

**Supporting Information (SI)**

Environmental Science & Technology

# Material Resources, Energy, and Nutrient Recovery from Waste: Are Waste Refineries the Solution for the Future?

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This SI document includes text, tables, and figures with details on the process data for the inventory analysis of the LCA. Additional information on waste materials and energy balances, and sensitivity analyses are also provided.

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## 1. LCA scenarios modelling and system boundary

This section provides additional information with respect to the modelling and system boundary of the investigated scenarios. As thoroughly detailed in the manuscript a total of 252 scenarios have been addressed. Figure S1-S14 illustrates the boundary conditions for the scenarios *0* and *I* for the case of waste composition *a* (only relevant flows; values are rounded to two significant digits). Notice that electricity and heat produced are net values (i.e., plants own consumptions have been subtracted, including energy for pre-treatments). The modelling for the case of scenarios *II* and for the case of waste composition *b* is similar. For the latter, additional information on waste materials and energy balances can be found in section 4. Table S3 provides an overview of technologies and pre-treatments involved in the scenarios under assessment.

### 1.1 Waste composition

As detailed in the main manuscript, two sets of waste compositions (*a* and *b*) were used in the assessment. These are shown in Table S1-S2 along with the composition of the residual MSW (rMSW representing the left-over after source-segregation of recyclable materials and organic waste) in the assessed scenarios *0*, *I*, and *II*. Details regarding waste composition, water content, and chemical composition of the individual waste material fractions can be found in (1, 2). For details regarding amounts of individual material fractions, source-segregated MSW (ssMSW), and residual MSW (rMSW), please refer to section 4 and Table S9. Please note that not all individual waste materials within waste type mentioned in Table S2 were assumed source-segregated, e.g., dirty paper and dirty cardboard were considered routed to the rMSW for further treatment. The share of dirty paper was estimated to ca. 1.9% (of the total) in waste *a* and to 1.4% (of the total) in waste *b*. Dirty cardboard was 0.98% in *a* and 1.4% in *b*. For plastic, non-recyclable plastic items (e.g., plastic toys) were estimated to 5.1% in *a* and 1.5% in *b*. The low share of recyclable plastic in *a* can be largely attributed due to the Danish return system (in Denmark a large share of the plastic bottles are collected through a separate collection-refunding system, and were not considered as part of the generated household waste in the investigations performed by (1), i.e. not part of the household-bin). For glass, the non-recyclable share was 0.24% in *a* and 0.82% in *b*. All materials in the fractions ferrous metal and aluminium (Table S1-S2) were considered recyclable. As an example, the amount of paper source-segregated in the case of waste *a* would equal the amount of clean paper multiplied by the separation efficiency

(0.8), i.e., on 1000 kg MSW: (250 kg-19 kg)\*0.8=184.8 kg (180 kg with two significant digits, as reported in Table S9 and Figures S1-S14).

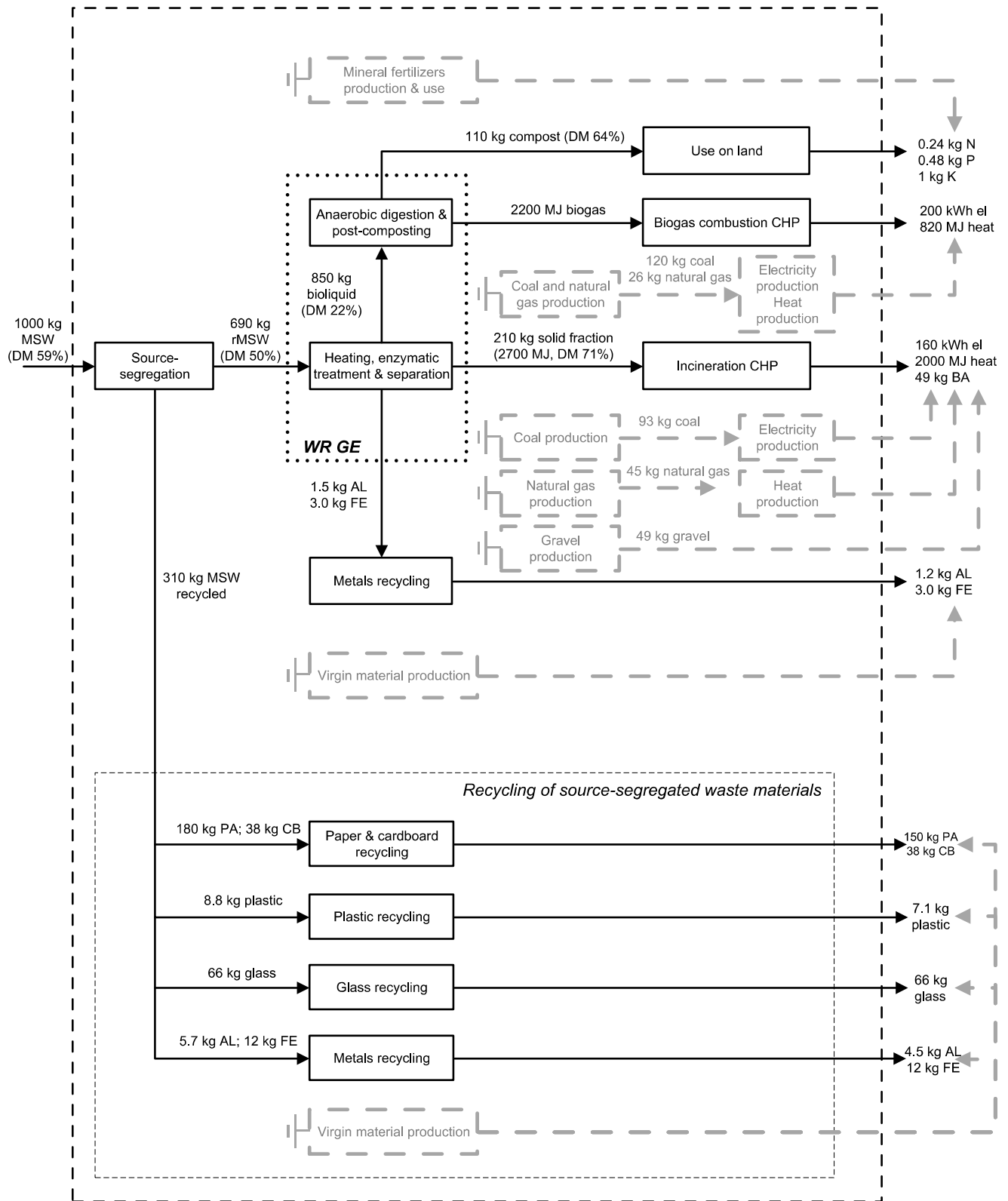
**Table S1.** Overview of the MSW composition datasets used in the study; *a*: (1); *b*: (2). The rMSW composition (as modelled after source-segregation of selected recyclables and organic waste) is also presented; rMSW *0*: no organic source-segregation; rMSW *I*: organic source-segregation (efficiency 70%); rMSW *II*: organic source-segregation (efficiency 100%). Any inconsistencies are due to rounding (values are rounded to two significant digits).

Waste material fraction	MSW		rMSW <i>0</i>		rMSW <i>I</i>		rMSW <i>II</i>	
	<i>a</i>	<i>b</i>	<i>a</i>	<i>b</i>	<i>a</i>	<i>b</i>	<i>a</i>	<i>b</i>
1. Organic waste	37	60	54	72	30	45	9.5	8.4
2. Paper	25	9.6	9.4	3.7	15	7.1	19	12
Clean paper	23	8.2						
Dirty paper	1.9	1.4						
3. Cardboard	5.7	4.1	2.8	2.3	4.3	4.5	5.6	7.6
Clean cardboard	4.7	2.7						
Dirty cardboard	0.98	1.4						
4. Cartons and alike	2.4	1.2	3.4	1.4	5.3	2.8	6.8	4.7
5. Plastic	6.9	9.3	8.8	6.5	13	13	17	21
Recyclable plastic	1.8	7.8						
Non-recyclable plastic	5.1	1.5						
6. Textile	1.8	3.5	2.6	4.2	4.0	8.2	5.1	14
7. Glass	9.0	3.3	3.3	1.0	5.0	1.9	6.5	3.2
Recyclable glass	8.8	2.5						
Other glass	0.24	0.82						
8. Ferrous metal	1.6	2.5	0.57	0.75	0.88	1.5	1.1	2.4
9. Aluminium	0.76	0.50	0.28	0.15	0.42	0.29	0.55	0.49
10. Other	10	6.6	14	8.0	22	15	29	26
Total (1 to 10)	100	100	100	100	100	100	100	100

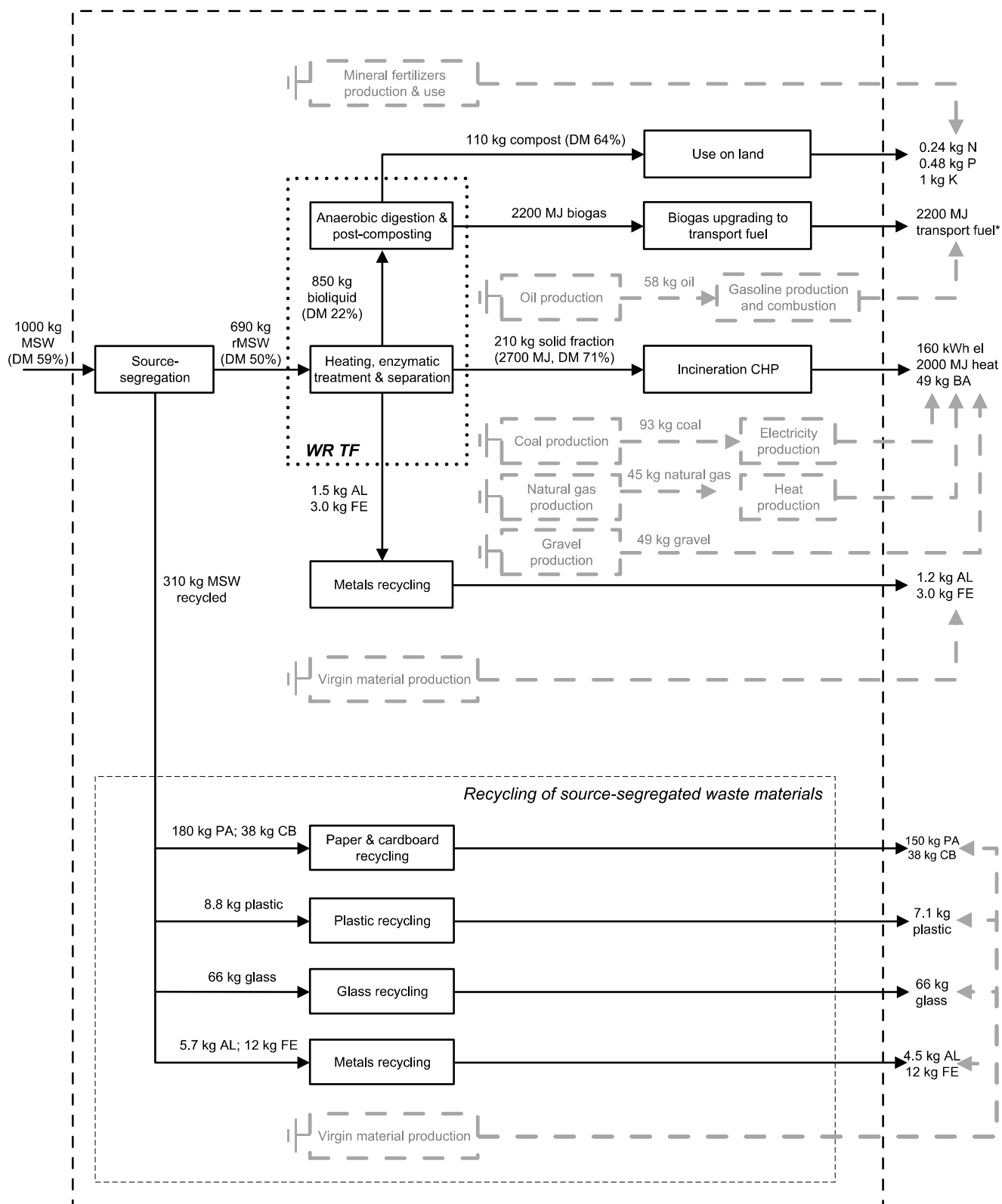
**Table S2.** Overview of the waste material fractions considered in the assessment. Lower heating values (LHV) assumed for waste materials as generated are provided as MJ kg DM<sup>-1</sup> for waste *a* and *b*.

Macro-waste material	LHV ( <i>a/b</i> )	Waste material fractions
Organic waste	19.1/15	Vegetable food, animal food, kitchen tissues, wood-like materials (yard waste)
Paper	15.4/14.7	Newspapers, magazines, advertisements, books and phonebooks, office paper, other clean paper, dirty paper
Cardboard	17.2/14.7	Cardboard containers, dirty cardboard
Cartons and alike	22/23.6	Milk cartons and alike, cartons with aluminium foil
Plastic	33.3/38.5	Soft plastic, plastic bottles, hard plastic, non-recyclable plastic
Textile	20.7/23	Textiles, shoes, and leather
Glass	0.0/0.0	Clear glass, green glass, brown glass, other glass
Ferrous metal	0.0/0.0	Metal containers, other mixed ferrous metals
Aluminium	3.7/3.7	Aluminium containers and tins, aluminium trays/foils
Other	15/28.9	Office articles, cigarettes butts, diapers, sanitary towel, bandages, cotton, tampons, vacuum cleaner bags, soil, stones, batteries, animals gravel and litter, ceramics, ashes, plastic-coated aluminium foils, other non-combustibles

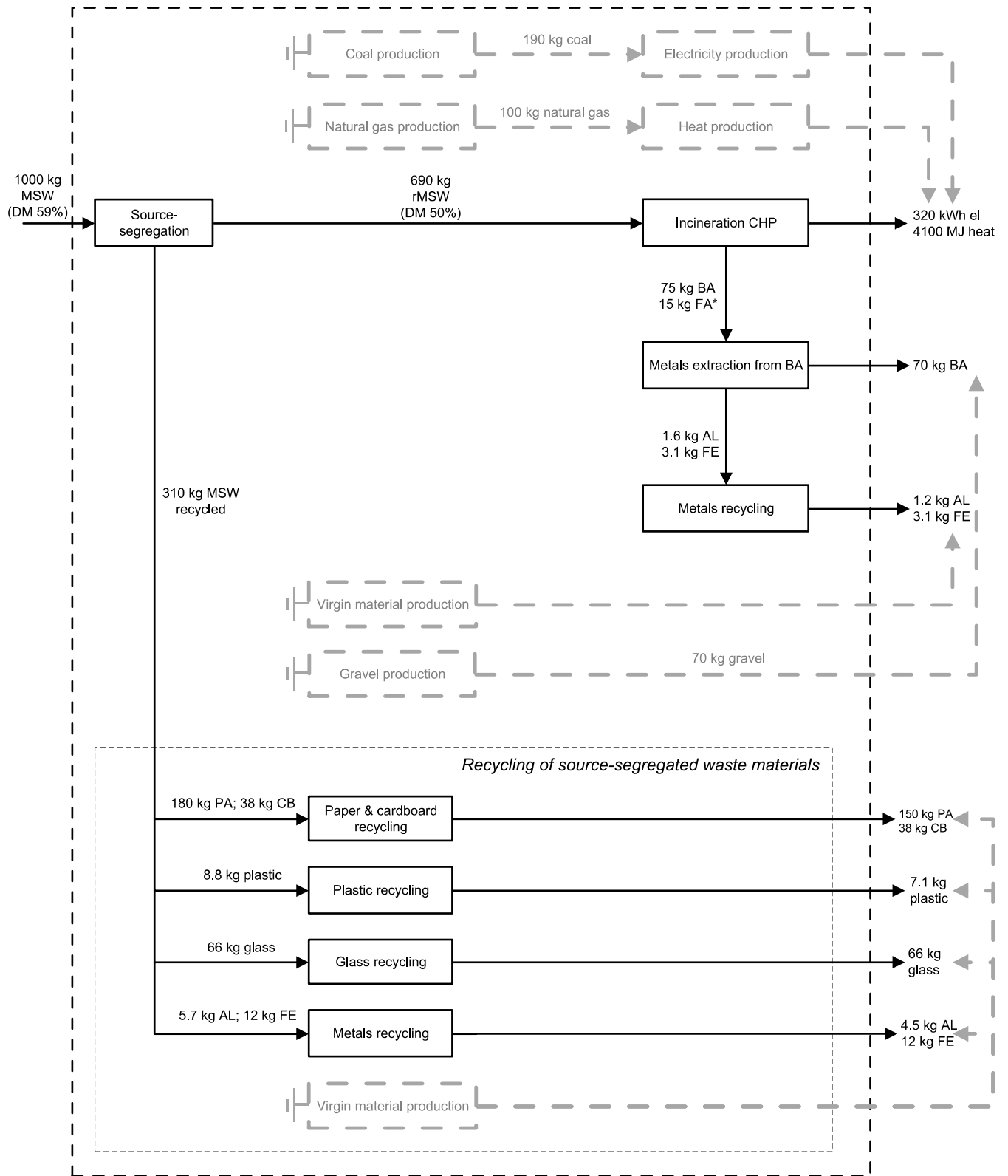
Figures S1-S14 display the main (selected) mass and energy flows involved in the scenarios under assessment. Please, notice that the amount of aluminium (AL) recovered from bottom ash accounts for all the aluminium (Al) that is found in the waste incinerated and not only for the Al found in the non-segregated share of the macro-fraction “Aluminium” reported in Table S1-S2 (for example plastic-coated aluminium foil and cartons with aluminium foil also contain Al). In addition, recovery of ferrous metal and aluminium from bottom ash was considered for all scenarios involving incineration (i.e., *WR GE*, *WR TF*, *INC*, *MBT AC*, *MBT DC*), although this is not specifically displayed for *WR GE*, *WR TF*, *MBT AC*, and *MBT DC* due to space limitation in the associated Figures. In these, the recovery of ferrous metal and aluminium from incineration bottom ash corresponded to ca. 0.4 kg and 0.2 kg, respectively.



