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SPECIAL FEATURE: ORIGINAL ARTICLE

Recent researches on Thermal Treatment and Emission Control (9th i-CIPEC)

Energy recovery and greenhouse gas reduction potential from food waste in Japan

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Abstract Waste-to-energy is one effective waste management approach for a sustainable society. The purpose of this study was to clarify the potential for energy recovery and greenhouse gas (GHG) reduction that could be achieved by introducing anaerobic digestion (AD) facilities in the process of reconstructing aging incineration facilities in Japan. Using statistical data from 1068 incineration facilities, four future scenarios were considered and compared with the current situation. As results, compared with the current situation the amount of electricity generated could increase by 60 % in 2030, by combining AD facilities for food waste with new, high-efficiency incineration facilities for remaining municipal solid waste (MSW). From a life cycle perspective, net energy recovery in 2030 was approximately three times greater than in 2011, and GHG emission could be reduced by 27 %. The introduction of AD facilities is attractive for small authorities, which currently treat <100 t/day of MSW through incineration facilities without energy recovery. An AD facility is also beneficial for large authorities. On the contrary, in middle-scale authorities that treat 100–299 t/day of MSW, the reconstruction of incineration facilities to include electricity production capabilities requires careful consideration, because it will significantly influence energy recovery and GHG reduction effects.

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Keywords Food waste · Municipal solid waste · Anaerobic digestion · Incineration · Waste-to-energy · Life cycle analysis

Introduction

In 2013, the EU incinerated 25 % of its municipal solid waste (MSW) [1]; in comparison, in 2014, Japan incinerated 81 % of its MSW [2]. Japan has historically adopted incineration as the MSW treatment method of choice, due to lack of landfill space. Approximately 1100 incineration facilities are currently in operation in Japan [3]. However, some aging incineration facilities will have to be reconstructed in the near future. Approximately one-third and two-thirds of all incineration facilities will need to be replaced by 2020 and 2030, respectively; at these points, these would have been in operation for approximately 30 years [3]. Future mid- and long-term MSW management strategies therefore need to be developed.

Food waste management is a global concern. The Food and Agriculture Organization (FAO) of the United Nations has estimated that roughly one-third of global food produced for human consumption is lost or wasted between production and consumption stages, the equivalent of approximately 1.3 billion t/year [4]. Food waste generation in the EU was estimated at 89 million t/year [5], compared to 60 million t/year in China and 17 million t/year in India [6]. In Japan, 17.1 million t of food waste were generated in 2010, of which 2.9 million t were derived from the industrial sector, 3.5 million t from the business sector, and the remaining 10.7 million t from households [7]. The Law for Promotion of Recycling and Related Activities for the Treatment of Cyclical Food Resources came into force in 2001 and was revised in 2007. However, it only covers businesses and industries that generate more than 100 t/year of food waste [8]. As per this law, in 2010, food waste recycled by feed production, composting, and energy recovery [including anaerobic digestion (AD)] accounted for 2.0, 0.6, and 0.4 million t, respectively [7].

Approximately 78.9 % of business food waste and 93.8 % of household food waste was not recycled in 2010; this was consequently incinerated as mixed waste without separation [7]. The management of food waste in MSW must therefore be carefully improved.

Waste-to-energy (WTE) is one effective waste management approach for a sustainable society. WTE contributes to the reduction of fossil fuel consumption and consequently of greenhouse gas (GHG) emissions. Incineration with power generation and/or heat recovery is one of the typical WTE methodologies. AD of food waste enables more efficient electricity and heat recovery than incineration alone. Landfill disposal of food waste can lead to large volumes of CH_4 emissions. Although biogas could be collected from landfill sites for energy recovery, landfilling is known to result in negative impacts on the environment and human health, and should be avoided.

There have been case studies of life cycle assessment (LCA) of WTE conducted in Uppsala, Stockholm, Älvdalen [9], and Augustenborg in Sweden [10], in Aarhus [11, 12] and for Denmark in general [13], in Rome [14, 15] and Milan [12, 16] in Italy, in London in the UK [17], in Thessaloniki in Greece [18], in Barcelona in Spain [19], in Jungnang-gu in the Republic of Korea [20], and in Jakarta in Indonesia [21]. In Japan, some previous studies have estimated the reduction in environmental impacts achieved by the comanagement of incineration and AD facilities [22–24]. As shown in Table 1, the wide range of net GHG emissions reported in these studies points to the difficulty of comparing results, due to differences in assumptions related to aspects, such as type of waste, system boundary, and considered substitutions. However, comparisons conducted within studies indicate that WTE using incineration and AD is effective in reducing environmental impact. A review of 25 and 82 LCA studies on food/organic waste by Bernstad et al. [25] and Morris et al. [26], respectively, also indicated that AD showed good environmental performance.

Mid- and long-term MSW management strategies in Japan should therefore include the introduction of AD facilities and the renewal of incineration facilities. The purpose of this study was to clarify the potential for energy recovery and GHG reduction that could be achieved by introducing AD facilities in the process of reconstructing aging incineration facilities in Japan, based on a LCA perspective. Using statistical data from more than 1000 incineration facilities in Japan, the results highlight differences in waste generation and composition among the waste collection areas.

Materials and methods

Number of facilities

Statistical data from 1068 operational incineration facilities that treat MSW were used for the analysis [3]. Facility specifications, such as treatment capacity, electricity production efficiency, and the composition of treated MSW (as reported for 2011) were used as starting values.

For purposes of the time series, we considered conditions for Japan in 2011, 2020, and 2030. Statistical data [3] indicated the year of construction of each facility. The lifespan of incineration facilities was assumed to be 30 years. In scenarios for 2020 and 2030, incineration facilities at the end of their lifespan were assumed to have been reconstructed, with or without an AD facility. The total number of incineration facilities was assumed to be constant. Consequently, 324 and 825 incineration facilities would be reconstructed as of 2020 and 2030, respectively.

Scenarios construction

A total of four future scenarios were considered for the reconstruction of aging incineration facilities and were compared with the current situation (*Inc2011*). In two of the scenarios, only an incineration facility (*Inc2020* and *Inc2030*) was reconstructed, while in the other two, an AD facility was constructed in addition to a high-efficiency incineration facility (*IncAD2020* and *IncAD2030*). To estimate the maximum potential, all (100 %) food waste was assumed to be source separated and treated in AD facilities in *IncAD2020* and *IncAD2030* scenarios.

