burn.

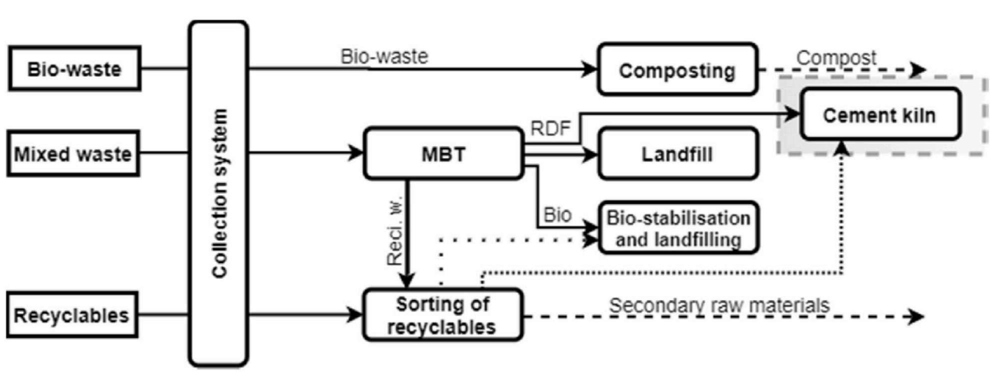

Fig. 10. Material recovery.

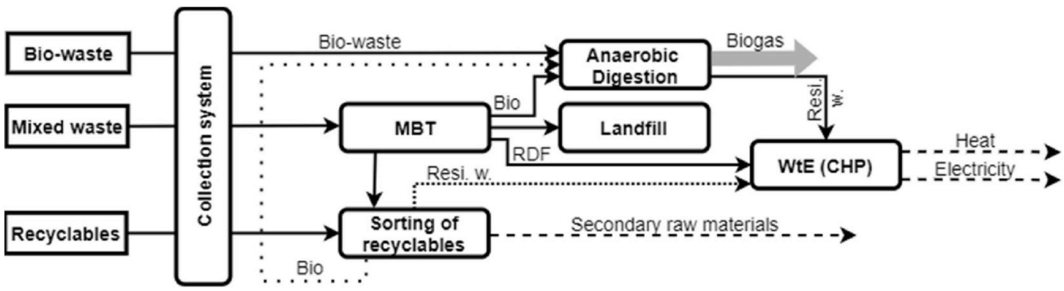

Fig. 11. Energy recovery.

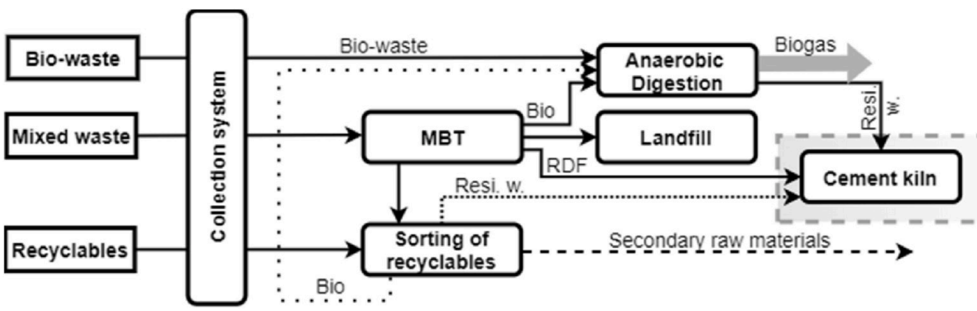

Fig. 12. Material recovery with AD

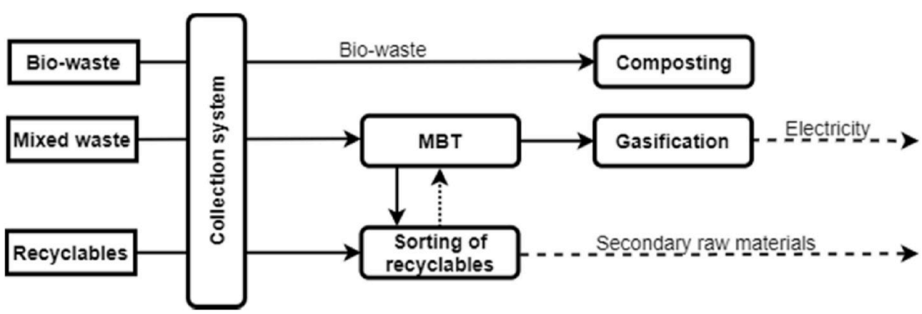

Fig. 13. Gasification.

according to the presented methodology and the results are shown and discussed in the following subchapters.

3.1. Input mass flows

Each considered technology input (waste) flow is calculated from the collection phase flows (Table 4) and the LCI UPR data for the technologies that precede the analysed technology – Tables 7 and 8. The same principle was used for calculating the secondary raw materials production, which is placed on the market – Table 9.

Input flow data for sorting technologies (Table 7) show that an increase in separate collection reduces the required capacity of the MBT facility. On the other hand, progress in primary separation increases the required capacity of plants for sorting separately collected waste, which is further increased by secondary separation of mixed waste. Exceptions are waste plastics (PET and HDPE) in the Energy Recovery and Gasification scenarios because they are classified as an energy recovery component. Also, the integration of MBT did not increase the amount of recycled paper where waste paper from mixed waste is routed to energy recovery due to its contamination/degradation. These effects are also visible at the thermal treatment facility in the Mass Burn scenario (Table 8) where primary waste separation reduces its input flows, which are further decreased with the integration of secondary waste separation. This reduction can be compensated through the co-incineration of AD sludge which must be properly disposed of and thermal treatment is one of the possible solutions. Due to the mixed waste separation, the amount of bio-waste is also continuously increasing through both time periods. Separated bio-waste may, in accordance with scenarios, be used to produce compost or biogas in the AD plant.

In the scenarios Existing System and Mass Burn, there is no difference in the quantity of secondary materials entering the secondary waste materials market (Table 9), since the same amount of waste is collected, and mixed waste is not secondary separated. The only difference is in the quantity of metals put on the market due to a metal separation in the WtE plant.

Regarding the scenario Material Recovery, the separated RDF is routed to the cement kiln, while the part which is not suitable for thermal treatment is disposed of. The share of disposed waste in all scenarios is less than 10%, thus all scenarios meet EU goals. In the scenarios Energy Recovery, Material Recovery and Gasification, there is an increase in material flows that are put on the market due to secondary waste separation.

Table 7
Input streams for sorting technologies.

Scenario/Input stream	Glass [t]	Metals [t]	Paper [t]	Plastics [t]	Mixed waste [t]
Existing System - 2020	2305	220	10,071	3642	0
Existing System - 2025	5500	3400	42,900	26,922	0
Existing System - 2035	12,900	7600	68,800	64,292	0
Mass Burn - 2020	2305	220	10,071	3642	0
Mass Burn -2025	5500	3400	42,900	26,922	0
Mass Burn - 2035	12,900	7600	68,800	64,292	0
Material Recovery - 2020	8349	5289	10,071	38,953	216,631
Material Recovery - 2025	12,913	8117	42,900	62,667	115,200
Material Recovery - 2035	18,567	11,021	68,800	92,232	105,300
Energy Recovery - 2020	8349	5289	10,071	3664	216,631
Energy Recovery - 2025	12,913	8117	42,900	26,949	115,200
Energy Recovery - 2035	18,567	11,021	68,800	64,312	105,300
Material Recovery with AD - 2020	8349	5289	10,071	38,953	216,631
Material Recovery with AD - 2025	12,913	8117	42,900	62,667	115,200
Material Recovery with AD - 2035	18,567	11,021	68,800	92,232	105,300
Gasification - 2020	8349	5289	10,071	3664	216,631
Gasification - 2025	12,913	8117	42,900	26,949	115,200
Gasification - 2035	18,567	11,021	68,800	64,312	105,300

3.2. Energy production

The production of energy vectors by energy recovery technologies are shown in Table 10, while the results of biogas transformations, as well as the total data, are shown in Appendix (Tables A1 – A3).

The biggest energy production through the landfill gas collection is in the Existing system, biogas in the Material Recovery with AD/Energy Recovery, and WtE plant in the Mass Burn scenario. In the scenarios Energy Recovery and Material Recovery with AD, energy production from biogas is the largest, and together with WtE plant production (in the scenario Energy Recovery) leads to the largest total energy production (Table A1 and Table 11). The biogas yield is higher only in the scenario Existing System with the actual amount of landfilled waste.

3.3. Socioeconomic analysis

In addition to the previously modelled system costs, the system income needs to be quantified in order to carry out the socioeconomic analysis. The previously described LCI based approach to mass and energy flows modelling was used to calculate the production of materials and energy vectors in the analysed scenarios. Revenues calculation from the sale of recovered materials were based on the market prices for the collected material (Table 3), revenue calculation from electricity was based on the average electricity market price in 2018 (CROPEX, 2019) and amounts to 60 €/MWh, while revenues from produced heat were calculated using the district heating distributor price list, amounting to 34 €/MWh (HEP, 2018). For the initial cost of thermal treatment of RDF in cement kiln, zero value is taken. Quantities of produced energy vectors from waste, for the variant of biogas conversion in the CHP plant with biogas engine, are shown in Table 11.

All calculations are made for a period that is covered by EU legislation. Capital investment is financed through a loan with a real interest rate of 5.5%, and investment costs are amortized due to tax reductions. In the economic calculation, the cash flow is kept on the value of positive zero on a yearly basis, and the additional system cost, which needs to be covered by system users, is tracked. The results in Fig. 14 show the dynamics of system price movements during the analysis period.

