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Approaches to solving China's marine plastic pollution and CO_2 emission problems

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ABSTRACT

Global contamination of the oceans by waste plastics is of increasing concern. Besides being the largest emitter of CO_2 in the world, China is suspected of being the largest contributor to marine plastic waste pollution. Responsible for the latter is the still inadequate management of waste in China, a significant improvement of which is necessary for addressing the issue of marine plastic pollution. Since plastics are hydrocarbons, submitting them to appropriate waste treatment/recycling technologies could contribute to mitigating the emission of CO_2, indicating the possibility of addressing the two environmental issues simultaneously. Based on the combined use of waste input–output and linear programming, we investigated options for mitigating CO_2 emissions under consideration of alternative waste treatment/recycling processes applied to waste plastics of China. It was found that of the nine processes considered, four could result in a net reduction in the emission: a win-win situation.

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Plastic waste; marine contamination; waste management; LP; WIO

1. Introduction

The millions of tons of plastic waste discarded annually in China can physically harm wildlife either because plastic waste is itself potentially toxic or because plastic waste absorbs other pollutants (Rochman et al., 2013). Due to the mismanagement of plastic waste, a significant amount of it ends up in the oceans via inland waterways, wastewater outflows, and transport by wind or tides, thus threatening the marine wildlife that ingests it or becomes entangled in plastic debris (Jambeck et al., 2015). Among the many types of plastic waste, microplastics in particular can accumulate in organisms' bodies after ingestion and can even come to be incorporated into tissue (Derraik, 2002; Lönnstedt and Eklöv, 2016; Qiu et al., 2015). The mismanagement of plastic waste, especially the low rate of proper treatment of household and business plastic waste (24% in 2010; Jambeck et al., 2015), creates significant amounts of marine plastic debris. In 2010, China generated 1.32–3.53 million metric tons (MMT) of marine plastic debris and was considered the largest contributor to such pollution (Jambeck et al., 2015). Meanwhile, China is also the largest CO_2 emitter in the world and is facing increasing pressure to reduce its CO_2

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emissions. Since plastics contain a significant amount of carbon, the two global environmental problems of marine plastic pollution and climate change are naturally linked. CO_2 reduction and marine debris mitigation may conflict with each other. For example, if we tried to prevent the disposal of plastic waste into the oceans by increasing the rate of recovery of household and business plastic waste for treatment, say, by incineration, the carbon contained in plastics would end up emitted into the atmosphere as carbon dioxide. However, the waste treatment strategy can be rearranged to avoid the conversion of carbon in plastic waste into carbon dioxide emissions. For example, instead of incinerating plastic waste, which is currently mostly the case in China, we could reuse plastic waste to produce refuse plastic fuel (RPF; Brunner and Rechberger, 2015; Kumar and Samadder, 2017). This motivates an analysis of technology selection to see what the best technology combination for plastic waste treatment is with respect to both CO_2 reduction and eventual marine debris mitigation. In summary, we are interested in finding a well-designed waste management system under which both CO_2 emission and marine plastic debris could be reduced simultaneously.

Treatment and recycling options for plastic waste in the management of municipal solid waste (MSW) in China has been the subject of a number of studies (Chen et al., 2011; Hong et al., 2010). Based on methods of process-based life-cycle assessment, these indicate great potential for CO_2 mitigation of the reorganization of waste management systems. The widespread use of plastics in all sectors of the economy, from agriculture, manufacturing, and services, to final consumers, implies that waste plastics occur also from almost every sector of the economy, more or less proportional to the level of its activity. Assessing the feasibility of waste management options requires a holistic approach based on a well-defined accounting system capable of representing the economy-wide intersectoral flow of plastics throughout a whole life cycle. Input–output analysis (IOA) emerges as the best analytical tool to consider the environmental effects of plastics. To our knowledge, studies that have used input–output frameworks to consider waste plastic issues are limited. Duchin and Lange (1995) considered the recycling of plastics in the United States within an IOA framework. Hoekstra et al. (2006) considered plastic waste and recycled plastics in a numerical example of a full physical input–output table. In their mathematical models, the former treated waste recycling as an exogenous sector while the latter required the one-to-one correspondence between endogenous waste types and treatment processes. To provide a flexible framework without the one-to-one correspondence between waste types and treatment processes, Nakamura and Kondo (2002, 2009) developed the waste input–output analysis (WIO), which is a comprehensive analysis framework that accounts for production, consumption, and waste treatment sectors (Lin, 2009; Tsukui et al., 2015). While IOA has become a standard tool of life-cycle inventory analysis, the use of WIO makes it possible to extend the system boundary by including the end-of-life phase in a highly general fashion.

