The recyclable waste recycling potential towards zero waste cities - A comparison of three cities in China

Binxian Gu [a, b, c, *], Xinyi Tang [a], Lingxuan Liu [c], Yuanyuan Li [a], Takeshi Fujiwara [d], Haohui Sun [e], Aijun Gu [f], Yanbing Yao [a], Ruiyang Duan [a], Jie Song [a], Renfu Jia [g,*]

[a] College of Environmental Science and Engineering, Yangzhou University, Yangzhou, 225127, China

[b] Jiangsu Collaborative Innovation Center for Solid Organic Waste Resource Utilization, Nanjing, 210095, China

[c] Management School, Lancaster University, LA1 4YX, Bailrigg, Lancashire, United Kingdom

[d] Graduate School of Environmental Science, Okayama University, 3-1-1 Tsushima-naka, Kuta-ku, Okayama, 700-8530, Japan

[e] Electrical Engineering and Automation, North China University of Water Resources and Electric Power, Zhengzhou, 450045, China

[f] College of Hydraulic, Energy and Power Engineering, Yangzhou University, Yangzhou, 225127, China

[g] College of Civil Science and Engineering, Yangzhou University, Yangzhou, 225127, China

*Corresponding Author: bxgu@yzu.edu.cn (B. Gu), rfjia@yzu.edu.cn (R. Jia)

[1]

1. Introduction

Urbanization, population growth, and industrialization are three key reasons for the increase of municipal solid waste (MSW) generation (Maghmoumi et al., 2020; Zhang et al., 2019). Over half of the world's population lives in urban areas, and almost all regions of the world will be predominantly urban by the mid-21st century (United Nations Department of Economic and Social Affairs, 2018). Coincident with industrialization and urbanization is the increase of MSW. By 2050, the world's

[1] municipal solid waste (MSW); zero waste (ZW); recyclable waste (RW); recyclable organic (RO); recyclable waste recycling potential (RWRP); high resource value recyclable material (HRVRM); low resource value recyclable material (LRVRM); no resource value recyclable material (NRVRM); school district housing (SDH); household consumption structure (HCS); individual consumption expenditures (ICE).

MSW generation is expected to increase by 70%, from 2.01 billion tonnes in 2016 to 3.40 billion tonnes annually (World Bank, 2018). In China, MSW generation increased from 31.3 million tonnes in 1980 to 228.0 million tonnes in 2018 (NBSC, 1981-2019). China was known as the world's largest waste generator in 2004 when it exceeded the amount generated by the U.S. In 2030, China will generate twice as much MSW as the U. S. (Hoornweg and Bhada-Tate, 2012), and the MSW will increase continuously to 480.0 million tonnes (World Bank, 2018). World Bank (2018) and Kaza et al. (2018) summed up the critical aspects of poorly managed waste as "contaminating the world's oceans, clogging drains and causing flooding, transmitting diseases, increasing respiratory problems from burning, and harming animals that consume waste unknowingly." These problems can affect millions of people personally, and it also affects economic development, thwarting the growth of tourism and businesses. But, if well-managed, the waste can be a source of resources and has potential to create social wealth (Ayodele et al., 2018; Hering, 2012). However, no matter how professional the MSW management system is, in some cases, China's MSW management system is faced with problems that are unavoidable and insurmountable. These factors include the high cost of exporting MSW due to limited land use, the cost of equipment and the large scale of waste disposal that exceeds the capacity of landfills or disposal sites (Chen et al., 2018; Huang et al., 2016; Wu et al., 2014).

Some previous efforts (Barrett and Scott, 2012; Fudala-Ksiazek et al., 2016; Geng et al., 2013; Paes et al., 2020; Pietzsch et al., 2017; Shahbazi et al., 2016; Wan et al., 2018; Xiao et al., 2020) emphasized the importance of Zero Waste (ZW) initiative to promote circular economy and sustainable social development, which no longer focuses solely on the disposal, but instead considers recycling. The concept of

ZW attracted worldwide attention in a press release issued by C40 in August 2018, which is a network of international megacities dedicated to supporting measurable and sustainable ways to achieve climate change. The mayors of 23 cities signed the C40's "Advancing Towards Zero Waste Declaration," with the overall goal of reducing the amount of waste in landfills and incineration facilities by 50% by 2030 as well as 15% reduction in waste per person (C40 Cities Climate Leadership UK, 2018). Furthermore, five cities signed the "Advancing Towards Zero Waste Declaration," in 2019 (C40 Cities Climate Leadership Group, U.S, 2019). The 28 cities (e.g., New York, London and Tokyo) are listed in the Table S1. The case for ZW acceleration on a global scale is clearly set forth in this ongoing movement. Obviously, recycling is an essential practice for ZW city acceleration, and has always been an important strategy. The minimum recovery rate in China's 13th Five-Year Plan was set at 35.0%. Moreover, in order to achieve this goal, 11 cities were designated as ZW pilot cities in China (Table S2), which represented a strong signal from central government.

Despite the government's promotion of recycling, the success in achieving the planned recovery rate with ZW cities acceleration by the end of this year (based on the time of this article's submission in November of 2020) is not optimistic. Since 2011, we have been conducting a field tracking survey of MSW stream generation in Suzhou (Gu et al., (2014; 2015; 2018)). We also reviewed China's MSW stream generation in 78 Chinese cities (Gu et al., 2017b). We found a significant gap between political slogans and practical implementation of recycling goals. Some local governments have produced their recycling policies or guidelines based on the experiences of other developed regions and cities, and their recycling initiatives have become a political task rather than a strategic planning and management. We also found that local policy-makers are task oriented when it comes to recycling because of the lack of reliable data on the local recyclable waste recycling potential (RWRP). Our investigation revealed that most recyclable waste (RW) information is currently unreliable; thus, alternative data have been widely adopted, causing poorly implemented and inefficient local recycling policies. A reliable RW database is urgently needed to provide credible information to national and local city authorities.