System cost (WM fee) is calculated as the total cost of the analysed system divided by the mass of mixed MSW produced by the system users. This method of charging the waste management fee defines an incentive for system users to primary separate MSW. It is noticeable that system costs can be negative, which is the result of comparative analysis that does not include the costs of the waste collection. This approach has been applied because the waste collection system is the same for all the considered scenarios, collects the same amounts of waste, and therefore generates the same cost across all scenarios. Since incorporating a waste collection system into a model would not result in a relative difference in overall results, it was not modelled.

By increasing the complexity of a local WMS, the cost of the overall system increases. When analysing the correlation between the recovery of waste materials and the cost of the system, it can be noticed that increasing energy recovery leads to the highest increase in system costs due to investment cost payback. This is most prominent in the case of thermochemical conversion plants (incineration and gasification plants) because they are the most investment intensive technologies. At the same time, integration of other energy recovery technologies such as AD of bio-waste reduces the cost of the system, as it can be seen by comparing the results of scenarios Material Recovery and Material

Table 8
Input waste streams for energy recovery and disposal technologies.

Scenario	Technology:	AD	Thermal treatment				Cem. kiln	Landfill
	Input [t]:	Bio	Mixed	AD sludge	Residual	RDF	RDF	Residual
Existing System - 2020		0	0	0	0	0	0	268,305
Existing System - 2025		0	0	0	0	0	0	163,227
Existing System - 2035		0	0	0	0	0	0	164,178
Mass Burn - 2020		0	218,264	0	0	0	0	0
Mass Burn –2025		0	126,410	0	0	0	0	0
Mass Burn - 2035		0	131,166	0	0	0	0	0
Material Recovery - 2020		0	0	0	0	0	70,384	21,663
Material Recovery - 2025		0	0	0	0	0	44,997	11,520
Material Recovery - 2035		0	0	0	0	0	49,593	10,530
Energy Recovery - 2020		86,040	0	114,269	1890	105,673	0	21,663
Energy Recovery - 2025		115,830	0	153,833	11,466	80,715	0	11,520
Energy Recovery - 2035		140,920	0	187,155	26,054	77,512	0	10,530
Material Recovery with AD - 2020		86,040	0	0	0	0	70,384	21,663
Material Recovery with AD - 2025		115,830	0	0	0	0	44,997	11,520
Material Recovery with AD - 2035		140,920	0	0	0	0	49,593	10,530
Gasification - 2020		0	0	0	107,563	0	0	21,663
Gasification - 2025		0	0	0	92,181	0	0	11,520
Gasification - 2035		0	0	0	103,567	0	0	10,530

Table 9
Output waste streams placed on secondary raw material market.

Scenario	Glass [t]	Steal [t]	Al [t]	Paper [t]	PE [t]	PET [t]	Compost [t]
Existing System - 2020	2134	204	34	9963	611	2105	4241
Existing System - 2025	5093	3072	386	42,422	4513	15,562	56,696
Existing System - 2035	11,944	6846	866	68,048	10,777	37,163	71,683
Mass Burn - 2020	2134	2736	334	9963	611	2105	4241
Mass Burn –2025	5093	5638	690	42,422	4513	15,562	56,696
Mass Burn - 2035	11,944	8965	1116	68,048	10,777	22,516	71,683
Material Recovery - 2020	7731	4737	599	9984	6530	22,516	4247
Material Recovery - 2025	11,957	7290	920	42,448	10,505	36,224	56,679
Material Recovery - 2035	17,192	9905	1255	68,068	15,461	53,313	71,656
Energy Recovery - 2020	7731	4737	599	9984	614	2118	0
Energy Recovery - 2025	11,957	7290	920	42,448	4517	15,577	0
Energy Recovery - 2035	17,192	9905	1255	68,068	10,781	37,175	0
Material Recovery with AD - 2020	7731	4737	599	9984	6530	22,516	0
Material Recovery with AD - 2025	11,957	7290	920	42,448	10,505	36,224	0
Material Recovery with AD - 2035	17,192	9905	1255	68,068	15,461	53,313	0
Gasification - 2020	7731	4737	599	9984	614	2118	4247
Gasification - 2025	11,957	7290	920	42,448	4517	15,577	56,679
Gasification - 2035	17,192	9905	1255	68,068	10,781	37,175	71,656

Table 10
Production of energy by technology - immediate production.

Technology	AD	WtE (incineration and gasification)		Landfill
Product	Biogas (10^6 Nm^3)	Electricity (GWh)	Heat (TJ)	Landfill gas (10^6 Nm^3)
Existing System - 2020	0.00	0.00	0.00	21.25
Existing System - 2025	0.00	0.00	0.00	8.62
Existing System - 2035	0.00	0.00	0.00	12.26
Mass Burn - 2020	0.00	89.87	1440.38	0.00
Mass Burn –2025	0.00	71.40	1144.35	0.00
Mass Burn - 2035	0.00	79.82	1417.98	0.00
Energy Recovery - 2020	14.78	81.65	1308.54	0.00
Energy Recovery - 2025	20.33	70.16	1094.86	0.00
Energy Recovery - 2035	24.81	74.35	1155.61	0.00
Material Recovery with AD - 2020	14.78	0.00	0.00	0.00
Material Recovery with AD - 2025	20.33	0.00	0.00	0.00
Material Recovery with AD - 2035	24.81	0.00	0.00	0.00
Gasification - 2020	0.00	124.81	0.00	0.00
Gasification - 2025	0.00	86.54	0.00	0.00
Gasification - 2035	0.00	106.44	0.00	0.00

Recovery with AD, which shows the best results. Also, from the presented results, it is evident that all scenarios except scenario Material Recovery based ones have a higher system cost compared to the Existing System. To carry out further economic analysis of these scenarios, system revenues and costs are separately shown in Figs. 15–19.

In these diagrams, the difference between the lines *Loan + O&M (Operation and Maintenance)* and *Income* represents the costs that are paid by system users. Because of reliance on primary waste separation to increase recycling rates, the income of the scenario Existing System is increasing with the increase of primary waste separation (increasing the income from secondary materials market), which at the same time increases the O&M costs. As the primary waste separation rate increases, the amount of landfilled waste is reduced, resulting in a reduction in the income from the landfill gas collection, which represents the only way of energy recovery in this scenario. As can be seen, this scenario, without waste collection, is operating near zero even without waste management fee, and after 2030 even has a positive economic result.

Most of the revenue in the scenario Mass Burn is realized through the WtE plant (Fig. 16), which results in a decrease in income with a reduction in the amount of incinerated waste due to an increase in primary separation. The income from material recovery is marginally better than in the scenario Existing System because of additional separation of metals from a produced ash, which boosts material recovery. As can be seen, this scenario generates significant losses that need to be paid by system users, but these losses significantly decrease with time.

The main economic problem of this scenario is a large initial investment that is concentrated in one facility (WtE plant) which could be a major problem for municipalities/cities. On the other hand, when the initial investment is paid off, the total system cost is reduced by an average of 60%, which drastically changes the economic situation.

The scenario Energy Recovery (Fig. 17) transfers most revenue to the side of energy sales while still meeting EU recycling targets through primary waste separation that is further increased through the secondary separation. This scenario further emphasizes the problem of a large initial investment whose annual payment amounts to more than 20 million euros. Investment in expensive facilities also leads to large O&M costs which are also around 20 million euros yearly but are (on average) covered by revenues from energy sales. However, high overall costs still require a high level of participation of citizens in system financing, which are highest from all analysed scenarios.

Income from materials sales represents the only revenue in the scenario Material Recovery (Fig. 18). In contrast to the previous two scenarios, due to the absence of energy recovery technologies, this scenario has the smallest investment cost that can easily be financed by municipalities/cities. The results of small investments are also relatively small O&M costs. The biggest problem with these systems is the placing of secondary materials on the market and the processing of residual waste (RDF) from the MBT plant. In these calculations, the price of the RDF treatment from the MBT plant is assumed to 0 €/t, which can be in the case of high-quality RDF with high LHV and other favourable characteristics (low content of Hg, Cl, moisture, ash, etc.). As it can be seen, income from materials (which is the same as overall income) rapidly grows over the years and in 2024 outgrows expenses resulting in overall positive economic result i.e, the negative cost for system users.

Since the income from the compost is lower than the one from the energy sales produced by the AD, the composting plant is replaced with the AD plant with the CHP biogas engine. This further increases the revenues, but at the same time investment (loan payback) and O&M costs are also increased due to the introduction of more complex technology (Fig. 19). Even though at first glance, final economic results are similar to the previous scenario, this change leads to overall better economic results.

When the integration of gasification in the WMS is looked upon, it

Table 11
Energy form waste – variant: Use of biogas in a cogeneration plant.

