Environmental impact assessment of China's waste import ban policies: An empirical analysis of waste plastics importation from Japan

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Environmental impact assessment of China's waste import ban policies: An empirical analysis of waste plastics importation from Japan

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Abstract

In January 2018, the Chinese government implemented a ban on importation of foreign waste, including waste plastics and other miscellaneous waste. Consequently, this ban has changed the global plastic recycling options. The aim of this study is to clarify the environmental impact of the ban between China and Japan, which is the largest exporter of plastic waste. The question of whether these policies are fundamentally helping to reduce the environmental burden has also been discussed from the viewpoints of China and Japan. Plastics have been classified into seven categories. Material flow analysis (MFA) and CO_2 emissions derived in the MFA were applied to examine the material flow of waste plastics and compare CO_2 emissions of virgin and recycled resins in Japan and China before, and after, the ban. A scope for the MFA including the recycling process of waste plastics, virgin resin production, and shipping between countries was proposed, and the transition of the material flow and CO_2 emissions caused by China's waste import ban were evaluated. Material flow relevant to both Japan and China was set as the scope. Material flow crossing the scope, such as import of waste plastic into China from countries other than Japan, and export of waste plastic from Japan to countries other than China before the ban was excluded. The main finding of MFA is that the transition in material flow between 2016 and 2018 in China demonstrated high virgin resin imports and exports in 2016, and 28% of recycled waste plastics in China were derived from imported waste plastic. After the ban, the amount of waste plastic imports decreased by more than 99%, causing domestic virgin resin production to increase. Other findings include that CO_2 emissions of the input of virgin materials increased by up to 10%, and the CO_2 emissions derived from virgin resin production increased by up to 11% in 2018 compared those reported in 2016. With this transition, the CO_2 emissions of 1 kg of raw materials in 2018 increased by 0.24 kg compared with 2016. In contrast, Japan's exports decreased, and domestic recycled plastic input increased. The CO_2 emissions of 1 kg of raw materials in 2018 decreased by 0.07 kg compared with 2016.

Keywords: China's waste import ban; Waste plastics; Virgin resin; Recycled resin; Material flow analysis; CO_2 emissions derived from material flow

1. Introduction

In 2017, China was the world's largest importer of waste plastics. From 2000–2017, more than 100 million tons of waste plastics were imported into China. This corresponds to more than half of the world's plastic waste trade (Brooks et al, 2018). In 2016, approximately 7.4 million tons of waste plastics from 43 countries were imported. The main exporting countries were the United States, Japan, and the European Union (EU). Recycled resin produced from imported waste plastics fills the raw material supply gap because of the shortage of natural resources in China. Additionally, importing waste plastics has several advantages, including lower labor costs compared to the collection and recycling of domestic waste plastics. As an economic outcome of shipping between China and other countries, manufactured products from China were exchanged with waste plastics from other countries. Although demand of waste plastics increased, many small and medium-sized recycling factories were in rural areas with inadequate environmental considerations (Japan Ministry of Environment, 2019a). In 2012, the Chinese government evaluated the economic growth patterns that adversely affected the national environment (Yoshida, 2019), and in 2017, the National Sword was enacted. The act targeted waste imports from overseas and by the end of December 2017, a waste import ban was officially enforced, resulting in the almost complete cessation of waste imports, including waste plastics since early 2018.

China's waste import ban has caused various global issues. One of the most prominent issues was the need to accelerate proper waste management in countries that were highly reliant on Chinese recycling. For example, in the United States, Japan, and EU countries there were shortages in their own recycling plant capacities (Japan Ministry of Environment, 2020).

Consequently, waste plastics were exported to Southeast Asian countries such as Malaysia, Thailand, Vietnam, and Taiwan, leading to a doubling in the import levels of waste plastics in these countries. This altered waste plastic flow has made these countries reconsider the standards for import measures and acceptance (Japan External Trade Organization, 2019). Furthermore, the impact of China's waste import ban has also had an enormous effect on the Chinese economy. Recycling factories that use waste plastics as raw materials face a shortage of raw materials due to the sharp decrease in waste plastic since 2018.

China's waste-import ban has triggered a major transition in the international material flow and recycling system of waste plastics globally. However, it is not yet clear whether this has resulted in a reduced environmental burden. Therefore, clarifying the transition from an environmental perspective is important for addressing the effects and limitations of China's waste import ban. Quantifying the impact of international material flow on the environment is important for presenting desirable plastic waste flows under the Basel Convention. The aim of this study is to empirically analyze whether China's waste import ban is effective in reducing the environmental burden when considering the international material flow between China and Japan. This study focuses on the production of recycled resin using waste plastics and virgin resin. The rational material flow of waste plastics is examined from an environmental perspective by 1) comparing and evaluating the material flow and CO_2 emissions of waste plastic recycling in China and Japan using a material flow analysis (MFA) and 2) evaluating the effect on CO_2 emissions derived from the MFA.

Many studies have examined the environmental impact assessment of international waste flows. Nakatani et al. (2008) conducted a Life Cycle Assessment (LCA) for domestic and Japan-China recycling for waste polyethylene terephthalate (PET) bottles. Yoshikawa et al. (2007) evaluated the material flow of e-waste between Japan and China. Yoshida et al. (2005) analyzed the flow of waste material between Japan and China, and Terazono et al. (2004) evaluated the environmental impact of recycling waste plastics, waste home appliances, and metals between Japan and China. In a Chinese study, Liu et al. (2018) conducted an empirical analysis of the greenhouse gas reduction effect of recycled resins, including overseas waste plastics, in the domestic waste plastic recycling industry. The impacts of China's waste import ban were also focused on scrap metal (Dong et al, 2020) and used paper (Shang et al, 2020). Focusing on the economy and trade, Brooks et al. (2018), Huang et al. (2020), and Wang et al. (2020) evaluated the change in supply and demand of recycled resin using waste plastics, and the impact on the circular economy caused by China's waste import ban. Ren et al. (2020) also adopted the LCA to analyze the environmental impact of international trade and the recycling of waste PET bottles. Kumamaru and Takeuchi (2021) adopted an economic surplus analysis to clarify the economic impacts of China and Japan. However, no studies have evaluated the transition in international material flow and environmental impacts on waste PET bottles and waste plastics. In this study, the CO_2 emissions related to the MFA of all waste plastics was conducted. The question of whether the policies are fundamentally helping to reduce the environmental burden from the standpoint of China and Japan has also been discussed.

2. Materials and methods

2.1 MFA

2.1.1 Data

An MFA was conducted before (in 2016) and after (in 2018) the enforcement of China's waste import ban. The three methods that were used to assess the impact of the ban on MFA in this study were: (1) recycled and/or virgin resin production and product manufacturing, (2) disposal of waste plastics, and (3) recycling of waste plastics, especially in China and Japan, with reference to the Plastic Waste Management Institute (2018, 2020). The target plastics were polyethylene terephthalate, polypropylene (PP), polyethylene (PE), polystyrene (PS), polyvinyl chloride (PVC), acrylonitrile butadiene styrene (ABS), and other resins. These plastics were assumed to be produced from virgin and recycled materials. Six types of plastics were selected because the machines used for processing differed, depending on the type of plastic.

