Greenhouse gas emissions from the treatment of household plastic containers and packaging: Replacement with biomass-based materials

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Abstract

The purpose of this study was to quantify the life cycle greenhouse gas (GHG) emissions reduction that could be achieved by replacement of fossil-derived materials with biodegradable, biomass-based materials for household plastic containers and packaging, considering a variety of their treatment options. The biomass-based materials were 100% polylactide or a combination of polybutylene succinate adipate and polylactide. A scenario analysis was conducted considering alternative recycling methods. Five scenarios were considered: two for existing fossil-derived materials (the current approach in Japan) and the three for biomass-based materials. Production and waste disposal of 1 m^3 of plastic containers and packaging from households was defined as the functional unit. The results showed that replacement of fossil-derived materials with biomass-based materials could reduce life cycle GHG emissions by 14–20%. Source separation and recycling should be promoted. When the separate collection ratio reached 100%, replacement with biomass-based materials could potentially reduce GHG emissions by 31.9%. Food containers are a priority for replacement, because they alone could reduce GHG emissions by 10%. A recycling system for biomass-based plastics must be carefully designed, considering aspects such as the transition period from fossil-derived plastics to biomass-based plastics.

Key words

biomass-based plastic, biodegradable plastic, polylactide (PLA), greenhouse gas (GHG), plastic containers and packaging, life cycle analysis, scenario analysis

Introduction

In accordance with the Containers and Packaging Recycling Law, local governments in Japan provide for separate collection of waste plastic containers and packaging from households. Fig. 1 shows a flowchart for the disposal and treatment of waste plastic containers and packaging from households in Japan. Municipalities collect the waste and outsource the recycling to recyclers after removing undesirable contaminated materials and baling (Nishijima *et al.*, 2012). Beverage bottles made of polyethylene terephthalate (PET) are collected separately from other plastic wastes and are not included the analysis. Plastic containers and packaging that are not collected separately are treated through incineration or landfilled as mixed waste by municipalities. According to a waste composition survey reported by the Japan Ministry of the Environment (MOE), the amount of daily household waste generated per capita was 543.8 g (1,326 g per household; average of fiscal years (FY) 2008 and 2010). Excluding PET bottles, plastic containers and packaging accounted for 6.7% of this amount (MOE, 2010 & 2011b). This resulted in 1.7 Mt of waste plastic containers and packaging being generated annually from households in Japan. Approximately 39.3% of this amount was collected separately and recycled.

Figure 1

To reduce greenhouse gas (GHG) emissions derived from waste plastic, effective approaches such as promotion of recycling and replacement of fossil-derived materials with biomass-based materials are required. Various biomass-based materials have been developed and are now being used for plastic containers and packaging in some parts of the world (Siracusa *et al*., 2008; Momani *et al*., 2009; Vink *et al*., 2010; Ammala *et al*., 2011).

A life cycle assessment (LCA) can be applied to quantify the environmental impact of waste plastic recycling (Perugini *et al.*, 2005; Astrup *et al.*, 2009; Nishijima *et al.*, 2012). LCA can be also used to evaluate the effects of using biomass-based plastic containers and packaging. Bohlmann (2004) conducted an LCA for two polymers: polylactide (PLA) packaging derived from corn and Polypropylene (PP) packaging. The LCA revealed that the fossil energy consumption of PLA packaging was lower than that of PP packaging and that the GHG emissions from PLA were nearly equal to those from PP when landfilled. Using LCA, Vidal *et al*. (2007) evaluated the environmental impacts of a multilayer film based on modified starch and PLA and concluded that global warming would be significantly affected compared to other categories such as acidification, eutrophication, and fossil energy depletion. The study also emphasized the importance of the disposal phase, because biomass-based plastics emit CH_4 after biodegradation in a landfill or composting facility, whereas they emit carbon-neutral CO_2 when incinerated (although some types of biomass-based plastic are non-biodegradable). Most previous studies have considered landfilling, incineration, and composting as the primary biomass-based waste plastic treatment methods (Athena Sustainable Material Institute, 2006; IFEU-Heidelberg, 2009; Madival *et al*., 2009), although Piemonte (2011) included anaerobic digestion. Although previous LCAs applied to biomass-based plastic have focused on specific items such as clamshells and bottles, an LCA focusing on biomass-based plastic containers and packaging with various mixed items has not yet been performed.

The purpose of this study was to quantify the life cycle GHG reduction that could be achieved by replacing fossil-derived materials with biodegradable, biomass-based materials for household plastic containers and packaging, considering a variety of treatment options. Pure (100%) PLA and mixtures of polybutylene succinate adipate (PBSA) with PLA were considered ‘biomass-based materials’ and are identified by this term unless otherwise specified. We first classified plastic containers and packaging and determined the material replacement rate. A scenario analysis was then conducted considering alternative treatment methods.

Materials and methods

Replacement with biomass-based materials

Fig. 2 summarized the procedures used to determine the material replacement rate. Plastic containers and packaging were classified into eight groups, consisting of 77 subgroups of items categorized based on their form and intended use (Kyoto City Environmental Policy Bureau, 2008). PET beverage bottles were excluded, because there is a separated collection and recycling system for these items in Japan.

Figure 2

Pure (100%) PLA and a mixture containing 30% PLA and 70% PBSA by weight were considered representative biomass-based materials. The function of PBSA is to reinforce the flexibility of items that are produced from a sheet or film, such as shopping bags. Although both are biodegradable materials, PLA is made from corn and PBSA is derived from fossil fuels. The lowest possible PBSA content of 70%

was assumed considering current technology. The potential for replacement of fossil-derived materials with biomass-based materials was qualitatively determined for each of the 77 subgroups of items based on the following functional properties: 1) durability, 2) ability to act as a gas barrier, 3) heat resistance, 4) impact resistance, and 5) flexibility.

Table 1 lists the 8 classifications and their material replacement rates based on Eq. 1:

$$R_{i,j} = \frac{\sum Wr_i}{\sum Wt_i} \times 100 \tag{1}$$

where i is the plastic group (see Table 1), j is the biomass-based material (100% PLA or PLA+PBSA), $R_{i,j}$ is the material replacement rate of the fossil-derived material of group i with biomass-based material j (wt%), Wr_i is the amount of replaceable plastic weight in plastic group i (t-wet), and Wt_i is the total plastic weight (before replacement) in plastic group i (t-wet).

Table 1

Most containers consist of 100% PLA, whereas most packaging consists of a mixture of PLA and PBSA. Of the total, 86.9% of the plastic containers and packaging was assumed to be replaceable by biomass-based plastics. ‘Other containers’, consisting of commodity cups and packs, were not considered to be replaceable from the standpoint of durability. Table 2 lists the material compositions of each plastic group before and after material replacement.

Table 2

Functional unit

Production and waste disposal of 1 m^3 of plastic containers and packaging from households was defined as the functional unit. The pellet densities of PLA and PBSA are higher than those of fossil-derived materials. Similarly, Madival *et al.* (2009) reported an increase in density of 22.3% per strawberry packaging container. IFEU-Heidelberg (2009) and Khoo *et al*. (2010a & 2010b) also considered the difference in weight between fossil-derived and biomass-based materials. Bohlmann (2004) used the product volume when determining the functional unit. Therefore, the functional unit in this study was based on volume to take into account density differences between the plastic materials. It was assumed that the density of the pellets was equal to the density of the products.