Year	Existing System		Mass Burn		Energy Recovery		Material Recovery		Material Recovery with AD		Gasification	
	El. [GWh]	Heat [TJ]	El. [GWh]	Heat [TJ]	El. [GWh]	Heat [TJ]	El. [GWh]	Heat [TJ]	El. [GWh]	Heat [TJ]	El. [GWh]	Heat [TJ]
2020	42.54	263.22	89.87	1440.38	117.29	1569.26	0	0	30.06	185.99	124.81	0
2021	37.42	231.56	86.18	1381.17	117.33	1539.56	0	0	32.39	200.44	117.16	0
2022	32.31	199.90	82.49	1321.97	117.37	1509.85	0	0	34.73	214.89	109.50	0
2023	27.19	168.24	78.79	1262.76	117.41	1480.15	0	0	37.06	229.33	101.85	0
2024	22.07	136.58	75.10	1203.56	117.45	1450.44	0	0	39.40	243.78	94.20	0
2025	16.96	104.92	71.40	1144.35	117.49	1420.74	0	0	41.73	258.22	86.54	0
2026	17.66	109.26	72.25	1171.72	118.90	1433.13	0	0	42.66	263.95	88.53	0
2027	18.36	113.59	73.09	1199.08	120.31	1445.53	0	0	43.58	269.68	90.52	0
2028	19.06	117.93	73.93	1226.44	121.73	1457.92	0	0	44.51	275.41	92.51	0
2029	19.76	122.26	74.77	1253.80	123.14	1470.31	0	0	45.44	281.13	94.50	0
2030	20.46	126.60	75.61	1281.17	124.55	1482.70	0	0	46.36	286.86	96.49	0
2031	21.16	130.93	76.45	1308.53	125.96	1495.10	0	0	47.29	292.59	98.48	0
2032	21.86	135.27	77.29	1335.89	127.37	1507.49	0	0	48.21	298.32	100.47	0
2033	22.56	139.60	78.13	1363.25	128.78	1519.88	0	0	49.14	304.04	102.46	0
2034	23.26	143.94	78.98	1390.61	130.20	1532.27	0	0	50.06	309.77	104.45	0
2035	23.96	148.27	79.82	1417.98	131.61	1544.67	0	0	50.99	315.50	106.44	0

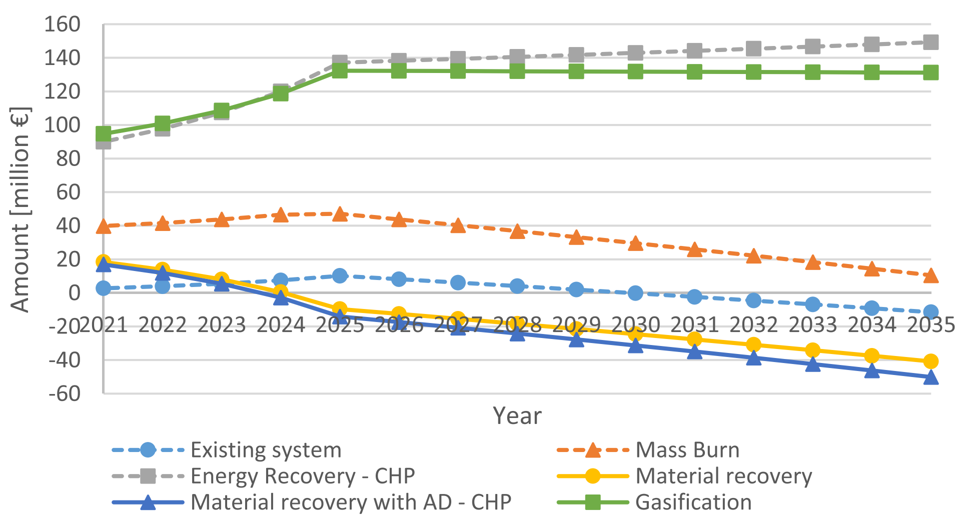

Fig. 14. System costs.

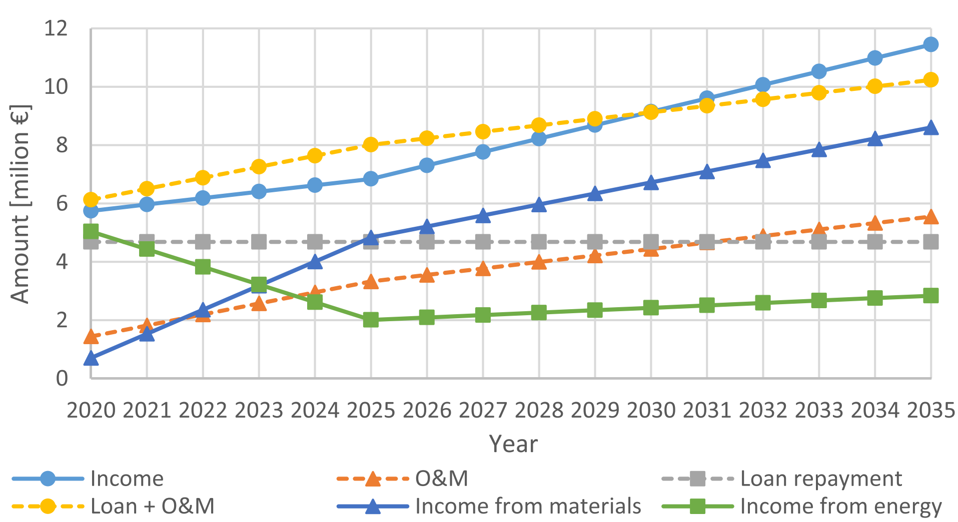

Fig. 15. Analysis of income and cost relationship – Existing System.

shows lower overall expenses then the Energy Recovery scenario, and larger then Material Recovery based scenarios (Fig. 20). On the other hand, its income is relatively close to Material Recovery with AD scenario. This leads to an overall economic result that is on the same level (slightly better) than in the case of the Energy Recovery scenario. This scenario has an overall lower income from energy sales when

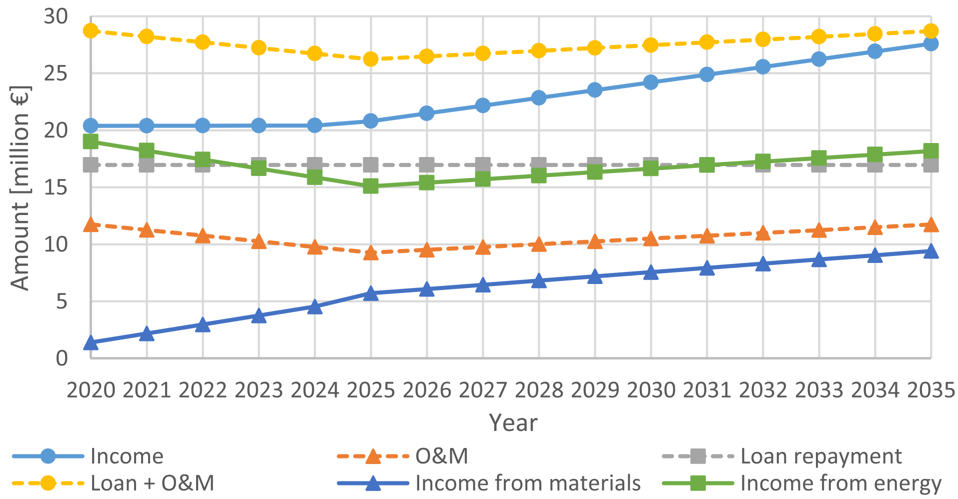

Fig. 16. Analysis of income and cost relationship – Mass Burn.

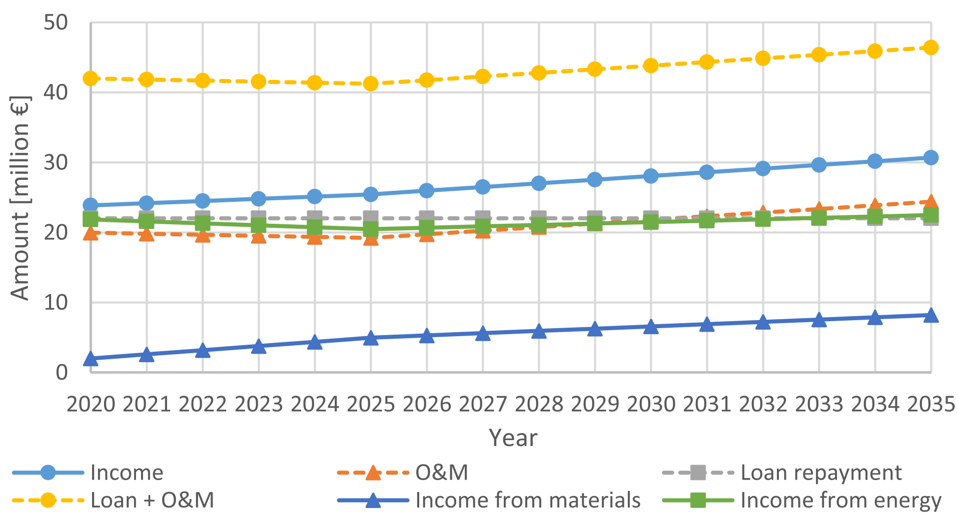

Fig. 17. Analysis of income and cost relationship - Energy Recovery.