The best way of mitigating the amount of waste plastics that become marine debris is to cut down at the point of origin, that is, to reduce the use of plastics in the economy. However, this requires substantial changes in both life-styles and technology. A strategy that would be easier to implement is to increase the share of waste plastics submitted to proper treatment via increases in the rate of collection. As previously mentioned, this, however, may result in additional CO_2 emission. There may thus exist a trade-off between the amount of waste plastics recovered to mitigate marine contamination and the CO_2

emission associated with that mitigation. We are interested in identifying the optimal, if any, combination of available waste processes that minimizes the emission of CO_2 for a given rate of waste plastics recovery. In other words, we are interested in obtaining the lowest Pareto frontier for the environmental burdens of improperly discarded plastic waste and CO_2 emission. The term 'improperly discarded plastic waste' refers to the part of plastic waste discarded directly into the natural environment without proper treatment. This leads us to mathematical optimization models aimed at finding the optimal selection of technology under 'what's best scenarios' (Pulido-Velazquez et al., 2008). For a linear model, such as an input–output model, linear programming (LP) is a useful and transparent tool for selecting the 'best scenario' (Azapagic and Clift, 1999; He et al., 2015; Vogstad, 2009; Ohno et al., 2017). The IO-based LP is frequently used to select optimal technologies among alternative options to address both theoretical and empirical problems (Duchin, 2005; Duchin and Lange, 1995; Leontief, 1985; Lin, 2011; Oliveira et al., 2016). Kondo and Nakamura (2005) introduced an LP model based on the WIO model to find an optimal solid waste management and recycling strategy from among a given set of alternative options.

With these backgrounds, this work is aimed at identifying the optimal treatment strategy, if any, of waste plastics to minimize both marine contamination and GHG emissions simultaneously. To account for the inter-sector linkages of the use and discard of plastics based on a rigorous system definition (Brunner and Rechberger, 2015), we developed the first WIO table (Nakamura and Kondo, 2002; 2009) of plastics and waste plastics for China that describes the flow of plastics and waste plastics among producing sectors, final consumers, and waste treatment sectors. This enables us to quantify, among others, the amounts of improperly discarded plastic waste and CO_2 emission resulting from different treatment programmes. Building upon previous work (Kondo and Nakamura 2005; Lin 2011), we use input–output-based LP to identify an optimal way to treat waste plastics in furtherance of the above goal. After a brief introduction to the methodology, we describe the data, and show the results and main findings, before concluding the paper.

2. Methods

2.1. Basic framework

In this study the WIO framework (Nakamura and Kondo, 2002) was applied to the flow of plastic in China, involving production, use, disposal, and recycling (Table 1). We consider n economic sectors, n_t waste treatment processes, and n_w types of wastes. Here, $\mathbf{X}_{I,I}$ ($n \times n$ matrix) refers to the matrix of goods and services flows, and $\mathbf{X}_{I,II}$ ($n \times n_t$ matrix) to the inputs of goods and services to treatment processes, and $\mathbf{W}_I$ ($n_w \times n$ matrix) and $\mathbf{W}_{II}$ ($n_w \times n_t$ matrix) to the waste flows of production sectors and treatment processes, respectively. The elements of these two matrices are positive when waste is generated and negative when waste is recycled. In this work, $\mathbf{x}_{I,df}$ ($n \times 1$ vector) is the vector of final demand for goods and services, $\mathbf{x}_{I,e}$ ($n \times 1$ vector) the vector of exports for goods and services, and $\mathbf{x}_{I,m}$ ($n \times 1$ vector) the vector of imports for goods and services. The total output of goods and services is denoted by $\mathbf{x}_I$. In addition, $\mathbf{w}_{fd}$ ($n_w \times 1$ vector) refers to waste generated by the domestic final demand, $\mathbf{w}_e$ ($n_w \times 1$ vector) to exported waste, and $\mathbf{w}_m$ ($n_w \times 1$ vector) to imported waste, while $\mathbf{w}$ ($n_w \times 1$ vector) denotes the total waste generation. Further $\mathbf{e}_I$ ($1 \times n$ vector) is the emission from production sectors, $\mathbf{e}_{II}$ ($1 \times n_t$ vector) the emission

Table 1. The framework of a Chinese WIO with plastic waste.

	Goods /services	Recycling/ waste treatment	Domestic final demand	Export	Import	Domestic production
Goods /services	$\mathbf{X}_{\mathrm{I,I}}$	$\mathbf{X}_{\mathrm{I,II}}$	$\mathbf{x}_{\mathrm{I,df}}$	$\mathbf{x}_{\mathrm{I,e}}$	$\mathbf{x}_{\mathrm{I,m}}$	$\mathbf{x}_{\mathrm{I}}$
Plastic Waste	$\mathbf{W}_{\mathrm{I}}$	$\mathbf{W}_{\mathrm{II}}$	$\mathbf{w}_{\mathrm{df}}$	$\mathbf{w}_{\mathrm{e}}$	$\mathbf{w}_{\mathrm{m}}$	$\mathbf{w}$
CO_2	$\mathbf{e}_{\mathrm{I}}$	$\mathbf{e}_{\mathrm{II}}$	e_{f}			e

from treatment processes, and e_{f} (1×1 scalar) the emission generated directly by the final consumption. The emissions from the production of goods and services and waste treatment/recycling are fully considered as far as they occur in China. However, the emissions outside China that are induced by Chinese demand for imports are not considered. Our analysis is static and does not consider any dynamics of waste generation and production. In particular, consideration of repeated cycles of recycling as in Nakamura et al. (2017) is outside the scope of this study.