The main data sources for estimating the material flow in China were from the 2017 and 2019 China Plastic Industry Yearbook (China Plastics Processing Industry Association, 2018 and the China Plastics Processing Industry Association, 2020), and the China Chemical Reporter (2020). The main data sources for estimating the material flow in Japan were the Japan Plastics Industry Federation (2020), the Plastic Waste Management Institute (2018, 2020), and the Council for PET Bottle Recycling (2018, 2019). The data sources for the international trade of waste plastics were comtrade.un.org, which was provided by the United Nations (UN) (2020) and the Global Trade Atlas (2020). In trade statistics, items are classified using the harmonized system (HS) code. However, the HS code numbers differ among Japan, China, and Hong Kong. For example, the Japanese HS code for PET waste has been classified by flakes and others since 2015. In contrast, China and Hong Kong recorded the same HS code, and therefore, the difference in items by HS code was considered when estimating the material flow. In addition, China's trade statistics classify waste PP and other resins using the same HS code. Dividing the HS code is difficult, and consequently, waste PP and other resins were analyzed as an item set.

2.1.2 Evaluation indexes for material flow

The amount of waste plastic collection, the amount of virgin resin production, and the self-sufficiency rate of resin materials were proposed as evaluation indexes using the following formulas:

$$P^{C,i} = P^{C,i}_{dom} - \left(P^{C,i}_{im} - P^{C,i}_{ex}\right), \tag{1}$$

$$P^{C} = \sum_i P^{C,i}, \tag{2}$$

$$M^{C,i}_{r} = P^{C,i} \times \alpha \tag{3}$$

$$M^{C}_{r} = \sum_i M^{C,i}_{r}, \tag{4}$$

$$M^{C,i}_{v} = M^{C,i}_{dom} - \left(M^{C,i}_{im} - M^{C,i}_{ex}\right), \tag{5}$$

$$M^{C}_{v} = \sum_i M^{C,i}_{v} \tag{6}$$

$$S^{C,i}_{v} = \frac{M^{C,i}_{dom}}{M^{C,i}_{dom} - \left(M^{C,i}_{im} - M^{C,i}_{ex}\right)}, \tag{7}$$

$$S^{C,i}_{v+r} = \frac{M^{C,i}_{dom} + P^{C,i}_{dom} \times \alpha}{M^{C,i}_{v} + M^{C,i}_{r}}, \tag{8}$$

where P is the total amount of plastic waste collection [kg], P_{dom} is the amount of domestic waste plastic collection [kg], P_{im} is the amount of imported waste plastics [kg], P_{ex} is the amount of exported waste plastics [kg], M_r is the total amount of recycled resin [kg], M_v is the total amount of virgin resin production [kg], M_{dom} is the domestic virgin resin production [kg], M_{im} is the imported virgin resin [kg], M_{ex} is the exported virgin resin [kg], α is the recycling ratio (=0.9), S_v is the self-sufficiency rate of virgin resin [–], S_{v+r} is the self-sufficiency rate of virgin and recycled resin [–], c is the country (China and Japan), and i is the type of plastic (PET, PP, PE, PS, PVC, and other resins).

The self-sufficiency rate is intended to represent the percentage of resin that can be supplied from within one individual country. The recycling ratio was calculated using data published by the

Plastic Waste Management Institute (2018, 2020). China has a small amount of exported proportion of plastics. For example, the amount of exported waste plastics in 2017 was 0.5% of the amount of imported waste plastics in the same year (UN, 2020). Therefore, the exported waste plastics were ignored in Equation (1).

2.2 CO_2 emissions derived from the MFA

2.2.1 Scope

In this study, CO_2 emissions derived from production of virgin resin and/or recycling of waste plastics between China and Japan and shipping during 2016 and 2018. Data required for each process to calculate CO_2 emissions are presented in the following sections. The scope for evaluating CO_2 emissions are shown in Fig. 1. In the Chinese case, the amount of domestic waste plastic collection was anticipated to increase because of China's waste import ban, and the shortage of recycled materials might be covered by virgin material. In the Japanese case, the amount of waste plastics exported to China was expected to decrease sharply, and the amount of domestic recycling and exports to Southeast Asian countries was anticipated to increase. As such, this study calculated CO_2 emissions focusing on three processes: (1) imported/exported waste plastics recycling, (2) domestic waste plastics recycling, and (3) domestic virgin resin production. This study excluded any material flow outside the scope of this study, as shown in Fig. 1 (eg. import of waste plastic from countries other than Japan to China, and export of waste plastic from Japan to countries other than China, including Hong Kong, before China's waste import ban.

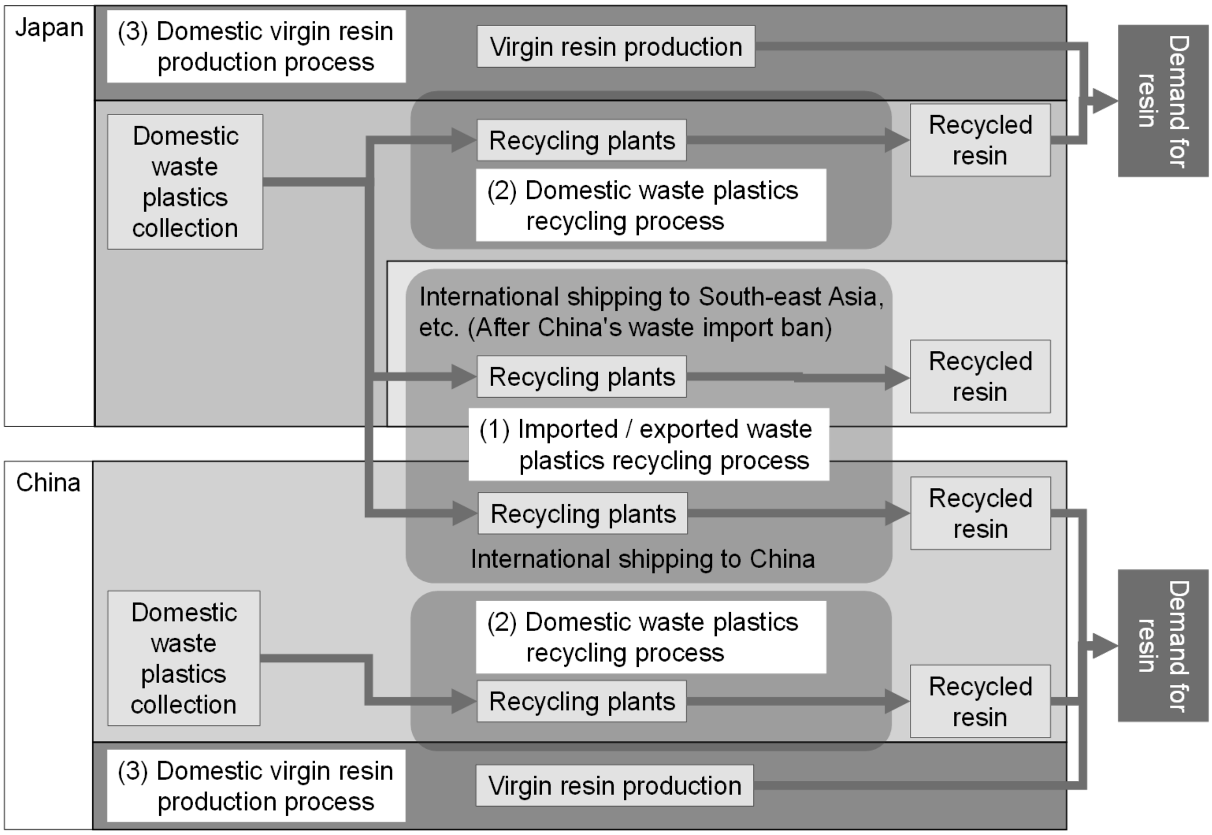


Fig.1.