Some previous efforts have focused on RWRP. At the case city level, Chang and Davila (2008) found that RW accounted for 93.0% of the MSW in Texas, U.S. Saeed et al. (2009) found that recyclable organics (RO, 58.0%) were the major contributor to MSW in Kuala Lumpur, Malaysia. Thanh and Matsui (2010) reported that RO and non-organic recyclable material (RM) accounted for (65.0-98.0)% and (2.5-45.4)%, respectively, in Ho Chi Minh City, Vietnam. Ibikunle et al. (2020) manually sorted the MSW and found that RO and RM accounted for 15.8% and 76.6%, respectively, in Ilorin, Nigeria. At the national level, the lead author of this paper and her research group (Gu et al., 2017b) adopted a stochastic simulation review based on 78 Chinese cities mentioned above, which indicated that the best estimate of RWRP was approximately 88.7%. Ma et al. (2020) extended the work of Gu et al. (2017b) and estimated the RO and RM of 135 prefecture-level cities in China as 53.7% and 34.9%, respectively, using the quantitative models established by a back-propagation neural network methodology. At the global level, Margallo et al. (2019) evaluated that RO was 44.0%, and RM was 38.0% with an average RWRP of 85.6% based on the results from 20 countries. The algorithm presented herein refers to previous studies (Gu et al, (2017b; 2018)), where food waste, wood and grass add up to RO, and paper, plastics, glass, metal and textile add up to RM. And, RO and RM add up to RW. However, those results are not sufficient to generate managerial insights and policy implication for ZW acceleration. Further local recyclable waste recycling potential (RWRP) data is required. Thus, more RWRP data is needed to verify the high-, low- and no-resource value of RW as it relates to the support and cooperation of community recycling enterprises and those who can provide better equipment or assistance of any kind, since many are not well funded or equipped with the basic set up needed for a successful operation (Ghanimeh et al., 2019; Jang et al., 2020; Ozcan et al., 2016). The characteristics of RW are related to city authorities' responsible management and national recycling strategy (Fernández-González et al., 2017). However, detailed quantitative results of RWRP in the resource value (RV) dimension are apparently lacking. Although high resource values (HRVs) and low resource values (LRVs) accounted for 64.8% and 26.0% of the quantified recyclables in Suzhou, China, respectively (Gu et al., 2018), the limitation lies in the fact that these RWRP represent a developed city. In China, there are 293 prefecture-level cities (CNBS, 2020), whose economic development levels and urbanization process are uneven. It is important to consider the local RWRP in each city based on the different stages of recycling and their relationship to each city's development of the ZW initiative or construction when formulating operational recycling policies.

The hypothesis presented here revolves around the lack of a level playing field when it comes to getting the help needed to advance the construction of a ZW city. A further hypothesis is that if the help is available that will be based on local RWRP's characterization. Therefore, we explored the data and methods of quantizing and recognizing the RWRP in three types of cities, based on case city classification of a high-, middle- and low-income-level. This effort represents an extension of our preliminary case study of Suzhou, a developed high-income level city (Gu et al., (2014; 2015; 2018)). Three key questions are answered in this study:

(1) What are the dynamic changes in RWRP in the case cities?

(2) What are the disparities of RWRP between intra-cities (inner-cities) and inter-cities (city-to city)?

(3) What are the key factors influencing RW stream generation?

To answer the above questions, systematic field tracking surveys covering multiple longitudinal case cities were implemented between 2016 and 2019. Results and discussion are presented in Section 3, including a discussion of dynamic changes in RWRP in case study cities. Moreover, RWRP disparities between inter-cities and in intra-cities are compared to other cities in the world. We also conducted fundamental driving factor analyses, followed by an analysis of the practical implications of this study (reported in Section 4). Finally, conclusions are drawn in Section 5.2.

2. Data and methods

2.1. Case cities and data collection

The RW samples of this study are from MSW. Generally, household solid waste, group institution waste, and street cleaning waste are recorded as MSW in China. Samples of waste for this survey were collected from a mixture of four designated sources (residential households, street cleaning, small businesses, and institutions). Collecting waste at generation sites and directly sorting them is one of the most accurate approaches for quantizing and characterizing RWRP (Chang and Davila, 2008; Ibikunle et al., 2020; Thanh et al., 2010). Any city's RW stream generation is affected by these factors, such as the social-economic level, urbanization development stage, local customs, culture, resident lifestyles, and geographical position (Gu et al., 2017a; Gu et al., 2018; Khandelwal et al., 2019). Since 2011, a systematic field tracking survey of the Suzhou case study was implemented, as mentioned in the introduction. Furthermore, from 2016 to 2019, extended surveys were implemented in the three representative Chinese cities, Suzhou, Yangzhou, and Suqian, which represent the cities of high-, middle- and low-income levels, respectively. In each city, four sampling locations were selected, two each in the old and new districts. Fig. 1 shows the 12 waste generation sites (①-⑫) that were sampled, which remained unchanged during the survey period. The definition of new and old districts is based on the development level/stage of a modular pattern. General information on the three case study cities, including GDP, population, MSW generation, etc. are provided in Table 1 and Figs. S1-2. Some contextual information on global cities is also provided so that non-Chinese readers can understand the size and status of our case study cities. More geographic location information is presented in Table S3.