MSW composition

Table 2 shows the composition of MSW treated in incineration facilities. While MSW composition differs between facilities, it was assumed that composition remained constant during the period 2011–2030. In all scenarios, the mass (three components and each element) and energy (lower heating value; LHV) of MSW treated at each facility were balanced. Values for average moisture, combustible matter, and ash content of MSW in all 1068 facilities were 46.7, 44.2, and 9.2 %, respectively. Food waste accounted for 41.5 % of MSW. In *IncAD2020* and *IncAD2030,* moisture content in remaining MSW after food waste separation was assumed to be constant at 21.4 %, to estimate moisture content in food waste. LHV was estimated using Steuer's model [30] after excluding sulfur, as described in Eq. (1)

Table 1 Net GHG emissions reported in previous LCA studies

References	Region/country	Waste type (unit)	Net GHG emissions (kg-CO_2eq/ton of *Unit*)			Remarks
			Incineration (Inc.)	Anaerobic digestion (AD)	Combined Inc. and AD	
Erikson et al. [9]	Uppsala, Stockholm, and Älvdalen, Sweden	MSW	885	–	793 (electricity, heat, N and P in mineral fertilizer) 793 (electricity, heat, N and P in mineral fertilizer, and diesel fuel for bus) 773 (electricity, heat, N and P in mineral fertilizer, and gasoline for vehicle)	70 % of biodegradable waste was treated by AD. The remaining fraction was incinerated System boundary was expanded for production process of substitutions
Cherubini et al. [14, 15]	Rome, Italy	MSW	297 (electricity)	−93 (natural gas by AD from organic fraction, electricity through RDF from inorganic fraction)	–	Net GHG emissions in AD included GHG reduction effect by RDF from inorganic fraction
Koroneos and Nanaki [18]	Thessaloniki, Greece	MSW	–	1.8 (electricity, heat, and inorganic fertilizer)	–	Collection process was excluded 50 % of food waste was treated by AD. The remaining fraction was landfilled
Gross et al. [16]	Milan, Italy	MSW	−160 (electricity and heat)	–	−220 (electricity and heat) −190 (heat, natural gas for distribution network) −180 (heat, natural gas for vehicle fuel)	Values were estimated from the scale given in figures by the authors
Møller et al. [27]	EU	OW	–	From −375 to 33 (electricity and heat) From −293 to 111 (natural gas for vehicle fuel)	–	Collection process was excluded
Morris et al. [26]	–	OW	–	From −590 to 100	–	Range shows 95 % confidence interval via review of 82 LCA studies
Evangelisti et al. [17]	London, UK	OW	−42 (electricity and heat)	−65 (electricity, heat, and inorganic fertilizer)	–	
IGES [6]	Developing Asia	FW	–	From 0 to 170	–	Collection process was excluded Calculated using minimum and maximum parameters reported in IPCC [28]
Clavreul et al. [29]	Denmark	FW	−357 (electricity and heat)	−301 (electricity, heat and inorganic fertilizer)	–	
Kim et al. [20]	Jungnang-gu, Republic of Korea	FW	–	33 (electricity)	–	
Bastian et al. [21]	Jakarta, Indonesia	FW	160 (No ER) 52 (electricity)	−230 (electricity)	–	

Product/energy written in brackets indicates substitutions

OW organic waste, *FW* food waste, *RDF* refuse-derived fuel, *ER* energy recovery

Table 2 Municipal solid waste (MSW) and composition of food waste fraction treated at incineration facilities in Japan ($N = 1068$)

	Unit	Average (Sd. value)
MSW (Inc2011, Inc2020, and Inc2030 scenarios)		
Three components		
Moisture	wt%	46.7 (Sd. 7.4)
Combustible matter	wt%	44.2 (Sd. 11.6)
Ash	wt%	9.2 (Sd. 9.9)
Elemental composition		
C (fossil-derived)	wt%	11.6 (Sd. 3.5)
C (biomass-derived)	wt%	15.4 (Sd. 2.4)
H	wt%	3.8 (Sd. 0.6)
N	wt%	0.5 (Sd. 0.1)
O	wt%	16.0 (Sd. 2.4)
Lower heating value	MJ/t-wet	10,572 (Sd. 2116)
Remaining MSW after food waste separation (IncAD2020 and IncAD2030 scenarios)		
Three components		
Moisture[a]	wt%	21.0 (Sd. 0.0)
Combustible matter	wt%	65.5 (Sd. 16.0)
Ash	wt%	13.2 (Sd. 16.0)
Elemental composition		
C (fossil-derived)	wt%	19.8 (Sd. 4.2)
C (biomass-derived)	wt%	21.1 (Sd. 2.3)
H	wt%	5.7 (Sd. 0.4)
N	wt%	0.4 (Sd. 0.0)
O	wt%	23.6 (Sd. 1.9)
Lower heating value	MJ/t-wet	17,272 (Sd. 1359)
Food waste (IncAD2020 and IncAD2030 scenarios)		
Food waste composition in MSW (before separation)	wt%	41.5 (Sd. 11.2)
Three components		
Moisture	wt%	82.6 (Sd. 11.1)
Combustible matter	wt%	14.2 (Sd. 9.1)
Ash	wt%	3.2 (Sd. 2.0)
Elemental composition		
C (fossil-derived)	wt%	0.0 (Sd. 0.0)
C (biomass-derived)	wt%	7.4 (Sd. 4.7)
H	wt%	1.0 (Sd. 0.7)
N	wt%	0.5 (Sd. 0.3)
O	wt%	5.2 (Sd. 3.3)
Lower heating value	MJ/t-wet	1027 (Sd. 2257)

Sd. standard deviation

[a] Moisture content in remaining MSW after food waste separation was assumed to be constant at 21.4 %, to estimate moisture content in food waste

$$\mathrm{LHV} = 339.4 \times \left(\mathrm{C} - \frac{3}{8}\mathrm{O}\right) + 238.8 \times \frac{3}{8}\mathrm{O} + 1445.6 \times \left(\mathrm{H} - \frac{1}{16}O\right) - 25 \times (9\mathrm{H} + W) \quad (1)$$

where LHV lower heating value (MJ/t-wet), C carbon content (wt%), H hydrogen content (wt%), N nitrogen content (wt%), O oxygen (wt%), *W* moisture content (wt%).