**Figure S1.** LCA system boundary of *WR GE 0* for the case of waste composition *a*. AL: aluminium; CB: cardboard; CHP: combined-heat-and-power; FE: ferrous metal; PA: paper; rMSW: residual MSW.

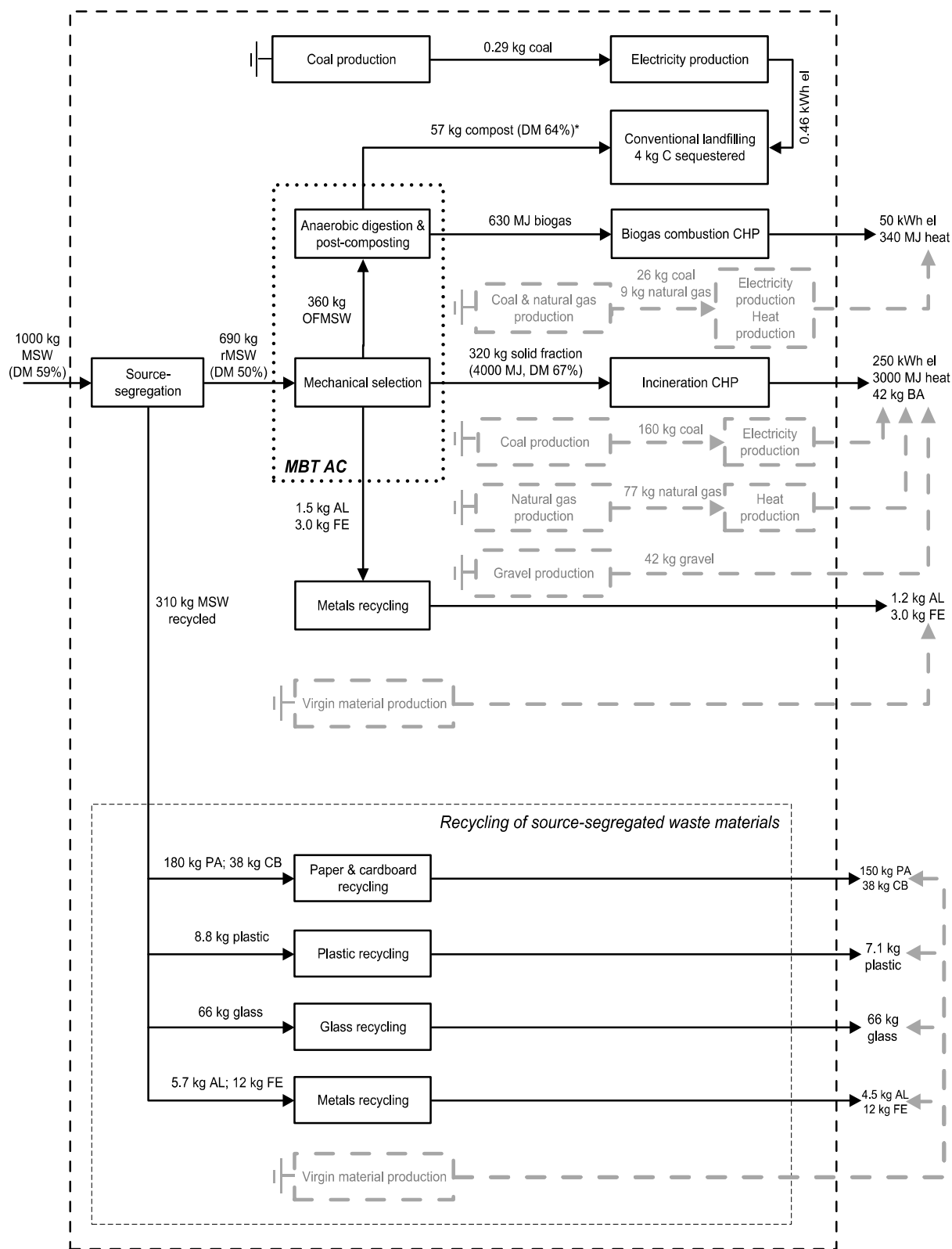


**Figure S2.** LCA system boundary of *WR TF 0* for the case of waste composition *a*. \*The energy consumed for the upgrading process corresponded to  $0.014 \text{ kWh MJ}^{-1} \text{ CH}_4$  (not visualized). AL: aluminium; CB: cardboard; CHP: combined-heat-and-power; FE: ferrous metal; PA: paper; rMSW: residual MSW.

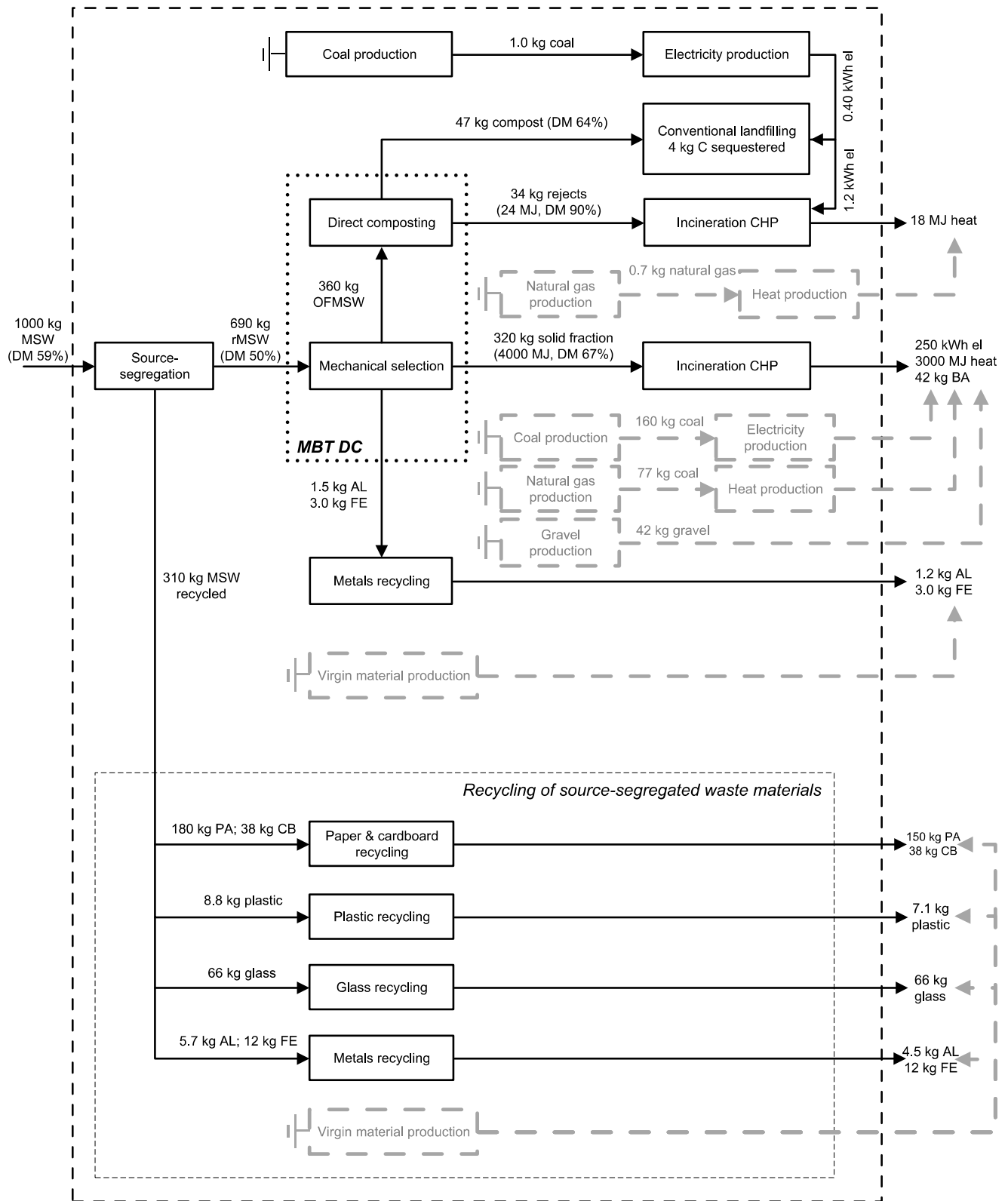


**Figure S3.** LCA system boundary of *INC 0* for the case of waste composition *a*. AL: aluminium; BA: bottom ash; CB: cardboard; CHP: combined-heat-and-power; FA: fly ash; FE: ferrous metal; PA: paper; rMSW: residual MSW. \*Used for backfilling of old salt mines.

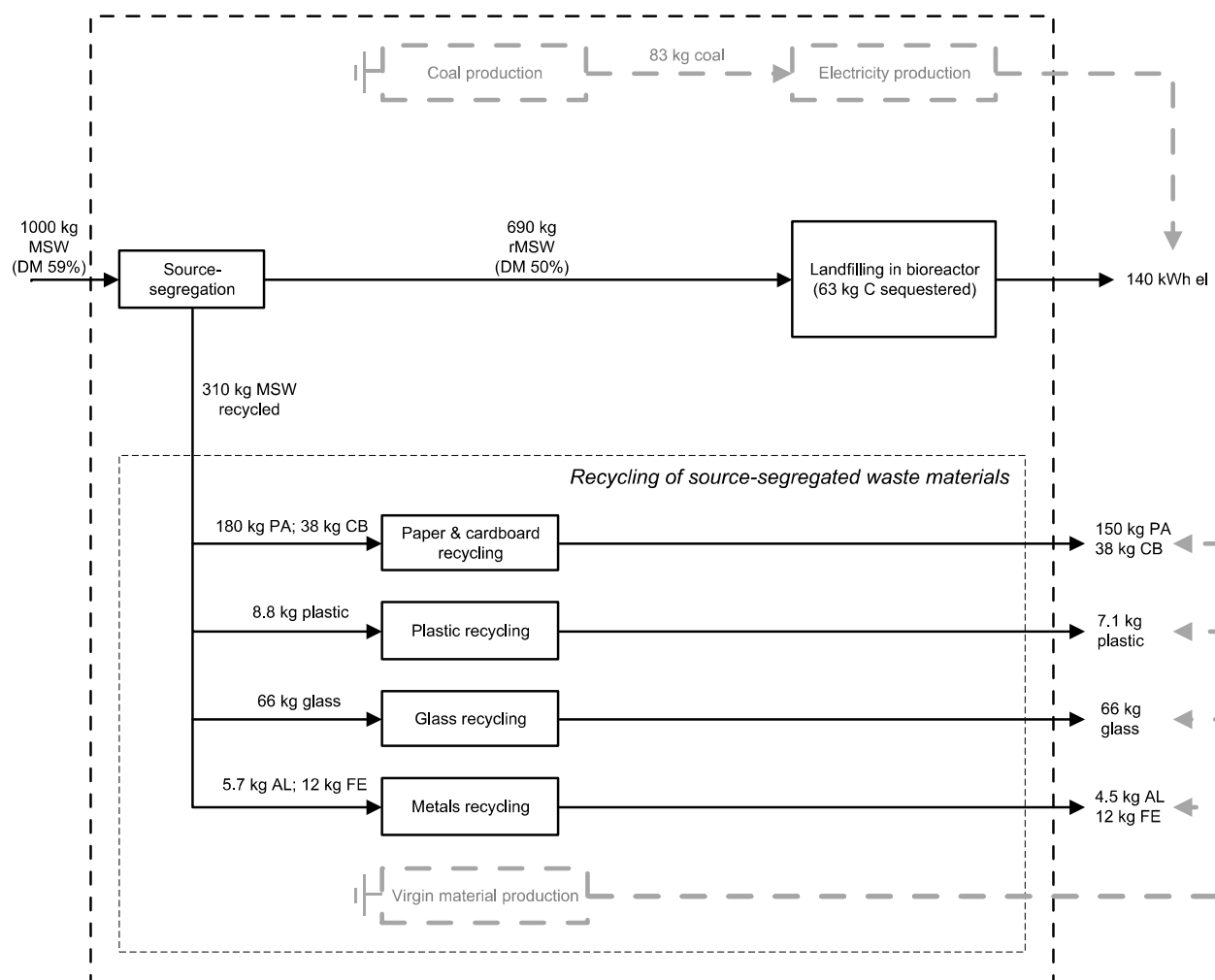




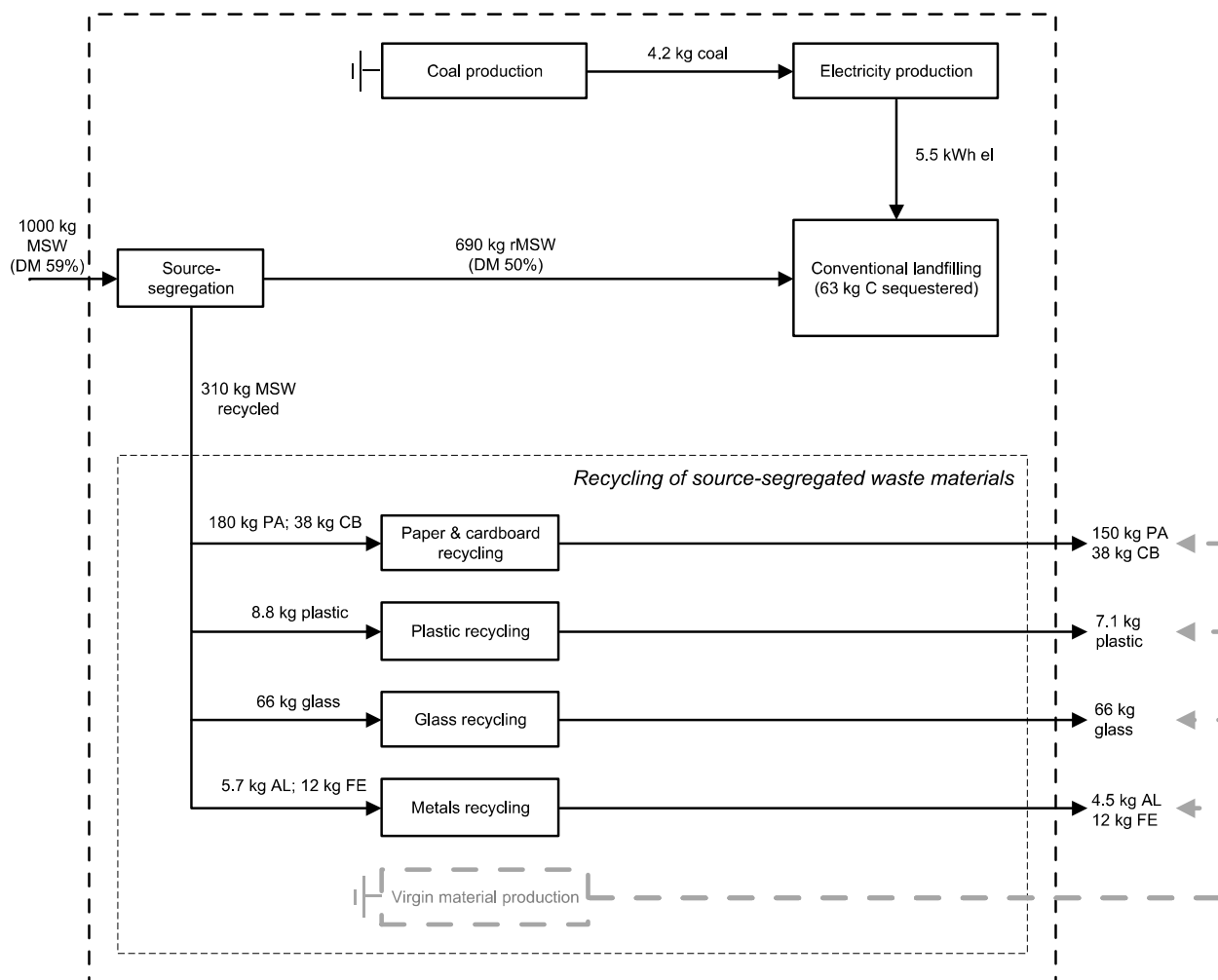
**Figure S4.** LCA system boundary of *MBT AC 0* for the case of waste composition *a*. AL: aluminium; CB: cardboard; CHP: combined-heat-and-power; FE: ferrous metal; PA: paper; rMSW: residual MSW.