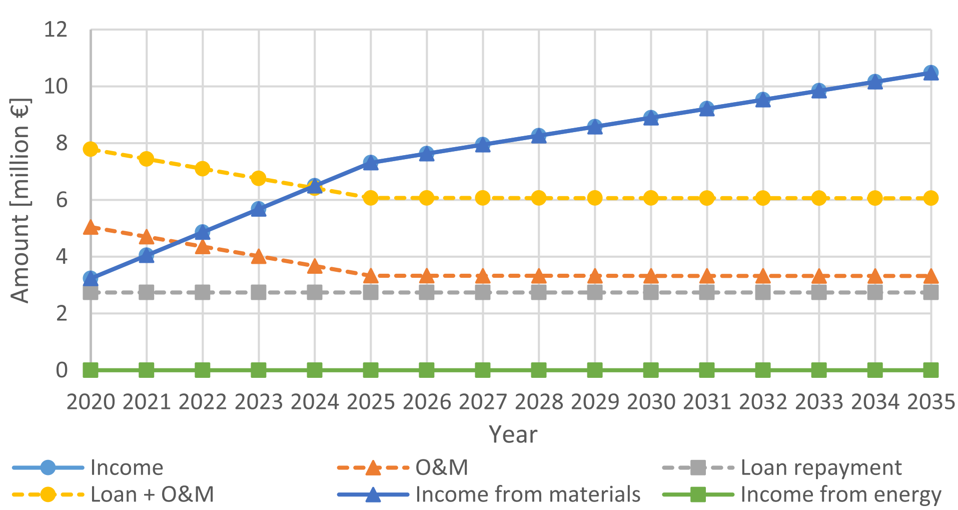

Fig. 18. Analysis of income and cost relationship - Material Recovery.

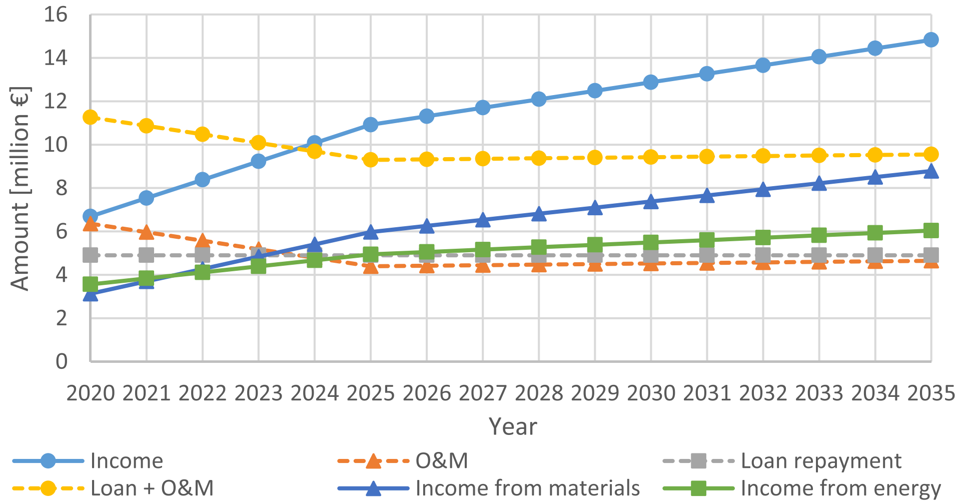

Fig. 19. Analysis of income and cost relationship - Material Recovery with AD.

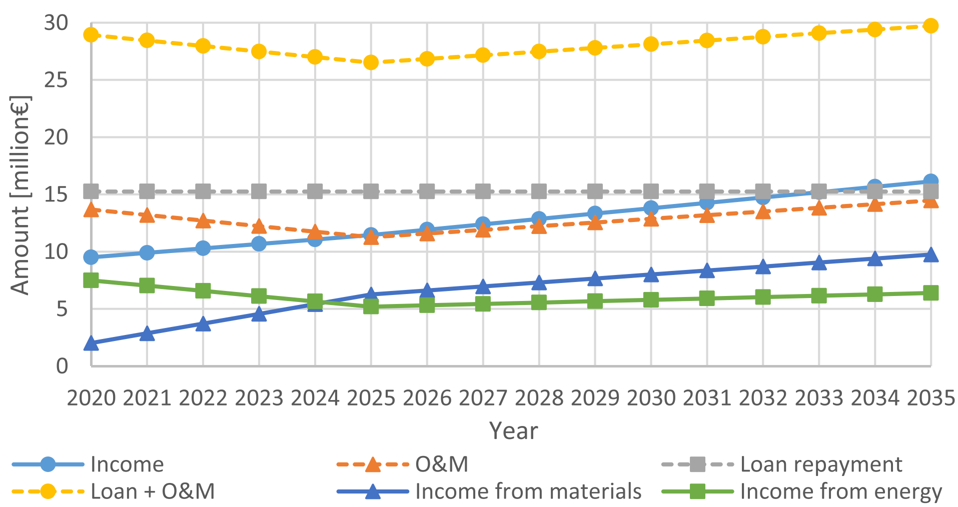

Fig. 20. Analysis of income and cost relationship - Gasification.

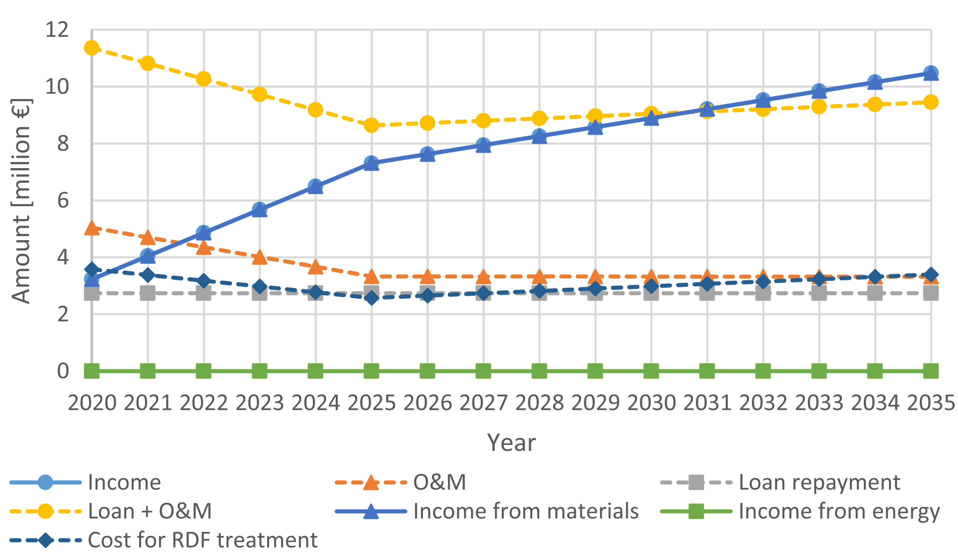

Fig. 21. Analysis of income and cost relationship - Material Recovery – included cost of thermal treatment of RDF.

compared to Energy Recovery scenario due to monogeneration and absence of AD of biowaste. At the same time, it does energy recover biowaste fraction from MBT resulting in not needing bio-stabilisation and landfilling of a fraction of biowaste which does not generate any income.

3.4. Influence of RDF treatment in cement kiln

The problem of burnable fraction (RDF) treatment from the MBT in a cement kiln is separately analysed. Cost of thermal treatment of RDF in cement kilns is on the level of 40–50 €/t in the peak of demand for such treatment (Government of India, 2018; RPS, 2014), while the price usually ranges from −20 to +20 €/t (Schäfer and Moser, 2012). Negative price makes sense because the replacement of coal in cement plants with RDF enables savings of up to 50 €/t for cement kiln which supply such a service (EcoMondis, 2018). Based on these data, the amount of 40 €/t is modelled which represents the situation when it is necessary to pay for the disposal of such waste. Based on this analysis, the maximum acceptable value of the fee will be estimated where such systems are still economically viable compared to other scenarios. According to this, scenarios Material Recovery and Material Recovery with AD are adjusted to encompass this cost as well – Figs. 21–23.

By introducing the cost for thermal treatment of RDF in a cement kiln, the overall cost of the system is increased in considered scenarios. Influence on waste management cost for the system users is shown in Fig. 23.

RDF treatment costs significantly increase the waste management fee. By comparing the results of analysed scenarios it can be seen that the overall system cost of these scenarios are now greater than the one of the Existing System, but the difference is reducing with the time, once again achieving negative difference after the years of 2028/2031. Scenarios Material Recovery and Material Recovery with AD show worse economic results than the scenario Mass Burn only in case of an increase of the above-mentioned fee of over 81.3 €/t, i.e. over 89.3 €/t.

3.5. Comparison of results and discussion

From previously shown variable specific system cost results (i.e. variable waste management fee), average system costs are calculated. The comparison of the cost of the analysed systems is shown in Table 12.

When cost volatility is observed, it is the smallest in the scenario Existing System, while all other scenarios have higher volatility with the largest one recorded in the case of the Energy Recovery scenario. Cost volatility is increasing with system complexity, changes with waste quantity and is one of the parameters to look upon when planning WMS. Another side of volatility is its character. Systems with increasing costs are Energy Recovery and Gasification, while all other systems show decreasing system costs. The reason for increasing cost over time is due to investment in energy recovery facilities to cover peak demand, while in the majority of the time they prove to be oversized and thus underutilised to cover its cost with energy production. Other technologies do not have these problems because they are all far less investment intensive and their production and increased secondary raw materials production cover cost introduced by their construction.

While one type of WMS cannot meet the needs of every municipality due to their local characteristics, they can be adapted to meet a larger range of needs. One of the ways to do this is subsequent energy transformations of biogas to other types of energy vectors. In the case of upgrading of produced biogas to biomethane and its injection in the gas grid, average cost of the system increase to 19.8/182.51/65.78 €/t for the Existing System/Energy Recovery/Material Recovery with AD scenarios, while in the case of its use in a form of CNG for use in transport average system cost amounts to 14.48/139.10/26.53 €/t. As can be seen, these transformations increase system cost (wase management fee) for system users but can be justified if there is a local need for other types of energy vectors. ThsBased on the presented results, it can be seen that the energy recovery of waste generates higher income through the energy sale than it can be achieved through the sale of secondary waste materials. Thus, will the system which puts emphasis on energy recovery be more economically sustainable is only dependent on the costs it binds to itself. It is apparent that the integration of thermochemical conversion based WtE plant results in large annual costs that need to be covered by through WM fee, while AD has fewer costs resulting in a reduction in the cost for the system users. On the other hand, material recovery based scenarios are based on internal waste separation technologies that have low investment cost, while all other technologies are outsourced and connected to the analysed systems via corresponding markets. Because of this, these systems have low investment costs which can be easily covered through material and energy sales, especially in the future years because of an increase in primary recovery efficiency and thus income from sales of secondary raw materials.