2.2. The WIO model

Based on WIO modelling, the economy-wide emission of CO_2, e (1×1 scalar), that is induced by the final demand for goods and services and the discard of waste from the final demand is given by

$$e = \mathbf{r}(\mathbf{I} - \mathbf{A})^{-1} \begin{pmatrix} (\mathbf{I} - \hat{\mathbf{m}})\mathbf{x}_{\mathrm{I,df}} + \mathbf{x}_{\mathrm{I,e}} \\ \mathbf{S}\mathbf{H}_{\mathrm{f}} \odot (\mathbf{w}_{\mathrm{df}} - \mathbf{w}_{\mathrm{e}} + \mathbf{w}_{\mathrm{m}}) \end{pmatrix} + e_{\mathrm{f}}, \tag{1}$$

where

$$\mathbf{A} = \begin{pmatrix} (\mathbf{I} - \hat{\mathbf{m}})\mathbf{A}_{\mathrm{I,I}} & (\mathbf{I} - \hat{\mathbf{m}})\mathbf{A}_{\mathrm{I,II}} \\ \mathbf{S}(\mathbf{H}_{\mathrm{I}} \odot \mathbf{G}_{\mathrm{I}}^{\mathrm{out}} - \mathbf{G}_{\mathrm{I}}^{\mathrm{in}}) & \mathbf{S}(\mathbf{H}_{\mathrm{II}} \odot \mathbf{G}_{\mathrm{II}}^{\mathrm{out}} - \mathbf{G}_{\mathrm{II}}^{\mathrm{in}}) \end{pmatrix}.$$

The $1 \times (n + n_{\mathrm{t}})$ vector $\boldsymbol{r}$ is the emission coefficients. Subscript I refers to production sectors and II to waste treatment sectors (Table 1). Accordingly, $\mathbf{A}_{\mathrm{I,I}}$ ($n \times n$ matrix) is the matrix of the input coefficient of production sectors, and $\mathbf{A}_{\mathrm{I,II}}$ ($n \times n_{\mathrm{t}}$ matrix) is the matrix of the input coefficient of treatment processes. The model handles imports according to the Chenery–Moses model (Chenery, 1953; Moses, 1955). Therefore, the proportion of imports to domestic demands $\hat{\mathbf{m}}$ is used to determine the imports endogenously. The ith diagonal element of $\hat{\mathbf{m}}$ ($n \times n$ matrix) is given by $m_i = x^i_{\mathrm{I,m}}/(x^i_{\mathrm{I}} + x^i_{\mathrm{I,m}} - x^i_{\mathrm{I,e}})$. Here, $x^{\mathrm{i}}_{\mathrm{I,m}}$ and $x^{\mathrm{i}}_{\mathrm{I,e}}$ stand for the import and export of goods and services i, respectively, and $x^{\mathrm{i}}_{\mathrm{I}}$ indicates the total output of goods and services i. Also, $\mathbf{G}_{\mathrm{I}}^{\mathrm{out}}$ ($n_{\mathrm{w}} \times n$ matrix) and $\mathbf{G}_{\mathrm{I}}^{\mathrm{in}}$ ($n_{\mathrm{w}} \times n$ matrix), respectively, refer to the matrix of waste generation and waste absorbing (recycling) coefficients of production sectors, with the (i,j)-th element of $\mathbf{G}_{\mathrm{I}}^{\mathrm{in}}$ referring to the amount of waste i recycled per production unit of sector j. The counterparts of $\mathbf{G}_{\mathrm{I}}^{\mathrm{out}}$ and $\mathbf{G}_{\mathrm{I}}^{\mathrm{in}}$ for treatment processes are given by $\mathbf{G}_{\mathrm{II}}^{\mathrm{out}}$ ($n_{\mathrm{w}} \times n_{\mathrm{t}}$ matrix) and $\mathbf{G}_{\mathrm{II}}^{\mathrm{in}}$ ($n_{\mathrm{w}} \times n_{\mathrm{t}}$ matrix), respectively. The allocation matrix of plastic waste into these processes is denoted by $\mathbf{S}$ ($n_{\mathrm{t}} \times n_{\mathrm{w}}$ matrix), the proportion of plastic waste generated from economic sectors treated via proper treatment processes is denoted by $\mathbf{H}_{\mathrm{I}}$ ($n_{\mathrm{w}} \times n$ matrix), and the proportion of plastic waste generated from treatment processes treated via proper treatment processes by $\mathbf{H}_{\mathrm{II}}$ ($n_{\mathrm{w}} \times n_{\mathrm{t}}$ matrix). The Hadamard product is expressed by $\odot$. Improperly discarded waste refers to that portion of plastic waste discarded directly without being submitted to any proper treatment. Therefore, $\mathbf{H}_{\mathrm{I}} \odot \mathbf{G}_{\mathrm{I}}^{\mathrm{out}} - \mathbf{G}_{\mathrm{I}}^{\mathrm{in}}$ and $\mathbf{H}_{\mathrm{II}} \odot \mathbf{G}_{\mathrm{II}}^{\mathrm{out}} - \mathbf{G}_{\mathrm{II}}^{\mathrm{in}}$ give the net waste generation coefficients, whose (i,j)-th element refers to the amount of waste i allocated to proper treatment per production unit of sector j, and l_{f} (1×1 scalar) is the direct CO_2 emission from the final consumption.