The properties of the treated waste plastic containers and packaging for the eight groups are summarized in Table 3. Material replacement was assumed to have no influence on moisture or ash content when waste plastic containers and packaging were disposed. PLA and PBSA have higher oxygen content and lower carbon content than fossil-derived plastics. Therefore, the biomass-based materials had reduced lower heating values (LHVs) compared to those of the fossil-derived plastics.

Table 3

Scenario development

A comparison was made between the existing case, in which there was no material replacement, and the biomass-based replacement case (replacement case), in which some of the plastic items were replaced with biomass-based materials (100% PLA and a mixture containing 30% PLA and 70% PBSA) based on the material replacement rates shown in Table 2.

Table 4 shows the five scenarios and treatment methods: two for the existing case and the three for the replacement case. The *S1* scenario was based on the actual amounts treated in Japan in FY2008–2010 using each method (MOE, 2010 & 2011b; JCPRA web site; PWMI, 2012). The types of waste plastic containers and packaging were assumed to remain consistent among treatment methods. The percentage of the household plastic waste that was separately collected was defined as the separate collection rate. In *S1*, 39.3% of waste plastic containers and packaging were separately collected and treated in material recycling (MR) or chemical recycling (CR) facilities, and the remaining 60.7% were collected and treated as mixed waste. Production of recycled plastic pellets was considered MR, while coke oven chemical recycling, blast furnace feedstock recycling, and gasification were defined as CR. Liquefaction, which is a chemical recycling method, was excluded because only 0.5% of the collected plastic containers and packaging was treated using this process in FY2010. The treatment methods and their substitutions are listed in Table 5.

Table 4

Table 5

In the replacement case, the recycling system would need to be modified, because existing recycling methods such as chemical recovery via blast furnace reduction and energy recovery (ER) are not suitable for PLA materials due to their lower reductive capacity and lower LHV (Yano *et al*., 2011). Therefore, a PLA recycling system would be needed; two such alternatives were considered in this study. The first method was superheated steam treatment and ring-opening polymerization as proposed by the Japan Bioindustry Association (JBA, 2008 & 2010). This method enables recycling of PLA items into PLA pellets. The other method was hyperthermal hydrolysis combined with anaerobic digestion (AD), which was demonstrated by the bio-cycle project in Kyoto, Japan in FY2007–2009 (ASTEM, 2010). This method produces biogas from PLA items. We assumed that these two treatment methods replaced the current MR and CR methods for the *S3* scenario. Furthermore, we considered an *S5* scenario to estimate the GHG reduction potential of AD with hyperthermal hydrolysis. Previous studies have indicated that composting may also be an environmentally friendly method compared to landfilling (Vidal et al., 2007; IFEU-Heidelberg, 2009). However, we believe that consumers would find it difficult to distinguish biomass-based plastics from fossil-derived plastics. As contamination by the latter is not desirable for the quality of the produced compost, this method was excluded from the study.

Landfill and incineration without ER should be avoided according to the 3R approach (reduce, reuse, and recycle) and the concept of ER should be promoted irrespective of material replacement. Therefore, the *S2* and *S4* scenarios, in which waste plastic containers and packaging collected as mixed waste were treated in an incineration facility with ER, were considered.

System boundary

Fig. 3 shows a flowchart of the processes considered in this study. The system boundary extended from production of the plastic pellets to final disposal, excluding consumer use of the plastic items. Cultivation of corn was included with respect to production of the PLA pellets. Because PLA pellets are mainly imported to Japan from the USA, it was assumed that both corn cultivation and PLA pellet production were carried out in the USA. Production of plastic items, including forming and moulding, was excluded because both biomass-based and fossil-derived plastic items were assumed to be produced by the same manufacturing lines and therefore there would be no relevant differences for the LCA.

Figure 3

Environmental impacts

Global warming was considered an environmental impact and the increase or reduction in emissions of CO_2, CH_4, and N_2O were accounted for. Emissions were characterized using global warming potential (GWP) 100-yr values of 1 for CO_2, 25 for CH_4, and 298 for N_2O (IPCC, 2007). CO_2 emissions derived from biomass were not included because of their carbon-neutral status.

Unit processes and data collection

Important parameters used in this study are listed in Table 6 and each process is briefly introduced below.

NatureWorks developed the PLA production technology and the inventory data they reported (Vink *et al.*, 2010) was used for corn cultivation and PLA pellet production. GHG emissions for PLA pellet production including corn cultivation were reported as 1.80 kg-CO_2eq kg-PLA^{-1} in 2003 (Vink *et al.*, 2003) and 1.26 kg-CO_2eq kg-PLA^{-1} in 2009 (Vink et al., 2010); we used the latter value, as it reflects current technology. Importation from the USA to Japan (by ship) was also considered. Mean GHG emissions from fossil-derived plastic pellet production, 1.60 t-CO_2eq t^{-1} (PWMI, 2009), were used to represent GHG emissions from PBSA pellet production because of a lack of inventory data.

Diesel fuel consumption was calculated for the waste collection process after estimating the net collection distance. Waste plastic containers and packaging were collected separately once a week, whereas those included in mixed waste were collected twice a week.

In the MR process, fossil-derived plastic materials are recycled into plastic pellets, which were assumed to substitute for 50 % of the virgin pellets (MOE, 2011a). Inventory data reported by JCPRA (2007) and the JLCA-LCA database (2008) developed by Life Cycle Assessment Society of Japan (JLCA) were used for the calculations.

Coke oven chemical recycling, blast furnace feedstock recycling, and gasification were included as CR processes. Inventory data reported by JCPRA (2007) was used for these calculations. Coke oven chemical recycling substituted for coking coal, heavy oil, oil coke, and BTX (Benzene, toluene, and xylene). Blast furnace feedstock recycling substituted for coking coal and heavy oil.

In the super heated steam treatment and ring-opening polymerization process proposed by JBA (2008

& 2010), waste plastic containers and packaging are first exposed to super heated steam at 130 ^{o}C. The PLA is easily broken down into oligolactic acid (OLA) and the OLA is dissolved in lactic acid. Recycled PLA is then produced by the ring-opening polymerization process. The net yield of recycled PLA pellets is 90.3%. Residues such as fossil-derived plastic materials were assumed to be incinerated with ER. Energy consumption was calculated using data reported by JBA (2008).

In AD with hyperthermal hydrolysis, the PLA is first hydrolysed in a hydrolysis tank at 80 ºC. The hydrolysed fraction is then biodegraded in a digestion tank using a dry methane fermentation system at 55 ºC. AD does not readily biodegrade PLA or PBSA in the absence of hyperthermal hydrolysis, which enables biodegradation of 72.5% of the hydrolyzed PLA and PBSA. The biogas produced, approximately 590 m^3 m^{-3} per functional unit, is comprised mainly of CH_4 (57%) and can be used for gas engine power generation with an efficiency of 37.3%. The residue remaining after digestion was assumed to be incinerated with other combustible waste. Inventory data for these processes was based on the results of a pilot-scale study in Kyoto (ASTEM, 2010).