3.6. Sensitivity analysis

To make the results of this analysis applicable for a wider range of scenarios, sensitivity analysis is conducted where the influence of

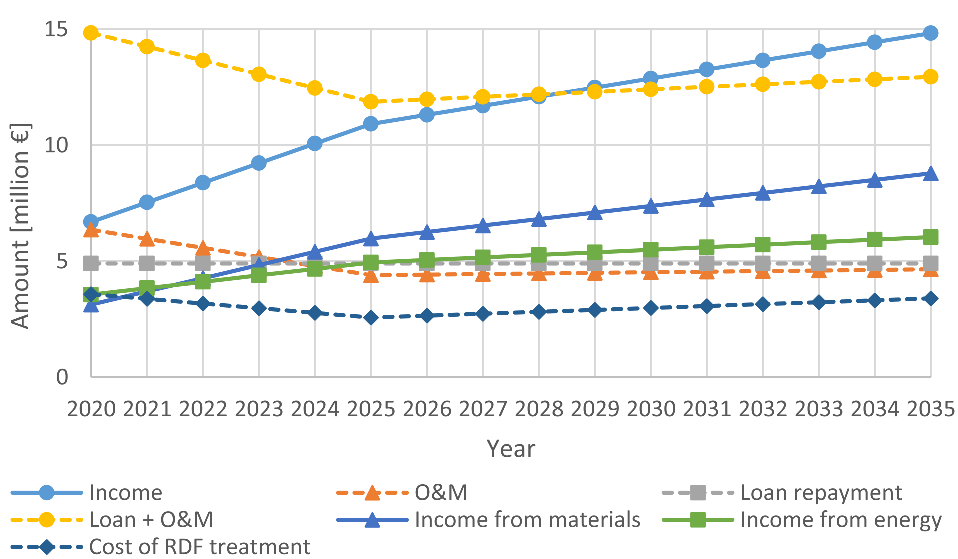

Fig. 22. Analysis of income and cost relationship - Material Recovery with AD – included cost of thermal treatment of RDF.

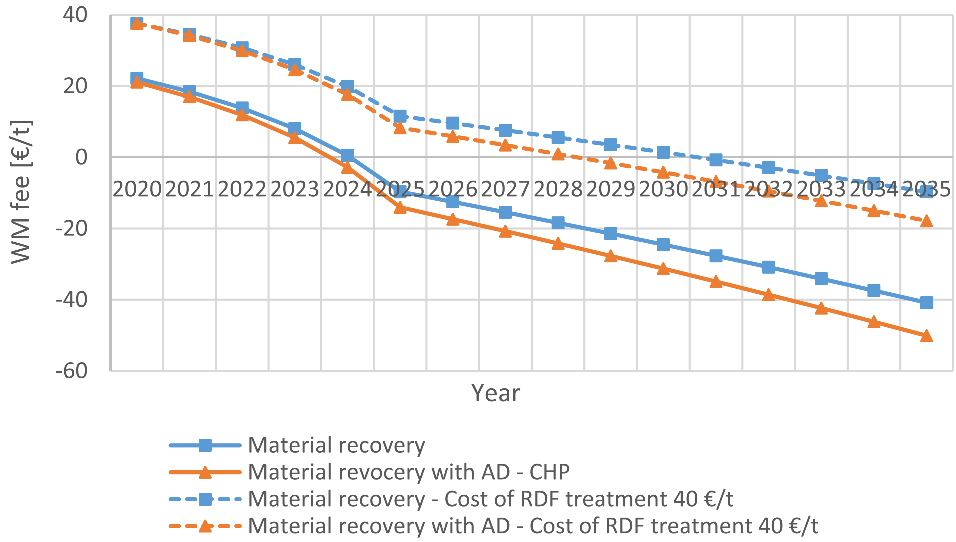

Fig. 23. System Costs with and without Cost of RDF treatment in Cement kiln.

Table 12
Comparison of costs of considered systems.

Year	Existing System - CHP		Mass Burn		Energy Recovery - CHP		Material Recovery[a]		Material Recovery with AD - CHP[a]		Gasification	
	Waste management fee [€/t]											
	Average	Volatile	Average	Volatile	Average	Volatile	Average	Volatile	Average	Volatile	Average	Volatile
2020	1.07	1.75	33.28	38.42	129.50	83.69	10.10	37.53	5.93	37.61	122.62	89.72
2021		2.74		39.83		89.96		34.48		34.16		94.72
2022		3.95		41.57		97.67		30.72		29.92		100.87
2023		5.48		43.76		107.39		25.99		24.58		108.62
2024		7.46		46.60		120.03		19.84		17.64		118.68
2025		10.14		47.13		137.11		11.52		8.25		132.30
2026		8.15		43.75		138.23		9.56		5.84		132.20
2027		6.12		40.31		139.37		7.57		3.40		132.09
2028		4.05		36.81		140.53		5.54		0.90		131.98
2029		1.95		33.25		141.71		3.48		−1.63		131.87
2030		−0.19		29.62		142.92		1.38		−4.22		131.76
2031		−2.37		25.93		144.14		−0.76		−6.85		131.65
2032		−4.58		22.17		145.39		−2.94		−9.52		131.53
2033		−6.84		18.35		146.66		−5.16		−12.25		131.41
2034		−9.14		14.45		147.96		−7.41		−15.03		131.29
2035		−11.49		10.48		149.27		−9.71		−17.86		131.17

[a] With a cost of thermal treatment of RDF in a cement kiln of 40 €/t.

changes in a market influenced parameters is assessed. Fig. 24 shows the influence of changes in energy and material prices on a specific cost of analysed systems.

As can be seen, next to showing the best economic results between all alternative scenarios, Material Recovery based scenarios also show the smallest sensitivity to market price changes (next to Existing System), where changes in market prices do not significantly change the results. At the same time, Mass Burn and Energy Recovery scenarios show the largest sensitivity to market changes. While Material Recovery based scenarios and Existing System scenarios can compete one with other, other scenarios cant become competitive from the socio-economic standpoint even with large changes in market prices.

4. Conclusion

Within the established legislative frameworks, the EU member states and decision-makers at local levels have to decide in which direction to continue the development of a municipal WMS. This is, even more, a problem for decision-makers in the younger EU member states which lag behind the older member states and therefore can expect large changes in the composition and amount of collected wastes in a short time.

Along with the goals set from the EU level, an important role in decision making on a local level plays the issue of the economic sustainability of the system, which is the most important aspect for local and regional governments. This issue is further emphasized since the issue of social acceptability is connected to it. The social acceptability is very important for the local decision-makers, which are elected by the citizens which at the same time finance the considered system through the WM fees.

Using LCA based system model in economic assessment can save time by using already calculated data in conjunction with economic functions. This way connection between LCA based system modelling, which is mostly used for conduction environmental sustainability analysis, and techno-economic analysis is established. As LCA approach is already recognized tool for environmental impact assessment on the EU level and is used for justification of deviations from the WM hierarchy, using the same approach for conducting an economic analysis can be valuable tool for decision-makers as time frame of the analysis for the economic part can be changed to encompass only economic payback/lifetime while the same model and system borders are used for both assessments.

From the results, it can be concluded that there are significant time-dependent cost and revenue changes for the analysed systems in the

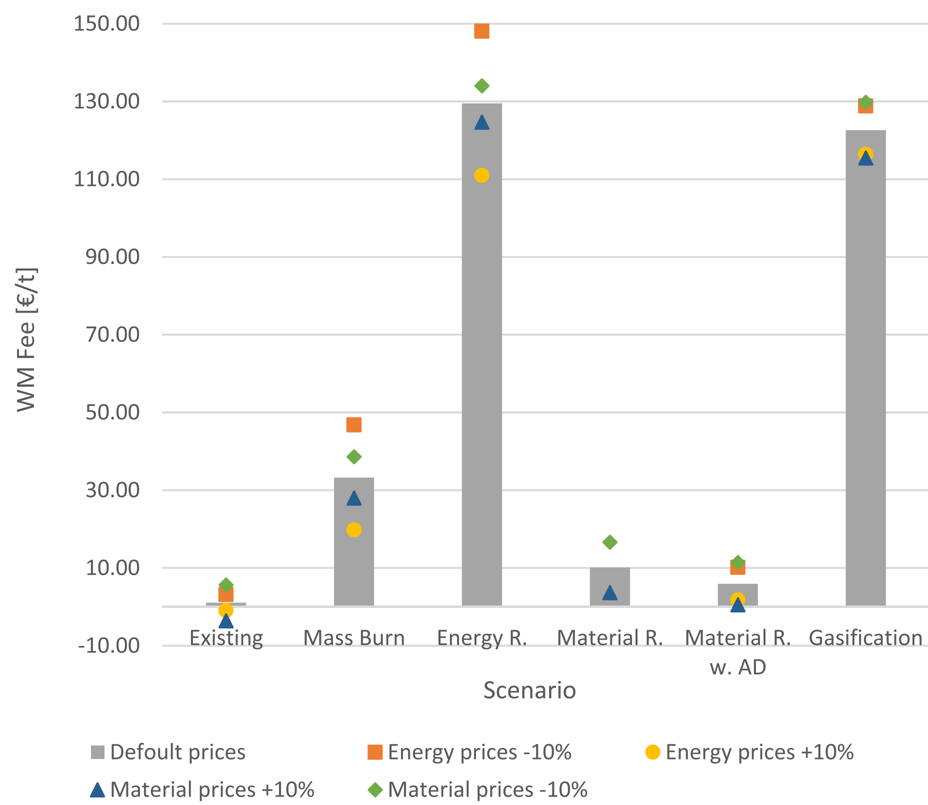

Fig. 24. Sensitivity analysis.

considered time frame. From a socio-economic point of view, any change in a complex system, such as a WMS, leads to a cost increase that needs to be paid by system users (i.e. citizens) through the MSW management fee. This is a situation during the investment repayment period after which economic situation can change significantly in favour of newly established systems, especially ones with higher investment costs.