The CO_2 emissions of the three processes were calculated using the following equations:

(1) Imported/exported waste plastic recycling processes

This process consists of recycling and international shipping.

$$CO_{2:r1}^{C} = \sum_i \left(P_{im}^{c,i} \ or \ P_{ex}^{c,i} \times E^{c,i} \times \beta\right), \tag{9}$$

$$CO_{2:s}^{C} = \sum_i \left(P_{im}^{c,i} \ or \ P_{ex}^{c,i} \times D_{d}^{c,i} \times \gamma\right), \tag{10}$$

where $CO_{2:r1}$ is the CO_2 emissions derived from recycling of waste plastics imported from overseas or exported overseas (kg-CO_2), E is the power consumption in recycling [kWh], β is the CO_2 intensity for power generation [kg-CO_2/kWh], D is the shipping distance [km],γ is the CO_2 intensity for international shipping [kg-CO_2/km], and d is the destination port.

(2) Domestic waste plastic recycling processes

$$CO_{2:r2}^{C} = \sum_i \left(P^{C,i} \times E^{c,i} \times \beta\right), \tag{11}$$

where $CO_{2:r2}$ is the CO_2 emissions derived from the recycling of domestic waste plastics [kg-CO_2].

(3) Production of domestic virgin resins

$$CO_{2:v}^{C} = \sum_i \left(M_v^{C,i} \times \delta\right),, \tag{12}$$

where $CO_{2:v}$ is CO_2 emissions derived from virgin resin production [kg-CO_2] and δ is the CO_2 intensity for virgin resin production [kg-CO_2/kg].

(4) Overall CO_2 emissions

Using the results obtained in each process, the CO_2 emissions derived from 1 kg of recycled resin or demand for resin were calculated as follows:

$$x_r^C = \frac{CO_{2:r1}^{C} + CO_{2:s}^{C} + CO_{2:r2}^{C}}{\sum_i M_r^{C,i}} \tag{13}$$

$$x_v^C = \frac{CO_{2:r1}^{C} + CO_{2:s}^{C} + CO_{2:r2}^{C} + CO_{2:v}^{C}}{M_v^C + M_r^C}. \tag{14}$$

The data used for the calculations are listed in Tables 1–4.

2.2.2 Waste plastics recycling data

The treatment of waste plastics was divided into pre-treatment and molding methods. The pre-treatment crushes and cleans waste plastics into flakes. Molding melts the flakes at a temperature of 200 °C or higher and produces a recycled resin through a molding machine. Although there are various uses for recycled materials, melting is an essential process. This study used inventory data up to the molding stage. The fundamental treatment flow is the same in China and Japan, although how the properties of waste plastics are disposed of, and the detailed treatment flow differs between the two countries. Here, it was assumed that waste plastics were recycled by the fundamental treatment flow in both countries. This assumption was based on interviews with business owners of waste plastic recycling companies in Japan. The power consumption of the pre-treatment and molding machines was collected based on interviews and online surveys.

Appendix Tables A–1 to A–9 show the power consumption data of the pre-treatment and molding machines. In this study, only power consumption was calculated to quantitate CO_2 emissions due to waste plastic recycling, because an interview survey revealed that machines that use electricity as an energy source are used in Japan and China. Some machine products vary the amount of, for example, treatment and power consumption. The average power consumption was calculated after anonymizing the machine products. The power consumption for recycling 1 kg of waste plastics for each type is summarized in Table 1. There are many types of PP and PE plastic products, and they are classified as soft plastic products, such as films and sheets, and hard plastic products. As with PP and PE, PVC has soft plastic products and hard plastic products. The pre-treatment of PVC was assumed to be the same as that of PP and PE. Other resins were also assumed to have PP and PE data, because other resins generally have the same treatment flow as PP and PE. If waste PET bottles were exported from Japan, PET flakes were exported in a pre-treated state. Therefore, the results of CO_2 emissions in Japan obtained through the process (1) were distributed to Japan and exporting countries.

Table 1 Power consumption for recycling 1 kg of waste plastics

	PET	PP and PE		PVC		PS
Unit	[kWh/kg]					
Pre-treatment	0.261	Soft products	0.326	Soft products	0.326	0.139
		Hard products	0.065	Hard products	0.065	
		Average	0.195	Average	0.195	
Molding	0.297	Single screw molding	0.278	Molding machine	0.297	0.254

		machine			
		Double screw molding machine	0.249		
		Average	0.263		
Total	0.558	0.459		0.492	0.392

Table 2 shows the CO_2 intensity derived from power generation. In this study, the CO_2 intensity derived from the power generation of each country in 2016 and 2018 was adopted. Obtaining data for each Association of Southeast Asian Nations (ASEAN) countries was difficult, and therefore, the average intensity was adopted.

Table 2 CO_2 intensity derived from power generation

	CO_2 intensity derived from power generation (Year) [kg-CO_2/kWh]	Source
Japan	0.41 (2010), 0.52 (2016), 0.46 (2018) and 0.49 (average of 2016 and 2018)	Japan Ministry of Environment (2019b)
China	0.42 (2010), 0.63 (2016), 0.64* (2018) and 0.64 (average of 2016 and 2018)	Calculated basing on International Energy Agency (2020)

ASEAN country (average)	0.71	National Institute of Advanced Industrial Science and Technology and Japan Environmental Management Association for Industry (2020)

*CO_2 intensity of 2017 was adopted because the 2018 data were not published as of February 2021.

2.2.3 Data for producing virgin resin

Table 3 shows the CO_2 intensity derived from the production of virgin resin. These data were obtained from the Japanese IDEA Ver.2.2 LCI database (National Institute of Advanced Industrial Science and Technology and Japan Environmental Management Association for Industry, 2020). However, the data were created in 2010. Using these data may have been inappropriate, and therefore, they were converted by multiplying the data from the IDEA Ver. 2.2, and the ratio obtained by dividing the average CO_2 intensity derived from the power generation of 2016 and 2018 values by the CO_2 intensity derived from power generation in 2010.

Table 3 CO_2 intensity derived from producing virgin resin

	Data from IDEA Ver.2.2	Converted data	
		Japan	China
Unit		[kg-CO_2/kg]	
PE	1.81	2.15	2.79
PS	2.95	3.49	4.53

ABS	3.31	3.93	5.09
PP	1.76	2.09	2.71
PVC	3.26	3.87	5.01
PET	2.71	3.22	4.17
Other resins	3.12	3.7	4.8

2.2.4 Data for international shipping

The CO_2 intensity was derived from international shipping adopted sea voyage data for international transport published by the Policy Research Institute for Japan Land, Infrastructure, Transport and Tourism (2012). The container ship to be used was set to 4,000 TEU (twenty-foot equivalent unit) or more, based on an interview survey with the business owner of a waste plastics recycling company in Japan. In this case, the CO_2 intensity was 0.0095 kg-CO_2/tkm (Policy Research Institute for Japan Land, Infrastructure, Transport and Tourism, 2012).

The data obtained from sea-distances.org (2021) were adopted for the shipping distance between countries. Numerous ports are used in China and Japan, and therefore, the shipping distance between China and Japan was calculated by weighted averaging the distances between each port (Table 4). The average distance between China and Japan was approximately 1,188 nautical miles (approximately 2,200 km). Shipping distances to countries other than Japan and China were also obtained from sea-distances.org (2021) after setting the major ports in Southeast Asian countries (see Appendix Table A-10).