**Figure S5.** LCA system boundary of *MBT DC 0* for the case of waste composition *a*. AL: aluminium; CB: cardboard; CHP: combined-heat-and-power; FE: ferrous metal; PA: paper; rMSW: residual MSW.

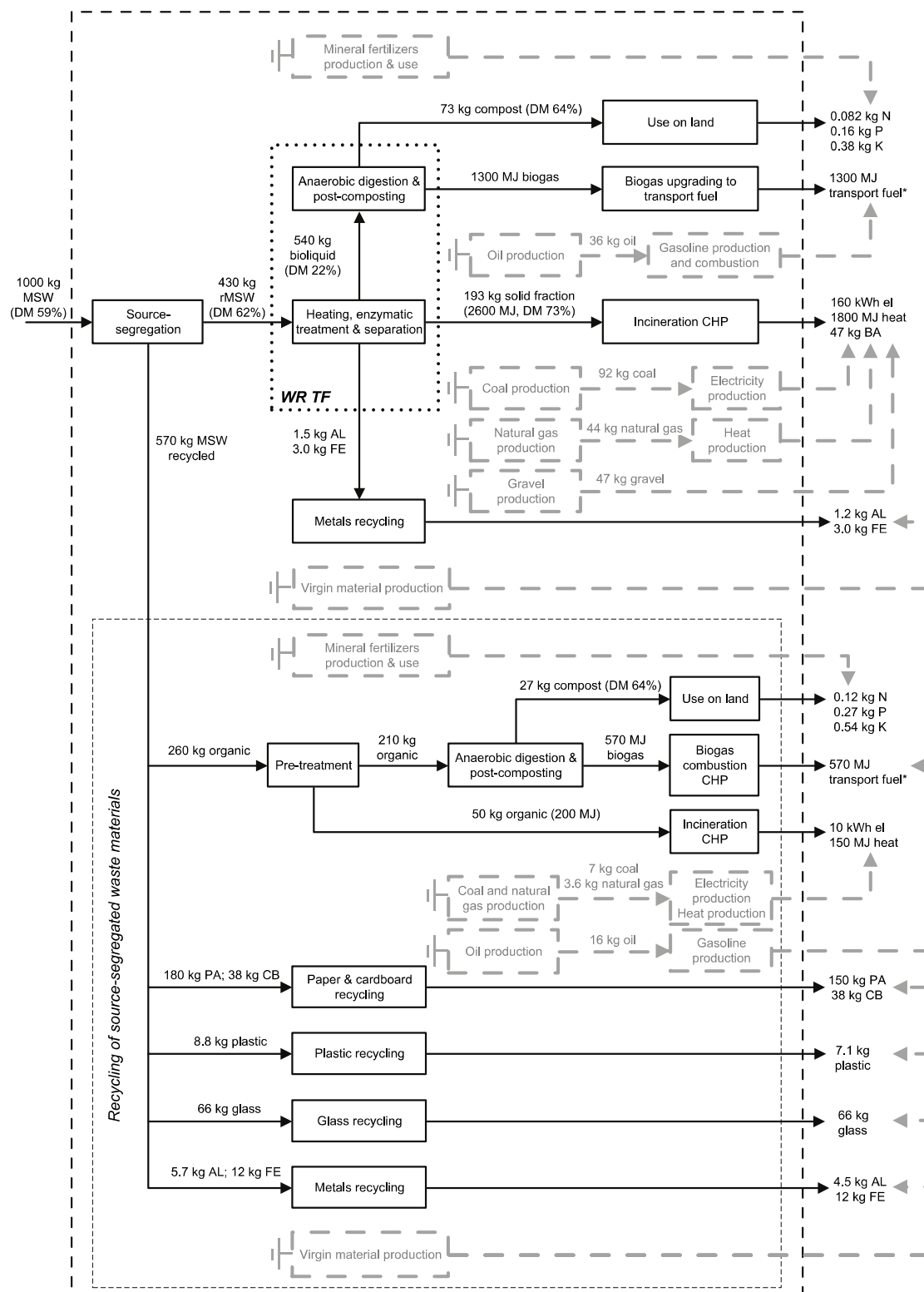


**Figure S6.** LCA system boundary of *BLF 0* for the case of waste composition *a*. AL: aluminium; CB: cardboard; CHP: combined-heat-and-power; FE: ferrous metal; PA: paper; rMSW: residual MSW.

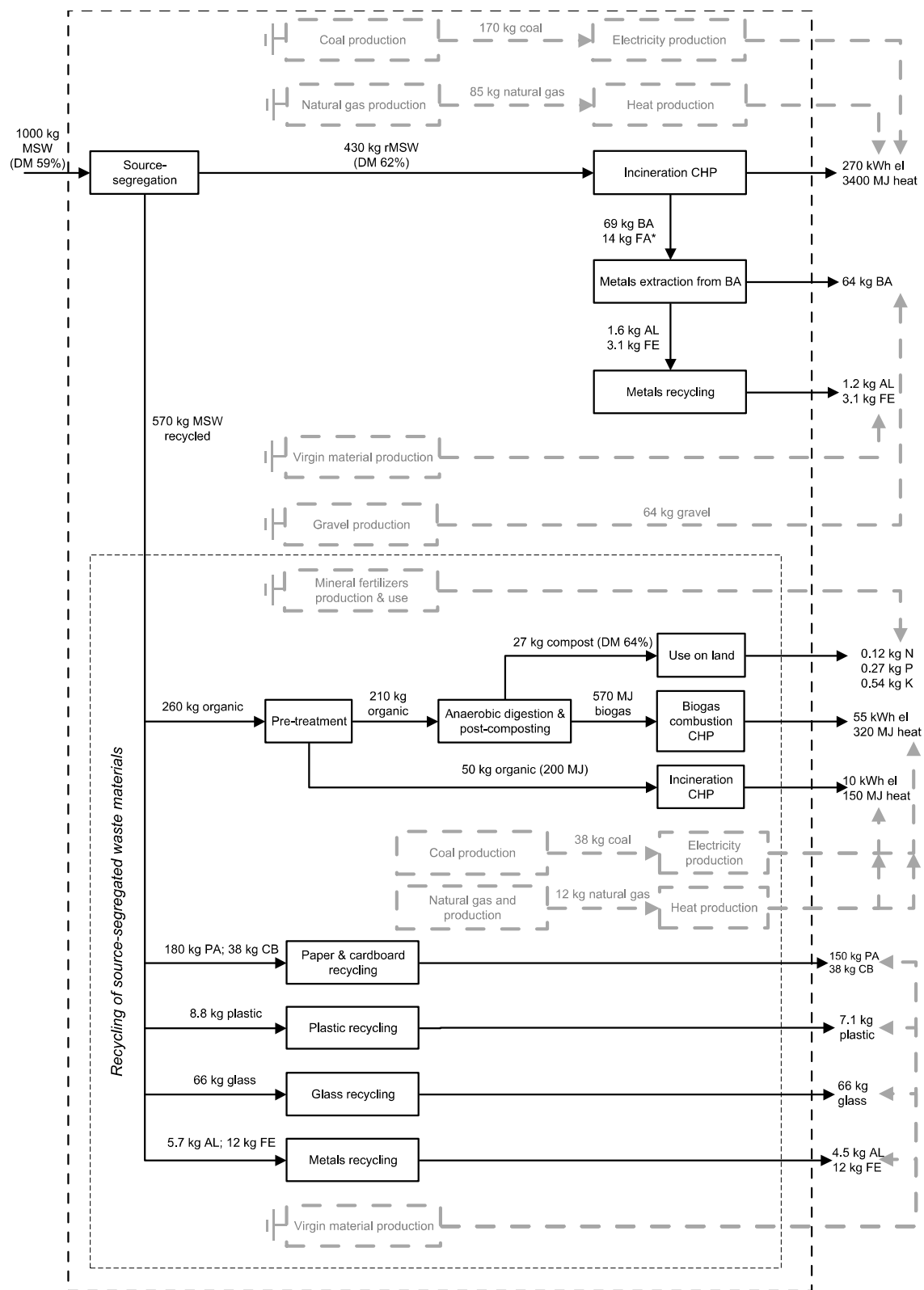


**Figure S7.** LCA system boundary of *CLF 0* for the case of waste composition *a*. AL: aluminium; CB: cardboard; CHP: combined-heat-and-power; FE: ferrous metal; PA: paper; rMSW: residual MSW.

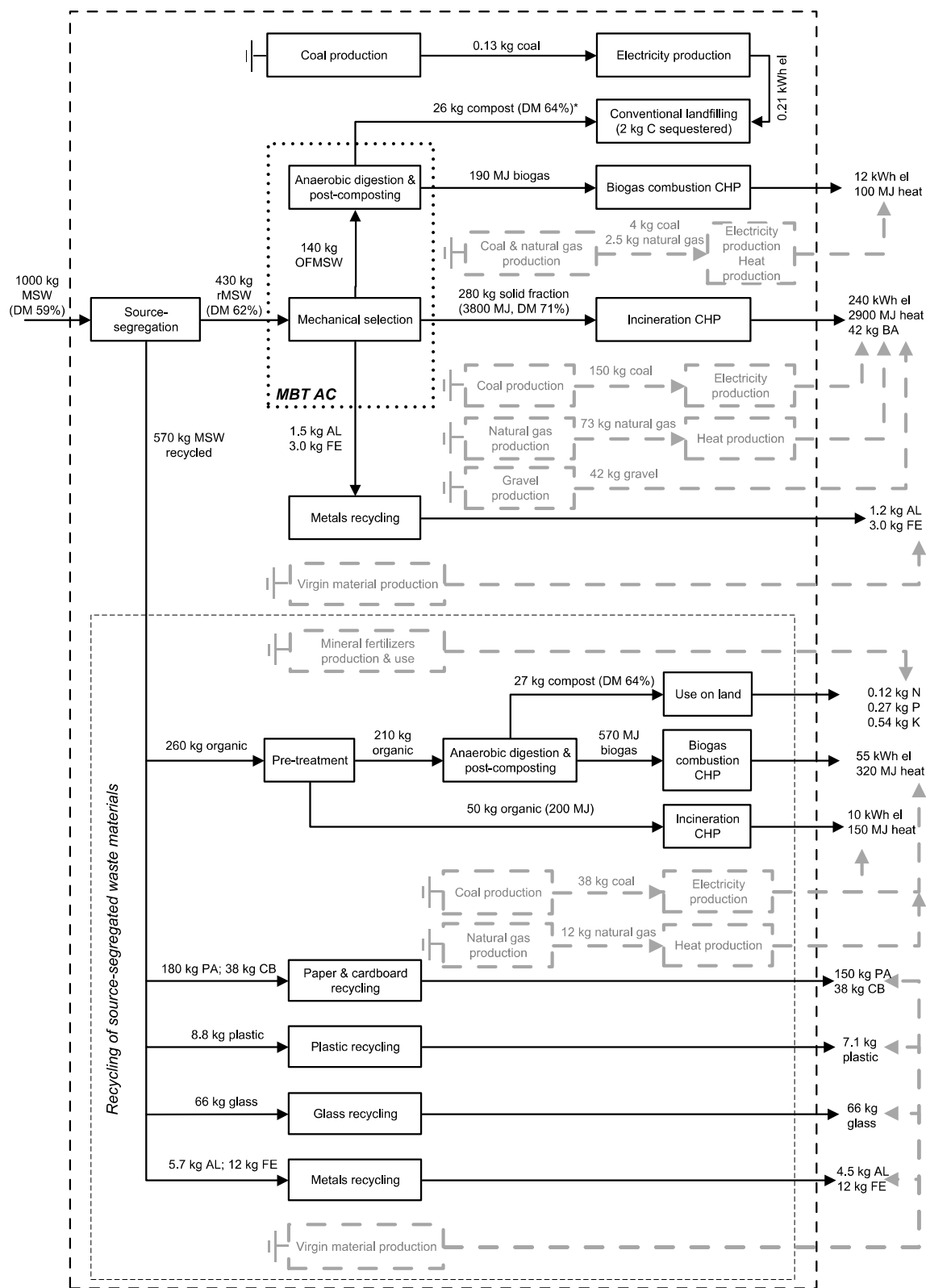




**Figure S9.** LCA system boundary of *WR TF I* for the case of waste composition *a*. \*The energy consumed for the upgrading process corresponded to  $0.014 \text{ kWh MJ}^{-1} \text{ CH}_4$  (not visualized). AL: aluminium; CB: cardboard; CHP: combined-heat-and-power; FE: ferrous metal; PA: paper; rMSW: residual MSW.



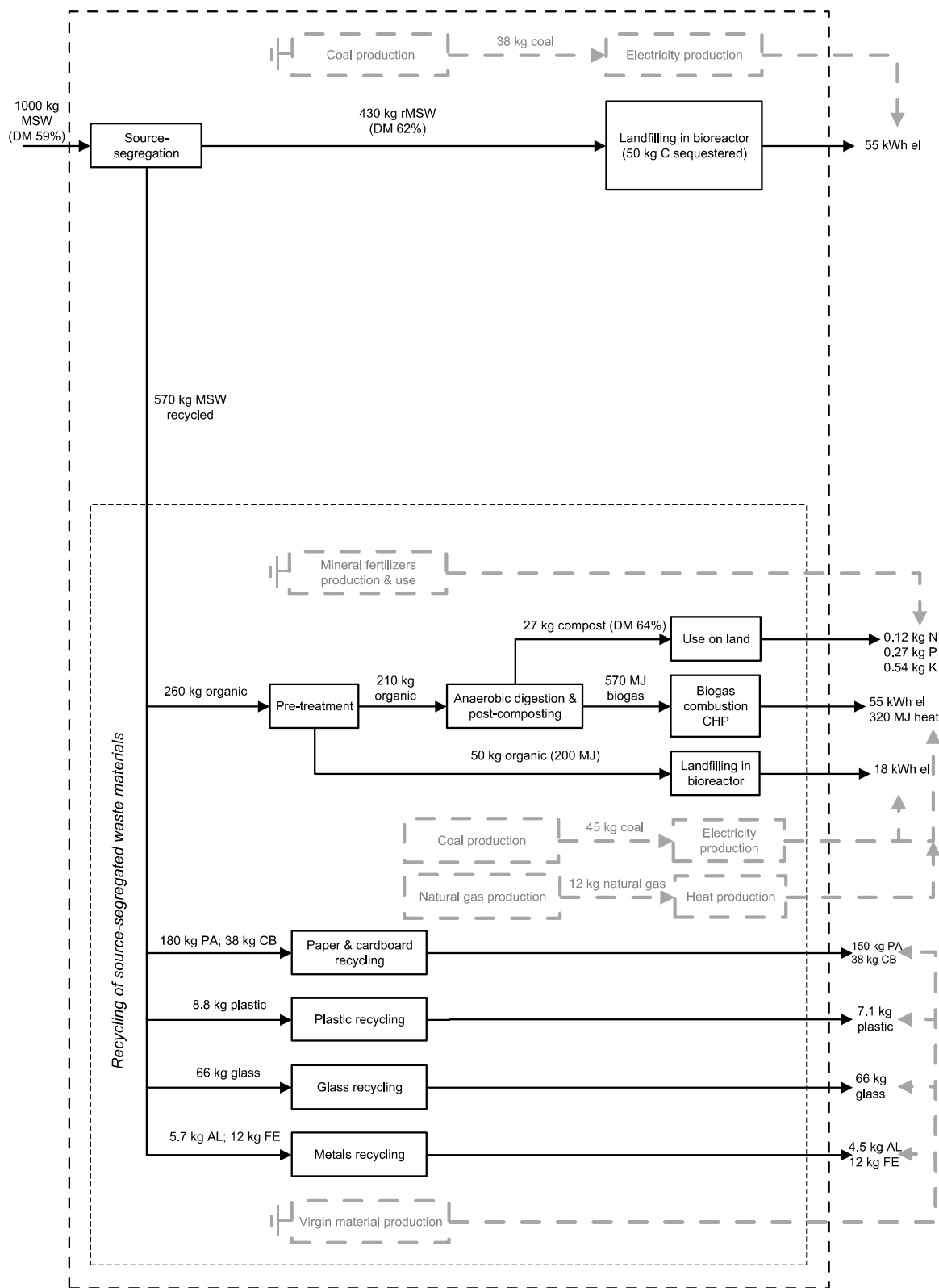
**Figure S10.** LCA system boundary of *INC I* for the case of waste composition *a*. AL: aluminium; BA: bottom ash; CB: cardboard; CHP: combined-heat-and-power; FA: fly ash; FE: ferrous metal; PA: paper; rMSW: residual MSW. \*Used for backfilling of old salt mines.