The quantity of waste treated via improper processes, d (1×1 scalar), can be attributed to three sources of origin, economic sectors, treatment processes, and final consumption:

$$d = ((\boldsymbol{\iota}\boldsymbol{\iota}' - \mathbf{H}_{\mathrm{I}}) \odot \mathbf{G}_{w,\mathrm{I}}^{\mathrm{out}})\mathbf{x}_{\mathrm{I}} + ((\boldsymbol{\iota}\boldsymbol{\iota}' - \mathbf{H}_{\mathrm{II}}) \odot \mathbf{G}_{\mathrm{II}}^{\mathrm{out}})\mathbf{x}_{\mathrm{II}} + (\boldsymbol{\iota}\boldsymbol{\iota}' - \mathbf{H}_{\mathrm{f}}) \odot \mathbf{w}_{\mathrm{f}}, \tag{2}$$

where $\mathbf{x}_{\mathrm{I}}$ ($n_{\mathrm{w}} \times 1$ vector) refers to the total output of economic sectors and $\mathbf{x}_{\mathrm{II}}$ ($n_{\mathrm{t}} \times 1$ vector) to the treatment quantity of treatment processes, and $\boldsymbol{\iota}$ to a column vector of unity with proper dimensions.

2.3. The WIO-LP model

We next turn to the identification of optimal (emission and marine contamination minimizing) treatment strategies from among the combination of alternative treatment technologies/options based on the WIO-LP methodology (Kondo and Nakamura, 2005). Writing $\mathbf{r}_{\mathrm{I}}$ ($1 \times n$ vector) for the vector of the emission coefficient whose j-th element refers to the CO_2 emission per unit of output of the j-th sector and $\mathbf{r}_{\mathrm{II}}$ ($1 \times n_{\mathrm{t}}$ vector) for the corresponding vector of CO_2 emission coefficients of treatment processes, we consider the following WIO-LP model:

Minimize

$$\mathbf{r}_{\mathrm{I}}\mathbf{x}_{\mathrm{I}} + \mathbf{r}_{\mathrm{II}}\mathbf{x}_{\mathrm{II}}, \tag{3}$$

subject to

$$\mathbf{x}_{\mathrm{I}} = (\mathbf{I} - \hat{\mathbf{m}})\mathbf{A}_{\mathrm{I,I}}\mathbf{x}_{\mathrm{I}} + (\mathbf{I} - \hat{\mathbf{m}})\mathbf{A}_{\mathrm{I,II}}\mathbf{x}_{\mathrm{II}} + (\mathbf{I} - \hat{\mathbf{m}})\mathbf{x}_{\mathrm{I,df}} + \mathbf{x}_{\mathrm{I,e}},$$

$$\mathbf{w} = (\mathbf{H}_{\mathrm{I}} \odot \mathbf{G}_{\mathrm{I}}^{\mathrm{out}} - \mathbf{G}_{\mathrm{I}}^{\mathrm{in}})\mathbf{x}_{\mathrm{I}} + (\mathbf{H}_{\mathrm{II}} \odot \mathbf{G}_{\mathrm{II}}^{\mathrm{out}} - \mathbf{G}_{\mathrm{II}}^{\mathrm{in}})\mathbf{x}_{\mathrm{II}} + \mathbf{H}_{\mathrm{f}} \odot (\mathbf{w}_{\mathrm{df}} - \mathbf{w}_{\mathrm{e}} + \mathbf{w}_{\mathrm{m}}),$$

$$\mathbf{x}_{\mathrm{II}} = \mathbf{S}\mathbf{w}.$$

with respect to

$$\mathbf{x}_{\mathrm{I}}, \mathbf{x}_{\mathrm{II}}, \mathbf{w}, \mathbf{S} \geq 0.$$

Lenzen and Reynolds (2014) and Fry et al. (2015) extended the WIO framework by incorporating a supply-use formalism, resulting in waste supply-use tables (WSUTs). The main multipliers provided by WSUTs were proven to be equivalent to standard WIO (Lenzen and Reynolds, 2014). The LP extension of WSUTs is also equivalent to standard WIO. Following Kondo and Nakamura (2005), Lin (2011), and Ohno et al. (2017), we chose to use the original WIO framework.

3. Data

Following the framework presented in Table 1, we developed a Chinese WIO table for plastic waste for 2007 with 138 plastic-producing sectors, seven types of plastic-related

Table 2. Composition of Plastic Waste.

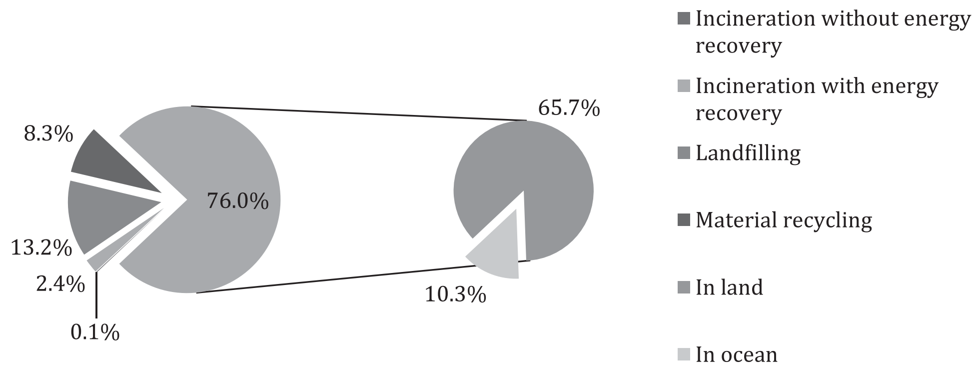
PE	PP	PS	PET	PVC	Other	Metal	Wood	Total
30.2%	21.1%	17.7%	13.8%	4.9%	2.4%	2.6%	7.3%	100.0%

Note: PE refers to polyethylene; PP refers to polypropylene; PS refers to polystyrene; PET refers to polyethylene terephthalate; and PVC refers to polyvinyl chloride. Data source: JCPRA[8].