Table 4 Operating distance between China and Japan

	Guangdong	South China	North China

Unit	[Nautical mile] (= 1,852 [m])					
Yokohama	Guangzhou	1,667	Shanghai	1,036	Qingdao	1,101
			Fuzhou	1,216	Tianjin	1,298
			Average	1,126	Average	1,200
Osaka	Guangzhou	1,459	Shanghai	793	Qingdao	815
			Fuzhou	1,003	Tianjin	1,012
			Average	898	Average	914
Port utilization ratio*	30%		45%		25%	
Weighted average			1,188			

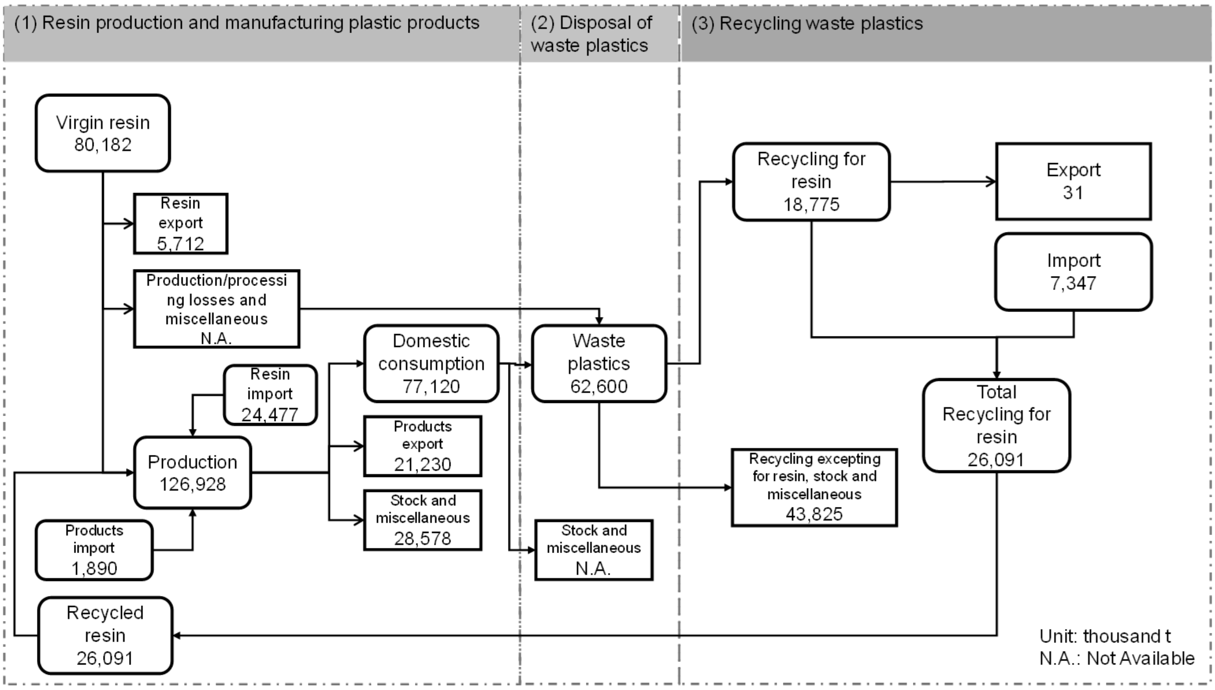
*: Port utilization ratio was calculated from the amount of waste plastics imported from each region (General Administration of Customs, P.R. China, 2020).

3. Results and discussion

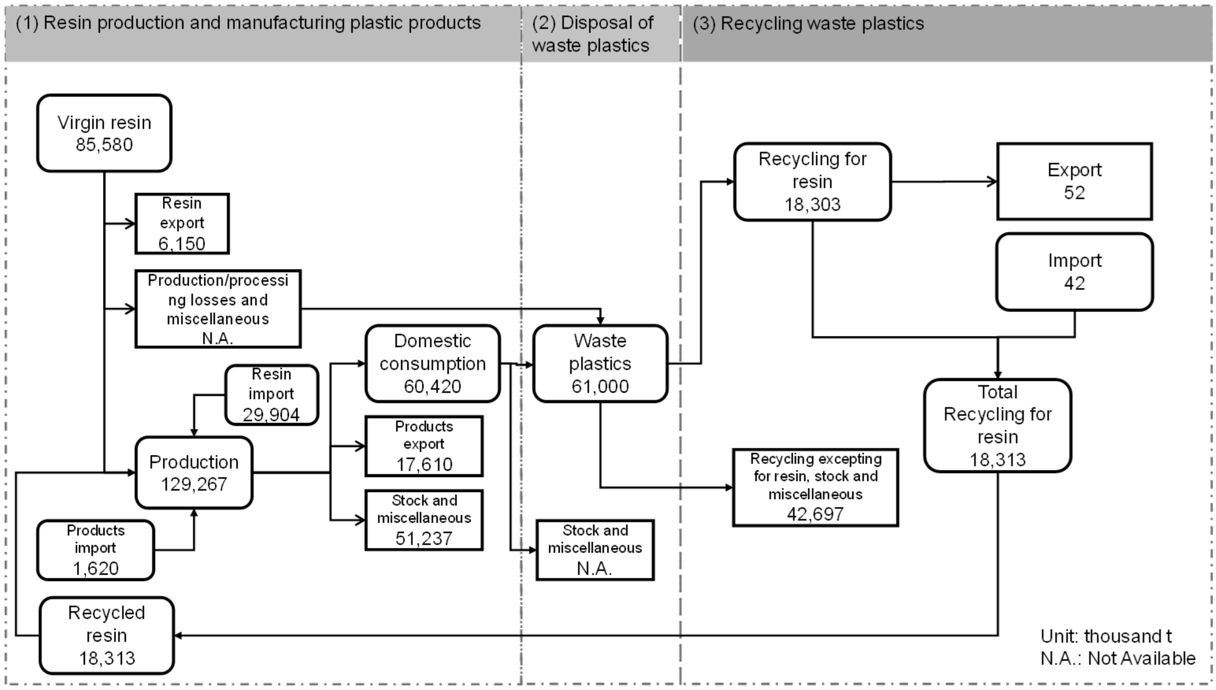
3.1 MFA

3.1.1 Chinese case

The transition of material flow for China and Japan was analyzed and discussed separately. Figure 2 shows the transition of the material flow between 2016 and 2018 in China. Tables 5 and 6 indicate the amount of each type of virgin resin production and the amount of each type of waste plastic collection in China. In process (1) in 2016, the amount of resin imported and exported was large. The self-sufficiency rate of virgin resin was 81% during this year. Approximately 20% of virgin resin was imported from other countries, and less than 30% of plastic products were exported overseas. These flows revealed that there is a delicate balance between virgin resources, waste plastic imports, and product exports. The self-sufficiency rate of the virgin and recycled resins was 78%, and 28% of the waste plastics recycled in China was from overseas. As a type of resin, PET has the highest rate of increase in virgin resin production, domestic waste plastic collection (excluding other resins), and waste plastic imports. According to Ren et al. (2020), 86% of waste PET imported in 2016 was used to produce polyester fiber. This suggests that China's waste import ban had a major impact on the domestic textile production industry, the world leader in chemical fibers (Ren et al, 2020). In process (2) in 2016, the amount of waste plastic disposal was 62,600,000 t. In process (3) in 2016, the amount of domestic recycled resin production was 26,091,000 t. This implies that China depended on more than 30% of the raw materials for products made from waste plastics.



(a) 2016



(b) 2018

Fig. 2.