Table 1. Description of Suzhou, Yangzhou and Suqian

City	GDP (per capita annual, $)	Urban residential population (million person)	Urban area (km^2)	ICE (per capita annual, $)	MSW generation (thousand tons)	RP (%)	Recycling rate (%)
Case cities							
Suzhou	25,508	815	4,653	5,501	234,000	87.0	6.4
Yangzhou	17,741	304	2,306	3,819	80,000	87.6	-
Suqian	8,212	296	2,153	2,553	41,000	72.8	-
Comparing with global major cities							
City	GDP (per capita annual, $)	City	GDP (per capita annual, $)	City	GDP (per capita annual, $)	City	GDP (per capita annual, $)
New York	75,092	Maribor	23,747	San Luis Potosi	15,303 [a]	Tlaxcala [a]	9,431
London	59,892	Webb	25,913	Miskolc	17,319	Bogota [a]	7,692
Sydney	44,930	Gunsan	26,494	Rancagua	19,398	Tapachula [a]	6,696

Note: 2018 GDP data, residential population, urban area, and individual consumption expenditures (ICE) were extracted from official statistics (Jiangsu Bureau of Statistics of China (JBSC), 2019). The GDP and ICE exchange rates between the RMB and the U.S. dollar were calculated at 6.8. The 2019 MSW generation data are from the local government. The 2019 recycling potential (RP) data was calculated using the "RW generation amount and the MSW generation amount" in a city as well as data from RW generation and MSW generation based on a survey sample. The Suzhou 2019 recycling rate was calculated based on 150,000 recyclable tons and an MSW generation amount of 2.34 million recyclable tons (http://www.suzhou.gov.cn/zszyhsly/index.shtml, accessed 2021.01.17). Additionally, 120,000 tons were recycled in 2018 (http://www.cn-hw.net/html/china/201811/63714.html, accessed 2020.01.17) with an annual recycling rate of 4.8%. In Yangzhou and Suqian, no clearly quantifiable recycling data were available; however, we extracted the data from local governments, recycling companies, and a community property company. For the global major cities from OECD (2018), the superscript “a” represents Bogota’s 2015, 2016, and 2019 data (Ghanimeh et al., 2019). Additionally, Case cities include New York, USA; London, UK; Maribor, Slovenia; Webb, NY, USA; Gunsan, South Korea; San Luis Potosi, Mexico; Rancagua, Chile; Tlaxcala, Mexico; Bogota, Columbia; Tapachula, Mexico.

A total of 5,538kg of sample waste was manually sorted and weighed as it was quantified into several categories, enabling first-hand RW data to be collected. Each category of waste was weighed, and all data was recorded on site (Fig. S3). Quantization steps were: 1) 11 physical components of MSW were quantified. 2) RO and five traditional RMs were quantified. 3) RMs were further quantified as HRV RM (HRVRM), LRV RM (LRVRM), and no RV RM (NRVRM) based on their local current market value. HRVRM (e.g., water bottles, plastic buckets) can be sold directly and recycled, which is favored by the recycling business, residents and scavengers. LRVRM (e.g., plastic packaging for yogurt, plastic packaging for instant noodles) cannot be recycled directly and must first be treated (sorted and cleaned). NRVRM (e.g., baby diapers, napkins) is designated as waste and is not counted as RW, which is an improvement in this study, compared to our previous study (Gu et al., 2018). More category information is presented in Table S4.

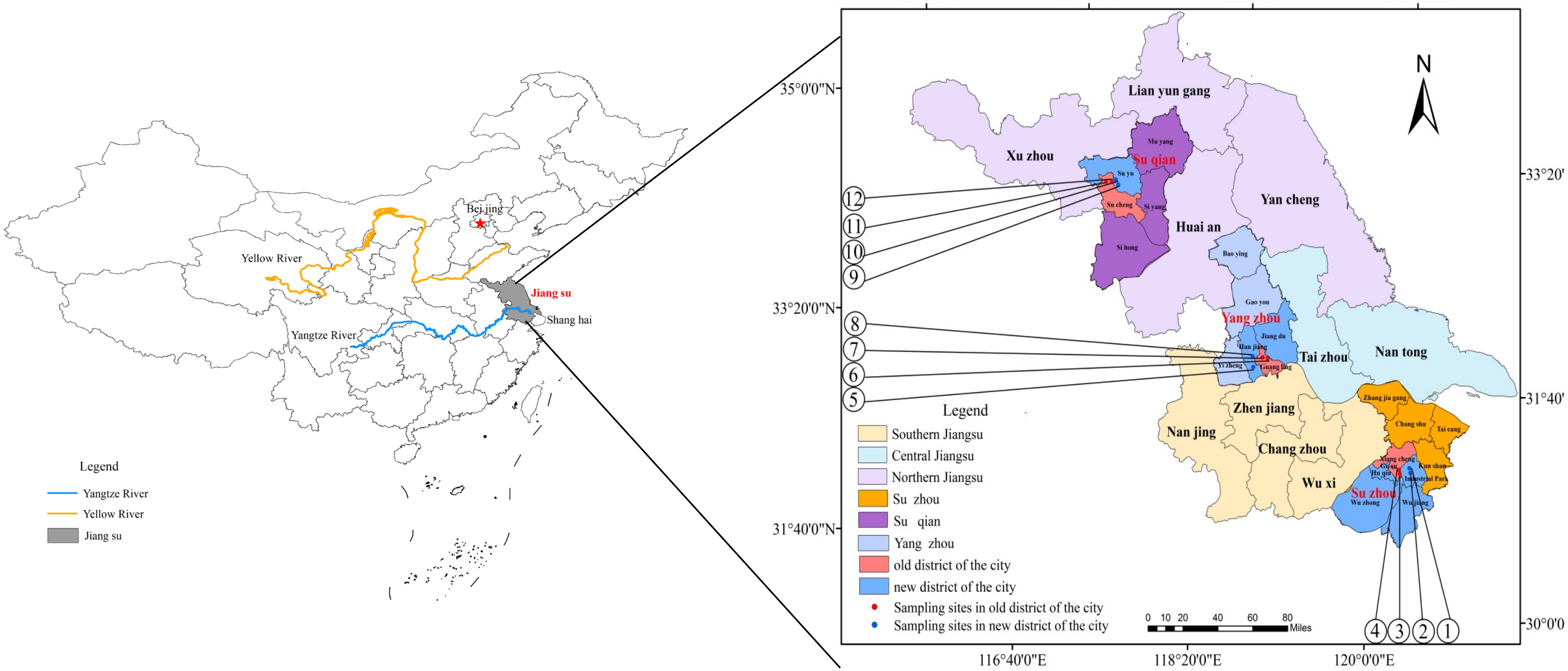


Fig. 1. Locations of Suzhou, Yangzhou, and Suqian, China

Note: Suzhou is shown in orange with old and new district of city in red and blue, respectively (1-4), Yangzhou is shown in a light blue with old and new district in red and darker blue, respectively (5-8), and Suqian is shown in purple with old and new district of city in red and blue, respectively (9-12). All are located in Jiangsu province, China.