waste/products, and nine waste treatment processes. The year 2007 was chosen as the benchmark because of the availability of waste input–output data from 2007 (Lin et al., 2014). The seven types of plastic-related waste/products included household and business plastic waste, industrial plastic waste, imported plastic waste, plastic waste pellets, plastic residue, plastic blast furnace (BFs) reductant, and RPF. The nine treatment processes included incineration without energy recovery, incineration with energy recovery, sanitary landfilling, RPF production, liquefaction, ammonia production, gasification (fuel), injection of plastic waste into BFs as a substitute for coke, and plastic pallet production. The last six treatment processes mentioned above are alternative technologies of plastic waste treatment, whereas the first three are traditional waste treatment processes. We considered those alternative technologies because they are frequently discussed in the literature and are considered feasible and efficient technology alternatives (Chen et al., 2011; Nakatani and Hirao, 2010). Specifically, incineration here refers to mixed incineration with other MSW; plastic pallet production converts plastic waste into plastic pallets as a replacement for wooden pallet boards; liquefaction produces light oil, medium oil, heavy oil, hydrochloric acid, and carbide using plastic waste; and gasification produces synthetic gas as fuel. Besides these proper treatment processes, we also considered improper treatment process of plastic waste that contributes to oceanic plastic pollution. We considered one type of emission, namely, CO_2. Here, CO_2 refers to carbon dioxide itself, but not to CO_2 equivalents.

The input coefficients of the treatment/recycling processes were obtained from the Japan Containers and Packaging Recycling Association (JCPRA, 2007). The inventory is related to the composition of plastic waste. To the authors' knowledge, there are no statistical data available on the average composition of plastic waste nationwide, and the composition varies across cities. We used the composition shown in Table 2, which was obtained from JCPRA (2007); these compositions are close to the results of sampling surveys in China on plastic composition (Zhang et al., 2007; Chen et al., 2010). Because the input coefficients of the treatment/recycling processes are mostly determined by the physical condition of plastic waste, we find it reasonable to use the Japanese input–output data for our study due to the similarity of the composition of plastic waste. Furthermore, many MSW treatment technologies used in China were imported from developed countries including Japan (Lu et al., 2017), providing another justification for using Japanese data. Chen et al. (2011) also used the same source of data from JCPRA to study plastic waste issues of China. The transportation of plastic wastes to treatment facilities was considered in this study. The average transportation distances of plastic waste to landfills, incineration facilities, and other recycling facilities were provided by Wang et al. (2001) and Liu (2011). For the details of the compilation methods and data sources, please refer to the appendix.

Figure 1. Allocation of household and business plastic waste in benchmark year.[1]



Note: 0.09% and 2.39% are the proportions of incineration without energy recovery and incineration with energy recovery, respectively.

Table 3. Treatment and recovery scenarios of plastic waste.

	Default scenario: the state of 2007	Optimization scenario
Objective	No optimization	CO_2 minimization
Recovery rate of discarded plastic waste	0–0.8	0–0.8

4. Results

4.1. Scenario settings

In 2007, 76% of household and business plastic waste was improperly discarded, with 21.05% of this waste littering the ocean (Jambeck et al., 2015). Of the remaining fraction (24%) that was treated properly, 0.09%, 2.39%, 13.39%, and 8.33% were treated through incineration without energy recovery, incineration with energy recovery, landfilling, and material recycling, respectively (Figure 1). In other words, 0.39%, 9.94%, 54.96%, and 34.71% of the properly treated plastic waste were treated through incineration without energy recovery, incineration with energy recovery, landfilling, and material recycling, respectively. As for plastic waste originating from industrial (pre-consumer) sources, it was assumed that 90% of it is recycled, which corresponds to the Japanese situation (PWMI, 2017). We justify this assumption by the fact that pre-consumer plastic waste such as industrial packaging is usually recycled at a much higher rate than post-consumer packaging, as it is relatively pure and available from a smaller number of sources with relatively higher volumes (Hopewell et al., 2009).

Because proper treatment of plastic waste may generate carbon emissions while improper treatment leads to marine debris, a trade-off relationship may result. We conducted the scenario analysis presented in Table 3 to assess the relationship between CO_2 emission and the recovery rate of improperly treated plastic waste as well as the effect of treatment strategy rearrangement. We considered two types of scenarios: default and optimization.

[1] Please refer to the Appendix A1 for the data source of the estimation

4.2. Comparison of scenarios

In the default scenario, recovered plastic waste is treated in the same way as the properly treated plastic waste in 2007 as mentioned above. In the optimization scenario, the CO_2 emission is minimized based on Equation (3) by allowing the plastic waste to be allocated among the nine treatment processes. To see the effects of a change in the recovery rate of plastic waste in these two scenarios, the recovery rate was altered from 0 to 0.8, which corresponds to the reduction in the fraction ending up in the oceans from 1.33 to 0.26 MMT.

4.2.1. Default: the state of 2007

The solid line in Figure 2 shows the relation between the recovery rate of improperly treated plastic waste and CO_2 emission when the plastic waste is treated under the default scenario. The horizontal axis shows the recovery rate of improperly discarded plastic waste. The vertical axis shows the difference in CO_2 emissions between the scenarios in question and the 2007 reality. In Figure 2, the lower-right area of the figure is enlarged to show the details. As seen in the figure, the increase in the recovery rate increases CO_2 emissions. The generation of plastic waste in China is expected to increase because of the current boom in the production and use of plastics. Many environmental initiatives by the Chinese government aim to increase the waste recovery rate (Zhang et al., 2010). However, our findings indicate that, under current conditions, increasing the recovery rate of plastic waste will increase CO_2 emissions, thus increasing pressures on China for carbon mitigation. Because the Chinese government has pledged to cap the country's carbon emissions by 2030, better strategies are needed for plastic waste management.