MSW generation

This study only considered MSW from household and business sectors sent to the incineration facility, because recycled components of MSW are already well managed. MSW generation and composition reported in statistical data [3] were used for *Inc2011*. In the case of four future scenarios, the amounts of MSW generated and treated in each facility were assumed to be proportional to the

Table 3 Rate of population change as a function of incineration capacity

Incineration capacity of the facility (t/day)[a]	Number of facilities in 2011 (*N*)	Average population size within the collection area compared to 2010 value (%)	
		2020	2030
<50	313	89.4 (Sd. 5.9)	78.7 (Sd. 9.9)
50–99	197	92.5 (Sd. 5.8)	83.7 (Sd. 10.0)
100–299	374	95.3 (Sd. 4.7)	88.4 (Sd. 8.1)
≥300	184	98.7 (Sd. 3.3)	94.0 (Sd. 6.0)
Total	1068	93.6 (Sd. 6.1)	85.6 (Sd. 10.3)

Sd. standard deviation

[a] Facilities in operation as of 2011

projected population in each managing authority domain (collection area) [31]. It is important to note that this assumption did not consider the effects of future waste prevention and source separation activities. Table 3 shows average population size in 2020 and 2030 in the collection area, compared with population in 2010. In 2030, the population decreased by only 6.0 % in the area where the facility treats over 300 t/day, while a reduction of 21.3 % can be seen in the area where the facility treats less than 50 t/day. It was found that the generated amount of MSW in smaller authorities tends to decline more rapidly. Table 4 lists the number of facilities and the amount of waste treated according to the treatment capacity of each scenario. The total amount of MSW treated was consequently estimated to decrease from 34.9 million t in 2011 to 31.9 million t in 2030, a reduction of 8.7 %. In *IncAD2020* and *IncAD2030*, 4.0 and 9.0 million t/year of food waste were treated in AD facilities, respectively. This accounted for 29.2 and 69.5 % of the total amount of food waste generated.

Treatment capacity of facilities and energy recovery

It was found that incineration facilities in *Inc2011* operated on average at 51.5 % of their maximum capacity, with a standard deviation of 16.7 % [3]. This value was assumed to be constant in all scenarios. The treatment capacity of the reconstructed incineration facilities was determined using this percentage and the amount of MSW incinerated. As shown in Table 4(a), the treatment capacity of each incineration facility tended to be scaled down following AD treatment of food waste.

As of 2011, 308 incineration facilities had electricity production equipment, with 11.7 % efficiency on average [3]. Efficiency in Japan was lower than in the EU (21.6 %) [32], due to the lower LHV of incinerated waste. The Japanese Government set an efficiency target for newly constructed incineration facilities of 21 % (average) for 2013–2017 [33]. For purposes of the four future scenarios, it was assumed that reconstructed incineration facilities with treatment capacity over 100 t/day could produce electricity, because as of 2011, almost all facilities that produce electricity treat over 100 t/day [3]. Electricity production efficiency at reconstructed incineration facilities in *Inc2020* and *Inc2030* was assumed to be 15 %. For *IncAD2020* and *IncAD2030*, it was assumed to be 15 % for facilities that treat 100–299 t/day and 20 % for those that treat ≥300 t/day. This was because electricity production efficiency loosely correlates with treatment capacity, as shown in the electronic supplementary material. Additionally, the higher LHV of the remaining MSW following large-scale food waste separation in *IncAD2020* and *IncAD2030* was expected to result in better WTE performance. AD facilities can also produce electricity using gas engines, with efficiency of 40 %, regardless of treatment capacity. Surplus heat recovery is also a significant function of WTE facilities. However, heat recovery was not considered, because providing surplus heat outside the facility was more difficult than providing electricity, due to lack of infrastructure for heat transport and to regional and seasonal demand gaps.

Environmental impact

GHG emissions and fossil fuel consumption were considered to be environmental impacts. CO_2, CH_4, and N_2O emissions were classified as GHGs and characterized using global warming potential (GWP) 100-year values of 1 for CO_2, 25 for CH_4, and 298 for N_2O [34]. CO_2 emissions derived from biomass were also excluded because of their carbon–neutral status.

Electricity and diesel fuel consumption were considered to comprise fossil fuel consumption. Both consumption and substituted electricity refer to commercial electricity from utility companies.

System boundary

Collection, treatment (incineration and AD), and final disposal (landfill of residue) were included within the

Table 4 (a) Number of facilities, and (b) amount of waste treated according to treatment capacity of each scenario

Treatment capacity at each time period (t/day)	*2011*	*2020*		*2030*	
	Inc	*Inc*	*IncAD*	*Inc*	*IncAD*
(a) Incineration facilities non-reconstructed					
<50	**313**	**240**	**240**	**52**	**52**
50–99	**197**	**133**	**133**	**39**	**39**
100–299	**374**	**246**	**246**	**98**	**98**
≥300	**184**	**125**	**125**	**54**	**54**
Subtotal	1068	744	744	243	243
Incineration facilities reconstructed					
<50	–	84	134	306	435
50–99	–	64	93	162	181
100–299	–	**128**	**79**	**257**	**168**
≥300	–	**48**	**18**	**100**	**41**
Subtotal	–	324	324	825	825
AD facilities					
<20	–	–	**80**	–	**307**
20–49	–	–	**92**	–	**218**
50–99	–	–	**80**	–	**162**
≥100	–	–	**72**	–	**138**
Subtotal	–	–	324	–	825
Total	1068	1068	1068	1068	1068
(b) Incineration facilities non-reconstructed					
<50	**1.27**	**0.81**	**0.81**	**0.15**	**0.15**
50–99	**2.76**	**1.76**	**1.76**	**0.51**	**0.51**
100–299	**12.04**	**7.87**	**7.87**	**3.27**	**3.27**
≥300	**18.84**	**13.80**	**13.80**	**5.73**	**5.73**
Subtotal	34.92	24.22	24.22	9.66	9.66
Incineration facilities reconstructed					
<50	–	0.42	0.59	1.19	1.61
50–99	–	0.92	1.23	2.33	2.44
100–299	–	**4.04**	**2.46**	**8.31**	**5.55**
≥300	–	**4.28**	**1.36**	**10.36**	**3.62**
Subtotal	–	9.66	5.64	22.22	13.22
AD facilities					
<20	–	–	**0.16**	–	**0.50**
20–49	–	–	**0.61**	–	**1.41**
50–99	–	–	**1.05**	–	**2.10**
≥100	–	–	**2.21**	–	**5.00**
Subtotal	–	–	4.02	–	9.00
Total	34.92	33.88	33.88	31.88	31.88

Facilities in bold values have energy recovery. However, the availability of incineration facilities depend on the starting situation in 2011; 308 facilities had electricity production equipment

system boundary. Construction of the facilities was not included. With regard to fossil fuel consumption, the system boundary considered the stages from raw material extraction to final use (combustion). The associated environmental impacts were allocated to the process consuming the fossil fuel.

Unit processes and data collection

Important parameters and assumptions employed in this study are listed in Table 5, and each process is briefly described. Uncertainties considered for some parameters are discussed in the "Sensitivity analysis" section.