2.2. Data consolidation and verification

The RWRP was obtained by calculating the proportion component of MSW:

$$\mathrm{Pro}_{i,j,k,t} = \frac{\mathrm{RW}_{i,j,k,t}}{\mathrm{MSW}_t} \times 100\%$$

$$\mathrm{MSW}_t = \sum_{i=1}^{11} \sum_{i=7}^{7} \sum_{i=17}^{17} \mathrm{RW}_{i,j,k,t} + \sum_{i=1}^{11} \mathrm{N}_{i,t}$$

where i refers to RW in kg; j is sub-RW in the physical component dimension, e.g., RO, paper, kg; k refers to sub-RW in the RV dimension, e.g., HRV paper and LRV plastic, kg; t is a year; $\mathrm{Pro}_{i,j,k,t}$ is the RP of $\mathrm{RW}_{i,j,k,t}$ in MSW, %; $\mathrm{RW}_{i,j,k,t}$ is the RW of the i, j, and k categories; MSW_t is the MSW generation sample amount in the survey of one city or one district or one year, kg. The weight unit, “kg” is flexible and could be converted to “g” or “tons.” There are eleven components in the physical dimension, seven components in the RW dimension, and 17 components in the RV dimension.

The survey is a random sample, but in a fixed sample source and fixed time intervals, undetected categories are considered no-generation and recorded as zero. Undetected categories and monitored categories are added together to estimate the mean value of every single RW categories. Both physical discrimination and statistical discrimination were combined as well as adopted for judging and removing outliers.

2.3. Statistical analysis

Linear regression analysis was performed between the RW stream generation and the assumed influence factors. The linear function model is represented as:

$$Y_{i,t} = \alpha + \sum_{i} \beta_{i,t} X_{i,t} + \varepsilon$$

where, $Y_{i,t}$ is the RW generation in year t in kg/tons; α is the intercept, which refers to the mean value of the response variable; $\beta_{i,t}$ is the slope indicating an average change in the response variable i ($i = 1, 2, \ldots, n-1, n$) in year t; $X_{i,t}$ are the variable factors i (i = 1, 2, … , n-1, n) in year t, i.e, the GDP per capita and ICE; ε is term of the average random error, and its expected value equals zero; t is the period between 2016 and 2019. Oracle's Crystal Ball as a Microsoft Excel add-in component was utilized to perform the uncertainty analysis. The Monte Carlo stochastic simulation approach was employed to model the probability distributions of key input parameters, and uncertainties were estimated. Monte Carlo sampling trials were set at 10,000. Standard deviation (Std. Dev.) represents the deviation between the primary data (the survey's first-hand data) and the mean value of uncertainties, which is expressed as:

$$\sigma = \sqrt{\frac{1}{N-1}\sum_{i,t=1}^{N} (X_{i,t} - \mu)^2}$$

where, σ is the Std. Dev. Value, a small value means a small deviation; vice versa. N denotes sample size; $X_{i,t}$ is the RWRP of i; μ is the mean value of $X_{i,t}$; t refers to one year.

3. Results and discussion

3.1. Dynamic changes in recyclable waste recycling potential (RWRP) in case cities

Our results show that case cities with different income levels have different dynamic changes trends in RWRP, in which, RO is a constant, and RM fluctuates. It should be noted that the results of this study are expected to be more useful to a wider range of recipients such as decision-makers, enterprise producers, and municipality planners. The dynamic changes of the RWRP in Suzhou are shown from 2011 to 2019, and the data between 2011 and 2017 are provided in our existing study (Gu et al., 2018). The corresponding dynamic changes of the RWRP in Yangzhou and Suqian between 2016 and 2019 are also presented herein.

As shown in Fig. 2, an important contributor to RWRP is RO, which can be converted into compost in the three case study cities with mean values of around 55.0%. The RO in Suzhou was 55.0% with a maximum of 56.2% in 2017 and a minimum of 54.0% in 2013. The RO in Yangzhou was 55.5% with a maximum of 56.5% in 2016 and a minimum of 54.1% in 2017. The RO in Suqian was 53.7% with a maximum of 55.5% in 2017 and a minimum of 52.9% in 2016. Notably, RO is dominated by food waste (Fig. S4). This is almost constant due to the Chinese preference for dominant food that is unprocessed and unpackaged, especially local green vegetables and fruit; thus, more dominant RO food waste is found in China than in Western cultures (Margallo et al., 2019). This implies that the RO value will continue to rise in the future, even as the economy and urbanization rapidly develop (Figs. S5-6). Composting has been recognized as a feasible policy that should be widely applied in China, and the implications are reported in Section 4.1. Results of mainstream RO agrees with previous Chinese studies (Chen et al., 2010; Gu et al., 2017b; Tai et al., 2011; Zheng et al., 2014); however, these studies are contrary to foreign findings indicating that lower economic level regions/cities produce more RO (Khandelwal et al., 2019; Vergara and Tchobanoglous, 2012).