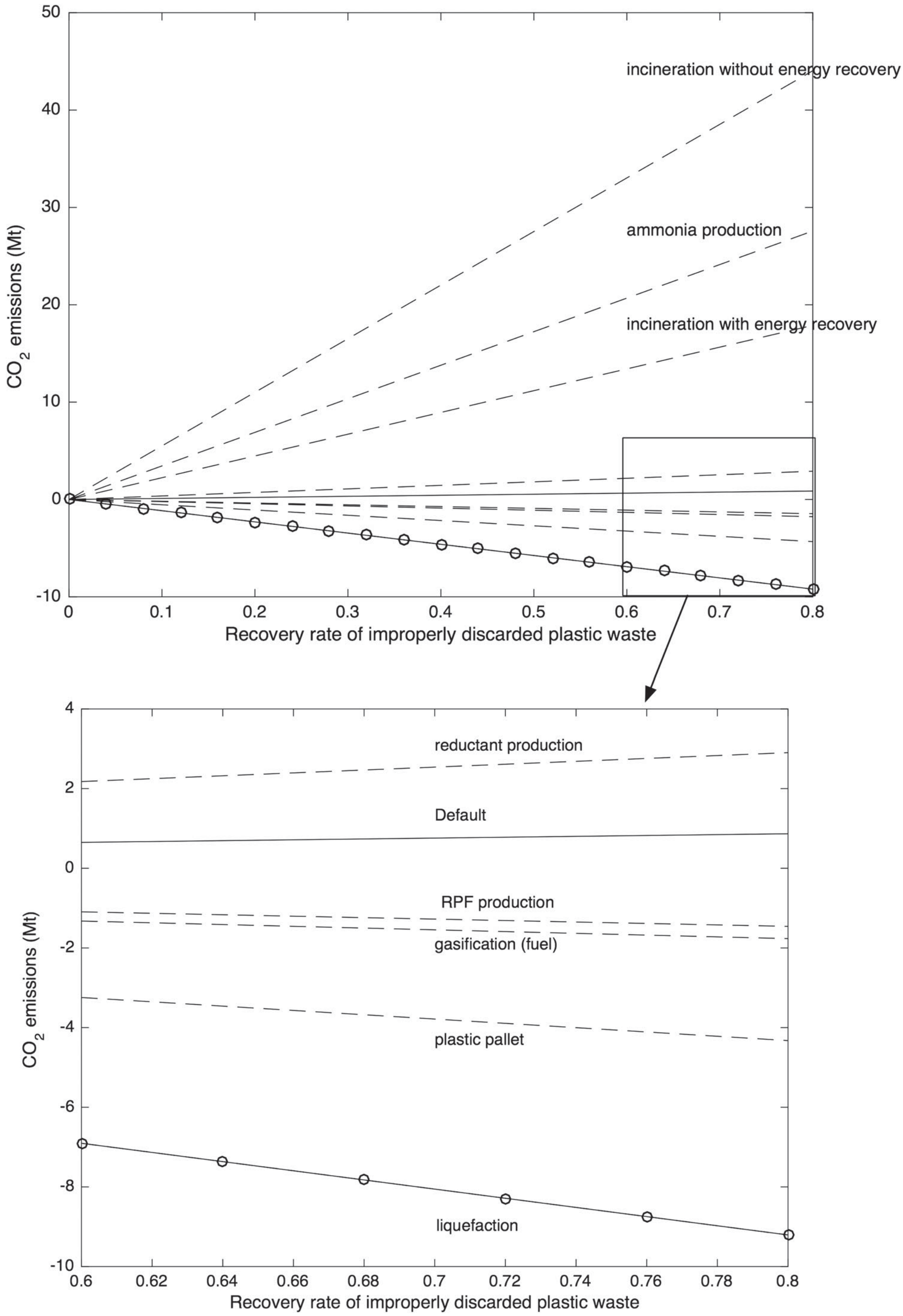
4.2.2. Optimization scenario

To address this challenge and determine the optimal plastic waste management strategy, we used the WIO-LP model (Equation (3)). Figure 2 (the lowest dash-circle line) shows the CO_2 emission under optimized strategies obtained by WIO-LP. The minimization of CO_2 emission was achieved by selecting liquefaction as the optimal treatment process, under which the CO_2 emission decreases up to 9 Mt with an increase in the recovery rate. The result supports the argument that the MSW is one of the most potentially renewable energy sources if waste to energy technologies are adopted that will not only reduce the dependency on conventional energy sources to meet the ever-increasing energy demand, but also reduce the problem of MSW (Kumar and Samadder, 2017).

The dashed lines in Figure 2 show CO_2 emissions when the plastic waste was treated solely by each of the alternative treatment processes. The second-best treatment process is plastic pallet production, and gasification and RPF production serve as the third and fourth best options. The slopes of the curves of these four processes are negative and imply that they create a win-win situation for the mitigation of CO_2 and marine debris; an increase in recovery rate of discarded plastic waste decreases the CO_2 emission.