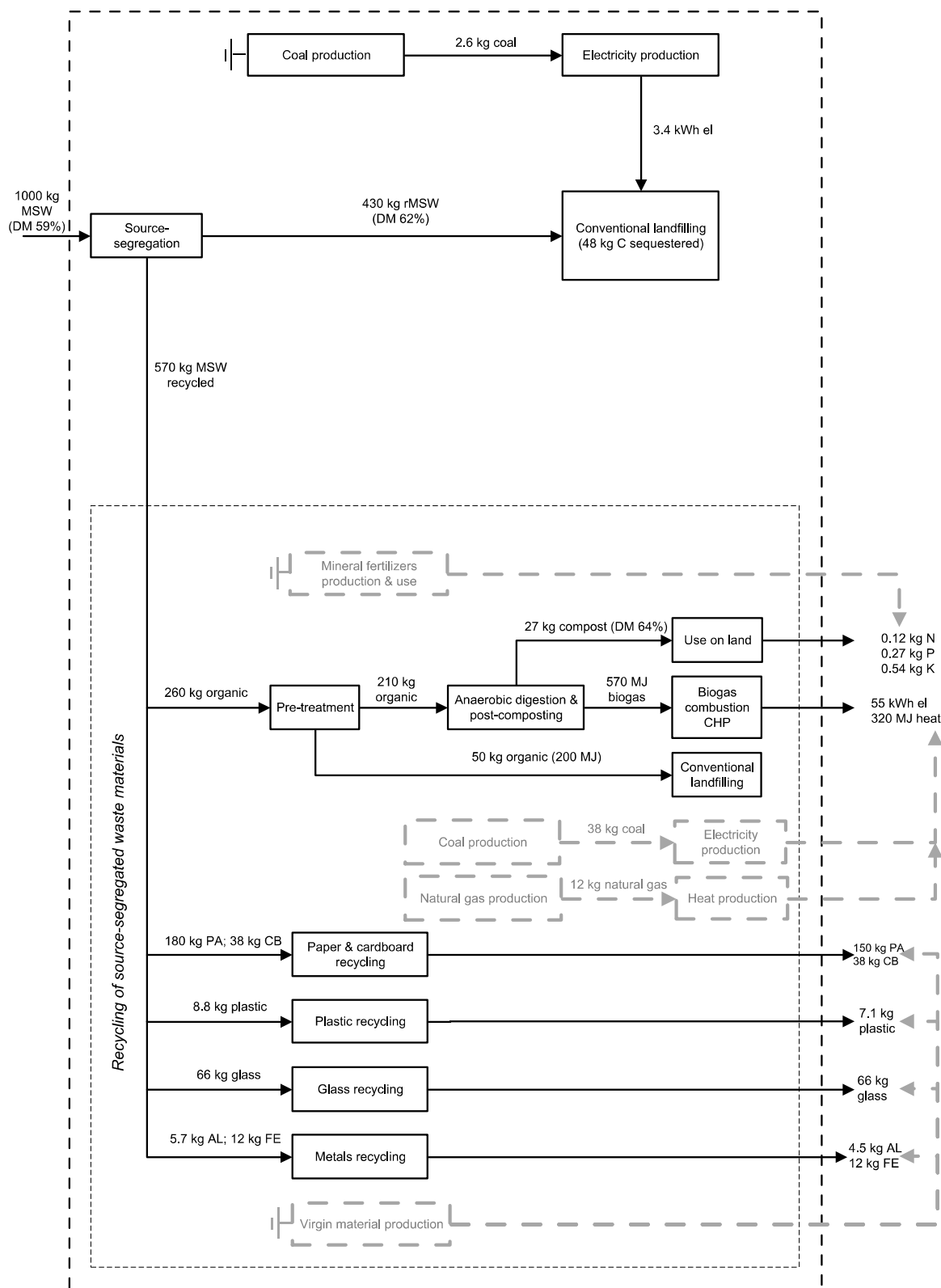
**Figure S11.** LCA system boundary of *MBT AC I* for the case of waste composition *a*. AL: aluminium; CB: cardboard; CHP: combined-heat-and-power; FE: ferrous metal; PA: paper; rMSW: residual MSW.







**Figure S13.** LCA system boundary of *BLF I* for the case of waste composition *a*. AL: aluminium; CB: cardboard; CHP: combined-heat-and-power; FE: ferrous metal; PA: paper; rMSW: residual MSW.



**Figure S14.** LCA system boundary of *CLF I* for the case of waste composition *a*. AL: aluminium; CB: cardboard; CHP: combined-heat-and-power; FE: ferrous metal; PA: paper; rMSW: residual MSW.

**Table S3.** Overview of the waste management scenarios considered in the LCA. Each individual scenario was assessed with two different types of waste composition, namely *a* and *b*. AD: anaerobic digestion; CHP: combined-heat-and-power; El: electricity; Enz: enzymatic; GE: natural gas engine; LF: landfill; Mec: mechanical; MSW: municipal solid waste; OFMSW: organic fraction of MSW; Res. solids: Residual solids; rMSW: residual MSW; SRF: Solid recovered fuel; SS OW: source-segregated organic waste.

Scenario		Source-segregation		Pre-treatment <sup>a</sup>			Associated energy conversion technologies (thermal/biological)			
		Selected materials	Organic waste	Mec	Enz	Outputs	SS OW	OFMSW/Bioliquid	SRF/Res. solids	rMSW/rejects
WR GE	<i>0</i>	x		x	x	Bioliquid/Res. solids		AD & GE (CHP)	Incineration (CHP)	Incineration (CHP)
WR TF		x		x	x	Bioliquid/Res. solids		AD & use for transport	Incineration (CHP)	Incineration (CHP)
INC		x								Incineration (CHP)
MBT AC		x		x		OFMSW/SRF		AD & GE (CHP)	Incineration (CHP)	Incineration (CHP)
MBT DC		x		x		OFMSW/SRF		Composting	Incineration (CHP)	Incineration (CHP)
BLF		x								Bioreactor LF (El)
CLF		x								Flaring
WR GE	<i>I</i>	x	x	x	x	Bioliquid/Res. solids	AD & GE (CHP)	AD & GE (CHP)	Incineration (CHP)	Incineration (CHP)
WR TF		x	x	x	x	Bioliquid/Res. solids	AD & GE (CHP)	AD & use for transport	Incineration (CHP)	Incineration (CHP)
INC		x	x	x			AD & GE (CHP)			Incineration (CHP)
MBT AC		x	x	x		OFMSW/SRF	AD & GE (CHP)	AD & GE (CHP)	Incineration (CHP)	Incineration (CHP)
MBT DC		x	x	x		OFMSW/SRF	AD & GE (CHP)	Composting	Incineration (CHP)	Incineration (CHP)
BLF		x	x	x			AD & GE (CHP)			Bioreactor LF (El)
CLF		x	x	x			AD & GE (CHP)			Flaring
WR GE	<i>II</i>	x	x	x	x	Bioliquid/Res. solids	AD & GE (CHP)	AD & GE (CHP)	Incineration (CHP)	Incineration (CHP)
WR TF		x	x	x	x	Bioliquid/Res. solids	AD & GE (CHP)	AD & use for transport	Incineration (CHP)	Incineration (CHP)
INC		x	x	x			AD & GE (CHP)			Incineration (CHP)
MBT AC		x	x	x		OFMSW/SRF	AD & GE (CHP)	AD & GE (CHP)	Incineration (CHP)	Incineration (CHP)
MBT DC		x	x	x		OFMSW/SRF	AD & GE (CHP)	Composting	Incineration (CHP)	Incineration (CHP)
BLF		x	x	x			AD & GE (CHP)			Bioreactor LF (El)
CLF		x	x	x			AD & GE (CHP)			Flaring

<sup>a</sup> Intended as prior to biological or thermal energy conversion (which could be, for example, anaerobic digestion or incineration).

## 2. Identification of marginals

### 2.1 Marginal energy technologies

Special attention was devoted to assumptions regarding the surrounding energy system as choices here may significantly affect the outcome of the LCA (3-7). In a long term perspective (e.g., beyond 15 years), it may be assumed that energy from waste contributes to the decommissioning of fossil-based energy production capacities (both electricity and heat) as these technologies are generally intended to be phased out in order to comply with political CO<sub>2</sub> reduction targets. The waste management scenarios were therefore credited with the environmental savings induced by substitution of fossil fuel-based energy production; such system boundary expansion to include the benefits deriving from replacement of fossil energy represents a typical approach in consequential LCA (e.g., (5-7) among the others). At a European level, coal and natural gas represent the two ends of the range with respect to CO<sub>2</sub> emissions per combustion unit of fossil fuel energy (ca. 95 kg CO<sub>2</sub> GJ<sup>-1</sup> coal and 55 kg CO<sub>2</sub> GJ<sup>-1</sup> natural gas). These are also expected to be the fuels reacting to increased electricity production from waste and biomass. For example, in the case of Denmark (7, 8) identified coal as marginal electricity source; in the case of Italy, (9) identified natural gas as marginal. In (10) natural gas was identified as marginal for Spain. In this study it was assumed substitution of coal-based electricity production for the baseline calculation. This assumption was tested in the sensitivity analysis by substituting natural gas-based electricity production (S1).

As opposed to electricity, the market for heat is rather local and substitution of district heating or heating fuels often depends on local conditions and production capacities connected to the district heating network in question (4). This means that when evaluating a system in a short-term perspective involving existing production capacities, substitution of district heating should reflect local conditions. However, it is viable to assume that in the long-term heat production from waste will contribute to phasing-out fossil fuels. In the European Union, the heat market for Western (Austria, Denmark, Finland, France, Germany, Italy, Netherlands, Norway, Sweden, Switzerland) and Southern Europe (France, Greece, Italy, Portugal, Spain) is dominated by natural gas (ca. 45% in Western and 61% in Southern EU, relative to the total heat supply) (11, 12). In Eastern Europe (Czech Republic, Hungary, Slovak Republic, Poland) instead, the largest share of the heat supply is attributed to hard coal (ca. 55% of the total supply), followed by natural gas (ca. 22%) (11, 12). In this study, 3

possible scenarios of heat substitution were assessed to exemplify three potential (and most likely) situations that could occur in Europe (and elsewhere): i) heat produced from natural gas boiler (most representative for Western and Southern Europe), ii) heat produced from coal boiler (most representative for Eastern Europe), and iii) heat produced from coal-fired CHP plants (as an example of heat produced from CHP plants connected to an existing district heating network). District heating currently owns a low share of the EU heat market (ca. 6%). However, in the future, expansion of district heating is expected (13). In addition, most Nordic Countries have a well developed district heating network. This is, for example, the case for Denmark where about 62% of the total heat supply is provided as district heating (of this about 76% is delivered by CHP plants which main purpose is the production of electricity with heat being a co-product) (14). The marginal heat production from CHP plants was calculated following the approach of (4). Based on this, the total emissions from a representative Danish coal-fired CHP plants were allocated to electricity and heat based on the exergy content. This approach determines that ca. 90% of the emissions are attributed to electricity (main product) and the remaining to heat. The background LCI data for the marginal heat and electricity processes used in this study are reported in Table S4. The baseline results presented in the main manuscript consider substitution of heat produced from natural gas boilers. The results for the substitution of heat from coal boilers (namely S2) and of (coal-based) district heating (namely S3) are presented in Figure 2 of the manuscript. Also, an additional sensitivity analysis (namely S4) was performed to evaluate the environmental performance of the scenarios in the case of no-heat recovery.

For the transport fuel scenario (scenario *WR TF*), gasoline was assumed as marginal. This choice was tested in the sensitivity analysis (namely S5) by substituting diesel fuel.

**Table S4.** Background LCI emission data for marginal electricity and heat production used in this study (unit kg kWh<sup>-1</sup><sub>el</sub> or kg kWh<sup>-1</sup><sub>heat</sub>). Only selected emission data are reported.

Parameter	Coal electricity <sup>α</sup>	Natural gas electricity <sup>β</sup>	Natural gas boiler <sup>γ</sup>	Coal boiler <sup>δ</sup>	Coal District heating <sup>α</sup>
As	6.2E-09	3.3E-09	-	1.1E-07	1.9E-10
CO	2.1E-04	3.7E-05	1.2E-04	4.5E-04	1.7E-05
Cd	8.2E-10	6.1E-10	1.5E-10	7.1E-09	2.1E-11
CO <sub>2</sub>	0.91	0.45	0.22	0.41	0.17
Cu	8E-09	6.3E-09	2.4E-10	2.4E-10	2.4E-10
Cr	6.4E-09	9.6E-09	5.8E-10	1E-10	5.8E-10
Dioxins	2.1E-16	1E-016	1E-016	7.2E-014	-
Methane	5.3E-03	1.2E-05	7.8E-06	4.5E-05	3.4E-07
Hg	8.9E-09	6.1E-10	1.2E-10	1.3E-08	1.2E-09
NM VOC	8.5E-10	1.2E-05	1E-05	7.7E-06	3.4E-07
Ni	2.9E-08	1.6E-06	3.9E-12	8.5E-08	6.7E-10
N <sub>2</sub> O	8.9E-6	7.7E-06	2E-06	4.5E-06	1.4E-06
NO <sub>x</sub>	1.3E-03	3.7E-04	7.9E-05	0.0009	3.7E-04
Pb	9.2E-09	6.3E-09	2.3E-011	3.41E-10	4.2E-010
SO <sub>2</sub>	6.9E-04	4.3E-06	2E-06	2.3E-03	1.5E-05
Unspecified particles	1.4E-06	7.0E-07	7.2E-07	2.3E-04	6.0E-06

<sup>α</sup> Details on data and technologies can be found in (15).

<sup>β</sup> Based on a Danish natural gas-fired CHP plant (16).

<sup>γ</sup> Natural gas boiler with low NO<sub>x</sub> emissions (from Ecoinvent v2.2).

<sup>δ</sup> A process for coal combustion in industrial furnaces was used as proxy (from Ecoinvent v2.2).

## 2.2 Marginal mineral N, P, and K fertilizers

The compost produced from biological treatment of source-segregated organic waste and bioliquid was used as a fertilizer (for N, P, and K), which avoided marginal mineral N, P, and K fertilizers to be produced and used, based on the content of N, P, and K of the compost. The marginal N, P, and K fertilizers considered were calcium ammonium nitrate, diammonium phosphate, and potassium chloride, respectively, conformingly with (17, 18). Based on these, calcium ammonium nitrate is the N-fertilizer that is most likely to react to additional use of N organic fertilizer in a EU perspective. Diammonium phosphate and potassium chloride are those, in a EU and World perspective, expected to react to an additional use of P and K organic fertilizer, respectively: the first because current market trends show that new investments on P-fertilizers producing capacities are mainly for diammonium phosphate production, and the second because it currently accounts for ca. 95% of the all K fertilizers used in agriculture (19).

### 3. LCI of waste treatment technologies

This section describes the waste treatment technologies used for the assessment of the 252 waste management scenarios. Relevant technologies efficiencies have been forecasted to consider future development and optimization: this was done with respect to energy efficiency, selected air emissions, leachate and gas treatment at the landfill, use of enzymes and energy at the waste refinery, and biogas yield. Table S7 provides an overview of energy and material input and output to and from the technologies used in the assessment. Table S8 highlights future technologies development (as modelled in this study) compared with current typical operational efficiencies. Data are based on a number of literature sources.

#### 3.1 Pre-treatments

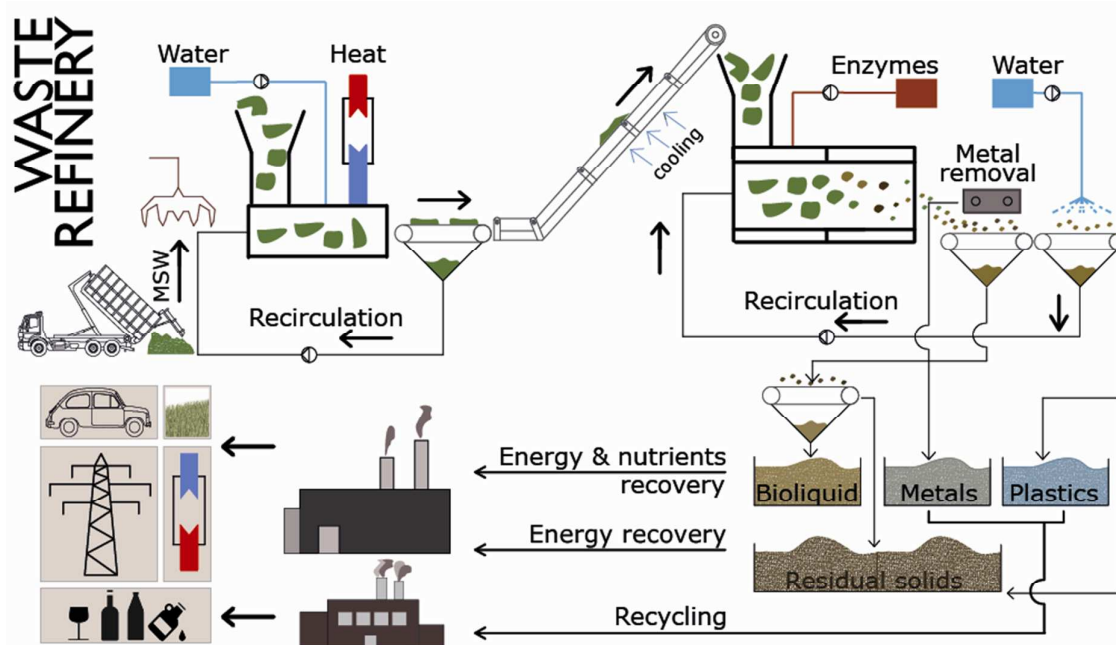
This section describes the waste pre-treatment technologies used in the assessment: i) waste refining, ii) mechanical-biological treatment (MBT), and iii) mechanical selection of source-segregated organic waste prior to biological treatment.