Table 5 Parameters and assumptions employed in this LCA study

Process	Parameters		Value	Unit	References
Collection	Diesel fuel consumption	MSW without food waste separation	4.05	L/t	[35] and calculation in this study
		Remaining MSW after food waste separation	4.22	L/t	[35] and calculation in this study
		Food waste	5.63	L/t	[35] and calculation in this study
Incineration	Electricity consumption	–	Empirical formula		[36] and calculation in this study
	Combustion ratio		98.9	%	[37]
	Moisture content in ash (residue)		20	%	[22]
	Emission factor	CH_4	0.96	g-CH_4/t-waste	[38]
	From exhaust gas	N_2O	56.5	g-N_2O/t-waste	[38]
Anaerobic digestion (AD)	Biodegradation rate		84	%	[39]
	CH_4 concentration in biogas		57.9	%	[24]
	Moisture content of AD residue		60	%	Assumed
	Electricity consumption	Pretreatment and digestion	357.5	kWh/t-TS	[39]
		Wastewater treatment	32.5	kWh/t-wastewater	[39]
	Emission factor	CH_4	5.9	g-CH_4/t-wastewater	[40]
	From wastewater	N_2O	4.5	g-N_2O/t-N	[40]
Transportation	Distance		50	km	Assumed
	Diesel consumption		2.99	km/L	[38]
Landfill	Electricity consumption		6.38	kWh/t	[41]
	Diesel consumption		0.763	L/t	[41]
Common	GHG emission factor	Electricity	0.439	kg-CO_2/kWh	[42]
		Diesel fuel	2.83	kg-CO_2/L	[43]
	Lower heating value	Diesel fuel	35.5	MJ/L	[43]
		CH_4	35.9	MJ/Nm^3	Assumed
	Primary energy conversion factor		9.76	MJ/kWh	[44]
	Secondary energy conversion factor		3.6	MJ/kWh	[44]
	GWP100 value	CH_4	25	t-CO_2eq/t-CH_4	[33]
		N_2O	298	t-CO_2eq/t-N_2O	[33]

GWP100 global warming potential 100-year values, *MSW* municipal solid waste

Collection

Diesel fuel consumption units, obtained through a case study of Kyoto City using a grid city model [35], were used for calculating annual diesel fuel consumption. The frequency of collection was assumed to be twice a week for MSW. In case of source separated food waste, the collection frequency was assumed to be twice a week for both food waste and remaining MSW. Because more vehicles are required for the collection of food waste in *IncAD* scenarios, the total collection distances in these scenarios are longer than those in *Inc* scenarios. CO_2 emissions from diesel combustion were then calculated.

Incineration

Direct emissions of CO_2 from fossil-derived carbon contained in waste were considered, as well as CH_4 and N_2O in exhaust gas. Electricity consumption was also calculated using an empirical formula (Eq. 2), which reflects differences in waste composition in different facilities [36]. In *IncAD2020* and *IncAD2030* scenarios, the digestion residue after the AD process was incinerated with remaining MSW. Ash residue from incineration was landfilled after transportation.

When electricity production equipment was present, the amount of electricity produced was calculated by multiplying LHV of waste with electricity production efficiency according to treatment capacity, as discussed in the "Treatment capacity of facilities and energy recovery" section

$$U_{EL\_consump} = 30.6 \times W + 0.0026 \times LHV + 0.015 \times G + 28.3 \times C_{ash} \quad (2)$$

where $U_{EL\_consump}$, electricity consumption unit (kWh/t-waste), W weight of incinerated waste (t-waste), LHV lower heating value (MJ/t-waste), G amount of wet exhaust gas (Nm^3/t-waste), C_{ash} ash content (wt%).

Anaerobic digestion

The AD technology was assumed to be dry methane fermentation at 55 °C. Inventory data for this process was based on the results of a demonstration project (Bio-cycle Project) in Kyoto, Japan in FY 2007–2009 [40].

Electricity consumption in AD was divided into digestion and wastewater treatment processes. Electricity consumption during the digestion process was assumed to be proportional to the total solid (TS) fraction, while electricity consumption during the wastewater treatment process was assumed to be proportional to the amount of wastewater generated. One of the problems with AD is that the wastewater treatment step must be tightly controlled. NH_3 concentration during digestion is a key parameter, because it counteracts the digestion of organic content. Volumes of digestive fluid and wastewater were therefore determined to maintain an NH_3 concentration below 2500 ppm. To meet this condition, the amount of wastewater generated was estimated using Eq. (3). CH_4 and N_2O emissions from wastewater were also considered using emission factors shown in Table 5.

Biogas generation was calculated by multiplying the amount of carbon in food waste with the biodegradation rate (84 %). The CH_4 concentration in biogas was assumed to be constant (57.9 %) [24]. Produced biogas was used for electricity production by means of a gas engine. It was assumed that there was no surplus heat removed from the facility

$$U_{wastewater} = 280 \times C_N - C_{ash} \times \frac{C_{moi}}{1 - C_{moi}} - C_{VS} \times \left(1 - R_{biodeg}\right) \times \frac{C_{moi}}{1 - C_{moi}} \quad (3)$$

$U_{wastewater}$: wastewater generation unit (t-wastewater/t-FW), FW: food waste, C_N: nitrogen content (wt%), C_{moi}: moisture content (wt%), C_{VS}: volatile solid content (wt%), R_{biodeg}: biodegradation rate of food waste (%).

Transportation

Distances between incineration facilities and landfill sites differ between authorities. However, this study assumed that in all cases, incineration residues were transported to a landfill site 25 km (as one way distance) away from the incineration facility. CO_2 emissions from diesel combustion were calculated.

Table 6 Electricity production by incineration and anaerobic digestion (AD) facilities for each scenario

	2011	*2020*		*2030*	
	Inc	*Inc*	*IncAD*	*Inc*	*IncAD*
Production capacity (MW)					
Inc.	1408	1805	1710	1998	1823
AD	0	0	180	0	435
Total	1408	1805	1890	1998	2259
Production amount (GWh/year)					
Inc.	8957	11,481	10,873	12,707	11,595
AD	0	0	1145	0	2769
Total	8957	11,481	12,018	12,707	14,365
Compared with *Inc2011*	1.00	1.28	1.34	1.42	1.60

Landfilling

For landfilling, electricity consumption for leachate treatment and diesel fuel consumption for heavy machinery were considered. It was assumed that the landfill was semi-aerobic and that there was no biogas generation, given that the majority of the biodegradable fraction in the residue would have been combusted before reaching the landfill.