Because of increased complexity, none of the scenarios has a lower system cost than the Existing System, i.e. lower system cost can be achieved only through the scenario Material Recovery with AD and only in case if the cost of thermal treatment of RDF in cement kiln is below 25 €/t, or average material price on secondary raw material market increase over 23%. While under the given conditions it seems that the existing system would be the best from the socio-economic point of view, but this system simply does not meet the goals from the implementation rate standpoint. To achieve a high degree of primary waste separation, local governments need to invest a lot of time and money in campaigns to raise awareness with system users, and without that, it is not possible to meet the legislative goals in a reasonable timeframe. In this regard, the secondary separation of waste plays an important role, especially in countries that do not neet EU legislation implementation timeframe.

Regarding energy recovery of waste, the introduction of the WtE technology based on thermochemical processes into the system automatically leads to a large increase in the cost of the system due to the high investment cost, but this kind of technology at the same time produces the largest amount of energy and generates the biggest revenue. This is the situation in the period of repayment of investment costs, after which this kind of system shows the best economic results. On the other hand, the energy from the AD plant reduces the cost of the system in all scenarios where it has been introduced, while the integration of material recovery of waste has low investment cost mainly due to outsourcing material recycling plants and because of that reduces the overall cost for the system users.

While some systems show high-cost volatility, its character can be ascending or descending. Both types of systems help municipalities to meet EU legislation goals, but systems with descending cost show long term economic sustainability as its system cost is lowering as primary and secondary waste separation are increased i.e. overall society becomes more ecologically aware. While WMS system planning is based around EU legislation goals, and need to help municipalities to meet them, one of the equally important of WMS is its long term economic sustainability.

Only systems that do not show this trend are systems that are generating less income as waste separation increases due to reduced quantity of mixed waste for thermochemical treatment. From this, it can be concluded that already built WtE plants need to be fully utilized before thinking about building new ones. Also, outsourcing of thermal treatment of residual (burnable) waste is an economically valid solution which is also beneficial from the side of the sustainability of the WMS and, from the ecological standpoint, decreases dependency on fossil fuels import. It can be concluded that the best economic results can be achieved with a combination of material recovery and energy recovery of waste which includes maximisation of material recovery with AD of biowaste and outsourcing of final disposal of the burnable waste fraction to cement kilns or pre-built WtE plants.

Through this, it is evident that legislative and socio-economic changes lead to unavoidable changes in the structure of municipal WMS, which results in an increase in the cost of the waste management service for the citizens. This socio-economic impact can be alleviated through smart WMS planning and parallel integration of the energy and material recovery technologies which leads to synergy and decreases WMS costs and thus increases socioeconomic acceptability of such changes.

Even though the EU legislative puts emphasis on material recovery, it can be concluded that the energy recovery of waste should not be neglected. Prebuilt incineration plants and cement kilns should be used to their full potential, and in the future, gasification should be considered when planning new final step waste disposal plants. At the same time, AD plants enable recovery of separated biological waste fraction from mixed waste which in other cases would be bio-stabilised and landfilled. This way, AD plays a role in securing the economic and social sustainability of the implementation of changes in WMS and, at the same time, reduce landfilling which is the last option in waste treatment hierarchy. Thus, integration of material and energy recovery technologies lead to a synergy that can contribute to wider European persistence towards the transformation of the existing economy into a low-carbon, competitive and resource-efficient economy as a part of circular economy and "closing the loop" concept.

Taking into account all presented results, it can be also concluded that in this legislative defined transition period to a more sustainable economy, from the WM point of view, it is better to plan systems with lower investment costs that more rely on markets which makes it less sensitive to changes in waste quantity and quality. In these circumstances relying on markets for material, energy and services in WM is a more secure option than investments in investment intensive WM plants.

In future work, the economic sustainability of each of analysed technologies will be analysed in relation to changing quantities of treated waste which could help in optimisation of a WMS. Also, the scope of analysed technologies will be expanded to include a wider range of thermochemical conversion technologies and multicriteria analysis will be conducted to establish a connection between environmental, energy and socio-economic sustainability.

CRediT authorship contribution statement

Tihomir Tomić: Conceptualization, Methodology, Software, Validation, Formal analysis, Investigation, Resources, Data curation, Visualization, Writing - original draft, Project administration. **Daniel Rolph Schneider:** Writing - review & editing, Supervision, Funding acquisition.

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APPENDIX

Table A1
Production of energy by energy product - variant: Use of biogas in cogeneration

Input	Product	Existing System - 2002	Existing System - 2025	Existing System - 2035	Energy Recovery - 2020	Energy Recovery - 2025	Energy Recovery - 2035	Material R. with AD - 2020	Material R. with AD - 2025	Material R. with AD - 2035
Landfill gas	Electricity (GWh)	42.54	16.96	23.96	0	0	0	0	0	0
	Heat (TJ)	263.22	104.92	.148,27	0	0	0	0	0	0
Biogas (AD)	Electricity (GWh)	0	0	0	35.64	47.33	57.26	35.64	47.33	57.26
	Heat (TJ)	0	0	0	260.72	325.88	389.06	260.72	325.88	389.06
Overall										

production*Electricity (GWh)
42.5416.9623.96117.29117.49131.6135.6447.3357.26Heat (TJ)
263.22104.92.148,271569.261420.741544.67260.72325.88389.06
* Including previous phases and energy production from other technologies.

Table A2
Production of energy by energy product - variant: Biomethane to grid

Input	Product	Existing System - 2002	Existing System - 2025	Existing System - 2035	Energy Recovery - 2020	Energy Recovery - 2025	Energy Recovery - 2035	Material R. with AD - 2020	Material R. with AD - 2025	Material R. with AD - 2035
Landfill gas	Methane (5 bar) (10^6 m3)	14.16	5.74	8.17	0	0	0	0	0	0
Biogas (AD)	Methane (5 bar) (10^6 m3)	0	0	0	9.86	13.56	16.54	9.86	13.56	16.54
Overall										

production*Electricity (GWh)00081.6570.1674.35000Heat (TJ)
0001308.541094.861155.61000Methane (5 bar)
(10^6 m3)14.165.748.179.8613.5616.549.8613.5616.54
* Including previous phases and energy production from other technologies.

Table A3
Production of energy by energy product - variant: Biomethane to CNG

Input	Product	Existing System - 2002	Existing System - 2025	Existing System - 2035	Energy Recovery - 2020	Energy Recovery - 2025	Energy Recovery - 2035	Material R. with AD - 2020	Material R. with AD - 2025	Material R. with AD - 2035
Methane (5 bar) from landfill gas	Methane (200 bar) (t)	425.51	172.66	245.56	0	0	0	0	0	0
Methane (5 bar) from AD	Methane (200 bar) (t)	0	0	0	295.89	407.22	496.91	295.89	407.22	496.91
Overall production[a]	Electricity (GWh)	0	0	0	81.65	70.16	74.35	0	0	0
	Heat (TJ)	0	0	0	1308.54	1094.86	1155.61	0	0	0
	Methane (200 bar) (t)	425.51	172.66	245.56	295.89	407.22	496.91	295.89	407.22	496.91

[a] Including previous phases and energy production from other technologies.

References

A2A Group, 2015. Brescia plant data. www.a2a.eu/gruppo/export/sites/default/a2a/en/plants_networks/waste_to_energy/documenti/folder_brescia.pdf.

Al Seadi, T., Rutz, D., Prassl, H., 2008. Biogas manual. Biogas for eastern Europe project. https://www.big-east.eu/croatia/handbook/Prirucnik_za_bioplin_w%5B1%5D.pdf.

Aleluia, J., Ferrão, P., 2017. Assessing the costs of municipal solid waste treatment technologies in developing Asian countries. Waste Manag. 69, 592–608. https://doi.org/10.1016/j.wasman.2017.08.047.

Armstrong, J.S., 2001. Principles of Forecasting: a Handbook for Researchers and Practitioners. Kluwer Academic Publishers, Boston.

ARUP, 2012. Bradford and calderdale residual waste management project. www.arup.com/Projects/Bradford_and_Calderdale_Residual_Waste_Management_Project.aspx.

Bel, G., Fageda, X., 2010. Empirical analysis of solid management waste costs: some evidence from Galicia, Spain. Resour. Conserv. Recycl. 54, 187–193. https://doi.org/10.1016/j.resconrec.2009.07.015.