##### 3.1.1 Waste refinery

The waste refinery process was based on a pilot-scale facility established in Copenhagen, Denmark. The waste refinery had a treatment capacity of about 1 t MSW h<sup>-1</sup> (wet weight) and has been in operation from 2010.

The waste refinery (Figure S15) aims at producing two primary products from the incoming mixed MSW: i) a bioliquid (i.e., slurry composed of enzymatically liquefied organic, paper, and cardboard) and ii) a residual solid (i.e., non-degradable waste materials). The refinery process consisted of two reactors: in the first reactor the waste was heated with hot water to about 75 °C for approximately 0.5-1 hours, then cooled to about 50-55 °C before entering the second reactor. In the second reactor enzymes were added (about 5 kg t<sup>-1</sup> MSW) resulting in hydrolysis and break-down of bonds in the organic materials thereby essentially suspending organic materials in a liquid phase. The retention time was about 10-16 hour. A detailed description of the enzymatic processing can be found in (20). After the second reactor, the liquid phase was separated from the remaining solids by a vibrating sieve. Further, another vibrating sieve separated the liquid phase into a bioliquid and a solid “fluff” (phase containing materials such as cotton and textiles, but also glass pieces, plastics, etc.).





**Figure S15.** Illustration of the waste refinery process.

The bioliquid consisted primarily of suspended organic matter (food waste, paper, and cardboard), while the solid fraction mainly consisted of non-degradable materials such as plastic, metals, textiles, soil, ceramics, etc. The bioliquid can be exploited for biogas production (option considered in this study and described in section 3.3 dedicated to biological treatments), co-combusted in coal-fired power plant or utilized for producing ethanol. This, compared with direct incineration, provides additional flexibility to the energy system as the energy production could be regulated and storage possible in form of bioliquid/biogas. This is important in the perspective of energy systems having high penetration of wind energy and other fluctuating renewables as illustrated in previous studies (21-25). The solid fraction can be further treated to separate and recover valuable materials such as metals and plastic. The remaining residual solids (mainly non-recyclable plastic, textiles, yard waste, undegraded organics, glass pieces, and fluff) can be combusted for energy recovery. The amount of bioliquid and residual solids fraction produced in the process vary upon the composition of the incoming waste. To this regard, a list of transfer coefficients used to model the materials transfer to bioliquid and residual solid fraction can be found in Table S5; these values were experimentally determined (after (20)). Table S6 reports average chemical composition data for the bioliquid as experimentally determined. Overall, the electricity and heat consumption for the operations of heating, enzymatic treatment, and sieving was about  $20 \text{ kWh t}^{-1} \text{ MSW}$  and  $490 \text{ MJ t}^{-1} \text{ MSW}$ . These were based on process data

from the operations of the pilot-scale plant. For metals and plastic (HDPE, HDPP, and PET) separation additional, respectively, 4.5 (magnet and ECS) and 8.2 kWh t<sup>-1</sup> MSW (optical separation of PET and HDPE plus baling) were considered based on the operational data for mechanical recovery facilities provided in (26). Additional information about the process and its environmental performance relative to a Danish incinerator can be found in (27).

**Table S5.** Bioliquid from waste refining: transfer coefficients of selected waste material fractions. After (20).

Waste material fraction	TC (% of input)
Vegetable and animal waste	100%
Paper and cardboard	100%
Cartons and alike	85%
Animals litter	50%
Soil/ash	50%
Yard waste	35%
Aluminium foil	30%
Diapers/sanitary towel/tampons/bandages/cotton	15%

**Table S6.** Bioliquid from waste refining: chemical composition in kg kg<sup>-1</sup> DM (unpublished results from analyses carried out at a Danish pilot-scale refinery). The LHV<sub>db</sub> and CH<sub>4</sub> pot are expressed as MJ kg<sup>-1</sup> DM and NL CH<sub>4</sub> kg<sup>-1</sup> VS, respectively. Values are rounded to two significant digits.

C <sub>biog</sub>	H	S	Cl	N	P	K	Fe	Al	Cd	Cr
4.2E-01	5.1E-02	1.0E-03	1.1E-02	2.0E-02	2.5E-03	8.4E-03	5.1E-03	3.0E-03	1.8E-07	1.9E-05
Cu	Ni	Sr	Mn	As	Hg	Pb	Sb	O	LHV <sub>db</sub>	CH <sub>4</sub> pot
3.5E-05	1.2E-05	1.0E-04	8.0E-05	1.4E-06	1.0E-07	1.2E-05	1.4E-06	4.7E-01	16	440

### 3.1.2 Mechanical-biological treatment (MBT)

Two types of mechanical-biological plants were modelled: i) *MBT type AC* with anaerobic digestion and post-composting of the OFMSW (organic fraction of municipal solid waste) and ii) *MBT type DC* with direct composting of the OFMSW. These technologies (and therefore the related scenarios) differed for the biological treatment, while the mechanical pre-treatment was the same. The data were based on a full-scale MBT plant operated in Spain (28). The plant had four main outputs: ferrous metals, aluminium, OFMSW and Solid Recovered Fuel (SRF). The recovery of ferrous and non-ferrous was set to 85% and 90%, respectively, in order to have consistency across the assumptions about metals recovery in MBT and waste refinery, being the mechanical selection based on the same type of

technology (magnets and Eddy Current Systems, that is, ECS). The authors are aware of that other materials (e.g., paper, plastic, and glass) could be further recovered in MBT plants; however, this was not considered in the present as the largest share of recyclables was assumed source-segregated from the MSW with high efficiency (see Figure S1-S14); this justified a ‘simpler’ MBT technology for the treatment of the rMSW where the primary concern is energy recovery and stabilization of organic waste along with production of a high-calorific value fraction (SRF) for further energy recovery in highly-efficient incineration.

The mechanical treatment consisted of shredding (bags opening), trommel for separation of the organic fraction (OFMSW), and of separation of ferrous and non-ferrous metals by means of magnets and ECS, respectively. The transfer coefficients for organic, ferrous and non-ferrous metals corresponded to 93%, 85%, and 90%. The residual solid fraction (SRF) was assumed to be incinerated for energy recovery. The consumption of electricity and diesel for the mechanical operations corresponded to 15 kWh t<sup>-1</sup> MSW and 0.32 t t<sup>-1</sup> MSW. The associated biological treatments for the mechanically separated OFMSW are detailed in section 3.3.

### *3.1.3 Mechanical-selection of source-segregated organic waste*

Prior to anaerobic digestion, source-segregated organic waste generally needs mechanical pre-treatment in order to remove unwanted items (e.g. plastic, large wood pieces, packaging materials, and other misplaced items), to reduce particle sizes before the reactor thereby minimizing mechanical problems, to mix several substrates, to enhance hygienization, and adjust waste properties (29). This leads to losses of organic matter as a consequence of inherent efficiencies of the mechanical separation. Based on recent field investigations on a number of existing anaerobic digestion plants with associated pre-treatment (29), these losses might fall in the range 13-39% (as percent of DM of the incoming waste). Based on these results, it was assumed that 20% (as average value) of the source-segregated organic waste input to the anaerobic digestion plant was diverted to the reject fraction during the pre-treatment. This was also in accordance with the findings of (30). The reject was further routed to the ‘reference technology’ for residual waste treatment (that is, incineration for *INC*, landfill for *CLF* and *BLF*, and incineration for *MBT AC*, *MBT DC*, *WR GE* and *WR TF*).

Electricity and diesel fuel consumption were assumed as for mechanical selection at MBT plants (15 kWh t<sup>-1</sup> OFMSW and 0.32 t t<sup>-1</sup> MSW). The associated anaerobic digestion

process for the pre-treated source-segregated organic waste is described in the section dedicated to biological treatments (section 3.3).

### 3.2 Waste incineration

The waste incineration plant was modelled as a grate-fired incinerator equipped with wet flue gas cleaning, selective catalytic reduction (SCR) of NO<sub>x</sub>, Hg and dioxin removal by activated carbon. The SCR-process for NO<sub>x</sub> reduction (currently not a legal requirement in many EU Countries) determines energy use for heating the flue-gas and electricity losses due to increased pressure loss. This typically causes a reduction of about 1% of electricity generated and ca. 2.5% of total energy recovered (31). Gross electricity and heat efficiencies were therefore assumed to 25% and 75%, respectively, relative to the LHV<sub>wb</sub> of the waste input conformingly with expected performances for 2015-2030 (31) (see Table S8). The total gross efficiency added up to 100%, relative to the LHV<sub>wb</sub>, as the plant was assumed provided with flue-gas condensation. Based on target cleaning efficiencies, the materials and resources consumption for operations and flue-gas cleaning were estimated: internal electricity consumption at the plant was 86 kWh t<sup>-1</sup> MSW plus an additional 0.63 L t<sup>-1</sup> MSW of oil as auxiliary fuel, 0.19 kg NaOH t<sup>-1</sup> MSW, 0.8 kg NH<sub>3</sub> t<sup>-1</sup> MSW, 0.5 kg activated carbon t<sup>-1</sup> MSW and 11 kg CaCO<sub>3</sub> t<sup>-1</sup> MSW for flue-gas cleaning.

Following the approach of (32), emissions were divided into either process-specific emissions (emissions independent of waste composition but proportional to the amount of waste incinerated) or waste-specific emissions (determined by output transfer coefficients). The emission of NO<sub>x</sub> was assumed 0.13 g t<sup>-1</sup> MSW (15 g GJ<sup>-1</sup> waste input) as expected with SCR (typically < 30 g GJ<sup>-1</sup> (31)). The SO<sub>2</sub> emission equalled 0.013 kg t<sup>-1</sup> MSW which assumed a degree of desulphurisation higher than 98.5% (31). This is in line with typical SO<sub>2</sub> emissions for Danish incinerators (33). Selected air emissions for relevant heavy metals were (as % of input): 0.2% (As), 0.1% (Cd), 0.01% (Cr), 0.0018% (Cu), 0.004% (Mn), 0.125% (Ni) and 0.015% (Pb) conformingly with available operational data from a Danish incinerator (34). The distribution of Al between bottom and fly ash was ca. 90.5% and 9%, respectively. Similar transfer coefficients were reported for another Danish waste-to-energy plant (27). Notice that the transfer coefficient of Hg to air was set to 0.5% (% of Hg input): this assumes improved removal thanks to absorption in filters as indicated in previous studies focusing on the historical development of air pollution control (35). The current performances indicate higher transfer emissions (ca. 3.5-5% of the Hg input) (27, 34).

Ferrous and non-ferrous metals were recovered from bottom ashes (BA) prior to utilization as construction material substituting natural gravel (36). The recovery efficiency was set to 85% for ferrous metals and 70% for non-ferrous (here assumed as aluminium). These values have to be considered as typical for state-of-the-art European recovery technologies based on the findings of a recent review (37). The electricity and diesel consumption for the operations equalled  $1.3 \text{ kWh t}^{-1} \text{ BA}$  and  $1.1 \text{ L t}^{-1} \text{ BA}$ . Fly ashes were assumed to be utilized in the backfilling of old salt mines (38).

### **3.3 Biological treatment**

#### *3.3.1 Biological treatment of source-segregated organic waste*

The source-segregated organic waste was digested in a one-stage mesophilic anaerobic digestion plant appropriate for OFMSW (39).

The methane yield was set to 75% of the potential in agreement with similar practice for organic waste (40-42) yielding about  $290\text{-}320 \text{ Nm}^3 \text{ CH}_4 \text{ t}^{-1} \text{ VS}$  (depending on the waste composition). The methane content in the biogas was assumed to be 60% (v/v). Internal energy consumption at the plant was: diesel ( $0.9 \text{ kg t}^{-1} \text{ OFMSW}$ ), electricity ( $40 \text{ kWh t}^{-1} \text{ OFMSW}$ ) and heat ( $100\text{-}110 \text{ MJ t}^{-1} \text{ OFMSW}$ ). The electricity was due to: digestion operations ( $18 \text{ kWh t}^{-1} \text{ OFMSW}$  for pumping, ventilation, etc.), dewatering by pressing (about  $7 \text{ kWh t}^{-1} \text{ OFMSW}$ ), and post-composting and sieving ( $15 \text{ kWh t}^{-1} \text{ OFMSW}$ ). These data were based on (42, 43). The amount of heat required was calculated based on the energy required to heat up the substrate (about 35-40% DM, depending on the considered waste composition) from  $8 \text{ }^\circ\text{C}$  to  $37 \text{ }^\circ\text{C}$  and to maintain constant temperature ( $37 \text{ }^\circ\text{C}$ ) in the reactor. The fugitive emissions of  $\text{CH}_4$  were assumed 1% of the methane produced in accordance with recent LCA studies (17, 43, 44).

The biogas was assumed combusted in a natural gas engine with electricity and heat efficiency of 40% and 60% relative to the LHV of the biogas. Similarly to incineration, the total efficiency added up to 100% as the plant was assumed provided with flue-gas condensation. This was done in order to have consistency regarding the assumptions about heat recovery. In fact, although not yet established, flue-gas condensation can be applied to natural gas engines achieving energy recoveries greater than 100% (relative to the LHV of the biogas) as for incineration (31). Yet, it has to be noted that this represents an upper limit as typically these installations have smaller capacity (5-8 MW per unit) than incinerators (typically larger than 100 MW) and this is an inherent limitation for the implementation of

expensive flue-gas condensation systems. Air emissions from biogas combustion in natural gas engines were assumed according to (33).

The digestate was post-composted in aerated tunnels provided with biofilters for exhaust gas cleaning; the degradation of the organic matter during post-composting was assumed between 15% (organic waste, e.g., vegetable, animal waste, animals litter, etc.) and 20% (paper, cardboard, cotton, textiles, wood, etc.). The fugitive emissions of CH<sub>4</sub> occurring during the process were assumed 0.2% of the C degraded, and N<sub>2</sub>O emissions to 1.4% of the N degraded. About 98.5% of the degraded N was in the form of NH<sub>3</sub> of which 99% was oxidized in biofilters in line with (39) for a similar composting technology. A final sieving of the unrefined compost was also modelled; this implied that unwanted materials such as diapers, cotton, wood, textiles and undegraded paper were rejected and sent to residual waste treatment (see Figures S1-S14). This constituted ca. 5% of the unrefined compost mass. The DM content of the refined compost was set to 64% conformingly with (39).

### 3.3.2 Biological treatment of OFMSW (MBT with anaerobic digestion and post-composting)

For MBT with anaerobic digestion and post-composting (*MBT AC*) anaerobic digestion of the OFMSW was modelled based on the data for a MBT plant operating in Spain (28). Accordingly, the methane yield was 55% of the potential of the incoming waste. The methane content in the biogas was 60% (v/v). The fugitive methane emissions from the reactor were set to 1% of the produced CH<sub>4</sub> based on recent LCA studies (17, 43, 44). The energy consumption was calculated as for source-segregated organic waste digestion and corresponded to: electricity (25 kWh t<sup>-1</sup> OFMSW for digestion operations and dewatering), heat (100-110 MJ t<sup>-1</sup> OFMSW, depending on water content) and diesel (0.9 kg t<sup>-1</sup> OFMSW).

The biogas was assumed combusted in a natural gas engine with electricity and heat efficiency of 40% and 60% relative to the LHV of the biogas. Similarly to incineration, the total efficiency added up to 100% as the plant was provided with flue-gas condensation. Air emissions from biogas combustion in natural gas engines were assumed according to (33).

After 15-20 days of digestion, the digestate was introduced to post-composting. This consisted of forced aeration in tunnels (15-20 days) followed by maturation in piles (about 2 months). The degradation of each material fraction was modelled as a percentage of the VS content of the incoming waste; this corresponded to about 35-40% for paper and organic materials, respectively. The electricity consumption was estimated to 15 kWh t<sup>-1</sup> OFMSW (adapted from (39)). The air emissions were modelled as for *MBT type DC* (see later

description). The rejects from the process were assumed incinerated. The compost was assumed landfilled according to current practices.

### *3.3.3 Biological treatment of OFMSW (MBT with direct composting)*

For MBT with direct composting of the OFMSW (*MBT DC*), the biological treatment was modelled as aerated tunnels provided with biofilters for exhaust gas cleaning conformingly with (28). The degradation of each material fraction was modelled as a percent of the VS content in the incoming waste; this corresponded to about 70% VS degradation for organic waste such as animal and vegetable food waste, 60% for yard waste, 10-20% for paper and cardboard fractions, and 5% for beverage cartons and textiles. The electricity consumption was 40 kWh t<sup>-1</sup> OFMSW similarly to source-segregated organic waste. The main emissions were CH<sub>4</sub>, N<sub>2</sub>O, and NH<sub>3</sub>. Conformingly with (28, 39), the fugitive CH<sub>4</sub> emissions were set to 0.2% of the degraded C, and N<sub>2</sub>O emissions to 1.4% of the degraded N. About 98.5% of the degraded N was in the form of NH<sub>3</sub> of which 99% was oxidized in biofilters (39) which are the current air treatment system at the considered plants. The rejects from the process were assumed incinerated. The compost was assumed landfilled according to current practices.