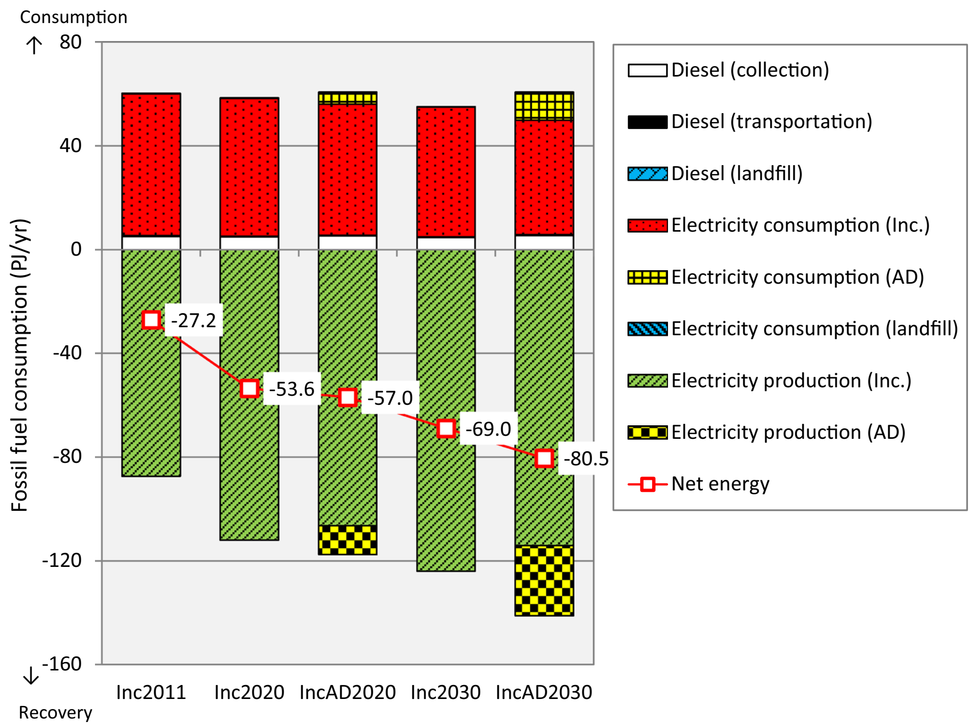
Results and discussion

Fossil fuel consumption

The results in Table 6 show trends in electricity production as a function of facility renewal. Compared with the current situation (*Inc2011*), the amount of electricity generated could increase by 34 % in *IncAD2020* and by 60 % in *IncAD2030*, through combining AD facilities for food waste with new, high-efficiency incineration facilities for remaining MSW. The energy recovery potential of AD in *IncAD2020* and *IncAD2030* was 1100 GWh/year (180 MW) and 2800 GWh/year (440 MW), respectively. The decrease in electricity production by the incineration facility between *Inc2020* and *IncAD2020* (or *Inc2030* and *IncAD2030*) was smaller than the increase from AD.

From a life cycle perspective, total electricity consumption and production in the 1068 incineration and AD facilities (Fig. 1) contributed considerably to overall fossil fuel consumption. Through reconstruction of aging incineration facilities, the net amount of recovered energy increased from 27.2 PJ/year in *Inc2011* to 53.6 PJ/year in

Fig. 1 Overall fossil fuel consumption in Japan for initial and four future scenarios



Inc2020. If AD facilities were also introduced with new, high-efficiency incineration facilities, then a further 3.4 PJ/year could be recovered in *IncAD2020*. In 2030, the gaps between *Inc2030* and *IncAD2030* widened. Consequently, net energy recovery in *IncAD2030* was estimated to be 80.5 PJ/year, approximately three times greater than in *Inc2011*. While the energy consumption of incineration and AD facilities did not differ greatly between the scenarios, electricity production gradually increased. Although recovered amounts from incineration facilities in *IncAD2020* and *IncAD2030* decreased compared with those in *Inc2020* and *Inc2030*, the recovery amounts from AD facilities exceeded reduction amounts. It is important to note that net energy recovery could increase, even if MSW generation decreases in future. Considering that energy consumption in a household is approximately 38.4 GJ/year, the net amount of energy recovery in *Inc2020*, *IncAD2020*, *Inc2030*, and *IncAD2030* equaled energy demand of 1.40, 1.49, 1.80, and 2.10 million households, respectively.

Greenhouse gas emissions

Table 7 presents summarized results of GHG emissions. Direct CO_2 emissions from MSW combustion, mainly derived from carbon in plastics, gradually decreased between 2011and 2030, because MSW generation decreased. Other GHG emissions increased slightly through the introduction of AD facilities. However, more GHG reduction effects were expected. Detailed results are provided in Fig. 2. In addition to direct CO_2 emissions, electricity consumption and other direct emissions (CH_4 and N_2O emissions from exhaust gas) in incineration contributed to GHG emissions. GHG emissions from electricity consumption in AD were relatively lower because the amount of treatment in AD facilities was lower than in incineration facilities. Reduced amounts of net GHG emissions (compared with *Inc2011*) were estimated to be 3.2 million t-CO_2eq/year in *Inc2030*. If AD facilities were also introduced with new, high-efficiency incineration facilities, then a further reduction of 0.64 million t-CO_2eq/year could be achieved by *IncAD2030*. Reduction ratios in *IncAD2020 and Inc2030AD* (compared with *Inc2011*) were 12.9 and 26.9 %; this indicates that the ratio could be doubled through reconstruction of incineration facilities during 2020–2030. Of these reductions, however, 2.9 and 8.6 % were derived from reductions in direct CO_2 emissions due to reduced MSW generation.

Net GHG emissions per 1 t of MSW were 412, 376, 370, 350, and 330 kg-CO_2eq/t-MSW in *Inc2011*, *Inc2020*, *IncAD2020*, *Inc2030*, and *IncAD2030*, respectively, with a 19.9 % reduction between *Inc2011* and *IncAD2030*. These values were similar to those obtained by Cherubini et al. [14, 15], i.e., 297 kg-CO_2eq/t-MSW (Table 1). Grosso et al. [16] indicate that AD can reduce GHG emissions by 30–60 kg-CO_2eq/t-MSW compared with incineration. In our study, GHG reduction effect compared with incineration (*Inc2011*) resulted in 36–42 and 62–82 kg-CO_2eq/t-

Table 7 GHG emissions in Japan for initial and four future scenarios

(Unit million t-CO_2eq/year)	*2011*	*2020*		*2030*	
	Inc	*Inc*	*IncAD*	*Inc*	*IncAD*
Direct CO_2 emission from MSW combustion (A)	14.83	14.40	14.40	13.55	13.55
Other GHG emissions (B)	3.49	3.38	3.43	3.18	3.31
GHG reductions (C)	−3.93	−5.04	−5.29	−5.58	−6.35
Net GHG emissions (A) + (B) + (C)	14.38	12.74	12.53	11.16	10.52
Reduction ratio compared with *Inc2011* (%)	–	11.4	12.9	22.4	26.9