The other processes, such as currently operated incineration with energy recovery, induce a trade-off relationship between CO_2 and marine debris mitigations. The results are consistent with those of other studies that have shown that the recycling processes are better than incineration with recovery with respect to CO_2 mitigation (Nakatani and Hirao, 2010, Chen et al., 2011). The seemingly superior performance of the default scenario over

Figure 2. Default and optimization scenarios and the performance of treatment processes.



Note: The vertical axis shows the difference in CO_2 emissions between the scenarios in question and the 2007 reality.

Table 4. Relation between waste composition and power output.

	Food	Paper	Textile	Plastics	Green waste	Power output (kWh/kg)
Beijing	0.72	0.11	0.02	0.13	0.02	1.29
Shanghai	0.73	0.04	0.02	0.20	0.01	1.57
Tianjin	0.75	0.09	0.02	0.12	0.02	1.24
Shenyang	0.85	0.08	0.01	0.05	0.02	0.55
Hangzhou	0.78	0.15	0.02	0.03	0.02	0.49
Qingdao	0.82	0.04	0.03	0.11	0.00	1.14
Tibet	0.75	0.06	0.07	0.12	0.00	1.27
Ningbo	0.83	0.05	0.03	0.08	0.01	0.92
Guanghan	0.84	0.09	0.01	0.06	0.00	0.64
Chongqing	0.64	0.10	0.06	0.16	0.04	1.45
Guangzhou	0.71	0.06	0.05	0.15	0.03	1.37
Shenzhen	0.65	0.17	0.05	0.13	0.00	1.36
Hong Kong	0.52	0.26	0.03	0.18	0.01	1.62

Data sources: Tanaka and Matsuto (1998) and Zhang et al. (2010).

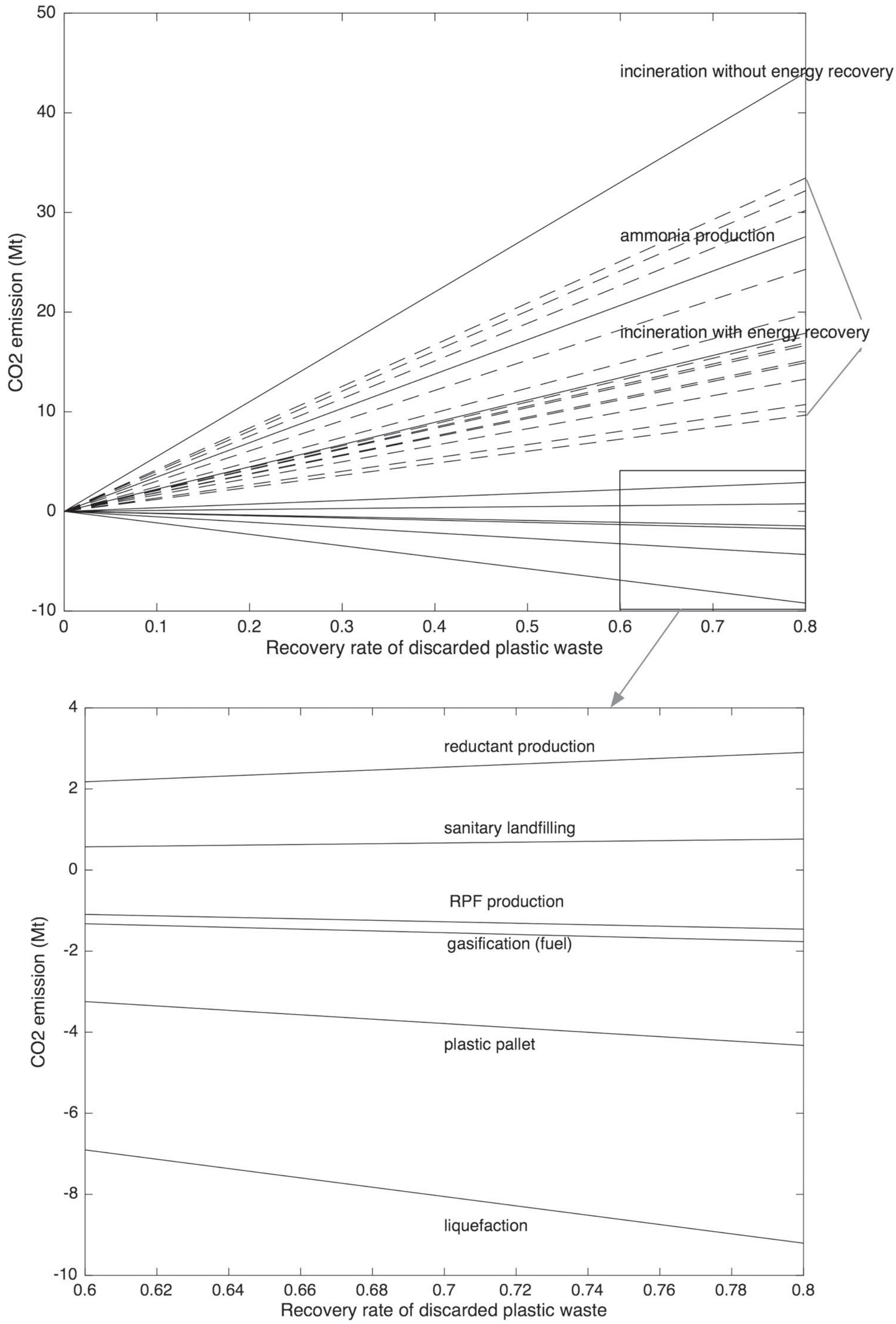
incineration, ammonia production and reductant production is realized by disposing most of recovered plastic waste in a landfill (Figure 1). Given that the construction of a landfill site is a controversial issue in China with local people opposing new construction (Che et al., 2013, Zhang and Klenosky, 2016), this scenario appears to be neither realistic nor sustainable.