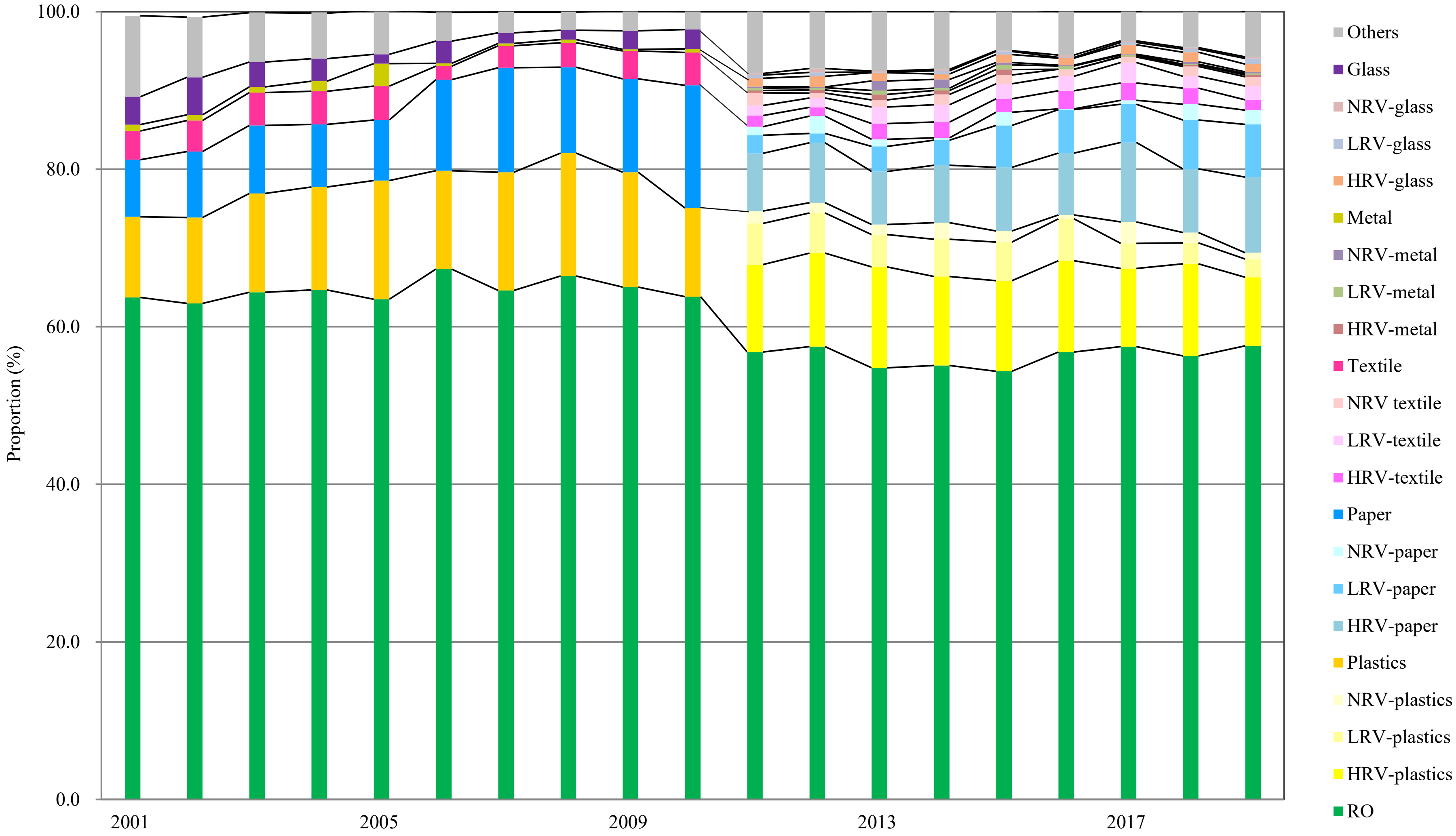
Another major contributor of RWRP is the RM in case study cities, where plastic and paper are the mainstream waste materials. On average, the RV of plastic and paper in Suzhou was 15.3% and 12.3%, respectively. The corresponding data on plastic and paper in Yangzhou is 15.1% and 8.3%, respectively. In Suqian, the proportion of plastic and paper are 5.9% and 5.6%, respectively. Additionally, significant dynamic changes in availability of plastic and paper were observed. In Suzhou, a slight decline in plastics occurred, and inversely, a slight increase in paper occurred. Plastic and paper were further defined in the RV dimension (Fig. S7 (a-c)). An interesting phenomenon was observed when the reduced amount of plastic with LRV was replaced by more paper with LRV. The dynamic changes in Yangzhou were similar to those in Suzhou, where it was noted that as the amount of plastic decreased, paper increased, especially LRV paper (Fig.2 (b) and Fig.S8 (a-c)). This could be attributed to the Chinese environmental protection policy and challenge initiatives. For instance, in 2008, after the "pay for plastic shopping bags" program was implemented, there was an immediate reduction in plastics, which was observed in Gu et al. (2017b). In 2013, a temporary restriction known as the "Green Fence" limited plastic waste entering China; in 2017, a policy permanently banning the import of nonindustrial plastic waste was announced. These policies revealed that plastics are undesirable; thus, the ban on imported plastics and cuts in overall use naturally resulted in plastic waste reduction (Brooks et al., 2018).

Notably, Suqian is different from Suzhou and Yangzhou. Plastic and paper have simultaneously grown rapidly due to the increase of NRV plastics and LRV paper (Fig. 2(c) and Fig. S9(a-c)). This could be due to the improvement of people's living standards. For instance, ICEs and individual consumption quantity of meat products, aquatic products, melons and fruits are growing rapidly (Fig. S10 (a-b)). A large number of packaged meat, fish, shrimp, and crabs as well as melons and fruit were found during the investigation in Suqian, but the packaging mainly consisted of LRV and NRV plastic and paper. Several similar studies that have analyzed packaging waste from different perspectives. Gu et al. (2015) reported packaging waste as comprising 66.8% of Suzhou's household solid waste (not counting home-grown food waste); hence, the largest proportion of packaging waste was for drinks, fast food and secondary-use shopping bags, which accounted for 19.7%, 12.9% and 6.6% of packaging waste, respectively. Geyer et al. (2017) reported that plastic packaging and containers are the largest component (36%) among industrial applications from worldwide plastic production (approximately 400 million tonnes annually). Liu et al. (2020) reported that food packaging accounts for 15.7% of the total MSW generated in the Jing-Jin-Ji region of China. It needs to be emphasized that the packaging waste mentioned by Gu et al. (2015) came from household solid waste, while the packaging waste in this study focuses on MSW. Additionally, the packaging waste of Geyer et al. (2017) analyzes industrial solid waste, while Liu et al. (2020) features food packaging waste in their study.