Unconformity was due to significant digits

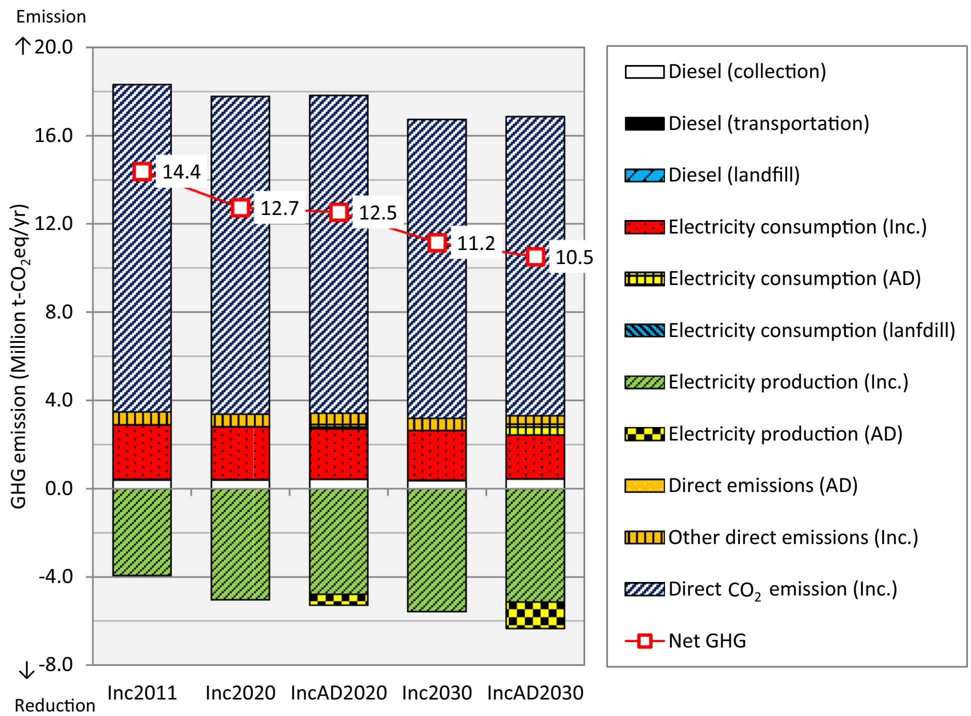


Fig. 2 GHG emissions in Japan for initial and four future scenarios

MSW as of 2020 and 2030, respectively. From comparison with the results of Koroneos and Nanaki [18] and Gross et al. [16], further GHG reductions would be expected if heat recovery is achieved in addition to electricity substitution. It can be estimated in our model that 1 % of heat recovery (LHV basis) additionally contributes to the reduction of 6.7 kg-CO_2eq/t-MSW.

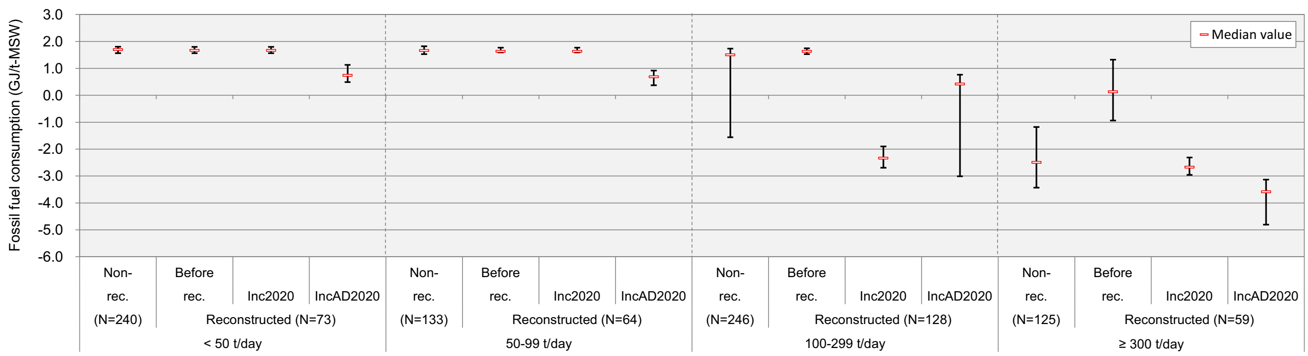
More detailed information about distribution of fossil fuel consumption and GHG emissions of each facility is provided in electronic supplementary material.

Influence of treatment capacity

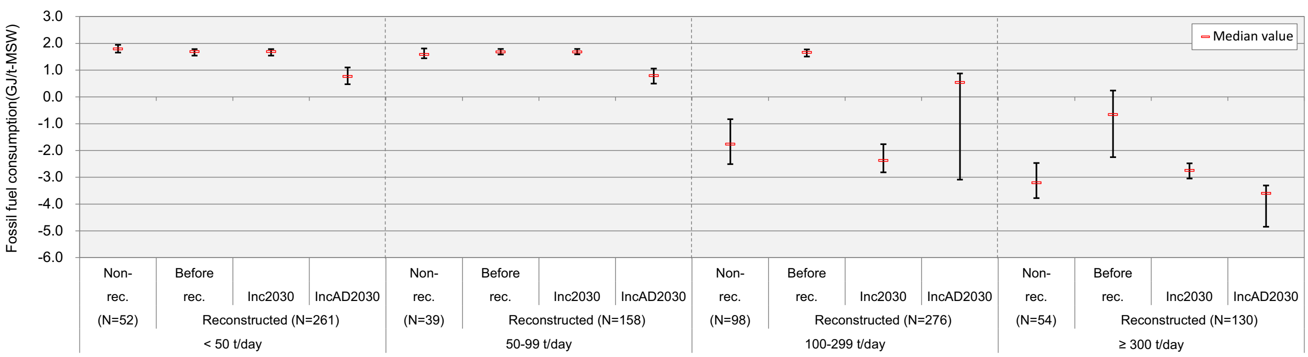
The results of fossil fuel consumption and GHG emissions of each facility differed with treatment capacity, or in other words, with the size of the authority. Figures 3 and 4 show the distributions of fossil fuel consumption and GHG emission units per 1 t of MSW, categorized by the treatment capacity of the incineration facility. Treatment capacity is not based on that of the renovated facility but on facilities available in 2011, because the MSW management strategy of each authority is considered from the standpoint of the current situation.

For incineration facilities with capacities of <50 and 50–99 t/day in 2011, reconstruction in *Inc2020* and *Inc2030* did not reduce either fossil fuel consumption or GHG emissions, because there were no electricity production facilities included. GHG emissions could be reduced with the introduction of AD facilities (*IncAD2020* and *IncAD2030*). However, fossil fuel consumption units in both *IncAD2020* and *IncAD2030* had positive values; the amount of energy recovery could thus not equal energy consumption.

For incineration facilities with capacities of 100–299 t/day, results were more complex. When only the incineration facility was reconstructed (*Inc2020* and



(a) 2020



(b) 2030

Fig. 3 Distribution of fossil fuel consumption unit in 2020 (**a**) and 2030 (**b**); treatment capacity categories are based on those of 2011 incineration facilities; range shows 25th–75th percentiles. *Non-rec.* non-reconstruction, *Before rec.* before reconstruction

Inc2030), fossil fuel consumption units had negative values; amounts of energy recovery thus exceeded energy consumption. On the other hand, units in *IncAD2020* and *IncAD2030* had wider distribution and their median values were positive because incineration facilities tended to be scaled down as a result of food waste separation, and electricity production capacity decreased. However, compared with the situation before reconstruction ("before rec." in Fig. 3), introducing AD facilities also resulted in reduced fossil fuel consumption. Because most incineration facilities did not have electricity production facilities before reconstruction, GHG emission units of the reconstructed facilities (*Inc2020*, *IncAD2020*, *Inc2030*, and *IncAD2030*) could be also reduced. As of 2030, both fossil fuel consumption and GHG emission units of non-reconstructed facilities ("non-rec." in Fig. 3) were lower than those of reconstructed facilities ("before rec."). This indicates that incineration facilities that were already reconstructed between 2000 and 2011 and that did not need further reconstruction after 2011 had good energy and environmental performance.