Electricity consumption was calculated for the incineration process using an empirical formula (NIES, 2008) based on waste composition. CO_2 emissions from fossil-derived carbon including PBSA in waste plastics were calculated using the elemental composition. For incineration with ER, a steam turbine was assumed to be used for production of electricity with 10.0% efficiency. The residue remaining after incineration was assumed to be landfilled.

For landfilling, energy consumption for leachate treatment was calculated. It was assumed that the landfill was semi-aerobic and there was no biogas collection. Biomass-based materials undergo partial biodegradation when placed in a landfill. CO_2 emissions from PBSA and CH_4 emissions from both PLA and PBSA biodegradation were calculated using the biodegradation rates reported by Kolstad *et al.* (2012).

Table 6

Results

In *S1*, GHG emissions from plastic pellet production, recycling of separately collected waste plastic containers and packaging, and treatment of mixed waste accounted for 46.6%, 14.9%, and 38.5% of the life cycle GHG emissions, respectively (Fig. 4). Use of ER resulted in an increase in GHG emissions from 1.24 t-CO_2eq m^{-3} (*S1*) to 1.44 t-CO_2eq m^{-3} (*S2*).

Figure 4

For the replacement case, *S3*, *S4*, and *S5* reduced total GHG emissions by 13.8–19.7% compared to *S1*. GHG emissions from plastic pellet production increased by 25.9% (to 1.90 t-CO_2eq m^{-3}), compared to that in *S1*. However, GHG emissions from recycling separately collected waste plastics and treatment of mixed waste decreased substantially. For the latter, GHG emissions decreased from 1.24 in *S1* to 0.77 t-CO_2eq m^{-3} in *S3*. This was because material replacement reduced GHG emissions from incineration with and without ER by 0.44 and 0.15 t-CO_2eq m^{-3}, respectively, while landfilling increased GHG emissions by 0.12 t-CO_2eq m^{-3}.

The lowest GHG emissions were estimated to be 2.60 t-CO_2eq m^{-3} for *S5*; in this scenario, anaerobic digestion reduced GHG emissions by 0.07 t-CO_2eq m^{-3} as a reduction effect.

Discussion

Comparison between plastic groups

Estimated GHG emissions varied among plastic groups (Fig. 5). Use of biomass-based plastic containers clearly reduced GHG emissions. In contrast, use of biomass-based plastic packaging (excluding other packaging) increased GHG emissions due to pellet production phase and the PBSA material used for packaging, which emits fossil-derived CO_2 when incinerated.

Replacement of food containers, which represent the second-highest proportion of plastic household waste, would be a priority, as this could result in reduction of 321 kg-CO_2eq per functional unit, 10.0% of total GHG emissions in *S1*.

Figure 5

Recycling and treatment systems

For the existing case, GHG emissions from treatment of mixed waste in *S2* are higher than those in *S1* (Fig. 4). This is because CO_2 is emitted from fossil-derived materials during incineration, whereas there are no direct GHG emissions from a landfill site. The reduction in GHG emissions by using ER in the incineration facility was not enough to cancel out these increased GHG emissions. Unlike in the existing case, in the replacement case there were no major differences in GHG emissions from treatment of mixed waste between *S3* and *S4*. This is because both incineration and landfilling result in GHG emissions, including CO_2 emissions derived from PBSA during incineration, and CO_2 emissions from PBSA and CH_4 emissions from the landfill site. Non-decomposed biomass-based carbon in the landfill site was not considered.

The effect of material replacement in the system varied with the separate collection rate. Fig. 6 shows the relationship between GHG emissions and the separate collection rate for each scenario. These results indicate that material replacement is advantageous regardless of the separate collection rate. If the separate collection rate reached 100%, GHG emissions for *S3* could potentially be reduced to 2.21 t-CO_2eq m^{-3}— a reduction of 31.9% compared to emissions for *S1*, although there are some limitations such as meeting the required quality for recycling.

Figure 6

The intensity of GHG emissions from recycling of separately collected waste plastics and treatment of mixed waste are listed in Table 7 for each scenario. The GHG emissions intensity for recycling was 1.22 t-CO_2eq m^{-3} for both *S1* and *S2*. In contrast, the GHG emissions intensity for treatment increased from 2.05 to 2.37 t-CO_2eq m^{-3} from *S1* to *S2*. These results imply that promoting ER for incineration of mixed waste would indirectly become a driving force for promoting separate collection of waste plastic containers and packaging to reduce GHG emissions. The GHG emissions intensity of 0.31 t-CO_2eq m^{-3} for recycling of separately collected plastc in *S3* was much lower than that for treatment of mixed waste, 1.27 t-CO_2eq m^{-3}. For all cases, source separation and recycling should be promoted; the recycling system for biomass-based materials needs to be developed similar to the existing recycling system for fossil-derived materials.

Table 7

The effects of material replacement depended on the combination of recycling methods. Therefore, a recycling system for biomass-based plastics needs to be designed carefully, considering various aspects. For instance, super heated steam treatment and ring-opening polymerization reduces the demand for virgin PLA pellets and does not compete with food demand for corn. The treatment of biomass-based plastic containers and packaging along with food waste is also possible.

This study compared an existing recycling case with a replacement case using various treatment methods. It is evident that some time will be required to introduce biomass-based plastic containers and packaging; while designing the treatment system, the transition period from fossil-derived plastics to biomass-based plastics should be considered.

Uncertainties

Several uncertainties were found to exist in this analysis. Clavreul *et al*. (2012) categorized uncertainties in LCAs for waste management systems using the framework introduced by Huijbregts *et al*. (1998): model uncertainty, scenario uncertainty and parameter uncertainty. For instance, Clavreul *et al*. (2012) identified the choice of a specific technology as a scenario uncertainty. The weighted average of electricity efficiency of 11.6% for incineration facilities with ER in Japan was lower than that of 21.6% in the EU (CEWEP, 2012) due to such factors as lower calorific value of waste. They also considered the substitution rate a parameter uncertainty. Nishijima *et al*. (2012) indicated that the substitution rate for MR was the most sensitive parameter when LCA was applied to recycling waste plastic containers and packaging. Some examples of parameters with uncertainties are as follow: the composition percentage of PBSA, material replacement rate, energy consumption of each process, and substitution rate for MR. These will be discussed further in future research.

In the present study, one uncertainty associated with GHG emissions intensity of PBSA production was evaluated. Replacement with packaging containing 70% PBSA tends to increase GHG emissions. However, it was assumed that GHG emissions from PBSA pellet production were equal to mean of those derived from fossil fuels. An uncertainty analysis was conducted using the range of GHG emissions from fossil-derived plastic pellet production, 1.33–1.94 t-CO_2eq t^{-1} (PWMI, 2009), because it was difficult to identify the actual GHG emissions of PBSA pellet production. Fig. 7 shows the range of GHG emissions for plastic packaging associated with the pellet production phase. These results confirmed that GHG emissions from the pellet production phase for the replacement case were higher than those for the existing case. The life cycle GHG emissions for *S3* ranged from 2.66 to 2.94 t-CO_2eq m^{-3}, which varies from *S1* emissions of 2.79 t-CO_2eq m^{-3} by -4.5% to 5.5%.