4.3. Sensitivity analyses

Note that the incineration of plastics mentioned above refers to a mixed incineration with other items of MSW. Accordingly, the composition of MSW is likely to affect the power generation efficiency of plastic waste (Tanaka and Matsuto, 1998). Therefore, we conducted a sensitivity analysis to test the robustness of our model with respect to possible changes in waste composition. Table 4 shows the relation between the composition of municipal waste from various Chinese cities and the theoretical power generation per unit of plastic content, obtained using the Hokkaido University model (Tanaka and Matsuto, 1998), a system engineering model that provides theoretical amounts of power generation from waste incineration under various parameter settings related to waste composition and facility specifications. The power output range varies widely depending on the composition of waste. We reran the optimization problems by varying the waste composition setting for the 13 cities/regions shown in Table 4. This resulted in the emergence of new trade-off lines (the dashed lines in Figure 3). With the waste composition changed, the emission from incineration with energy recovery showed values near those of ammonia production. The higher the plastic content, the higher the combustion efficiency is. While incineration with energy recovery outperformed ammonia production in an extreme case, it was not able to reduce the overall emission. On the other hand, liquefaction was the optimal choice across all the scenarios we considered. Plastic pallet production, gasification, and RPF production also turned out to produce win-win situations, though to lesser extents. It is concluded that we have quite a robust result that a proper selection of plastic waste recycling processes results in a win-win situation in terms of both CO_2 emission and the recovery of plastic waste.

Finally, we considered possible implications of transport distances for the above results. The above results were obtained by assuming fixed distances of waste transport given by

Figure 3. Default and optimization scenarios under varying waste composition.



Note: The vertical axis shows the difference in CO_2 emissions between the scenarios in question and the 2007 reality.

Wang et al. (2001) and Liu (2011). The transportation distance, however, can vary significantly among cities. To assess the robustness of our results, we conducted a sensitivity analysis considering these variations in transport distances and found[2] that adjusting the transportation distance did not change the optimal plastic waste treatment/recycling processes.

5. Discussion

The Premier Minister of China, Li Keqiang, in his report on government work for the 13th National People's Congress, promised to reinforce the waste-sorting system of China (Li, 2018). Also in this report, Li Keqiang promised to reduce the proportion of coal consumption to the total energy consumption by 8.1%. As shown by our study, the products of recycling processes of plastic waste, such as RPF production, gasification, and liquefaction, can replace fossil fuels, such as coal. Meanwhile, the waste-sorting system, actually, is the foundation of conducting the above-mentioned recycling processes of plastic waste. Therefore, our results suggest that the realization of the first promise by the Premier Minister (reinforcement of the waste-sorting system) can help China to achieve his second promise (reducing the proportion of coal consumption). For the same reason, the waste-sorting system is the key to achieving a win-win situation with regard to both CO_2 reduction and marine debris mitigation. The lack of well-organized waste-sorting systems, however, is a feature still common to most Chinese cities (Steuer et al., 2016). However, the introduction of liquefaction requires the sorting of plastics from other components of MSW. In Chinese cities, recyclable waste is largely collected by informal parties (Tai et al., 2011), mainly scavengers and junk-buyers, although according to the Law of China on the Prevention of Environmental Pollution Caused by Solid Waste (Chi et al., 2011), the local Infrastructure and Construction Bureau is responsible for the collection, storage, transportation, and final disposal of MSW. Informal recyclers use crude and polluting methods to separate reusable components from other MSW, entailing serious, negative environmental and health effects. The prevalence of informal recycling can be linked to the lack of formal recyclers (JCPRA, 2007). Therefore, a well-designed source-separated collection of waste plastics implemented and overseen by a formal institution would be necessary for the introduction of the optimal strategy proposed in this study.

We close this study by mentioning some of its major limitations and suggesting important directions for future research. This study was limited to a static analysis. Because plastic waste generation is related to the evolution of plastic stock, a dynamic analysis based on WIO (Nakamura et al., 2017) for plastic waste in China is an important future direction for research. Consideration of dynamics is also important to take into account possible adverse effects of recycling due to the unintentional concentration of hazardous materials in recycled plastics (Puype et al., 2015; Leslie et al., 2016; Turner, 2018). Besides limitation in terms of time, this study was also limited in terms of space because of its focus on China alone. A multi-regional extension based on an MRIO framework involving waste flows, such as in Tisserant et al. (2017), would be able to address this, although development of the relevant data is non-trivial. Our study focused on an approach to reducing both CO_2

[2] To save space, because there was no change in the selected treatment strategy or the shape of the Pareto frontier for environmental burdens when the transportation distance was changed within a reasonable range, we do not show the resulting figure here.

emissions and marine debris for a given amount of plastic waste consumption. Reducing consumption of plastic products, targeting 'single-use' plastics, for example, would be a highly effective way of mitigating those environmental problems (European Commission, 2018), although proper attention should be paid to the impact of alternative materials that may replace plastics.

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