For incineration facilities with capacities of over 300 t/day, introducing AD facilities (*IncAD2020* and *IncAD2030*) provided the best outcome, because even if facilities were scaled down as a result of food waste separation, these had a large enough treatment capacity to produce electricity.

These results show that the introduction of AD facilities is attractive for small authorities, which currently treat <100 t/day of MSW through incineration facilities without energy recovery. An AD facility is also beneficial for large authorities. However, for middle-scale authorities that treat 100–299 t/day, the inclusion of electricity production facilities must be carefully considered because it will significantly influence energy recovery and GHG reduction effects.

Sensitivity analysis

As shown in Figs. 3 and 4, statistical data from 1068 incineration facilities was used to derive results, with the degree of variation indicated through error bars. These variations describe uncertainties related to waste generation and composition differences between treated areas, and to energy recovery and GHG emission/reduction effects related to the introduction of AD facilities.

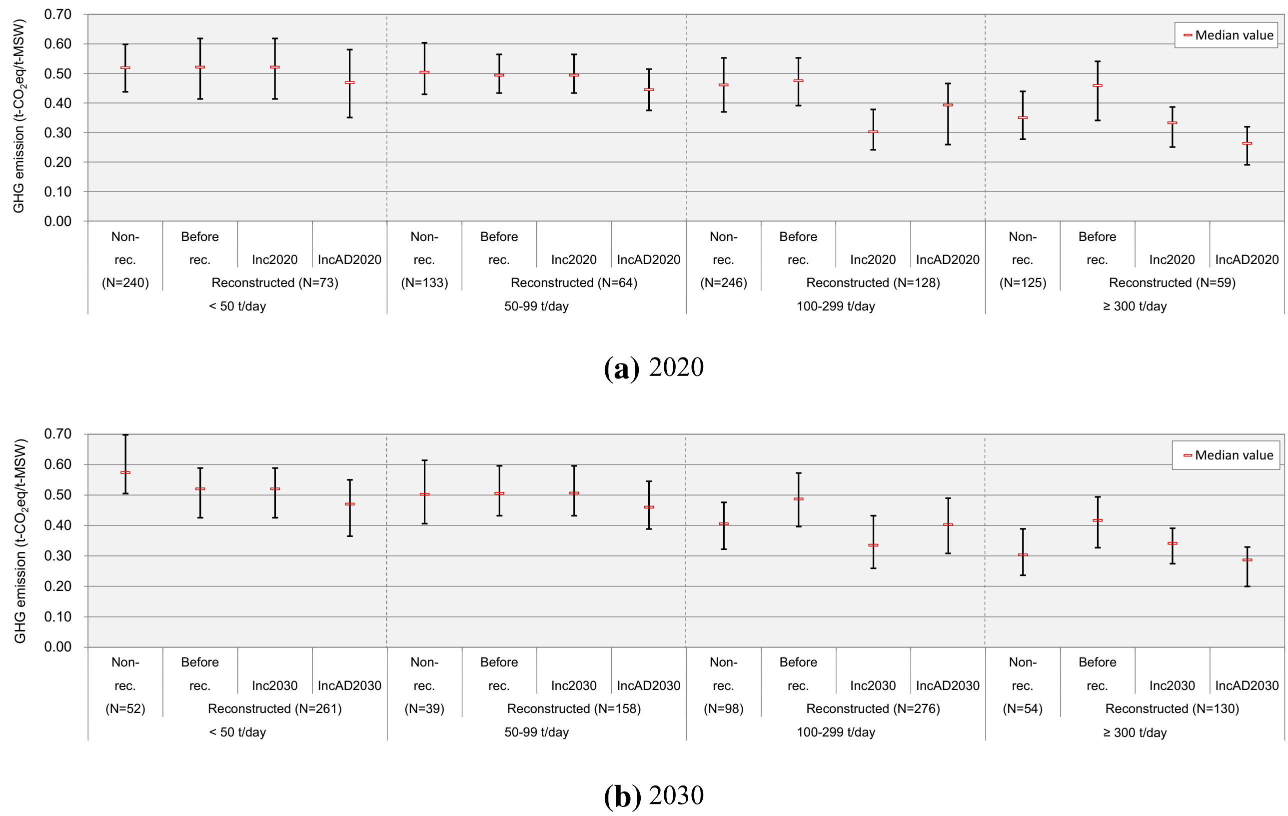


Fig. 4 Distribution of GHG emission unit in 2020 (**a**) and 2030 (**b**); treatment capacity categories are based on those of 2011 incineration facilities; range shows 25th–75th percentiles. *Non-rec.* non-reconstruction, *Before rec.* before reconstruction

However, several uncertainties were still noted in this analysis. Clavreul et al. [29] categorized uncertainties in LCAs for waste management systems using the framework introduced by Huijbregts [45]: model uncertainty, scenario uncertainty, and parameter uncertainty. With regard to parameter uncertainty, sensitivity analysis, which evaluates the influence of input changes on results, is useful [29]. Sensitive parameters for AD of food waste are related to the characteristics of food waste and energy recovery through biogas combustion and incineration [12, 16, 25, 46].

Perturbation analysis, a sensitivity analysis tool to assess the influence of parameter uncertainties [29, 47] (Heijungs and Kleijn, 2001; Clavreul et al. 2012), was applied in this study. Sensitivity ratios (SR) for fossil fuel consumption and GHG emission, defined in Eq. (4), were calculated. Table 8 lists the considered ten parameters. SRs were calculated when each parameter changed by 10 % against the default value

$$SR = \frac{\Delta \text{result}/|\text{default\_default}|}{\Delta \text{parameter}/|\text{default\_parameter}|} \quad (4)$$

Calculated SRs for Japan in general are listed in Table 9 and more detailed information is provided in electronic supplementary material. If a parameter has a SR of 2, this implies that when changing its value by 10 %, the final result is changed by 20 %. Negative value means that result value is decreased when parameter is increased. Overall, parameters related to incineration facilities tended to show higher sensitivity because the amount of MSW treated in these was higher than in AD facilities. For fossil fuel consumption, the three highest sensitivity parameters among all scenarios were LHV of MSW, moisture content in MSW, and electricity consumption in the incineration facility. Electricity production efficiencies for incineration (for treatment capacities of 100–299 and ≥300 t/day) also showed higher SRs (exceeding 0.5) in *Inc2030*. Although a similar tendency was noted in case of GHG emission, SRs were lower than those of fossil fuel consumption. This was because the dominant emission source, direct CO_2 emission from incineration of the plastic fraction in MSW, dulled sensitivity. In addition to three highest sensitivity parameters mentioned above, the CO_2 emission factor of electricity also showed higher sensitivity in all scenarios except for *Inc2011*. Among the ten parameters, change of its value by 10 % resulted in 0.62 million t-CO_2eq/year (19.6 kg-CO2 eq/t-MSW) in *IncAD2030* as maximum case.