Figure 7

Other uncertainties include: 1) public preferences for separate collection procedures will result in different collection rate among the plastic groups and 2) the technology used for biomass-based plastic pellet production and recycling methods are underdeveloped. Life cycle GHG emissions associated with replacement with biomass-based materials will decrease in the future if technology improves.

Conclusions

This study quantified the life-cycle GHG emissions reduction achieved by replacement of fossil-derived plastics with biomass-based materials for household plastic containers and packaging, considering various treatment options.

Our conclusions are as follows:

- Replacement with biomass-based materials could reduce life cycle GHG emissions by 14–20% compared to the *S1* scenario using fossil-derived materials.
- Source separation and recycling should be promoted. With a separate collection rate of 100%, replacement with biomass-based materials could potentially reduce GHG emissions to 2.21 t-CO_2eq m^{-3}— a reduction of 31.9% compared to *S1*.
- Food containers are the priority replacement groups because they could reduce GHG emissions by 10% compared to emissions in *S1*.

In future research, uncertainty analysis will be conducted to quantify uncertainties and identify critical parameters that should be carefully considered in development of biomass-based material treatment and recycling systems.

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Figure captions

Fig. 1: Flowchart of disposal of waste plastic containers and packaging from households in Japan (mean of FY2008–2010; MOE 2010 & 2011b; JCPRA website; PWMI 2012).

Fig. 2: Procedure for determining the material replacement rate.

Fig. 3: The processes considered in this study.

Fig. 4: GHG emissions from treatment of waste plastic containers and packaging.

Fig. 5: GHG emissions for *S1* and *S3* (a) per functional unit (1m^{-3} of plastic containers and packaging) and (b) total by plastic group.

Fig. 6: Relationship between GHG emissions and the separate collection rate.

Fig. 7: Uncertainties (error bar) in GHG emissions for certain plastic groups during the pellet production phase (a) per functional unit (1 m^{-3} of plastic containers and packaging) and (b) total by plastic group.

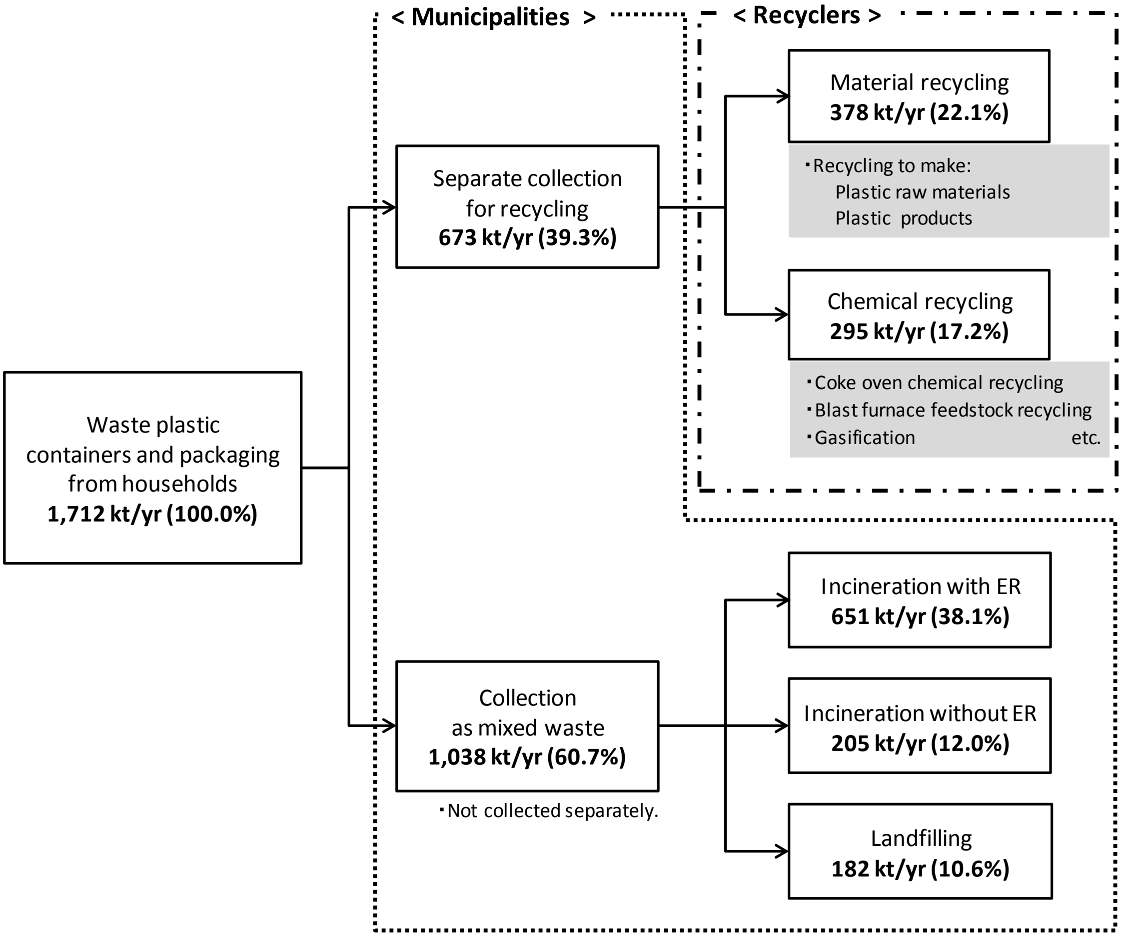
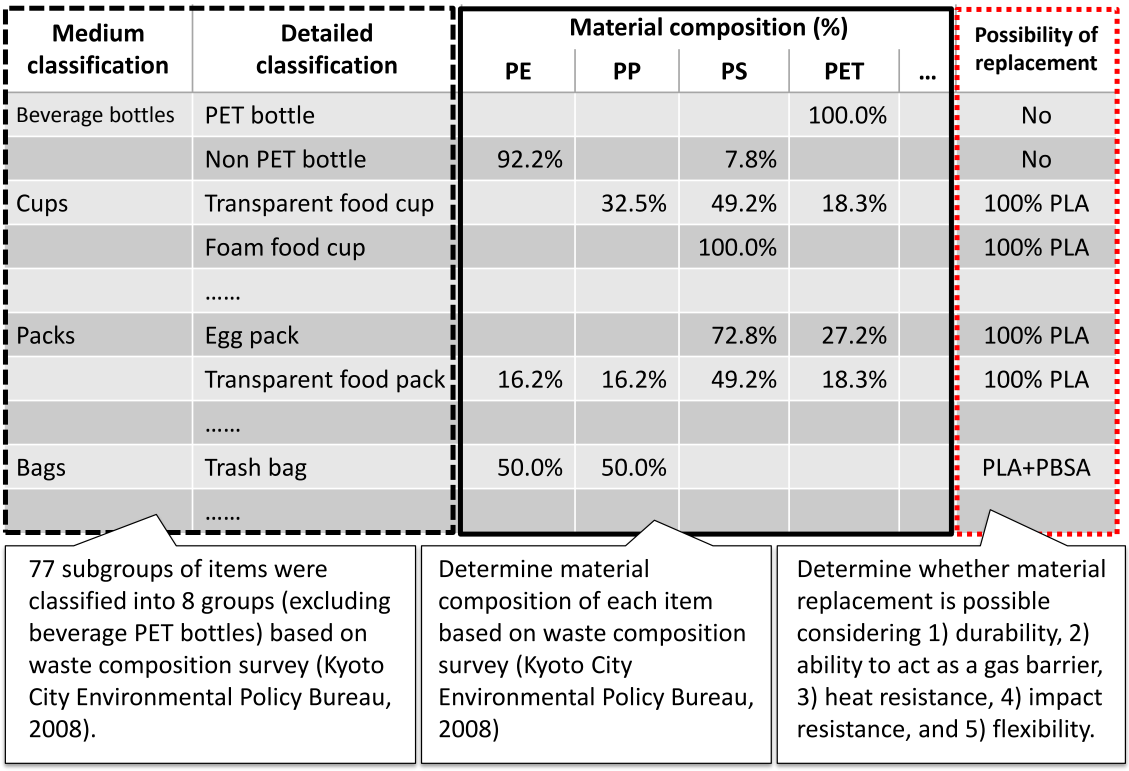