### *3.3.4 Biological treatment of the bioliquid from waste refinery*

For bioliquid from waste refining, the methane yield was set to 82% of the methane potential based on the results of tests conducted in a full-scale biogas plant (unpublished results) yielding about 365 Nm<sup>3</sup> CH<sub>4</sub> t<sup>-1</sup> VS. The methane content in the biogas was assumed to be 60% (v/v). Internal energy consumption at the plant was: diesel (0.9 kg t<sup>-1</sup> bioliquid), electricity (40 kWh t<sup>-1</sup> bioliquid) and heat (ca. 120 MJ t<sup>-1</sup> bioliquid). The amount of heat required was calculated based on the energy required to heat up the substrate (20% DM) from 8 °C to 37 °C. The fugitive emissions of CH<sub>4</sub> were assumed 1% of the methane produced in accordance with recent LCA studies (17, 43, 44).

The biogas was assumed combusted in a natural gas engine with electricity and heat efficiency of 40% and 60% relative to the LHV of the biogas. Similarly to incineration, the total efficiency added up to 100% as the plant was assumed provided with flue-gas condensation. Air emissions from biogas combustion in natural gas engines were assumed according to (33).

Similarly to the treatment of source-segregated organic waste, the digestate was post-composted in aerated tunnels provided with biofilters for exhaust gas cleaning. Emissions and process efficiencies were assumed accordingly, except that no rejects were considered from the process.

### 3.4 Landfilling in bioreactor

The inputs to the bioreactor landfill were electricity ( $10 \text{ kWh t}^{-1} \text{ MSW}$ ), diesel oil for vehicles operating on site ( $2 \text{ L t}^{-1} \text{ MSW}$ ), soil and clay (in total,  $0.5 \text{ t t}^{-1} \text{ MSW}$ ). The soil and clay were assumed to be transported 30 km on average. The landfill had an average filling depth of 20 m and a bulk waste density of  $1.2 \text{ t m}^{-3} \text{ MSW}$ .

The generated leachate was recirculated to the waste mass for a time period of 8 years, in order to optimize the waste degradation process and increase the rate of methane generation. Tap water was also added in order to achieve an amount of recirculated water equal to 6750 mm within 8 years. During this period, 65% of the overall methane potential was generated and 90% of this collected. The collected gas was extensively (90%) used for electricity production in a natural gas engine with an efficiency of 40% relative to the LHV of the biogas. Gas collection continued for 10 years after the end of the leachate recirculation period. During this period, 20% of the overall methane potential was generated and 90% of this collected. The collected gas was used (70%) for electricity production with the same efficiency as earlier. The collected gas which was not used for electricity generation was instead flared. The uncollected gas fraction received partial oxidation in the top soil cover. Oxidation efficiencies were specified for each gas constituent for all time periods. With respect to methane oxidation, efficiencies ranged from 40% to 80% depending on the actual flow-rate of gas through the cover: 40% was used until the end of the leachate recirculation period and 80% oxidation afterwards.

During the first 2 years of filling operations the leachate generation was set to  $450 \text{ mm y}^{-1}$ . After the first 2 years an impermeable surface liner was applied as top cover. During the following 13 years the leachate generation in the model was set to  $450 \text{ mm y}^{-1}$ : in reality this is equivalent to 8 years of leachate recirculation (6,750 mm including household water addition) followed by 5 years of draining operations. This way of modeling did not reflect how the water actually came out (this is not really important as the emissions in the modelling are integrated over 100 years) but it reflected that all the added water was supposed to come out before 15 years after the start of the landfill. After the end of the draining period, the leachate generation was set to  $20 \text{ mm y}^{-1}$ , due to the impermeable top sealing. The leachate collection system was assumed to capture 95% of the leachate generated for the first 40 years of the landfill life time. Thereafter, since only  $20 \text{ mm y}^{-1}$  of leachate was generated, the leachate collection efficiency was set to 0% and all the uncollected leachate migrated to the groundwater. The collected leachate was treated in a leachate treatment plant for purification. Treated leachate was discharged to surface water



bodies. Natural attenuation of leachate moving towards the groundwater has been disregarded for all leachate constituents except ammonia. It was assumed that half of the ammonia was converted into nitrates, as the redox conditions in the subsurface may be favourable to oxidation.

### **3.5 Conventional landfilling**

Conventional landfilling was modelled as an up-to-date conventional landfill with engineered measures to prevent emissions of gas and leachate to the environment. These included bottom liner, leachate collection system and leachate treatment prior to discharge of treated leachate to surface water bodies, and top soil cover, gas collection system and flaring. Also the provision of energy and materials to site and on-site operations were included. Data were adapted from (45).

The inputs to the landfill were electricity ( $8 \text{ kWh t}^{-1} \text{ MSW}$ ), diesel oil for vehicles operating on site ( $2 \text{ L t}^{-1} \text{ MSW}$ ), soil and clay (in total,  $0.5 \text{ t t}^{-1} \text{ MSW}$ ). The soil and clay were assumed to be transported 30 km on average. The landfill had an average filling depth of 20 m and a bulk waste density of  $1 \text{ t m}^{-3} \text{ MSW}$ .

The filling phase was divided into two parts. In the first part (years 1-2) no gas collection was practiced, while in the second part (years 3-5) gas was extracted. It has been assumed that gas collection and gas treatment was active until year 100. Gas generation and collection were modelled according to 4 time periods: during the first 2 years of sector filling, 1% of the overall methane potential was generated and no-collection was practiced. During the following 3 years of sector filling, 4% of the overall methane potential was generated and 50% of this was collected. This inherently assumed that from year-3 gas wells were constructed and operated as the landfill body grew higher. The final soil top cover was constructed at the end of the filling phase (year-5). The final cover did not include a surface liner. However, assuming a soil cover only is in compliance with EU regulations and represents a worst-case scenario allowing for leachate generation and migration of LFG through the cover. Within the following 25 years, 60% of the overall methane potential was generated and 50% of this was collected. During the remaining 70 years, 30% of the overall methane potential was generated and again 50% of this was collected. All the collected gas during 100 years was flared. The uncollected gas fraction received partial oxidation in the top soil cover. Oxidation efficiencies were specified for each gas constituent for all time periods. With respect to methane oxidation, efficiencies ranging from 40% to 80% were assumed depending on the actual flow-rate of gas through the cover.

During the first 2 years of filling operations the leachate generation was set to  $350 \text{ mm y}^{-1}$ . In the following 3 years of filling the leachate generation was set to  $300 \text{ mm y}^{-1}$ , assuming major water retention due to the increased waste mass. After the soil top cover construction (year-5) the leachate generation dropped to  $200 \text{ mm y}^{-1}$ . Afterward, from year 30 to year 100 the leachate generation was decreased to  $100 \text{ mm y}^{-1}$ , assuming an increased evapotranspiration due to vegetation growth on the top cover. Leachate collection and treatment were assumed operated until year 30. With respect to leachate collection, capture efficiencies were set to 95% during the first 30 years of the landfill life. Afterwards (from year 30 to year 100), an impermeable surface liner was constructed and the leachate collection stopped. The capture efficiency was then set to 0% and all the leachate generated was assumed to reach the groundwater.

The collected leachate was treated in a leachate treatment plant for purification. Treated leachate was discharged to surface water bodies. Natural attenuation of leachate moving towards the groundwater has been disregarded for all leachate constituents except ammonia. It was assumed that half of the ammonia is converted into nitrates, as the redox conditions in the subsurface may become favourable to oxidation.

**Table S7** Overview of energy and materials input and output to and from the main technologies used in the LCA. El: electricity; APC: air pollution control residue; MSW: municipal solid waste; OFMSW: organic fraction of MSW; Res. solids: Residual solids; rMSW: residual MSW; SRF: Solid recovered fuel; SS OW: source-segregated organic waste.

Technology		Input				Output					
		Materials		Energy		Materials/products			Energy		
		Waste materials	El (kWh t <sup>-1</sup> )	Heat (MJ t <sup>-1</sup> )	Fuel (L t <sup>-1</sup> )	Ferrous (% input)	Aluminium (% input)	Other outputs	El <sup>β</sup> (%)	Heat <sup>β</sup> (%)	Technology
Pre-treatment	Mechanical selection	rMSW, SS OW	15	-	0.32	85	90	OFMSW SRF	-	-	-
	Waste refining	rMSW	25-33 <sup>α</sup>	490	-	85	90	Bioliq Res. solids	-	-	-
Biological treatment	Anaerobic digestion of SS OW from source-segregation	SS OW	40	100-110	0.9	-	-	Biogas Compost	40	60 <sup>γ</sup>	Natural gas engine (CHP)
	Anaerobic digestion of OFMSW from MBT	OFMSW	40	100-110	0.9	-	-	Biogas Compost	40	60 <sup>γ</sup>	Natural gas engine (CHP)
	Direct composting of OFMSW from MBT	OFMSW	40	-	0.9	-	-	Compost	-	-	-
	Anaerobic digestion of bioliq from waste refinery	Bioliq	40	120	0.9	-	-	Biogas Compost	40	60 <sup>γ</sup>	Natural gas engine (CHP)
rMSW treatment	Incineration	rMSW, SRF, Res. solids	86	-	0.63 <sup>δ</sup>	85	70	Bottom ash APC	25	75 <sup>γ</sup>	Steam cycle (CHP)
	Conventional landfilling	rMSW	8	-	2	-	-	Biogas Leachate	-	-	-
	Bioreactor landfilling	rMSW	10	-	2	-	-	Biogas Leachate	40	-	Natural gas engine (El)

<sup>α</sup> The value 33 kWh includes plastic separation (ca. 8.2 kWh). <sup>β</sup> Gross efficiency, expressed as percent of LHV<sub>wb</sub> of the incoming waste (except for biogas for which the energy content is expressed as LHV<sub>db</sub>). <sup>γ</sup> Flue-gas condensation is assumed (total energy recovery 100%; thermal energy losses due to SCR are accounted for). <sup>δ</sup> Auxiliary fuel (fuel oil).

**Table S8.** Overview of future expected performances of the technologies used in the assessment (selected efficiencies and/or operational data). For the purpose of comparison, current (typical) gross efficiencies and/or operational data are reported based on different sources. The reported total efficiencies are to be considered at full-load (i.e., dismissing of surplus heat in cooling towers during summer is not considered).

Technology	Current (typical) performance (2012)	Expected future performance (2015-2030)	Note
Mechanical selection <sup>α</sup>	Ferrous metal recovery: 65-90% (46) Aluminium recovery: 60-90% (46)	Ferrous metal recovery: 85% Aluminium recovery: 90%	Optimized material recovery.
Waste refining	El consumption: 33-51 kWh t <sup>-1</sup> (27) Heat consumption: 590 MJ t <sup>-1</sup> (27) Enzymes consumption: 12-24 kg t <sup>-1</sup> (27)	El consumption: 25-31 kWh t <sup>-1</sup> Heat consumption: 490 MJ t <sup>-1</sup> Enzymes consumption: 5 kg t <sup>-1</sup>	Optimized energy and enzyme consumption (updated data from pilot-scale plant).
Biological treatment (anaerobic digestion and biogas combustion in a natural gas engine CHP)	Air emissions: CH <sub>4</sub> (fugitive): 0-3% of CH <sub>4</sub> produced (47, 48) CH <sub>4</sub> yield: 60-90% of potential (49) El efficiency: 34-42% (50) Total efficiency: 85-90% (31)	Air emissions: CH <sub>4</sub> (fugitive): 1% of CH <sub>4</sub> produced (17, 43, 44) CH <sub>4</sub> yield: 75% of potential (40-42) El efficiency: 40-50% (31) Total efficiency: 88-100% (31)	Improved insulation of the reactor. Optimized energy recovery by means of flue-gas condensation.
Biological treatment (direct and post-composting)	Air emissions: CH <sub>4</sub> : 0.8-13.5% of C degraded (51) <sup>β</sup> N <sub>2</sub> O: 0.1-1.8% of input N (51) <sup>β</sup>	Air emissions: CH <sub>4</sub> : 0.2% of C degraded (39) N <sub>2</sub> O: 1.4% of input N (39)	Post-composting is operated in aerated tunnels instead of outdoor piles. This allows for reducing CH <sub>4</sub> emissions as the exhaust air is oxidized in biofilters.
Incineration	El consumption: 67 kWh t <sup>-1</sup> (34) El efficiency: 14-21% (52) Total efficiency <sup>θ</sup> : up to 102% (31) Air emissions: Hg: 3.5-5% of Hg input (27, 34) NO <sub>x</sub> : 80-180 g GJ <sup>-1</sup> (input fuel) (31)	El consumption: 86 kWh t <sup>-1</sup> El efficiency: 25-30% (31) Total efficiency <sup>γ</sup> : 100% (31) Air emissions: Hg: 0.5% of Hg input (35) NO <sub>x</sub> : < 30 g GJ <sup>-1</sup> (input fuel) (31) <sup>γ</sup>	Optimized technology equipped with SCR, activated carbon system, and flue-gas condensation. This allows for increased efficiency for energy recovery and NO <sub>x</sub> and Hg removal.
Bioreactor landfilling	-	See section 3.4 (45)	State-of-the-art bioreactor landfill
Conventional landfilling	-	See section 3.5 (45)	State-of-the-art conventional landfill

<sup>α</sup> Mechanical selection is used in MBT prior to biological treatment and as pre-treatment for source-segregated organic waste prior to digestion. However, in the latter case ferrous metal and aluminium recovery were not modelled as these are typically disposed of in landfill since the materials are very dirty (53). <sup>β</sup> Review of a number of different studies.

<sup>γ</sup> From 2020 application of the SCR-process (selective catalytic reduction) is assumed for NO<sub>x</sub> reduction (currently not a legal requirement in many EU Countries). Energy use for heating the flue-gas and electricity losses due to increased pressure loss typically causes a reduction of about 1% of electricity generated and ca. 2.5% of total energy recovered (included in the forecasted values for those parameters). <sup>θ</sup> Depends upon heat demand and presence of a district heating network (the value 102% represents a potential maximum total recovery with flue-gas cleaning and with SNCR for NO<sub>x</sub> reduction for Danish conditions according to (31)).

### 3.6 Recycling

Ferrous metal recycling was assumed to substitute 98% virgin production (i.e., 2% material loss) with a market substitution ratio of 100% for the produced metal. Ferrous metal recycling included re-melting of scraps and rolling of new steel sheets from the melted metal waste, minus the avoided virgin production. The benefit of metal recycling was primarily savings in energy consumption, corresponding to a net saving of 1,700 kg CO<sub>2</sub>-eq. t<sup>-1</sup> ferrous metal input (when coal was the marginal energy in the recycling process).

Aluminium recycling was assumed to substitute 79% virgin production (i.e., 21% material loss) with a market substitution ratio of 100% for the produced aluminium. An overall material loss of 21% was assumed due to the sorting process. Aluminium recycling included re-melting of aluminium scrap and alloying, minus the avoided virgin production. The benefit of aluminium recycling was primarily savings in energy consumption, corresponding to a net saving of 7,700 kg CO<sub>2</sub>-eq. t<sup>-1</sup> aluminium input (when coal was the marginal energy in the recycling process).

Plastic recycling was assumed to substitute 90% virgin production (i.e., 10% material loss) by re-melting with a market substitution ratio of 90%. The latter value was a rough estimate of the potential decrease in material quality. According to (54), the loss of material quality can be as high as 20%. However, in (54) it is also stated that the loss highly depends on the field of application of the secondary plastic. For instance, in the case that the recovered plastic is utilized as an admixture in the production of primary plastic there may be no loss. Thus, in this study, 10% loss in material quality was assumed. Plastic recycling included the granulation and re-melting for production of polyethylene (PE) plastic from waste plastic minus the avoided virgin production. The benefit of plastic recycling was primarily savings in energy consumption, corresponding to a net saving of 810 kg CO<sub>2</sub>-eq. t<sup>-1</sup> plastic input (when coal was the marginal energy in the recycling process).

The recycling dataset used for paper reflected European paper recycling processes around year 2000, although not necessarily statistically based. Newspaper and magazines were recycled into newspaper. The technical substitution was 82% (i.e., 18% material loss) and the market substitution 100% (54). The paper recycling dataset included the paper production from waste paper minus the avoided virgin production. The benefit of paper recycling was primarily savings in energy consumption, corresponding to a net saving of 1,800 kg CO<sub>2</sub>-eq. t<sup>-1</sup> paper input (when coal was the marginal energy in the recycling process).