Table 8 Parameters considered in sensitivity analysis

No.	Parameters		Default value
1	LHV	MSW	Depends on the facility
2	Moisture content	MSW	Depends on the facility
3		Remaining MSW after food waste separation	21.4 %
4	Electricity consumption	Incineration	Depends on the facility
5		AD	Depends on the facility
6	Electricity production efficiency	Incineration: 100–299 t/day	*Inc2020* and *Inc2030*: 15 % *IncAD2020* and *IncAD2030*: 20 %
7		Incineration: ≥300 t/day	20 %
8		AD	40 %
9	Biodegradation rate in AD		84 %
10	GHG emission factor	Electricity	0.439 kg-CO_2eq/kWh

AD anaerobic digestion, *MSW* municipal solid waste

Table 9 Sensitivity ratios (SRs) of overall fossil fuel consumption and GHG emission in Japan

No.	Fossil fuel consumption					GHG emissions				
	2011	2020		2030		2011	2020		2030	
	Inc	Inc	IncAD	Inc	IncAD	Inc	Inc	IncAD	Inc	IncAD
1	−2.86	−1.92	−1.23	−1.67	−0.47	−0.24	−0.36	−0.25	−0.46	−0.16
2	1.90	1.48	1.59	1.35	1.41	−0.68	−0.64	−0.62	−0.61	−0.59
3	–	–	−0.14	–	−0.11	–	–	−0.03	–	−0.03
4	2.01	0.99	1.62	0.72	0.74	0.17	0.19	0.33	0.20	0.26
5	–	–	0.08	–	0.13	–	–	0.02	–	0.04
6	–	−0.31	−0.30	−0.52	−0.49	–	−0.06	−0.06	−0.14	−0.17
7	–	−0.36	−0.22	−0.69	−0.42	–	−0.07	−0.05	−0.19	−0.15
8	–	–	−0.20	–	−0.34	–	–	−0.04	–	−0.12
9	–	–	−0.17	–	−0.28	–	–	−0.03	–	−0.10
10	–	–	–	–	–	−0.10	−0.21	−0.22	−0.30	−0.37

–, not affected by the parameter

Discussions and limitations

This section discusses key results obtained and some limitations of the study:

- The results indicate that net energy recovery could increase, even if MSW generation decreases in future, through reconstruction of aging incineration facilities. However, the effects of future waste prevention and source separation activities were not considered in the scenarios developed in this study. Food waste is generally given higher priority for waste prevention [48, 49]. If MSW generation decreases in future, net energy recovery would also decrease. However, this does not justify deterring waste prevention, because the latter produces several environmental benefits.
- In this study, the residue remaining after digestion was assumed to be incinerated with MSW. However, the use of AD residue, which can be used as a chemical fertilizer, would contribute to a reduction in GHG emissions [27]. The supply and demand balance for each MSW treatment area needs to be considered; making use of AD residue would be more difficult in large local authorities than in small, rural authorities.
- In 2011, 379 facilities were managed by multiple authorities, while the rest were managed by a single authority. The number of incineration facilities was assumed to be constant. However, this study showed that energy recovery from AD and incineration of remaining MSW after food waste separation were significant for reducing fossil fuel consumption and

GHG emissions. To ensure energy recovery from the remaining MSW in incineration facilities, merging facilities managed by local authorities would therefore be one effective option.

- This study assumed that all food waste was source separated and treated in AD facilities in *IncAD2020* and *IncAD2030* to estimate the maximum potential. It was acknowledged that the efficiency of source separation could significantly affect the results. However, it was found challenging to determine appropriate source separation efficiency because most of the local authorities in Japan incinerate food waste without source separation and there is a lack of knowledge in this regard. Nevertheless, it would be beneficial to clarify the effects of source separation efficiency in a further study.
- The treatment methods considered in this study are only some of the options available. Dry methane fermentation could treat not only food waste but also paper and garden waste. Wet methane fermentation could also contribute to mid- and long-term MSW treatment strategies. In this case, instead of combining food waste with incineration, sewage treatment might be applicable. As mentioned before, the introduction of a heat recovery system could be considered, as this would also be effective for further energy recovery and GHG reductions. When considering available options, MSW management systems must therefore be independently evaluated for each area.
- This study mainly focused on energy recovery and GHG reduction effects. However, WTE plants can contribute towards a sustainable society in a number of ways. Incineration could destroy hazardous organic substances and concentrate toxic metals in relatively small amounts of residue. Valuable metals could also be collected from residues [50]. Apart from energy recovery, WTE facilities can thus play a variety of additional roles in waste management.

Conclusions

The purpose of this study was to investigate the potential for energy recovery and GHG emission reduction that can be achieved by introducing AD facilities during reconstruction of aging incineration facilities in Japan. A total of four future scenarios were considered and compared with the current situation. The total amount of MSW treated was assumed to decrease by 8.7 % between 2011 and 2030.

Our conclusions are as follows:

- Compared with the current situation (*Inc2011*), the amount of electricity generated could increase by 34 % in *IncAD2020* and 60 % in *IncAD2030*, by combining AD facilities for food waste with new, high-efficiency incineration facilities for remaining MSW.
- From a life cycle perspective, net energy recovery and GHG emissions in *IncAD2030* were estimated to be 80.5 PJ/year and 10.5 million t-CO_2eq/year, respectively. Net energy recovery was approximately three times greater than in *Inc2011*. A further 3.9 million t-CO_2eq/year could be reduced, compared with *Inc2011*.
- When 1068 facilities were compared based on their treatment capacities, it was found that energy recovery from AD of food waste and from incineration of remaining MSW significantly affected energy and GHG reductions. This was particularly true for middle-scale authorities that treat 100–299 t/day of MSW. In these authorities, the reconstruction of incineration facilities to include electricity production capabilities requires careful consideration, because it will significantly influence energy recovery and GHG reduction effects.

Currently, most household food waste is not utilized. Thus, from the perspective of energy recovery and GHG emission reduction, introducing AD facilities would be an attractive treatment option for local municipalities, where energy recovery is difficult with incineration. It would also be important for mid- and long-term national MSW management strategies.

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