Fig. 1: Flowchart of disposal of waste plastic containers and packaging from households in Japan (mean of FY2008–2010; MOE 2010 & 2011b; JCPRA website; PWMI 2012.

ER: energy recovery. Beverage bottles made of polyethylene terephthalate (PET) are collected separately from other plastic wastes and were excluded. We assumed that the distribution of household waste between incineration with ER, incineration without ER, and landfills was the same as that for municipal solid waste (including both household and business waste).



Medium classification	Detailed classification	Material composition (%)					Possibility of replacement
		PE	PP	PS	PET	…	
Beverage bottles	PET bottle				100.0%		No
	Non PET bottle	92.2%		7.8%			No
Cups	Transparent food cup		32.5%	49.2%	18.3%		100% PLA
	Foam food cup			100.0%			100% PLA
	……						
Packs	Egg pack			72.8%	27.2%		100% PLA
	Transparent food pack	16.2%	16.2%	49.2%	18.3%		100% PLA
	……						
Bags	Trash bag	50.0%	50.0%				PLA+PBSA
	……						

77 subgroups of items were classified into 8 groups (excluding beverage PET bottles) based on waste composition survey (Kyoto City Environmental Policy Bureau, 2008).

Determine material composition of each item based on waste composition survey (Kyoto City Environmental Policy Bureau, 2008)

Determine whether material replacement is possible considering 1) durability, 2) ability to act as a gas barrier, 3) heat resistance, 4) impact resistance, and 5) flexibility.

Fig. 2: Procedure for determining material replacement rate.

PE: polyethylene, PP: polypropylene, PS: polystyrene, PET: polyethylene terephthalate, PLA: polylactide, PBSA: polybutylene succinate adipate.

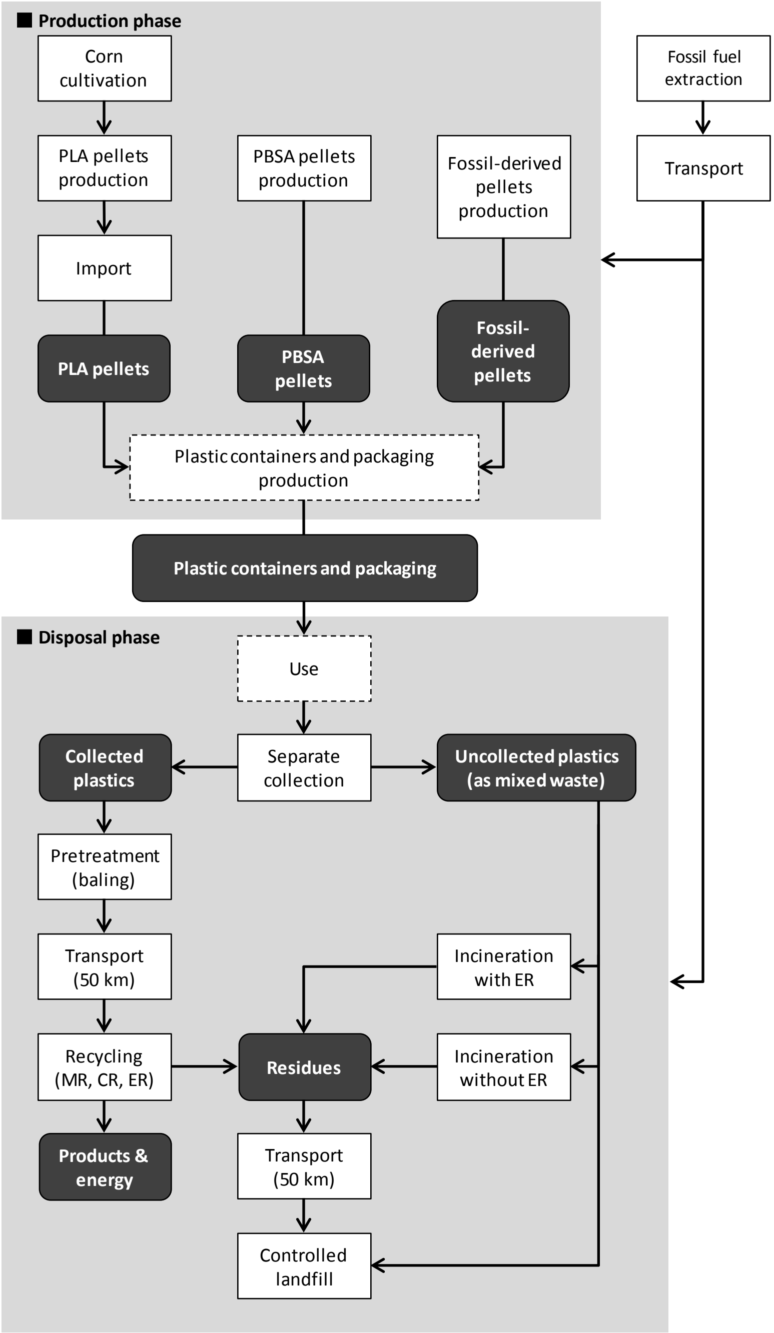


Fig. 3: The processes considered in this study.

White squares: processes; black squares: products and energy; dotted white squares: excluded processes.