Bertanza, G., Ziliani, E., Menoni, L., 2018. Techno-economic performance indicators of municipal solid waste collection strategies. Waste Manag. 74, 86–97. https://doi.org/10.1016/j.wasman.2018.01.009.

Boer, E.D., Boer, J.D., Jager, J., 2005. Waste Management Planning and Optimisation (LCA IWM). Obidem-Verlag, Stuttgart.

Chifari, R., Lo Piano, S., Matsumoto, S., Tasaki, T., 2017. Does recyclable separation reduce the cost of municipal waste management in Japan? Waste Manag. 60, 32–41. https://doi.org/10.1016/j.wasman.2017.01.015.

Christensen, T.H., Simion, F., Tonini, D., Møller, J., 2009. Global warming factors modelled for 40 generic municipal waste management scenarios. Waste Manag. Res. 27, 871–884. https://doi.org/10.1177/0734242X09350333.

Commission of the European Communities, 2003. COM(2003) 302 final – integrated product policy - building on environmental life-cycle thinking. http://ec.europa.eu/transparency/regdoc/rep/1/2003/EN/1-2003-302-EN-F1-1.Pdf.

Council of the European Union, 2009. 17271/1/08 –brussels European Council 11 and 12 december 2008 – presidency conclusions. www.consilium.europa.eu/uedocs/cms_Data/docs/pressdata/en/ec/104692.pdf.

Council of the European Union, 2014. EUCO 169/14 – European Council (23 and 24 october 2014) – conclusions. http://data.consilium.europa.eu/doc/document/ST-169-2014-INIT/en/pdf.

Croasdell, S., Watson, S., How, E., 2015. Review of 2012 metal packaging statistics. http://metalpackagingeurope.org/wp.

CROPEX, 2019. Croatian power exchange. https://www.cropex.hr/.

Domanovac, T., Fundurulja, D., Mrkoci, S., Franolić, V., Jurkić, I., Vrbanek, A.M., Novak Mujanović, S., Mužinić, M., Pašalić, M., 2018. Analysis of Waste Composition and Morphological Characteristics Based on Field Tests for 2018. IPZ Uniprojekt TERRA.

Ducharme, C., 2010. Technical and Economic Analysis of Plasma-Assisted Waste-To-Energy Processes. Columbia University, pp. 1–75.

EC, 1999. Directive 1999/31/EC of 26 April 1999 on the landfill of waste, 1999. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=celex%3A31999L0031.

EC, 2006. Integrated pollution prevention and control reference document on the best available techniques for waste incineration. http://eippcb.jrc.ec.europa.eu/reference/BREF/wi_bref_0806.pdf.

EC, 2008. SEC(2008) 2741 – communication from the Commission to the European Parliament and the Council - the raw materials initiative: meeting our critical needs for growth and jobs in Europe. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:52008DC0699.

EC, 2010. COM/2010/2020 final – EUROPE 2020 A strategy for smart, sustainable and inclusive growth. https://eur-lex.europa.eu/legal-content/en/ALL/?uri=CELEX%3A52010DC2020.

EC, 2011. COM/2011/571 – Roadmap to a resource efficient Europe. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:52011DC0571.

EC, 2011. COM/2011/0112 final – a Roadmap for moving to a competitive low carbon economy in 2050. https://eur-lex.europa.eu/legal-content/EN/ALL/?uri=CELEX%3A52011DC0112.

EC, 2013. The European innovation partnership (EIP) on raw materials. https://ec.europa.eu/growth/tools-databases/eip-raw-materials/en/content/european-innovation-partnership-eip-raw-materials.

EC, 2015. Capital Factsheet on Separate Collection - under the Project „Assessment of Separate Collection Schemes in the 28 Capitals of the EU". Brussels. https://www.municipalwasteeurope.eu/sites/default/files/BG%20Sofia%20Capital%20factsheet.pdf.

EC, 2015. Assessment of separate collection schemes in the 28 capitals of the EU – final report. Brussels. http://ec.europa.eu/environment/waste/studies/pdf/Separate%20collection_Final%20Report.pdf.

EC, 2018. Eurostat. Material flow accounts (online data codes: env_ac_mfa). http://ec.europa.eu/eurostat.

EC, 2018. Eurostat. Material flow accounts (online data codes: env_ac_mfa). http://ec.europa.eu/eurostat.

EC, 2018. Eurostat. Simplified energy balances - annual data (online data codes: nrg_100a). http://ec.europa.eu/eurostat.

Ecoinvent Centre, 2016. Ecoinvent database v3.2. Inf. Syst. Sustain. Dev. https://doi.org/10.4018/978-1-59140-342-5.ch003.

EcoMondis, 2018. Waste derived alternative fuels. http://ecomondis.com/brochure.pdf.

Eew, 2013. Knapsack standort. www.eew-energyfromwaste.com/fileadmin/content/Materialbestellung/Knapsack_Standortflyer.pdf.

Eew, 2014. Thermal waste treatment and energy generation plant (TREA) Breisgau. www.eew-energyfromwaste.com/en/our-sites/breisgau.html.

Energia, Vantann, 2015. Waste-to-energy gives a new life for rubbish. www.vantaanenergia.fi/en/waste-to-energy-gives-a-new-life-for-rubbish/.

Energinet, dk, 2012. Technology data for energy plants. https://energiatalgud.ee/img_auth.php/4/42/Energinet.dk._Technology_Data_for_Energy_Plants._2012.pdf.

EPEM, 2014. Database of waste management technologies. www.epem.gr/waste-c-control/database/html/costdata-00.htm.

Eunomia, 2009. EU waste management. http://ec.europa.eu/environment/waste/studies/pdf/euwasteman-agement_annexes.pdf.

European Parliament, 2018a. Waste management in the EU: infographic with facts and figures. http://www.europarl.europa.eu/news/en/headlines/society/20180328STO00751/eu-waste-management-infographic-with-facts-and-figures.

European Parliament, 2018b. Briefing EU legislation in progress from july 2018 - circular economy package four legislative proposals on waste. http://www.europarl.europa.eu/RegData/etudes/BRIE/2018/625108/EPRS_BRI(2018)625108_EN.pdf.

European Parliament and Council, 2015. COM/2015/0614 final – communication from the commission to the European parliament, the Council, the European economic and social committee and the committee of the regions closing the loop - an EU action plan for the circular economy. https://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:52015DC0614.

Fei, F., Wen, Z., Huang, S., De Clercq, D., 2018. Mechanical biological treatment of municipal solid waste: energy efficiency, environmental impact and economic feasibility analysis. J. Clean. Prod. 178, 731–739. https://doi.org/10.1016/j.jclepro.2018.01.060.

Franchetti, M., 2013. Economic and environmental analysis of four different configurations of anaerobic digestion for food waste to energy conversion using LCA for: a food service provider case study. J. Environ. Manag. 123, 42–48. https://doi.org/10.1016/j.jenvman.2013.03.003.

Franckx, L., Van Acoleyen, M., Hogg, D., Gibbs, A., Elliott, T., Sherrington, C., Ballinger, A., O'Brien, S., Lister, D., Beswick, C., 2010. Assessment of the Options to Improve the Management of Biowaste in the European Union.

Franzese, P.P., Rydberg, T., Russo, G.F., Ulgiati, S., 2009. Sustainable biomass production: a comparison between gross energy requirement and emergy synthesis methods. Ecol. Indicat. 9, 959–970. https://doi.org/10.1016/j.ecolind.2008.11.004.

Friotherm, A.G., 2015. www.friotherm.com/webautor-data/41/sysave006_uk.pdf.

Frischknecht, R., Jungbluth, N., Althaus, H.-J., Bauer, C., Doka, G., Dones, R., et al., 2007. Implementation of life cycle impact assessment methods. Ecoinvent report No. 3. https://inis.iaea.org/search/search.aspx?orig_q=RN:41028089.

Gentil, E.C., Damgaard, A., Hauschild, M., Finnveden, G., Eriksson, O., Thorneloe, S., Kaplan, P.O., Barlaz, M., Muller, O., Matsui, Y., Ii, R., Christensen, T.H., 2010.