Table 5 Amount of each type of virgin resin production in China

	Domestic production		Export		Import		Total of production		
	2016	2018	2016	2018	2016	2018	2016	2018	Increase rate
Unit	[thousand t]								
PET	7,570	10,170	1,983	2,744	44	62	5,631	7,488	0.33
PP	18,106	20,419	240	312	3,017	3,280	20,883	23,387	0.12
PE	14,355	10,420	299	229	9,943	14,025	23,999	24,216	0.010

PS	1,958	1,757	318	320	684	1,153	2,324	2,590	0.11
PVC	16,899	18,739	1,173	774	867	938	16,593	18,903	0.14
ABS	3,098	3,258	28	48	1,686	2,013	4,756	5,223	0.10
Other resins	18,196	20,817	1,671	1,723	8,236	8,433	24,761	27,527	0.11
Total	80,182	85,580	5,712	6,150	24,477	29,904	98,947	109,334	0.10

Table 6 Amount of each type of waste plastics collection in China

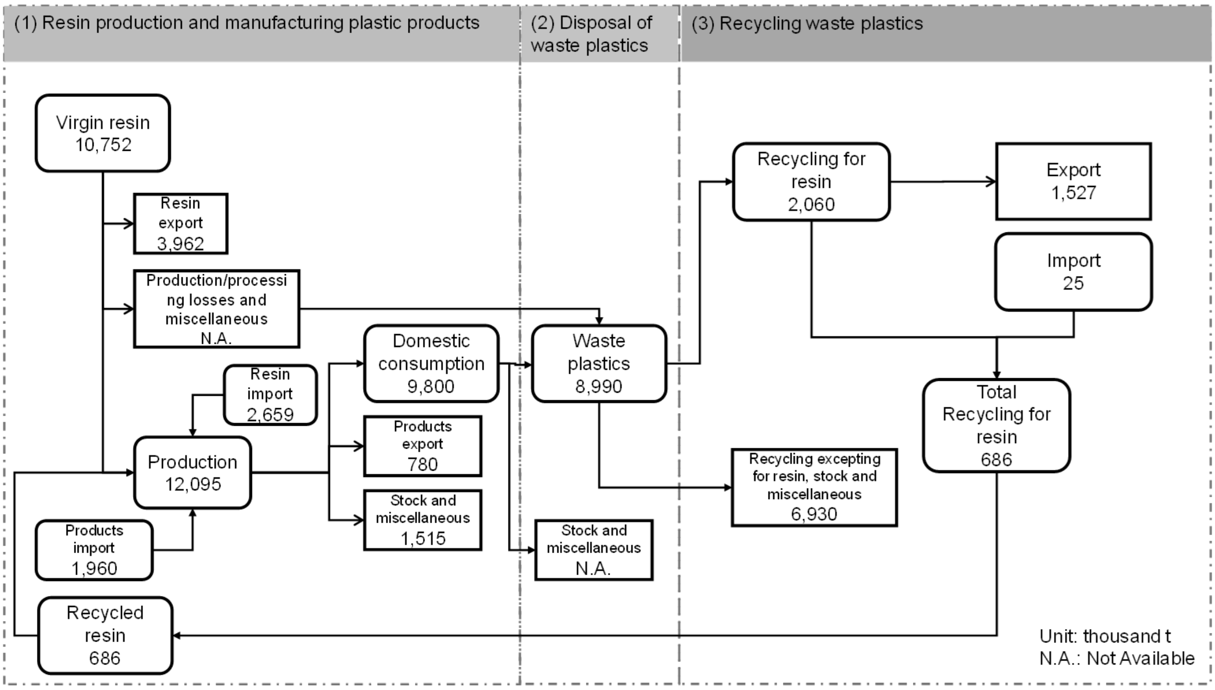
	Domestic collection		Export		Import		Total of collection		
	2016	2018	2016	2018	2016	2018	2016	2018	Increase rate
Unit	[thousand t]								
PET	6,200	6,040	0.02	0.05	2,533	14	8,733	6,054	-0.31
PE	3,235	2,915	2	2	2,532	13	5,766	2,926	-0.49
PS	313	312	0.1	0.2	92	2	405	314	-0.23
PVC	2,237	2,276	3	5	446	0.2	2,680	2,271	-0.15
PP + other resins	6,790	6,760	26	35	1,744	23	8,508	6,748	-0.21
Total	18,775	18,303	31	42	7,347	52	26,091	18,313	-0.30

In contrast, the self-sufficiency rate of virgin resin decreased to 78% in 2018. After China’s waste import ban, the amount of waste plastic imports decreased by more than 99%, while domestic virgin resin production increased. In particular, the import of virgin resin increased by 174%. Virgin resin production decreased by approximately 30% compared to 2016. The amount of domestic waste plastic collection also decreased slightly. This may be due to the decreased number of companies and employees because of the reorganization of the recycling industry. An improvement in the amount of waste plastic collection and the rate of recycling cannot be expected in the short-term.

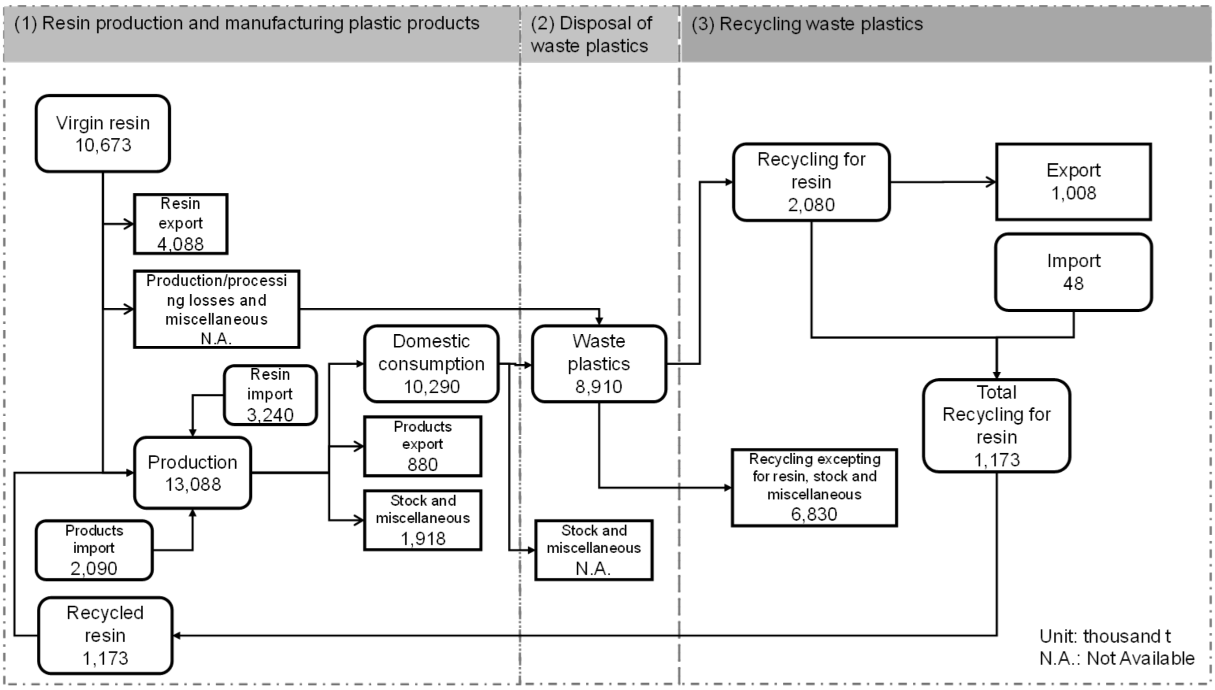
Alternatively, Hong Kong exports more than 99% of the waste plastics imported in 2016 to China. Japan, the United States, and the EU were the top three exporters to China, including exports to Hong Kong. The importation of waste plastic to China and Hong Kong dropped sharply from 7,347,000 t in 2016 to 52,000 t in 2018.

3.1.2 Japanese case

Figure 3 demonstrates the transition in the material flow between 2016 and 2018 in Japan. Tables 7 and 8 indicate the amount of each type of virgin resin production and the amount of each type of waste plastic collection in Japan. There is little transition in the material flow in processes (1) and (2). In process (3), the amount of waste plastic exported overseas, and the export destination changed significantly. In 2016, 1,500,000 t of waste plastics disposed of domestically were exported. After China's waste import ban, Japan's exports decreased to 1,000,000 t. China was the largest exporter in 2016. In 2018, the key exporters changed from China to other Asian countries, for example, Malaysia, Thailand, and Taiwan. In addition, waste plastics that cannot be exported are recycled in Japan. The amount of waste PET exported is equivalent to 35% of the amount of domestic waste plastic recycling. PVC is subject to international movement restrictions under the Basel Convention, and its export ratio is small. These situations suggest that Japan has a low market demand for recycled resin for waste plastics other than PET and PVC or lacks the recycling capacity for these types of waste plastics.