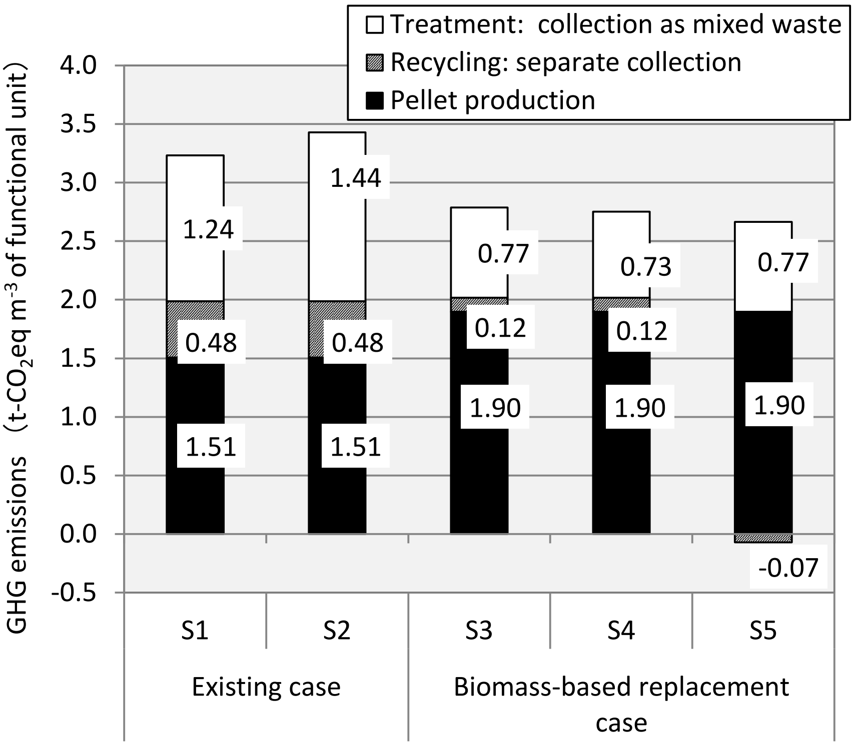
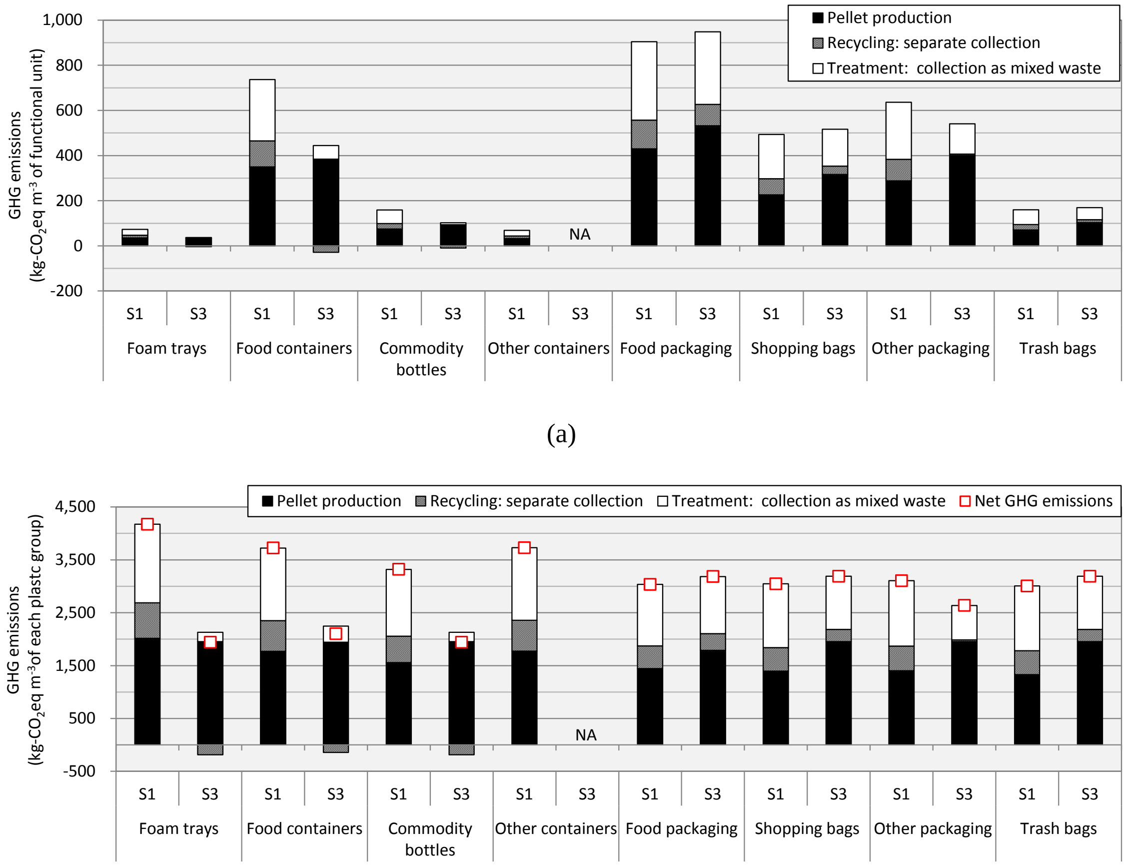


Fig. 4: GHG emissions from treatment of waste plastic containers and packaging.



(a)

(b)

Fig. 5: GHG emissions for *S1* and *S3* (a) per functional unit (1m^{-3} of plastic containers and packaging) and (b) total by plastic group. There was no material replacement for ‘other containers’ (indicated by ‘NA’).

Fig 5a considered the composition of the plastic containers and packaging (see Table 1).

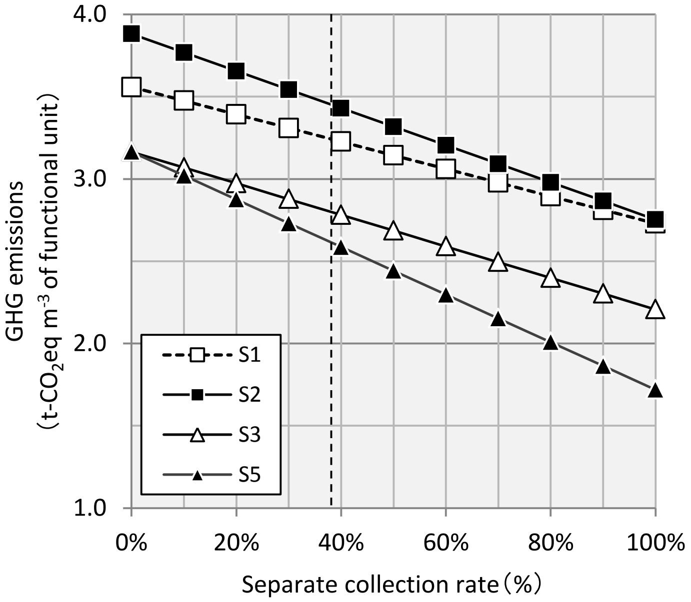
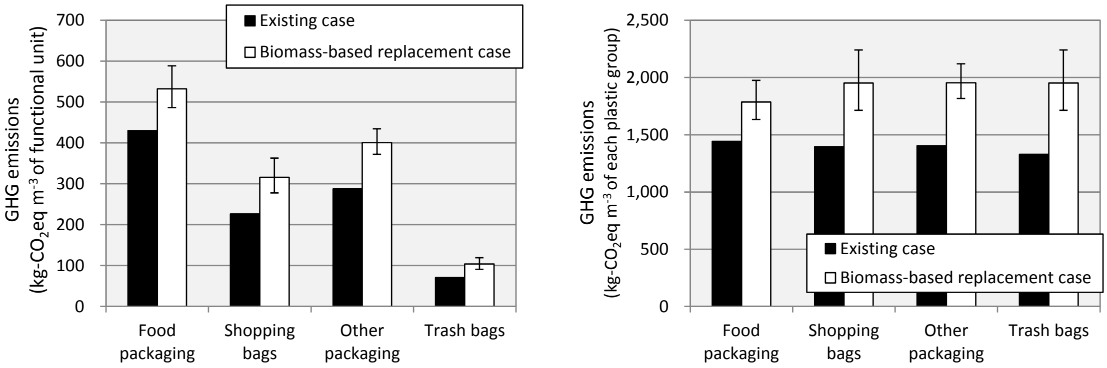


Fig. 6: Relationship between GHG emissions and the separate collection rate.