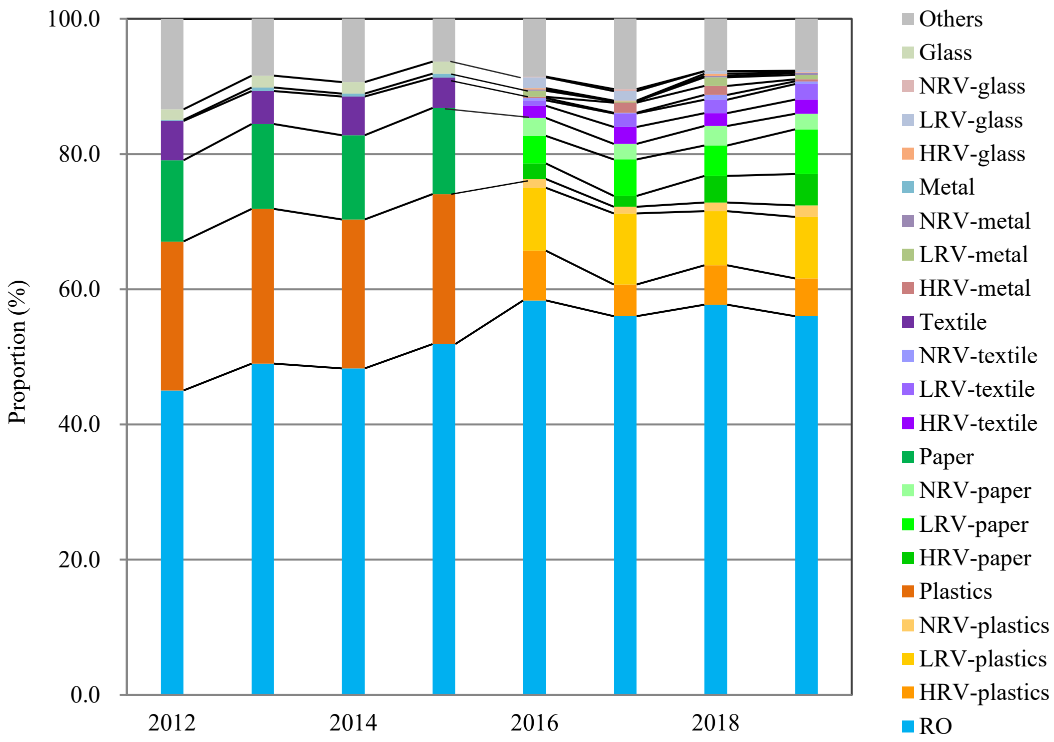
Some Chinese food markets feature open-top packaging, especially in developing cities. This type of packaging is extremely cheap for the vendor, and it is free to consumers. It needs to be emphasized that food market packaging is different from the "pay for plastic shopping bags." Paid packaging is defined as HRVRM in this study (Table S4). The results recorded in Suqian agree with the World Bank (2018) and Hoornweg and Bhada-Tate (2012), which noted that the trend toward increased plastic and paper generation is growing faster in China's MSW. It is expected that

LRV-plastics will continue to grow for a while in developing Chinese cities but will soon be replaced by degradable paper and textiles. However, the dynamic changes of textile in these case study cities were not obvious, averaging 5.0%. Furthermore, the combined proportions of metal and glass were no more than 3.0% in each case study city.

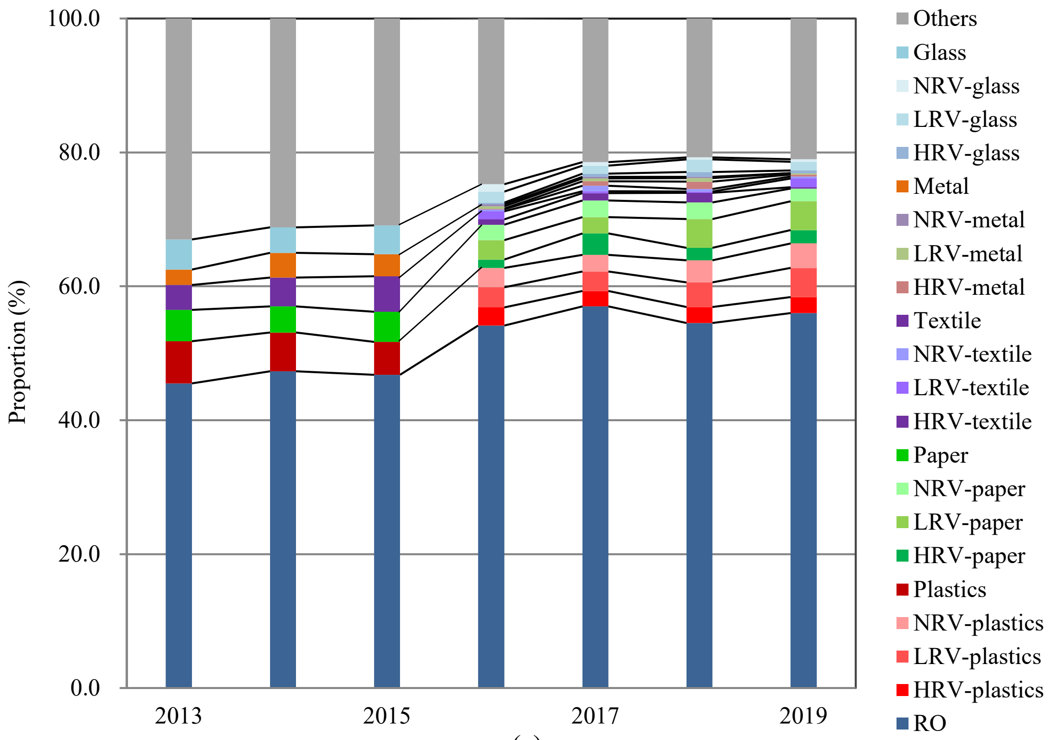
335


100.0
80.0
60.0
40.0
20.0
0.0
Proportion (%)
2001
2005
2009
2013
2017
Others
Glass
NRV-glass
LRV-glass
HRV-glass
Metal
NRV-metal
LRV-metal
HRV-metal
Textile
NRV textile
LRV-textile
HRV-textile
Paper
NRV-paper
LRV-paper
HRV-paper
Plastics
NRV-plastics
LRV-plastics
HRV-plastics
RO