The recycling dataset used for cardboard represented shredding and reprocessing of mixed cardboard and paper materials into cardboard cores and tubes, based on the process operated at *Skjern Paprifabrik*, in Denmark (55). The technical substitution was 100% (55) and the market substitution was assumed 100%, as for paper (54). The cardboard recycling dataset included the cardboard production from waste cardboard minus the avoided virgin production. The benefit of cardboard recycling was primarily savings in energy consumption, corresponding to a net saving of 1,200 kg CO<sub>2</sub>-eq t<sup>-1</sup> cardboard input (when coal was the marginal energy in the recycling process).

The recycling dataset used for glass represented European glass processes around year 1990, although not necessarily statistically based. It was assumed that cullet technically substituted 99% virgin production (i.e., 1% material loss) by remelting and that the market substitution ratio was 100%. In practice the mixing of cullet and virgin resources for glass production may use a different ratio. The glass recycling dataset included the glass production from cullet minus the avoided virgin production. Any saving in providing virgin resources for glass production was not included. The benefit of glass recycling was primarily savings in energy spending corresponding to a net saving of 260 kg CO<sub>2</sub>-eq. t<sup>-1</sup> glass input (when coal was the marginal energy in the recycling process).

All inventory data for the recycling processes were based on the EDIP database. Additional information on the GHG accounting for these processes can be found in (56-59).

### **3.7 Use-on-land**

The amount and composition of the compost derived from anaerobic digestion of the source-segregated organic waste and of the bioliquid was calculated based on a mass balance approach, i.e., as the difference between the initial nutrients and dry matter fed to the digestion process and the amount transferred to biogas and lost during the post-composting process. The compost was assumed to be applied on land substituting mineral fertilizers following the approach of (60).

In order to quantify the amount of mineral fertilizers replaced it is necessary to assume a type of soil and a specific legislative context as the application of organic fertilizers is typically regulated differently in each Country. In this study the Danish legislation for application on land of organic fertilizers was assumed as reference for the modelling (61). According to that, the amount of N that can be brought into the field is limited, so that the N cannot be applied in excess. However, not all the N applied translates into avoided mineral fertilizer, as the regulation considered an efficiency of only 20% for compost, i.e. a

substitution rate of 20%. The substitution of P and K was assumed 100% (that is, all P and K applied substituted corresponding amount of mineral fertilizers) conformingly with the approach of (60). This is in accordance with similar studies (e.g. (39)). The distribution of N into  $\text{NH}_3$ ,  $\text{NO}_3^-$ , and organic N was set to 13%, 0.2%, and 86.8% conformingly with (60). The air emission of  $\text{NH}_3$  and  $\text{N}_2\text{O}$  were assumed to 0.21% (i.e., 1.6% of the  $\text{NH}_4^+$  content of the compost) and 1.5%, respectively, of the N applied conformingly with (60, 62). The emission of  $\text{NO}_3^-$  to water bodies was set to 20% of the applied N according to the findings of (62). It should be noted that N-leaching varied according to soil type, livestock density, and organic waste treatment and was found within a range of 3%-87% of the N added with the waste (61). Based on the results from the same study, the C bound in soil (over the considered 100 years LCA horizon) was assumed 14% of the C initially applied with the compost.

### **3.8 Collection and transportation**

Waste collection was modelled as fuel consumption per tonne of wet waste. The specific fuel consumption has been determined by direct measurements on a large range of Danish collection routes (63). These were ( $\text{L t}^{-1} \text{ ww}$ ): 3 for residual waste, 4 for source-segregated paper, 6 for source-segregated glass, 8 for source-segregated plastic and 6 for source-segregated organic waste.

The fuel consumption for transport was represented by “transportation” to the point of unloading, e.g., at the treatment or disposal facilities. The fuel consumption was expressed in fuel consumption per tonne of waste per km (one-way distance) according to previous studies (63, 64). Transport distances were: 15 km for the residual waste sent to treatment; 40 km for stabilized organic (compost from MBT) sent to landfilling; 25 km for compost from anaerobic digestion of source-segregated organic waste and bioliquid (sent to use-on-land); 70 km for bottom ash (including post-treatment and reuse); 500 km for fly ash (sent to backfilling of old salt mines). As recyclables generally enter a global market, transport distances were unknown in this case but average European values were used for approximation (64): 100 km for plastic, glass and paper, and 500 km for aluminium and ferrous metals.

### **3.9 Other processes**

With respect to incineration residues, bottom ashes from incineration were assumed utilized as construction material substituting natural gravel (36). APC residues were assumed to be utilized in the backfilling of old mines following the approach of (38).

The upgrading of biogas to methane (96% v/v, corresponding to a LHV of 35 MJ Nm<sup>-3</sup>) was modelled based on (44): The electricity use was reported to 0.5 kWh Nm<sup>-3</sup> CH<sub>4</sub> (0.014 kWh MJ<sup>-1</sup> CH<sub>4</sub>). The main direct air emissions were CO (1.5E-04 kg CO MJ<sup>-1</sup> CH<sub>4</sub>), CH<sub>4</sub> (1.5E-05 kg CH<sub>4</sub> MJ<sup>-1</sup> CH<sub>4</sub>), NO<sub>x</sub> (6.7E-06 kg NO<sub>x</sub> MJ<sup>-1</sup> CH<sub>4</sub>), and SO<sub>2</sub> (2.9E-07 kg SO<sub>2</sub> MJ<sup>-1</sup> CH<sub>4</sub>). Overall, the upgrading process equalled a GHG emission of 0.031 kg CO<sub>2</sub>-eq. MJ<sup>-1</sup> CH<sub>4</sub> (produced).



#### 4. Waste materials and energy balance

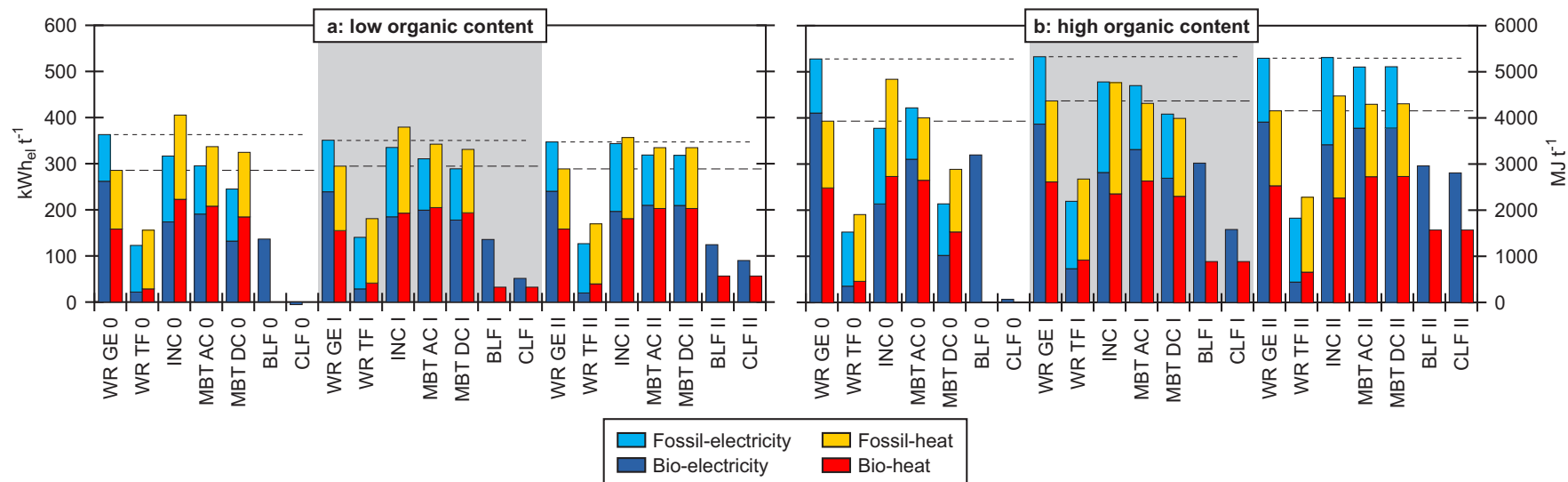
The results of a waste material balance of the assessed waste management scenarios is reported in Table S9. The waste material balance reports the amount of the individual flows of waste material fractions across the investigated scenarios. As reported in Table S9, the amount of source-segregated recyclable materials was the same for all the scenarios assessed (but differed between *a* and *b*). The amount of source-segregated organic waste varied from 0 (*0*) over 70% (*I*) to 100% (*II*). As a result, the amount of rMSW equalled 690, 430, and 310 kg ww for scenarios *0*, *I*, and *II* for the case of waste composition *a*, and 830, 420, 240 kg ww for scenario *0*, *I*, and *II* for the case of waste composition *b*.

Figure S16 illustrates the total amount of electricity and heat recovered in the investigated scenarios. This is further distinguished between bioenergy (from organic, papers, and other biogenic materials) and fossil energy.

**Table S9.** Waste material fractions balance of the individual scenarios assessed (unit: kg ww). Organic, PA (paper), CB (cardboard), Plastic, Glass, AL (aluminium), and FE (ferrous metal) refer to the source-segregated amount of these waste material fractions; ssMSW: source-segregated MSW; rMSW: residual MSW. Eventual inconsistencies are due to rounding (values rounded to two significant digits).

	Scenarios <i>I</i>		Scenarios <i>I</i>		Scenarios <i>II</i>	
	WR GE, WR TF, INC, MBT AC, MBT DC, BLF, CLF		WR GE, WR TF, INC, MBT AC, MBT DC, BLF, CLF		WR GE, WR TF, INC, MBT AC, MBT DC, BLF, CLF	
	<i>a</i>	<i>b</i>	<i>a</i>	<i>b</i>	<i>a</i>	<i>b</i>
MSW	1,000	1,000	1,000	1,000	1,000	1,000
Organic	0	0	260	410	370	590
PA <sup><i>α</i></sup>	180	66	180	66	180	66
CB <sup><i>β</i></sup>	38	22	38	22	38	22
Plastic <sup><i>γ</i></sup>	8.8	39	8.8	39	8.8	39
Glass <sup><i>θ</i></sup>	66	18	66	18	66	18
AL	5.7	3.8	5.7	3.8	5.7	3.8
FE	12	19	12	19	12	19
<b>ssMSW</b>	<b>310</b>	<b>170</b>	<b>570</b>	<b>580</b>	<b>680</b>	<b>760</b>
<b>rMSW</b>	<b>690</b>	<b>830</b>	<b>430</b>	<b>420</b>	<b>320</b>	<b>240</b>

<sup>*α*</sup> Does not include dirty paper which is routed to rMSW. <sup>*β*</sup> Does not include dirty cardboard which is routed to rMSW. <sup>*γ*</sup> Does not include non-recyclable plastic which is routed to rMSW. <sup>*θ*</sup> Does not include non-recyclable glass which is routed to rMSW.



**Figure S16.** Energy balance: net electricity and heat production in the investigated scenarios. Electricity production refers to the left axis ( $0\text{--}600 \text{ kWh}_{\text{el}} \text{t}^{-1}$ ), while heat production refers to the right axis ( $0\text{--}6000 \text{ MJ t}^{-1}$ ). The energy produced is distinguished between bio- and fossil-based. The dotted lines represent the net electricity and heat produced in the waste refining CHP scenarios (*WR GE*), here used as comparative reference.

## 5. Sensitivity analysis: parameter uncertainty (waste refinery vs. incineration)

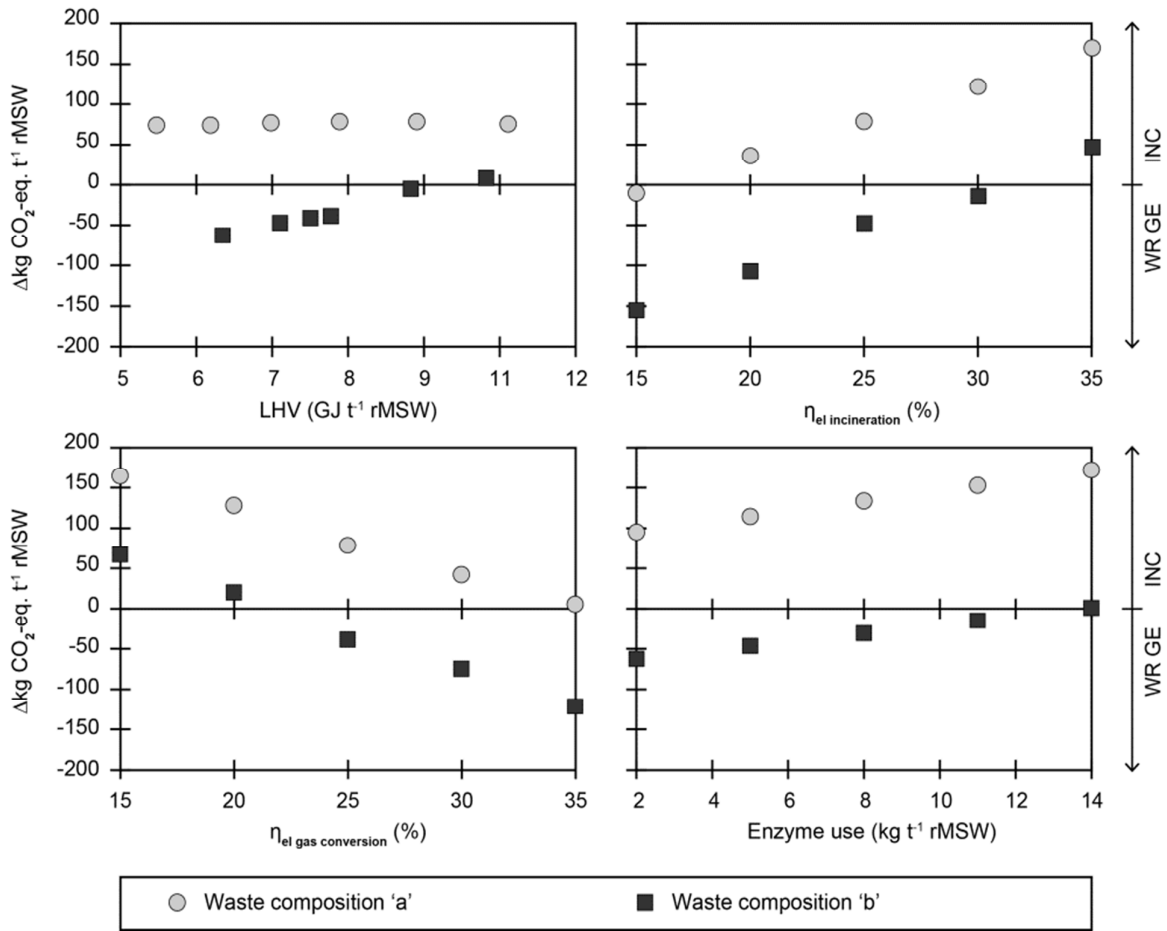
This section presents the results of a number of sensitivity analyses where the GW performance of two selected scenarios, namely *INC 0* and *WR GE 0*, was compared by individually varying relevant parameters; these were: (i) LHV (i.e., obtained by varying the share of organic waste in the initial MSW), (ii) electricity efficiency of incineration, (iii) electricity efficiency of biogas-to-electricity conversion and (iv) enzymes consumption. The objective was to identify the conditions (with respect to waste composition and technology efficiency) when one waste management scenario allowed for greater GHG savings than the other (i.e., thresholds or break-even points). The results are displayed in Figure S17 for both waste compositions *a* and *b*. The results are expressed as net difference ( $\Delta$ ) between the GW performance of *WR GE 0* and *INC 0*. Therefore, when the indicator is below zero *WR GE 0* performs better than *INC 0*, and viceversa.

With respect to (i) the results were totally dependent upon the selected waste composition: with waste composition *a* incineration was always better than waste refining. This was also true for very low LHV; the reason for this lied in the waste composition as determined by (1): an increase on the share of organic waste determined a decrease of LHV and thus a diminished energy recovery of incineration. However, also the performance of the waste refining scenario suffered as the amount of bioliquid would decrease proportionally to the decrease in the share of paper materials (due to the increased share of organic waste) which dry matter content was significantly higher than that of organic waste. In the case of waste composition *b*, instead, the results were opposite and only for a LHV above  $9.5 \text{ GJ t}^{-1}$  MSW incineration performed better than waste refining. In (2), in fact, the measured water content for the organic waste was significantly lower; this benefitted the waste refinery scenario when increasing the share of organic waste (i.e., relatively more dry matter was transferred into the bioliquid compared with the case of *a*).

The result for (ii) again highlighted the importance of the waste composition: in the case of *a* waste refining performed better than incineration only when the electricity efficiency was lower than 15%. In the case of *b*, instead, this occurred always until exceeding 32% efficiency at the incinerator.

Similar results were found for (iii): in the case of *a* waste refining achieved a better performance than incineration only for biogas-to-electricity efficiency greater than 50%. In the case of *b* 36% was instead sufficient.

Lastly, the analysis (iv) revealed that in the case of *a* any decrease in enzymes consumption would not change the ranking between the scenarios. As opposite to this, in the case of *b* waste refining became worse than incineration only for consumptions greater than 15 kg t<sup>-1</sup> MSW.