The vertical dotted line is the mean of the separate collection rates (39.3%) in Japan for FY2008–2010.



(a) (b)

Fig. 7: Uncertainties (error bar) in GHG emissions for certain plastic groups during the pellet production phase (a) per functional unit (1 m^{-3} of plastic containers and packaging) and (b) total by plastic group.

Table captions

Table 1: Material replacement rate for each type of plastic container and packaging [wt%].

Table 2: Material composition of each plastic container and packaging group [wt%].

Table 3: Properties of plastic containers and packaging from household waste.

Table 4: Scenarios and treatment methods.

Table 5: Treatment methods and their substitutions.

Table 6: Important parameters for unit processes.

Table 7: Intensity of GHG emissions for treatment and recycling of waste plastic containers and packaging.

Table 1: Material replacement rate for each type of plastic container and packaging [wt%].

Group		Composition	Material replacement rate		No replacement
			100% PLA	PLA+PBSA	Fossil-derived
Containers	Foam trays	1.9	100.0	0.0	0.0
	Food containers	21.0	91.2	0.0	8.8
	Commodity bottles	5.2	100.0	0.0	0.0
	Other containers	2.0	0.0	0.0	100.0
Packaging	Food packaging	30.2	0.0	69.0	31.0
	Shopping bags	15.2	0.0	100.0	0.0
	Other packaging	19.5	43.0	57.0	0.0
	Trash bags	5.1	0.0	100.0	0.0
Total		100.0	34.6	52.3	13.1

Composition is the percentage that each group makes up of the total household plastic waste before separate collection.

Table 2: Material composition of each plastic container and packaging group [wt%].

		Before material replacement								
		LDPE	HDPE	PP	PS	EPS	PET	PVC	100% PLA	PLA+PBSA
	Group Density:	0.92	0.95	0.90	1.04	1.04	1.29	1.30	1.26	1.24
Containers	Foam trays	0.0	0.0	0.0	0.0	100.0	0.0	0.0	0.0	0.0
	Food containers	0.0	15.4	18.2	25.3	25.3	15.7	0.1	0.0	0.0
	Commodity bottles	0.0	31.5	31.5	0.0	0.0	37.0	0.0	0.0	0.0
	Other containers	0.0	16.2	16.2	49.2	0.0	18.4	0.0	0.0	0.0
Packaging	Food packaging	22.7	22.7	34.4	0.0	0.0	0.2	20.0	0.0	0.0
	Shopping bags	100.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Other packaging	23.3	23.3	41.8	11.1	0.0	0.4	0.1	0.0	0.0
	Trash bags	50.0	50.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
		After biomass-based replacement								
		LDPE	HDPE	PP	PS	EPS	PET	PVC	100% PLA	PLA+PBSA
Containers	Foam trays	0.0	0.0	0.0	0.0	100.0	0.0	0.0	0.0	0.0
	Food containers	0.0	1.4	1.6	2.2	2.2	1.4	0.0	91.2	0.0
	Commodity bottles	0.0	0.0	0.0	0.0	0.0	0.0	0.0	100.0	0.0
	Other containers	0.0	16.2	16.2	49.2	0.0	18.4	0.0	0.0	0.0
Packaging	Food packaging	7.0	7.0	10.7	0.0	0.0	0.1	6.2	0.0	69.0
	Shopping bags	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	100.0
	Other packaging	0.0	0.0	0.0	0.0	0.0	0.0	0.0	43.0	57.0
	Trash bags	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	100.0

LDPE: low-density polyethylene, HDPE: high-density polyethylene, PP: polypropylene, PS: polystyrene, EPS: expanded polystyrene, PET: polyethylene terephthalate, PVC: polyvinyl chloride, PLA: polylactide, PBSA: polybutylene succinate adipate.

Table 3: Properties of plastic containers and packaging from household waste.

			Existing case	Biomass-based replacement case
Material replacement rate	PLA100%	dry%	0.0	34.6
	PLA+PBSA	dry%	0.0	52.3
Density of pellets		tonne m^{-3}	0.98	1.21
Lower heating value		MJ $t\text{-}wet^{-1}$	38,254	23,490
Moisture content		wt%	7.7	7.7
Ash content		wt%	1.9	1.9
Combustible content		wt%	90.4	90.4
	Fossil-derived carbon	wt%	74.6	32.6
	Biomass-based carbon	wt%	0.0	22.7

Table 4: Scenarios and treatment methods.

Existing case			*S1*	*S2*	
Separate collection					
	MR	Plastic pellet production	22.1%	22.1%	
	CR	Coke oven chemical recycling	1.9%	1.9%	
		Blast furnace feedstock recycling	10.7%	10.7%	
		Gasification	4.6%	4.6%	
Collection as mixed waste					
	ER	Incineration with ER	38.1%	60.7%	
	No use	Incineration without ER	12.0%	0.0%	
		Landfilling	10.6%	0.0%	
Biomass-based replacement case			*S3*	*S4*	*S5*
Separate collection					
	CR	Super heated steam treatment and ring-opening polymerization	22.1%	22.1%	0.0%
	ER	Hyperthermal hydrolysis and anaerobic digestion	17.2%	17.2%	39.3%
Collection as mixed waste					
	ER	Incineration with ER	38.1%	60.7%	38.1%
	No use	Incineration without ER	12.0%	0.0%	12.0%
		Landfilling	10.6%	0.0%	10.6%

MR: material recycling; CR: chemical recycling; ER: energy recovery.

Table 5: Treatment methods and their substitutions.

	Treatment method	Substitution for	Specific features
Existing case			
MR	Plastic pellet production	Virgin plastic pellets	Substitution rate: 50%
CR	Coke oven chemical recycling	Coking coal, heavy oil, oil coke, BTX	
	Blast furnace feedstock recycling	Coking coal, heavy oil	
	Gasification	Ammonia, carbon-rich gas	
Biomass-based replacement case			
CR	Super heated steam treatment and ring-opening polymerization	Virgin PLA pellets	Substitution rate: 100%
ER	Hyperthermal hydrolysis and anaerobic digestion	Electricity	Power generation by gas engine (efficiency: 37.3%)
Both cases			
ER	Incineration with ER	Electricity	Power generation by steam turbine (efficiency: 10.0%)
-	Incineration without ER	Nothing	
	Landfilling	Nothing	

MR: material recycling; CR: chemical recycling; ER: energy recovery; BTX: benzene, toluene, and xylene; PLA: polylactide.