(a)



(b)



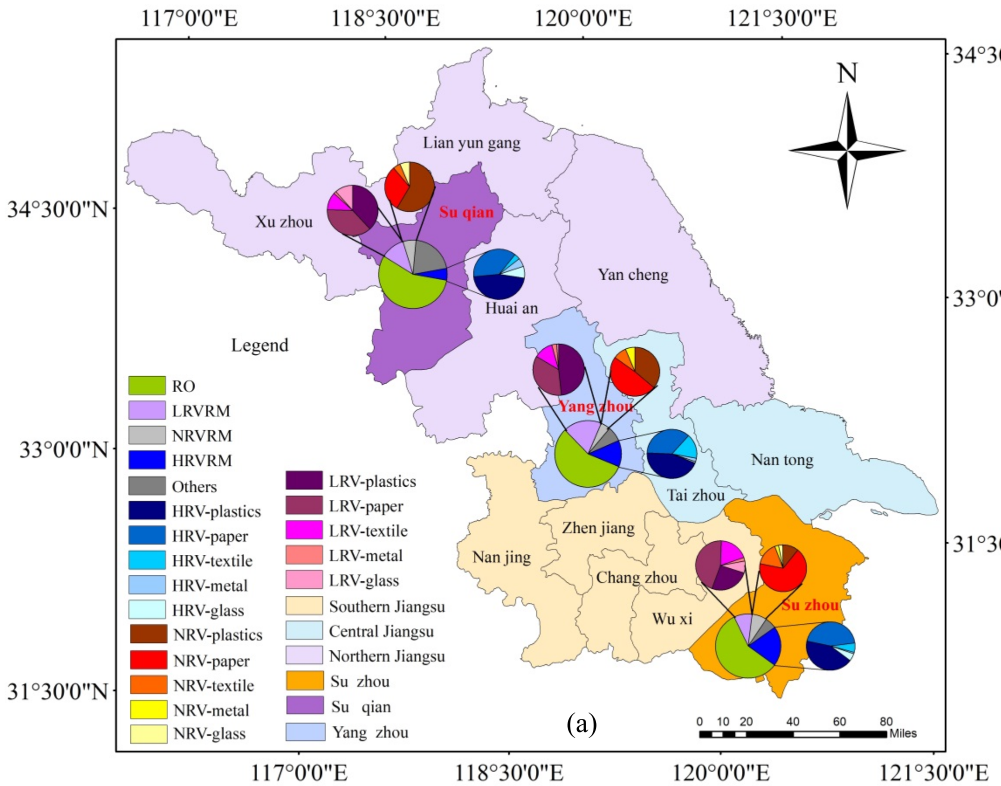
(c)

Fig. 2. Dynamic changes of RWRP in (a) Suzhou, (b) Yangzhou, and (c) Suqian.

3.2. Recyclable waste recycling potential (RWRP) disparities in 2019

Obvious spatial disparities of RWRP were observed between inter-city and within intra-city in these case studies. This was verified by the latest data for 2019 based on our study of the three cities via surveys shown in Fig. 3. The inter-city disparities of RW stream generation are presented in Fig. 3(a). First, the RWRP was analyzed. Suzhou (87.0%) is on a par with Yangzhou (88.0%). Furthermore, both cities had a higher RWRP than Suqian (72.8%). Second, mainstream RO was analyzed, which showed similar levels in the three case cities (57.6% for Suzhou, 56.0% for Yangzhou, and 56.0% for Suqian) due to the high food waste contribution, mentioned in Section 3.1. Third, RVRM was analyzed and results showed 29.4% in Suzhou, 31.6% in Yangzhou, and 16.8% in Suqian, respectively. Furthermore, more HRVRMs (20.3%) were observed in Suzhou, in which, plastics, paper, textile, metal and glass accounted for 8.7%, 9.1%, 1.3%, 0.2% and 1.0%, respectively. One of the main reasons for this phenomenon is that citizens in Suzhou tend to consume more multiple pre-packed products made of plastic and paper. It was found that packaging with multiple container products of MSW accounted for 70.7% in Suzhou (Gu et al., 2018). More LRVRMs (18.9%) were observed in Yangzhou, in which, plastics, paper, textile, metal and glass accounted for 9.1%, 6.6%, 2.3%, 0.7% and 0.2%, respectively. This could be attributed to Yangzhou's development as a small and exquisite tourism city. In 2018, the number of inbound tourists reached 76.4 million people, and tourism revenue was $8.3 million, which accounted for 16.6% of the local GDP that year (YBSC, 2019). A lot of local snack packages (e.g., Yangzhou old goose, Beggar's chicken) were found during the investigation, which were mainly made of LRV-paper and LRV-plastic. The waste sampling locations were located near tourist attractions, which indicated the tourist effect on RW stream generation.