Models for waste life cycle assessment: review of technical assumptions. Waste Manag. 30, 2636–2648. https://doi.org/10.1016/j.wasman.2010.06.004.
Government of India, 2018. Ministry of housing and urban affairs – guidelines on usage of refuse derived fuel in various industries. http://164.100.228.143:8080/sbm/content/writereaddata/Guidelines%20on%20RDF%20Usage.pdf.
Granatstein, D.I., 2000. Technoeconomic Assessment of Fluidized Bed Combustors as Municipal Solid Waste Incinerators: A Summary of Six Case Studies, IEA Task 23. CETC, Nepean, Canada. https://www.scribd.com/document/227844300/A-Summary-of-Six-Case-Studies.
Greco, G., Allegrini, M., Del Lungo, C., Gori Savellini, P., Gabellini, L., 2015. Drivers of solid waste collection costs. Empirical evidence from Italy. J. Clean. Prod. 106, 364–371. https://doi.org/10.1016/j.jclepro.2014.07.011.
Green Investment Bank, 2013. Investment in new waste to energy project. www.greeninvestmentbank.com/news-and-insight/2013/green-investment-bank-announces-investment-in-new-waste-to-energy-project/.
HEP, 2018. Pricelist 2018. http://www.hep.hr/toplinarstvo/en/customers/price.aspx.
HZ INOVA, 2014. Lausanne/Switzerland energy-from-waste plant. www.hz-inova.com/cms/wp-content/uploads/2014/11/hzi_ref_lausanne-en.pdf.
Ichinose, D., Yamamoto, M., Yoshida, Y., 2015. The decoupling of affluence and waste discharge under spatial correlation: do richer communities discharge more waste? Environ. Dev. Econ. 20, 161–184. https://doi.org/10.1017/S1355770X14000370.
Kalogirou, E., 2010. MVB (MSW & Biomass) Waste to Energy Plants and the AVG Hazardous WTE Plant in Hamburg, Germany. WtERT Technical Report. Columbia University, New York. http://www.wtert.com.br/home2010/arquivo/noticias_eventos/HAMBURG_VISIT_of_Dr_KALOGIROU.pdf.
Kalogirou, E., 2011. Oslo Norway WtE Plant, WtERT Technical Report, Earth Engineering Center. Columbia University, New York. http://www.seas.columbia.edu/earth/wtert/sofos/Kalogirou_report_OSLO_WtE.pdf.
Kalogirou, E., 2013. Report of Participation at Meeting of MatER. Synergia, Torino, Italy. http://www.seas.columbia.edu/earth/wtert/newwtert/Research/sofos/Report%20of%20GWC%20participation%20at%20Torino.pdf.
Karavezyris, V., 2001. Prognose von Siedlungsabfällen: Untersuchungen zu determinierenden Faktoren und methodischen Ansätzen. TK Verlag, Neuruppin.
Karellas, S., Boukis, I., Kontopoulos, G., 2010. Development of an investment decision tool for biogas production from agricultural waste. Renew. Sustain. Energy Rev. 14, 1273–1282. https://doi.org/10.1016/j.rser.2009.12.002.
Khan, M.M.U.H., Jain, S., Vaezi, M., Kumar, A., 2016. Development of a decision model for the techno-economic assessment of municipal solid waste utilization pathways. Waste Manag. 48, 548–564. https://doi.org/10.1016/j.wasman.2015.10.016.
Letsrecycle, 2018. Secondary material market prices indicators. https://www.letsrecycle.com/.
Lima, P.D.M., Colvero, D.A., Gomes, A.P., Wenzel, H., Schalch, V., Cimpan, C., 2018. Environmental assessment of existing and alternative options for management of municipal solid waste in Brazil. Waste Manag. 78, 857–870, 0.1016/j.wasman.2018.07.007.
Lončar, D., et al., 2009. Developer Support - Best Practices for Wood Biomass Cogeneration. Technology Transfer Center - CTT, Zagreb. http://oie.mingo.hr/UserDocsImages/BIOCHP_HR.pdf.
Luz, F.C., Rocha, M.H., Lora, E.E.S., Venturini, O.J., Andrade, R.V., Leme, M.M.V., Del Olmo, O.A., 2015. Techno-economic analysis of municipal solid waste gasification for electricity generation in Brazil. Energy Convers. Manag. 103, 321–337. https://doi.org/10.1016/j.enconman.2015.06.074.
Magrinho, A., Semiao, V., 2008. Estimation of residual MSW heating value as a function of waste component recycling. Waste Manag. 28, 2675–2683. https://doi.org/10.1016/j.wasman.2007.12.011.
Mazzanti, M., Zoboli, R., 2008. Waste generation, waste disposal and policy effectiveness. Resour. Conserv. Recycl. 52, 1221–1234. https://doi.org/10.1016/j.resconrec.2008.07.003.
Mazzanti, M., Montini, A., Zoboli, R., 2008. Municipal waste generation and socioeconomic drivers: evidence from comparing Northern and Southern Italy. J. Environ. Dev. 17, 51–69. https://doi.org/10.1177/1070496507312575.
McDougall, F.R., Franke, M., Hindle, P., White, P.R., 2001. Integrated Solid Waste Management: a Life Cycle Inventory. Blackwell Science.
Minoglou, M., Komilis, D., 2013. Optimizing the treatment and disposal of municipal solid wastes using mathematical programming - a case study in a Greek region. Resour. Conserv. Recycl. 80, 46–57. https://doi.org/10.1016/j.resconrec.2013.08.004.
Mitchell, G., 2015. Building a Business Case for Compressed Natural Gas in Fleet Applications. National Renewable Energy Laboratory. https://afdc.energy.gov/files/u/publication/business_case_cng_fleets.pdf.
Montejo, C., Tonini, D., Márquez, M., del, C., Fruergaard Astrup, T., 2013. Mechanical-biological treatment: performance and potentials. An LCA of 8 MBT plants including waste characterization. J. Environ. Manag. 128, 661–673. https://doi.org/10.1016/j.jenvman.2013.05.063.
Mužinić, M., Pašalić, G., Martina, C., Fundurulja, D., Domanovac, T., 2007. Waste Management Plan of the City of Zagreb.
Norris, G.A., 2001. Integrating economic analysis into LCA. Environ. Qual. Manag. 10, 59–64. https://doi.org/10.1002/tqem.1006.
Požgaj, Đ., Puntarić, E., Kušćević-Vukšić, M., Kufrin, J., 2019. Municipal Solid Waste Report for 2018. Ministry of Environment and Energy.
RPS, 2014. Dublin waste to energy – waste market assessment. https://www.indymedia.ie/attachments/oct2014/rpsreportfinal.pdf.
SACE, 2011. www.eib.org/epec/resources/presentations/sace-guarantees-psf-may2011.pdf.
Schäfer, A., Moser, A., 2012. Dispatch optimization and economic evaluation of distributed generation in a virtual power plant. IEEE Energytech. https://doi.org/10.1109/EnergyTech.2012.6304655, 2012.
Sharma, B.K., 2006. Analytical Chemistry: Comprehensively Covering the UGC Syllabus. Krishna Prakashan Media, Meerut, Delhi.
State of Green, 2011. Waste-to-Energy CHP. EGE in Oslo Norway. https://stateofgreen.com/en/profiles/ramboll/solutions/waste-to-energy-chp-ege-in-oslo-norway.
Syazwina, F., Shukor, A., Mohammed, A.H., Abdullah, S.I., Awang, M., 2011. A Review on the Success Factors for Community Participation in Solid Waste Management, pp. 963–976.
Tan, S.T., Ho, W.S., Hashim, H., Lee, C.T., Taib, M.R., Ho, C.S., 2015. Energy, economic and environmental (3E) analysis of waste-to-energy (WTE) strategies for municipal solid waste (MSW) management in Malaysia. Energy Convers. Manag. 102, 111–120. https://doi.org/10.1016/j.enconman.2015.02.010.
Tomić, T., Schneider, D.R., 2017. Municipal solid waste system analysis through energy consumption and return approach. J. Environ. Manag. 203, 973–987. https://doi.org/10.1016/j.jenvman.2017.06.070.
Tomić, T., Schneider, D.R., 2018. The role of energy from waste in circular economy and closing the loop concept – energy analysis approach. Renew. Sustain. Energy Rev. 98, 268–287. https://doi.org/10.1016/j.rser.2018.09.029.
Tomić, T., Ćosić, B., Schneider, D.R., 2016. Influence of legislative conditioned changes in waste management on economic viability of MSW-fuelled district heating system: case study. Therm. Sci. 20, 1105–1120. https://doi.org/10.2298/TSCI160212114T.
Tomić, T., Dominković, D.F., Pfeifer, A., Schneider, D.R., Pedersen, A.S., Duić, N., 2017. Waste to energy plant operation under the influence of market and legislation conditioned changes. Energy 137, 1119–1129. https://doi.org/10.1016/j.energy.2017.04.080.
UK Government, 2013. Projects in procurement, 2013. www.gov.uk/government/uploads/system/uploads/attachment_data/file/230869/Annex_B_-_pfi_projects_in_procurement_march_2013.xlsx.
UK Government, 2013b. Incineration of municipal waste. www.gov.uk/government/uploads/system/uploads/attachment_data/file/221036/pb13889-incineration-municipal-waste.pdf.
USEPA, 2019. AP-42 chapter 2.4 Municipal solid waste landfills. https://www3.epa.gov/ttnchie1/ap42/ch02/final/c02s04.pdf.
Warren, W.K.E.H., 2012. A Techno-Economic Comparison of Biogas Upgrading Technologies in Europe. Jyväskylä. University of Jyväskylä. . 2012. citeseerx.ist.psu.edu/viewdoc/download?doi=10.1.1.456.1353&rep=rep1&type=pdf.
Waste Management World, 2014. 320,000 TPA recycling, biogas and waste to energy plant secures finance in yorkshire. www.waste-management-world.com/articles/2014/11/320-000-tpa-recycling-biogas-and-waste-to-energy-plant-secures-finance-in-yorkshire.html.
Wood, S., et al., 2013. Review of State-Of-The-Art Waste-To-Energy Technologies, Stage Two Report – Case Studies. WSP, London. http://www.wtert.com.br/home2010/arquivo/noticias_eventos/WSP%20Waste%20to%20Energy%20Technical%20Report%20Stage%20Two.pdf.
Zhang, R., Ma, X., Shen, X., Zhai, Y., Zhang, T., Ji, C., Hong, J., 2020. PET bottles recycling in China: an LCA coupled with LCC case study of blanket production made of waste PET bottles. J. Environ. Manag. 260 https://doi.org/10.1016/j.jenvman.2019.110062.
Zhou, Z., Tang, Y., Dong, J., Chi, Y., Ni, M., Li, N., Zhang, Y., 2018. Environmental performance evolution of municipal solid waste management by life cycle assessment in Hangzhou, China. J. Environ. Manag. 227, 23–33. https://doi.org/10.1016/j.jenvman.2018.08.083.