Electricity refers to commercial electricity from utility companies.

Table 6: Important parameters for unit processes.

Processes and parameters			**Value**	**Unit**	**Specific features**	**Reference**
Fossil-derived pellet production						
	GHG emissions intensity	LDPE pellets	1.52	t-CO$_2$eq tonne^{-1}		PWMI (2009)
		HDPE pellets	1.33	t-CO$_2$eq tonne^{-1}		PWMI (2009)
		PP pellets	1.48	t-CO$_2$eq tonne^{-1}		PWMI (2009)
		PS pellets	1.92	t-CO$_2$eq tonne^{-1}		PWMI (2009)
		EPS pellets	1.94	t-CO$_2$eq tonne^{-1}		PWMI (2009)
		PET pellets	1.58	t-CO$_2$eq tonne^{-1}		PWMI (2009)
		PVC pellets	1.45	t-CO$_2$eq tonne^{-1}		PWMI (2009)
Biomass-based pellet production						
	GHG emissions intensity	PLA pellets	1.26	t-CO$_2$eq tonne^{-1}		Vink *et al*. (2010)
		PBSA pellets	1.60	t-CO$_2$eq tonne^{-1}	Mean for fossil-derived pellet production.	Assumed
Importation of PLA pellets						
	GHG emissions intensity		0.29	t-CO$_2$eq tonne^{-1}	Assumed equal to corn imports from the USA	Nakata (2011), Estimated
Separate collection						
	Diesel consumption	Source separation	12.3	L m^{-3}	Pressed volume basis (0.13 tonne m^{-3}).	Estimated
		Collected as mixed waste	4.1	L m^{-3}	Pressed volume basis (0.37 tonne m^{-3}).	Estimated
Incineration						
	Electricity consumption	Regression coefficient	30.6	kWh tonne^{-1} waste		NIES (2008)
		Regression coefficient	0.0026	kWh MJ^{-1} waste		NIES (2008)
		Regression coefficient	0.0150	kWh m^{-3} wet gas		NIES (2008)

		Regression coefficient	28.3	kWh tonne^{-1} ash		NIES (2008)
	CH_4 emissions intensity	Exhaust gas	0.96	g-CH_4 tonne^{-1}		MOE (2009)
	Moisture content of residue		18.1	%		NIES (2008)
	Electricity production efficiency		10.0	%	For energy recovery case; produced by steam turbine	Assumed
Landfilling						
	Electricity consumption		6.38	kWh tonne^{-1}		Dote *et al*. (1999)
	Diesel consumption		0.763	L tonne^{-1}		Dote *et al.* (1999)
	Biodegradation rate	PLA	85.8	%	Semi-aerobic conditions	Kolstad *et al.* (2012)
		PBSA	85.8	%	Semi-aerobic conditions; assumed to equal to be the rate for PLA	Assumed
	CH_4 content in landfill gas	PLA	14.0	%	Semi-aerobic conditions	Kolstad *et al.* (2012)
		PBSA	14.0	%	Semi-aerobic conditions; assumed to equal to be the rate of PLA	Assumed
Pretreatment (baling of plastic containers and packaging)						
	Electricity consumption		138	kWh tonne^{-1}		Inaba *et al.* (2005)
	Yield of plastic bales		92	%		Inaba *et al.* (2005)
Material recycling						
	Electricity consumption		419	kWh tonne^{-1} of bales		JCPRA (2007)
	Diesel consumption		11	L tonne^{-1} of bales		JCPRA (2007)
	COG consumption		17	m^3 tonne^{-1} of bales		JCPRA (2007)
	Yield of recycled pellets		52	%		JCPRA (2007)
	Substitution rate		50	%		JCPRA (2007)
	GHG emissions reduction		0.378	t-CO_2eq tonne^{-1} of bales		JLCA (2008),

	through substitution				Estimated
Chemical recycling: Coke oven chemical recycling					
	Electricity consumption		307	kWh $tonne^{-1}$ of bales	JCPRA (2007)
	COG consumption		104	m^3 $tonne^{-1}$ of bales	JCPRA (2007)
	GHG emissions reduction through substitution		3.38	t-CO_2eq $tonne^{-1}$ of bales	JCPRA (2007), Esimated
Chemical recycling: Blast furnace feedstock recycling					
	Electricity consumption	Reducing agent production	291	kWh $tonne^{-1}$ of bales	JCPRA (2007)
	COG consumption	Reducing agent production	7.0	m^3 $tonne^{-1}$ of bales	JCPRA (2007)
	Kerosene consumption	Reducing agent production	8.0	L $tonne^{-1}$ of bales	JCPRA (2007)
	Coking coal	Pig iron production	42,500	kg $tonne^{-1}$ of bales	JCPRA (2007)
	Oil coke	Pig iron production	14.0	kg $tonne^{-1}$ of bales	JCPRA (2007)
	GHG emissions reduction through substitution		106	t-CO_2eq $tonne^{-1}$ of bales	JCPRA (2007), Estimated
Chemical recycling: Gasification					
	Electricity consumption		583	kWh $tonne^{-1}$ of bales	JCPRA (2007)
	Diesel consumption		0.428	L $tonne^{-1}$ of bales	JCPRA (2007)
	Natural gas consumption		0.005	m^3 $tonne^{-1}$ of bales	JCPRA (2007)
	GHG emissions reduction through substitution		7.15	t-CO_2eq $tonne^{-1}$ of bales	JCPRA (2007), Estimated
Super heated steam treatment and ring-opening polymerization					
	Electricity consumption	Super heated steam treatment	27.0	kWh $tonne^{-1}$ of bales	JBA (2008)
		Ring-opening	2,120	kWh $tonne^{-1}$ of PLA	JBA (2008)

		polymerization and PLA pellet production				
	LPG consumption	Super heated steam treatment	14.2	m^3 $tonne^{-1}$ of bales		JBA (2008)
	Yield of recycled PLA pellets		90.3	%		JBA (2008), Estimated
Hyperthermal hydrolysis and anaerobic digestion						
	Electricity consumption	Anaerobic digestion	440	kWh $tonne^{-1}$ of total solids		ASTEM (2010)
		Wastewater treatment	32.5	kWh m^{-3} of wastewater		ASTEM (2010)
	Biodegradation rate	PLA	72.5	%		ASTEM (2010)
		PBSA	72.5	%	Assumed to equal to be the ratio of PLA.	Assmp.
	CH_4 emissions intensity	Wastewater	5.9	g-CH_4 m^{-3} of wastewater		MOE (2009)
	Electricity production efficiency		37.3	%	Produced by gas engine.	ASTEM (2010)

COG: coke oven gas; LPG: liquefied petroleum gas.

Table 7: Intensities of GHG emissions of treatment and recycling of waste plastic containers and packaging ($t\text{-}CO_2eq\ m^{-3}$ of waste plastic containers and packaging).

	Existing case		Biomass-based replacement case		
	S1	*S2*	*S3*	*S4*	*S5*
Treatment: collection as mixed waste	2.05	2.37	1.27	1.21	1.27
Recycling: separate collection	1.22	1.22	0.31	0.31	-0.18