(a) 2016



(b) 2018

Fig. 3.

Table 7 Amount of each type of virgin resin production in Japan

	Domestic production		Export		Import		Total of production		
	2016	2018	2016	2018	2016	2018	2016	2018	Increase rate
Unit	[thousand t]								
PET	418	393	98	139	1,075	1,163	1,395	1,417	0.020
PP	2,466	2,358	324	396	245	504	2,387	2,465	0.030
PE	2,987	2,860	588	539	443	587	2,842	2,909	0.020
PS	754	784	164	220	52	47	642	611	-0.050
PVC	1,651	1,690	646	642	15	16	1,019	1,064	0.040
ABS	430	453	130	136	53	56	353	373	0.060
Other resins	2,046	2,135	2,012	2,016	776	867	810	987	0.22
Total	10,752	10,673	3,962	4,088	2,659	3,240	9,449	9,825	0.040

Unit: thousand t

Table 8 Amount of each type of waste plastics collection in Japan

	Domestic collection		Export		Import		Total of collection		
	2016	2018	2016	2018	2016	2018	2016	2018	Increas

									e rate
Unit					[thousand t]				
PET	530	560	382	303	6	11	186	288	0.55
PE	330	330	377	198	9	24	-9	152	-17.59
PS	130	150	180	128	0	2	-32	35	-2.10
PVC	310	280	85	65	4	0	233	222	-0.05
PP + other resins	760	760	503	314	6	11	307	477	0.55
Total	2,060	2,080	1,527	1,008	25	48	686	1,173	0.71

3.2 CO_2 emissions derived from the MFA

3.2.1 Transition of CO_2 emissions before, and after, China's import waste ban

Figure 4 demonstrates the transition of CO_2 emissions derived from recycling waste plastics in China, and Figure 5 demonstrates the transition of CO_2 emissions derived from producing virgin resin in China. The CO_2 emissions derived from recycling waste plastics decreased by 34%, from 8,690,000 t-CO_2 in 2016, to 4,760,000 t-CO_2 in 2018 because of the sharp decline in waste plastic imports. The CO_2 emissions derived from the recycling of imported waste plastics and international shipping were almost zero. In contrast, the amount of virgin resin production increased by 10% from 2016 to 2018, and the CO_2 emissions derived from virgin resin production also increased, from 383,620,000 t-CO_2 in 2016, to 427,190,000 t-CO_2 in 2018.

Fig. 4.

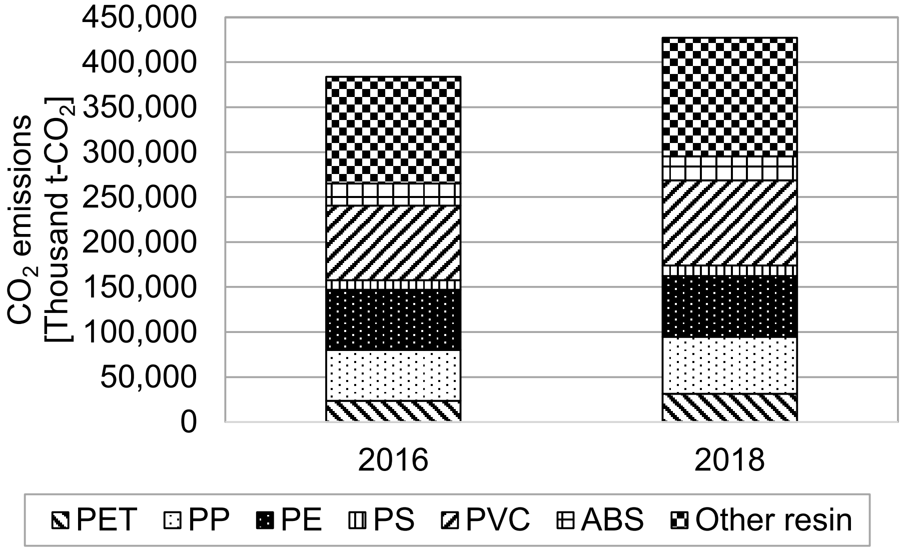


Fig. 5.

Figure 6 The CO_2 emissions derived from recycling waste plastics in Japan. In Japan, the amount of recycled waste plastics increased by approximately 500,000 t from 2016 to 2018 because of China's waste import ban. The CO_2 emissions derived from the recycling waste plastics also increased by 67%, from 190 thousand t-CO_2 in 2016, to 310,000 t-CO_2 in 2018.

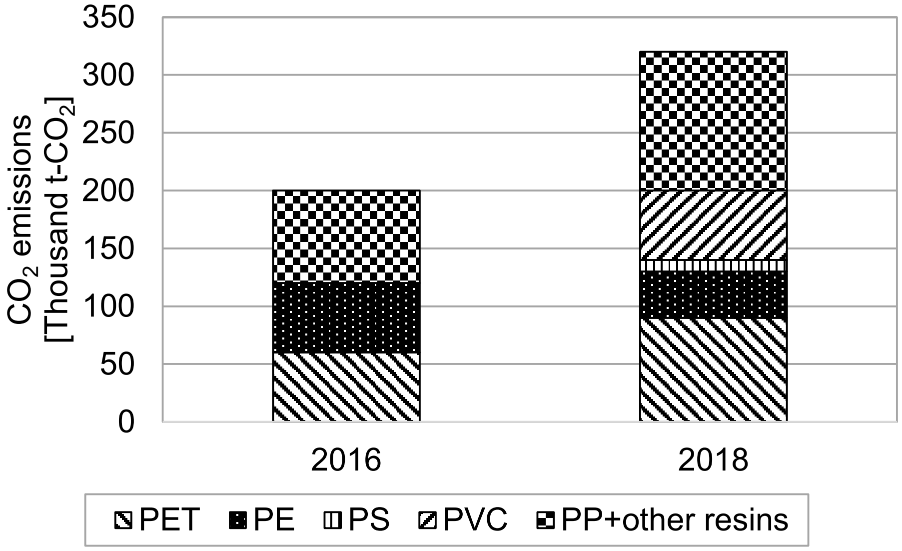


Fig. 6.

Figure 7 The CO_2 emissions derived from exported waste plastics from Japan. The overall CO_2 emissions decreased by 23%, from 470,000 t-CO_2 in 2016, to 360,000 t-CO_2 in 2018. The CO_2 emissions derived from waste plastics exported from China and recycled in 2018 was zero because of China's waste import ban. The CO_2 emissions derived from international shipping increased slightly because of the change in export routes from China to Southeast Asian countries.