**Figure S17.** Individual comparison between incineration (*INC*) and refining with CHP (*WR GE*) for the scenario without organic waste source-segregation (*0*). Four selected parameters (LHV, electricity efficiency of incineration, efficiency of biogas-to-electricity conversion, enzyme use) were varied and the corresponding GHG performance of the two scenarios was quantified. The results are expressed as net difference ( $\Delta$ ) between the GW performance of *WR GE 0* and *INC 0*. Therefore, when the indicator is below zero *WR GE 0* performs better than *INC 0*, and viceversa.

## References

1. Riber, C.; Petersen, C.; Christensen, T.H. Chemical composition of material fractions in Danish household waste. *Waste Manage.* **2009**, *29*, 1251-1257.
2. Montejo, C.; Costa, C.; Ramos, P.; Marquez, M.C. Analysis and comparison of municipal solid waste and reject fraction as fuels for incineration plants. *Appl. Therm. Eng.* **2011**, *31*, 2135-2140.
3. Fruergaard, T.; Astrup, T.; Ekvall, T. Energy use and recovery in waste management and implications for accounting of greenhouse gases and global warming contributions. *Waste Manage. Res.* **2009**, *27*, 724-737.
4. Fruergaard, T.; Christensen, T.H.; Astrup, T. Energy recovery from waste incineration: Assessing the importance of district heating networks. *Waste Manage.* **2010**, *30*, 1264-1272.
5. Finnveden, G.A.; Hauschild, M.Z.; Ekvall, T.; Guinee J.B., J.; Heijungs, R.; Hellweg, S.; Koehler, A.; Pennington, D.; Suh, S. Recent developments in Life Cycle Assessment. *J. Environ. Manage.* **2009**, *91*, 1-21.
6. Ekvall, T. and Weidema, B.P. System boundaries and input data in consequential life cycle inventory analysis. *Int. J. Life Cycle Assess.* **2004**, *9*, 161-171.
7. Weidema, B.; Frees, N.; Nielsen, A.M. Marginal production technologies for Life Cycle Inventories. *Int. J. Life Cycle Assess.* **1999**, *4*, 48-56.
8. *Market Information in Life Cycle Assessment*, Environmental Project No. 863; Danish Ministry of the Environment, Environmental Protection Agency: Copenhagen, Denmark, 2003; [www2.mst.dk/udgiv/publications/2003/87-7972-991-6/pdf/87-7972-992-4.pdf](http://www2.mst.dk/udgiv/publications/2003/87-7972-991-6/pdf/87-7972-992-4.pdf) (accessed January 30, 2013).
9. Turconi, R.; Butera, S.; Boldrin, A.; Grosso, M.; Rigamonti, L.; Astrup, T. Life cycle assessment of waste incineration in Denmark and Italy using two LCA models. *Waste Manage. Res.* **2011**, *29*, 78-90.
10. Meneses, M.; Pasqualino, J.; Castells, F. Environmental assessment of the milk life cycle: The effect of packaging selection and the variability of milk production data. *J. Environ. Manage.* **2012**, *107*, 76-83.
11. *Electricity information 2010*; International Energy Agency (IEA): Paris, France, 2010.
12. *Europe in figures - Eurostat yearbook 2011*. Eurostat: Luxembourg, European Union, 2011; [http://epp.eurostat.ec.europa.eu/portal/page/portal/publications/eurostat\\_yearbook\\_2011](http://epp.eurostat.ec.europa.eu/portal/page/portal/publications/eurostat_yearbook_2011) (accessed January 30, 2013).
13. *The European heat market*; Euroheat & Power: Brussels, Belgium, 2006; [http://www.euroheat.org/Files/Filer/eoheatcool/documents/Ecoheatcool\\_WP1\\_Web.pdf](http://www.euroheat.org/Files/Filer/eoheatcool/documents/Ecoheatcool_WP1_Web.pdf) (accessed January 30, 2013).
14. *Energy statistics 2011*; Danish Energy Agency (DEA): Copenhagen, Denmark, 2012; [http://www.ens.dk/en-US/Info/FactsAndFigures/Energy\\_statistics\\_and\\_indicators/Annual%20Statistics/Documents/Energy%20Statistics%202011.pdf](http://www.ens.dk/en-US/Info/FactsAndFigures/Energy_statistics_and_indicators/Annual%20Statistics/Documents/Energy%20Statistics%202011.pdf) (accessed January 30, 2013).

15. *Miljøevurdering af affaldsforbraending og alternative*; DTU Environment, Department of environmental engineering, Technical University of Denmark: Lyngby, Denmark, 2008; <http://www.affalddanmark.dk/docs/udgivelser/Miljovurdering.pdf> (accessed January 30, 2013).
16. *H.C. Oersted Vaerket Miljoergnskab 2010* (H.C. Oersted Plant Green Account 2010); DONG energy: Copenhagen, Denmark, 2011; [http://www.dongenergy.com/SiteCollectionDocuments/business\\_activities/generation/Miljoeregnskaber\\_2010/HC\\_Oerstedvaerket\\_2010.pdf](http://www.dongenergy.com/SiteCollectionDocuments/business_activities/generation/Miljoeregnskaber_2010/HC_Oerstedvaerket_2010.pdf) (accessed January 30, 2013).
17. Hamelin, L.; Wesnaes, M.; Wenzel, H.; Petersen, B.M. Environmental Consequences of Future Biogas Technologies Based on Separated Slurry. *Environ. Sci. Technol.* **2011**, *45*, 5869-5877.
18. Hamelin, L.; Joergensen, U.; Petersen, B.M.; Olesen, J.E.; Wenzel, H. Modelling the carbon and nitrogen balances of direct land use changes from energy crops in Denmark: a consequential life cycle inventory. *GCB Bioenergy* **2012**, *4*(6), 889-907.
19. *Current world fertilizer trends and outlook to 2015*; Food and Agriculture Organization of the United Nations (FAO): Rome, Italy, 2011; <ftp://ftp.fao.org/ag/agp/docs/cwfto15.pdf> (accessed January 30, 2013).
20. Jensen, J.W.; Felby, C.; Jørgensen, H.; Roensch, G.Oe.; Noerholm, N.D. Enzymatic processing of municipal solid waste. *Waste Manage.* **2010**, *30*, 2497-2503.
21. Tonini, D. and Astrup, T. Life-cycle assessment of biomass-based energy systems: a case study for Denmark. *Appl. Energy* **2012**, *99*, 234-246.
22. Mathiesen, B.V.; Lund, H.; Karlsson, K. 100% Renewable energy systems, climate mitigation and economic growth. *Appl. Energy* **2011**, *88*, 488-501.
23. Lund, H. and Mathiesen, B.V. Energy system analysis of 100% renewable energy systems: The case of Denmark in years 2030 and 2050. *Energy* **2009**, *34*, 524-531.
24. Mathiesen, B.V.; Lund, H.; Noergaard, P. Integrated transport and renewable energy systems. *Utilities Policy* **2008**, *16*, 107-116.
25. Lund, H. Renewable energy strategies for sustainable development. *Energy* **2007**, *32*, 912-919.
26. Combs MC. *Life Cycle Analysis of Recycling Facilities in a Carbon Constrained World*. Master Thesis, NC State Department of Civil, Construction, and Environmental Engineering, NC, 2012.
27. Tonini, D. and Astrup, T. Life-cycle assessment of a waste refinery process for enzymatic treatment of municipal solid waste. *Waste Manage.* **2012**, *32*, 165-176.
28. Montejo, C.; Tonini, D.; Márquez, M.C.; Astrup, T. Mechanical-biological treatment: performance and potentials. An LCA of 8 MBT plants including waste characterization. *J. Environ. Manage.* **2013**. DOI: <http://dx.doi.org/10.1016/j.jenvman.2013.05.063>.
29. Bernstad, A.; Malmqvist, L.; Truedsson, C.; Jansen, la Cour J. Need for improvements in physical pre-treatment of source-separated household food waste. *Waste Manage.* **2013**, *33*, 746-754.

30. Pognani, M.; Barrena, R.; Font, X.; Sánchez, A. A complete mass balance of a complex combined anaerobic/aerobic municipal source-separated waste treatment plant. *Waste Manage.* **2012**, *32*, 799-805.
31. *Technology data for energy plants - Generation of Electricity and District Heating, Energy Storage and Energy Carrier Generation and Conversion*. Energistyrelsen: Copenhagen, Denmark, 2012; [http://www.ens.dk/Documents/Netboghandel%20-%20publikationer/2012/Teknologikatalog\\_2012.pdf](http://www.ens.dk/Documents/Netboghandel%20-%20publikationer/2012/Teknologikatalog_2012.pdf) (accessed January 30, 2013).
32. Riber, C.; Bhandar, G.S.; Christensen, T.H. Environmental assessment of waste incineration in a life-cycle-perspective (EASEWASTE). *Waste Manage. Res.* **2008**, *26*, 96-103.
33. *Emissions from decentralised CHP plants 2007*. Environmental project no. 07/1882. Project report 5–Emission factors and emission inventory for decentralised CHP production. National Environmental Research Institute: Aarhus University, Aarhus, Denmark, 2010; <http://www.dmu.dk/Pub/FR786.pdf> (accessed January 30, 2013).
34. *Amagerforbraending Miljøredogørelse 2011* (Amagerforbraending Green Account 2011). Amagerforbrænding I/S: Copenhagen, Denmark, 2012; <http://www.amfor.dk/~media/files/aarsrapport%20og%20miljoeredogørelser/amfor%20miljoeredogørelse%202011.ashx> (accessed January 30, 2013).
35. Damgaard, A.; Riber, C.; Fruergaard, T.; Hulgaard, T.; Christensen, T.H. Life-cycle-assessment of the historical development of air pollution control and energy recovery in waste incineration. *Waste Manage.* **2010**, *30*, 1244-1250.
36. Birgisdottir, H.; Bhandar, G.; Hauschild, M.Z.; Christensen, T.H. Life cycle assessment of disposal of residues from municipal solid waste incineration: Recycling of bottom ash in road construction or landfilling in Denmark evaluated in the ROAD-RES model. *Waste Manage.* **2007**, *27*, S75-S84.
37. Grosso, M.; Biganzoli, L.; Rigamonti, L. A quantitative estimate of potential aluminium recovery from incineration bottom ashes. *Resour. Conserv. Recy.* **2011**, *55*, 1178-1184.
38. Fruergaard, T.; Hyks, J.; Astrup, T. Life-cycle assessment of selected management options for air pollution control residues from waste incineration. *Sci. Total Environ.* **2010**, *408*, 4672-4680.
39. Boldrin, A.; Neidel, T.L.; Damgaard, A.; Bhandar, G.S.; Moeller, J.; Christensen, T.H. Modelling of environmental impacts from biological treatment of organic municipal waste in EASEWASTE. *Waste Manage.* **2011**, *31*, 619-630.
40. Davidsson, Å.; Gruvberger, C.; Christensen, T.H.; Hansen, T.L.; Jansen, la Cour J. Methane yield in source-sorted organic fraction of municipal solid waste. *Waste Manage.* **2007**, *27*, 406-414.
41. Pognani, M.; D'Imporzano, G.; Scaglia, B.; Adani, F. Substituting energy crops with organic fraction of municipal solid waste for biogas production at farm level: A full-scale plant study. *Process Biochemistry* **2009**, *44*, 817-821.
42. Moeller, J.; Christensen, T.H.; Jansen, la Cour J. Anaerobic Digestion: Mass Balances and Products, In *Solid Waste Technology & Management*; Christensen T.H. Ed.; John Wiley & Sons, Ltd: 2010; pp. 618-627.



43. Boerjesson, P. and Berglund, M. Environmental systems analysis of biogas systems - Part I: Fuel-cycle emissions. *Biomass Bioenergy* **2006**, 30, 469-485.
44. *Life Cycle Inventories of Bioenergy*; Data v 2.0 (2007), Ecoinvent report No. 17; Swiss Centre for Life Cycle Inventories, ESU-services: Uster, Switzerland, 2007.
45. Manfredi, S. and Christensen, T.H. Environmental assessment of solid waste landfilling technologies by means of LCA-modeling. *Waste Manage.* **2009**, 29, 32-43.
46. Tchobanoglous, G. and Kreith, F. *Handbook of solid waste management*, 2<sup>nd</sup> edition. McGraw-Hill: New York, USA, 2002.
47. *2006 IPCC guidelines for national greenhouse gas inventories*. Volume 5 Waste. Intergovernmental panel on climate change: Hayama, Japan, 2006; [www.ipccnggip.iges.or.jp/public/2006gl/index.html](http://www.ipccnggip.iges.or.jp/public/2006gl/index.html).
48. Reeh, U. and Møller, J. Evaluation of different biological waste treatment strategies, In *Proceedings from NJF-seminar No. 327, Urban Areas - Rural Areas and Recycling - the Organic Way Forward?* Magid, J., Granstedt, A., Dýrmundsson, O., Kahiluoto, H., Ruissen, T., Eds.; Danish Research Centre for Organic Agriculture, DARCOF: Tjele, Denmark, 2002; pp. 147-156; [http://orgprints.org/3065/1/dar\\_3.pdf](http://orgprints.org/3065/1/dar_3.pdf) (accessed January 30, 2013).
49. Angelidaki, I. and Batstone, D.J. Anaerobic Digestion: Process, In *Solid Waste Technology & Management*; Christensen T.H. Ed.; John Wiley & Sons, Ltd: 2010; pp. 583-600.
50. *The Viability of Advanced Thermal Treatment of MSW in the UK*. Fichtner Consulting Engineers Ltd: London, United Kingdom, 2004; [http://www.esauk.org/reports\\_press\\_releases/esa\\_reports/thermal\\_treatment\\_report.pdf](http://www.esauk.org/reports_press_releases/esa_reports/thermal_treatment_report.pdf) (accessed April 30, 2012).
51. Boldrin, A.; Andersen, J.K.; Møller, J.; Christensen, T.H.; Favoino, E. Composting and compost utilization: accounting of greenhouse gases and global warming contributions. *Waste Manage. Res.* **2009**, 27, 800-812.
52. *CEWEP Energy Report II (status 2004-2007)*. CEWEP: Bamberg, Germany, 2009; [http://avfallnorge.web123.no/article\\_docs/09\\_04\\_29%20final%20version%20CEWEPReport.pdf](http://avfallnorge.web123.no/article_docs/09_04_29%20final%20version%20CEWEPReport.pdf) (accessed November 30, 2012).
53. Jansen, la Cour J. Anaerobic Digestion: Technology, In *Solid Waste Technology & Management*; Christensen T.H. Ed.; John Wiley & Sons, Ltd: 2010; pp. 601-617.
54. *Genanvendelse i LCA – systemudvidelse*. Danish Environmental Protection Agency: Copenhagen, Denmark, 2006; <http://www2.mst.dk/udgiv/publikationer/2006/87-7052-165-4/pdf/87-7052-166-2.pdf> (accessed January 30, 2013).
55. *Sustainability report 2008*. Skiern Papirfabrik: Skiern, Denmark, 2008; <http://www.skjernpaper.com/wp-content/uploads/sustrep2008.pdf> (accessed June 30, 2013).
56. Damgaard, A.; Larsen, A.W.; Christensen, T.H. Recycling of metals: accounting of greenhouse gases and global warming contributions. *Waste Manage. Res.* **2009**, 27, 773-780.

57. Larsen, A.W.; Merrild, H.; Christensen, T.H. Recycling of glass: accounting of greenhouse gases and global warming contributions. *Waste Manage. Res.* **2009**, *27*, 754-762.
58. Merrild, H.; Damgaard, A.; Christensen, T.H. Recycling of paper: accounting of greenhouse gases and global warming contributions. *Waste Manage. Res.* **2009**, *27*, 746-753.
59. Astrup, T.; Fruergaard, T.; Christensen, T.H. Recycling of plastic: accounting of greenhouse gases and global warming contributions. *Waste Manage. Res.* **2009**, *27*, 763-772.
60. Hansen, T.L.; Bhandar, G.S.; Christensen, T.H.; Bruun, S.; Jensen, L.S. Life cycle modelling of environmental impacts of application of processed organic municipal solid waste on agricultural land (Easewaste). *Waste Manage. Res.* **2006**, *24*, 153-166.
61. Bekendtgørelse nr. 1650 af 13 December 2006 om anvendelse af affald til jordbrugsformaal (Slambekendtgørelsen). Public law, 2006; <https://www.retsinformation.dk/Forms/R0710.aspx?id=13056> (accessed January 30, 2013).
62. Bruun, S.; Hansen, T.L.; Christensen, T.H.; Magid, J.; Jensen, L.S. Application of processed organic municipal solid waste on agricultural land - A scenario analysis. *Environ. Model Assess.* **2006**, *11*, 251-265.
63. Larsen, A.W.; Vrgoc, M.; Christensen, T.H.; Lieberknecht, P. Diesel consumption in waste collection and transport and its environmental significance. *Waste Manage. Res.* **2009**, *27*, 652-659.
64. Eisted, R.; Larsen, A.W.; Christensen, T.H. Collection, transfer and transport of waste: accounting of greenhouse gases and global warming contribution. *Waste Manage. Res.* **2009**, *27*, 738-745.