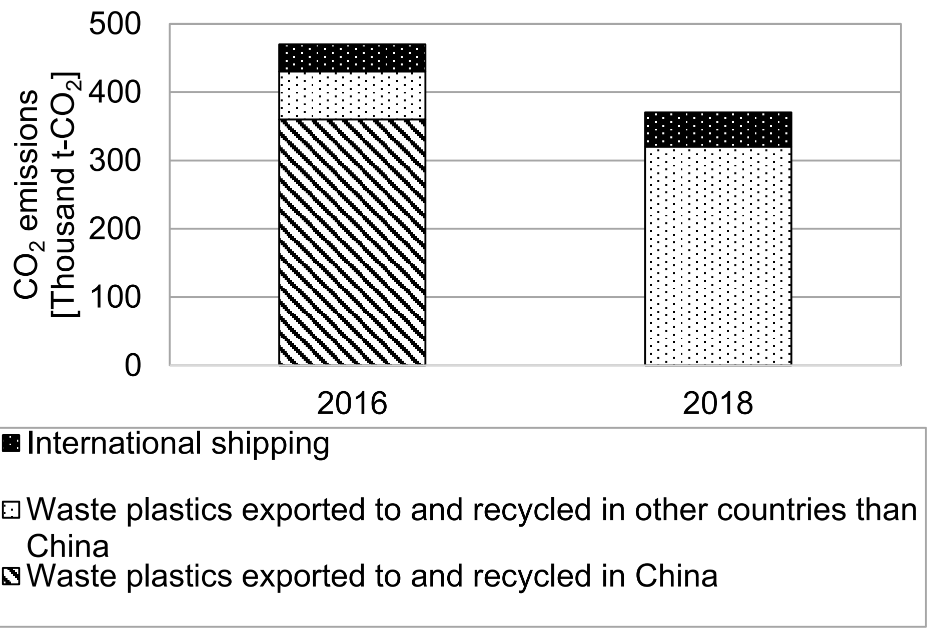


Fig. 7.

Figure 8 The CO_2 emissions derived from producing virgin resin in Japan. The amount of virgin resin decreased slightly compared to 2016. Conversely, an increase in the production of virgin resin with high CO_2 intensity such as PS, PVC, and ABS caused CO_2 emissions to increase slightly compared to 2016.

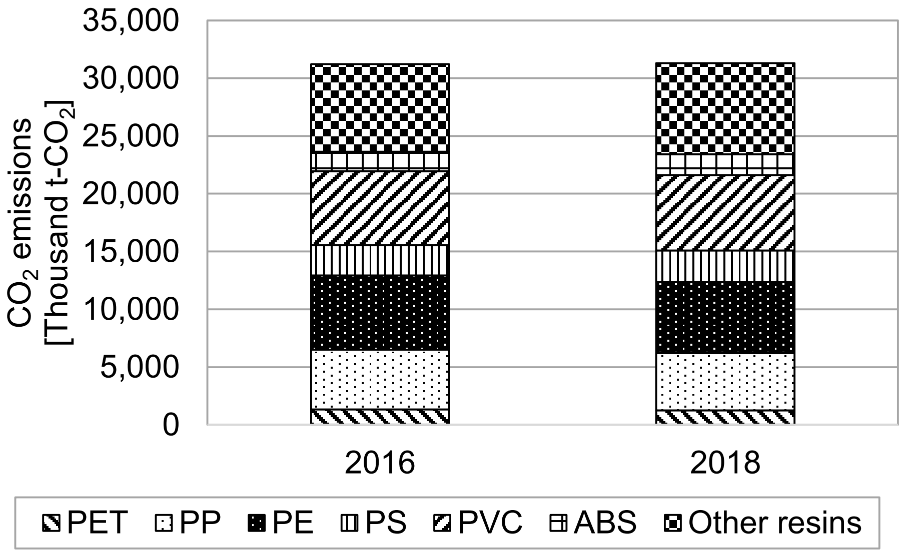


Fig. 8.

3.2.2 CO_2 emissions per kg of recycled resin and demand for resin

Table 9 demonstrates the transition of CO_2 emissions per 1 kg of recycled resin and the demand for resin in China and Japan. The result of totaling the CO_2 emissions of China and Japan revealed that the CO_2 emissions in 2018 are higher than those in 2016. China's waste import ban has led to an increase in demand for virgin resins, and this increased demand has contributed to an increase in CO_2 emissions.

In the Chinese case, the CO_2 emissions per 1 kg of recycled resins decreased in 2018 by 0.02 kg compared to 2016. This result is discussed from three aspects, namely, domestic waste plastic collection, waste plastics import, and international shipping. There was no change in CO_2 emissions from the domestic recycling of waste plastics. In contrast, the CO_2 emissions derived from recycling imported waste plastics decreased slightly, from 0.28 kg in 2016, to 0.27 kg in 2018. The CO_2 emissions derived from international shipping also decreased by 50%, from 0.06 kg in 2016, to 0.03 kg in 2018, and therefore, international shipping was the main factor in reducing CO_2 emissions. A reason for the decrease in CO_2 emissions for international shipping is the shortening of shipping distances. In 2016, most imported waste plastics were from North America and EU countries. However, only three countries outside Asia were in the top 10 countries that imported into China in 2018.

Table 9 Transition of CO_2 emissions per 1 kg of recycled resin and demand for resin in China and Japan

		2016	2018	Increase rate
Unit		[kg-CO_2/kg]		[%]
China	Recycled resin	0.300	0.283	-5.67
	Demand for resin	3.204	3.441	7.40
	Sub-total	3.504	3.724	6.28
Japan	Recycled resin	0.272	0.265	-2.57
	Demand for resin	2.745	2.669	-2.77
	Sub-total	3.017	2.934	-2.75
Total		6.521	6.658	2.10

While the CO_2 emissions derived from recycled resin decreased, those from virgin resin production increased by 0.24 kg in China. This increase was due to a 10% increase in the production of virgin resin. PET resin production increased CO_2 emissions by 33% in 2018 compared to 2016. These results suggest that the domestic demand for PET has increased since China's waste import ban.

In the Japanese case, there was almost no change in the CO_2 emissions derived from recycled resins. Conversely, the CO_2 emissions derived from virgin resin production decreased by 0.07 kg in 2018 compared to 2016. This suggests that the amount of domestic recycled plastics increased due to a decrease in waste plastic imports to China. The CO_2 emissions derived from recycled resin increased by 0.1% between 2016 and 2018 and included recycling, not only in Japan, but also in Southeast Asian countries, where the CO_2 intensity of power generation is higher than that of Japan and China. These CO_2 emissions results indicate that Japan relied on waste plastic recycling for recycling companies in Southeast Asian countries and not for domestic recycling companies.

Table 10 shows the transition of CO_2 emissions for the demand for resin in China and Japan. In China, the amount of virgin resin production increased, and the demand for resin in 2018 increased by 2.1% compared to 2016. The CO_2 emissions derived from the demand for resin also increased by 9.6%. In Japan, CO_2 emissions increased by 5.5% due to the large proportion of recycled resin, although the demand for resin was higher than in China.

Table 10 Transition of CO_2 emissions of demand for resin in China and Japan

		2016	2018	Increase rate [%]
China	Demand for resin [million t]	125	128	2.1

	CO_2 emissions derived from demand for resin [million t-CO_2]	401	439	9.6
Japan	Demand for resin [million t]	10	11	8.5
	CO_2 emissions derived from demand for resin [million t-CO_2]	28	29	5.5

3.2.3 Sensitivity analysis

CO_2 emissions derived from the recycling of waste plastics were calculated based on the average power consumption data of the pre-treatment and molding machines shown in the Appendix table. The power consumption varies depending on the machine product. On this basis, a sensitivity analysis was conducted to investigate the CO_2 emissions if the power consumption was changed. To conduct the sensitivity analysis, the minimum and maximum power consumption of the pre-treatment and molding machine products were used to calculate the range of the minimum and maximum CO_2 emissions. Table 11 indicates the results for Japan and China if the average results (the results shown in Figs. 4 and 5) were set as 1.00. The results revealed that CO_2 emissions in both Japan and China were reduced by nearly 10% below the average if the minimum power consumption was applied. In contrast, the increase in CO_2 emissions was up to 20% if the maximum power consumption was applied.

Table 11 Sensitivity analysis

	2016			2018		
	Average	Minimum	Maximum	Average	Minimum	Maximum
Japan	1.00	0.90	1.20	1.00	0.92	1.15

China	1.00	0.92	1.11	1.00	0.92	1.12

4. Conclusions

This study empirically analyzed the impact of China's waste import ban that was enforced at the end of 2017. The ban was imposed on the material flow and CO_2 emissions of waste plastics between China and many other countries. In this study, we analyzed the impact of the ban on Japan and the main findings are as follows:

(1) The analysis of the transition in material flow between 2016 and 2018 in China demonstrated that the amount of virgin resin imports, as well as exports, was high in 2016, and the self-sufficiency rate of virgin and recycled resins was 78%. Additionally, 28% of the recycled waste plastics in China were derived from imported waste plastic. The examination of the types of plastics, revealed that the demands for PET resin has increased and accounted for 34% of the plastic waste imports in 2016. After China's waste import ban, the amount of waste plastic imports decreased by more than 99%, while domestic virgin resin production increased.

(2) In Japan, there has been few changes to the flow charges related to domestic resin production and waste plastic disposal. In 2016, 1,500,000 t of waste plastics were exported to China. After China's waste import ban, exports decreased to 1,000,000 t and domestic recycling and imports of Southeast Asian countries increased.

(3) As the changes in the CO_2-emission levels before and after the implementation of the ban were also evaluated in this study, the CO_2 emissions 1 per kg of recycled resin production in China increased from 0.348 kg in 2016 to 0.349 kg in 2018. The CO_2 emissions per 1 kg of virgin resin production also increased from 3.204 kg in 2016 to 3.441 kg in 2018. This increase may be caused by a compensating for a decrease in the supply for waste plastic obtained from imports in China.

With globalization, the production of recycled resin in some countries and their subsequent export to China might be considered desirable from the perspective of reducing CO_2 emissions. However, the international supply-demand balance of resin materials should be considered. International cooperation by sharing roles among countries based on the Basel Convention is also required. Furthermore, waste segregation policies implemented throughout China are expected to increase the amount of domestic waste plastic collection in the future. Such policies might be expected to decrease the use of virgin resources and CO_2 emissions derived from virgin resin production. To realize this scenario, the development of waste plastic recycling plants that meet the potential supply of domestic waste plastic collection is needed. If this is realized, a resourceful recycling system might be formed by local production for the local consumption of waste plastics in China. Developed countries should acknowledge that they exported waste plastics to China, to prioritized economic rationality, and had resulted in an increase in the CO_2 emissions of China, which were due to waste plastic recycling. Therefore, building infrastructure to promote waste plastic recycling in their own country is necessary. At the same time, energy-saving machines and plants should be introduced to mitigate the increase in CO_2 emissions due to recycling.

China's waste import ban has had a major impact on China and many developed countries, including Japan. A limitation of this study is that the system boundary focuses primarily on material flows in Japan and China. To conduct a more realistic evaluation, the material flow of waste plastics, including not only China and Japan, but also other developed countries such as those in European countries and the United States is necessary. This makes it possible to evaluate the impact of the legislation more specifically in China and the impact in each country. Furthermore, discussing how to import and/or export waste in a globalized economy and how to build a recycling infrastructure when the economy shifts from globalization to a block economy is also important for future work.

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Figure Captions

Fig.1. Scope showing (a) the Chinese case and (b) the Japanese case

Fig. 2. Transition of material flow in China (Unit: thousand t)

Note: The amount of production and processing losses in China is unknown. The numerical values may not agree because of the estimation. (a) 2016 and (b) 2018

Fig. 3. Transition of material flow in Japan (Unit: thousand t)

Fig. 4. CO_2 emissions derived from recycling waste plastics in China

Note: Numerical values may not match because of rounding (a) 2016 and (b) 2018

Fig. 5. CO_2 emissions derived from producing virgin resin in China

Fig. 6. CO_2 emissions derived from recycling waste plastics in Japan

Fig. 7. CO_2 emissions derived from exporting waste plastics from Japan

Fig. 8. CO_2 emissions derived from producing virgin resin in Japan

Appendix

Table A-1 Pre-treatment of waste PET bottle

	Power consumption [kWh/kg]		Power consumption [kWh/kg]
Unraveling machine	0.030	Hot water washing / specific gravity sorting	0.036
Belt conveyor	0.0020	Screw transport	0.0020
Label peeler	0.022	High speed friction washer	0.022
Hand sorting container	0.0020	Screw transport	0.0020
Metal detector	0.00040	Washing tank	0.0070
Belt conveyor	0.0020	Screw transport	0.0020
Crushing machine	0.055	Vertical dehydrator	0.011
Screw transport	0.0020	Dryer	0.043
Washing tank	0.0070	Air extraction system	0.011
Screw transport	0.0020	Saving tank	0
		Total	0.261

Table A-2 PET flake molding machine

	Screw size [mm]	Amount of treatment [kg/h]	Power consumption [kWh/kg]

Product A	63	250–300	0.25
Product B	72	300–400	0.225
Product C	92	500–600	0.417
Average			0.297

Table A-3 Pre-treatment of waste plastics (soft products)

	Power consumption [kWh/kg]		Power consumption [kWh/kg]
Belt conveyor	0.002	High speed friction washer	0.037
Crushing machine	0.055	Screw transport	0.003
Screw transport	0.003	Vertical dehydrator	0.057
Belt conveyor	0.002	Dryer	0.052
Crushing machine	0.055	Interprocess connection	0
Screw transport	0.003	Dryer	0.045
Washing tank	0.007	Saving tank	0.002
Screw transport	0.003	Total	0.326

Table A-4 Pre-treatment of waste plastics (hard products)

	Power consumption		Power consumption

	[kWh/kg]		[kWh/kg]
Belt conveyor	0.001	Friction washer	0.007
Crushing machine	0.03	Film bleaching tank	0.001
Friction washer	0.007	Screw transport	0.002
Film bleaching tank	0.001	Dry system	0.015
Interprocess connection	0	Total	0.065

Table A-5 Single screw molding machine

	Amount of production [kg/h]	Power consumption [kWh/kg]
Product A	20-80	0.275
Product B	60-200	0.275
Product C	120-400	0.275
Product D	250-600	0.3
Product E	600-1200	0.263
Average		0.278

Table A-6 Double screw molding machine

	Amount of production [kg/h]	Power consumption [kWh/kg]
Product A	160	0.25

Product B	300	0.263
Product C	500	0.256
Product D	700	0.231
Product E	1,000	0.245
Average		0.249

Table A-7 Conical molding machine for PVC

	Amount of production [kg/h]	Power consumption [kWh/kg]
Product A	70	0.479
Product B	180	0.278
Product C	220	0.227
Product D	250	0.26
Product E	450	0.244
Product F	600	0.292
Average		0.297

Table A-8 Pre-treatment of waste plastics (ABS and PS)

	Amount of treatment [kg/h]	Power consumption [kWh/kg]
Product A	2,500	0.152

Product B	4,000	0.125
Average		0.139

Table A-9 Molding machine for ABS

	Screw size [mm]	Rated power consumption [kW]	Power consumption [kWh/kg]
Product A	90	140	0.264
Product B	110	180	0.25
Product C	120	220	0.25
Product D	140	300	0.25
Average			0.254

Table A-10 Operation distance between Japan and Southeast Asian countries

	Major ports	Nautical mile with Yokohama [Nautical mile] (= 1,852 [m])
Indonesia	Anyer, Jakarta	4,209
Malaysia	Klang	3,955
Philippines	Manila, North	2,294
Thailand	Bangkok	3,940

Vietnam	Ho Chi Minh City	3,072

Note: A port with a large amount of cargo handled was set.