Suzhou is a developed tourist city with a wider urban area than Yangzhou (Table 1); thus, scenic areas and tourist stops are scattered, causing corresponding tourism waste to be diluted to the point where snack packages are not obvious in Suzhou. MSW "tracks" the whereabouts of humans and scatters in every corner of the city (Gu et al., 2017b), so does RW, because RW is an important physical component of MSW. More NRVRMs (27.3%) were observed in Suqian, which could be attributed to the improvement of people's living standards, mentioned in Section 3.1. However, HRVRMs and LRVRMs were 5.3% and 11.5%, respectively. Suqian's RVRMs are expected to increase in the future, along with local economic development.



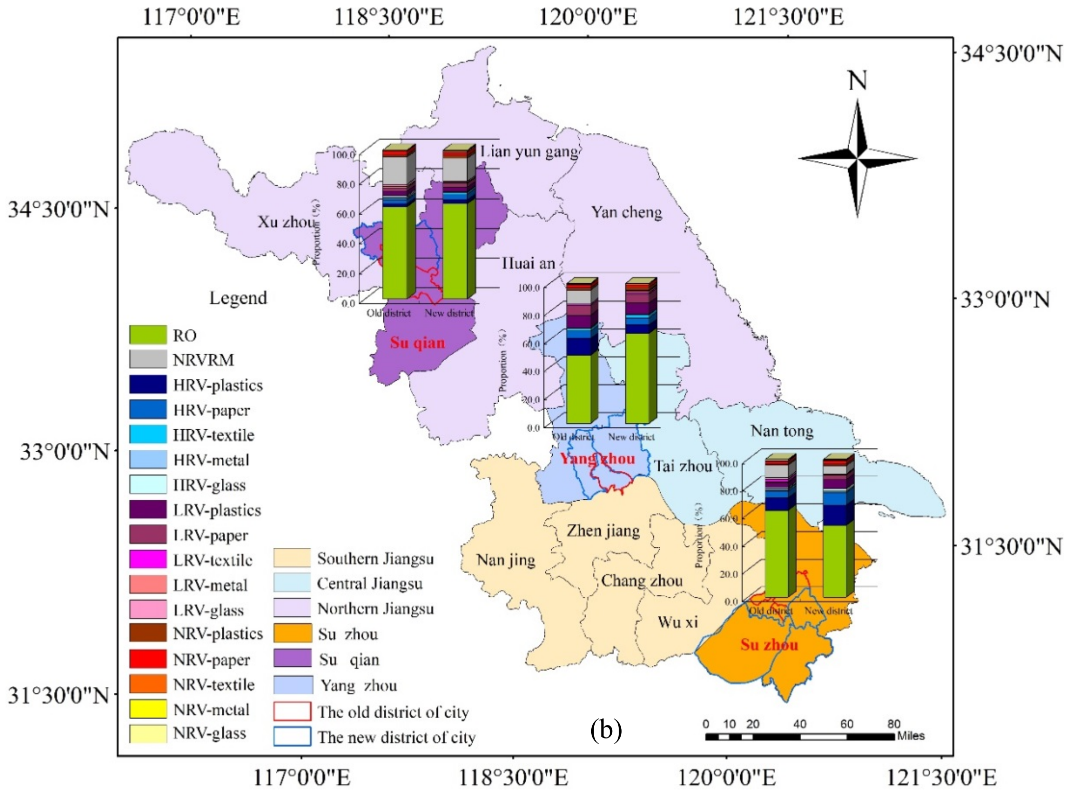


Fig. 3. RW stream generation in Suzhou, Yangzhou, and Suqian in 2019.

Note: (a) This illustration uses pie charts to show the old districts' the inter-city disparities of RW stream generation within the scope of intra-city Geographic Information Software, while (b) shows the differences in the intra-city RWRP of the new districts. More data information is presented in Tables S5-6.

The intra-city disparities of RW stream generation are presented in Fig. 3(b). In Suzhou, more RO (63.0%) were generated in old districts, and more RM (37.3%) was generated in new districts. Additionally, more HRV-plastics (14.7%) and HRV-paper (9.0%) were generated in new districts. The reasons behind these increases could be because: 1) more than 65.0% of Suzhou's restaurants and attractions are located in old districts; 2) many adult Suzhou children eat in an old district and live in a new district; 3) people living in a new district are mainly young white-collar workers who prefer to go shopping, resulting in the generation of more plastics and paper packaging (Gu et al., (2015; 2018)). Generally, these packages are made of HRV-plastics and HRV-paper.

In Yangzhou, more RO (64.2%) were generated in new districts due to food waste (64.0%). In Yangzhou's old districts, food waste was 47.0% and was different from that in Suzhou. Yangzhou's schools, supermarkets, and government offices are concentrated in the old districts. Moreover, most residential buildings in old districts are school district housing (SDH). In most Chinese cities, a primary school student's assigned school district is based on his or her household registration. If a student can enter a high-level primary or secondary school, his/her domicile house is called an SDH, which is a special phenomenon in China. Generally, Chinese parents buy an SDH to keep their children "winning at the starting line." In Yangzhou, the housing area of SDH is greater than or equal to 30 square meters. In fact, the SDH is uninhabitable. Thus, most primary school students are registered in SDH in old districts, but they actually live in new districts. In other words, many SDHs in old districts are vacant. Thus, household registrants in old districts eat and live in new districts. As a result, more ROs are generated in the new Yangzhou districts. Additionally, more RVRM (36.9%) was generated in the old Yangzhou district, in which plastics and paper were 21.0% and 12.1%, respectively. This is because the old district has primary, secondary and university campuses, and they adjoin areas where supermarkets and government offices are concentrated; thus, the old district generated more recycled plastic and paper. RW stream generation is influenced by prominent characteristics referred to as “small but delicate” in Yangzhou.

There is no significant difference in RO and RM in the intra-city of Suqian. The only explanation is that the level of economic development is relatively low, and local residents’ lifestyle and consumption preferences are similar.

3.3. Comparison with other cities in the world

This study used the RWRP data of Suzhou, Yangzhou, and Suqian to compare the regional discrepancies in the RWRP of RW data from 69 international cities reported in established English/Chinese journals. The addition of the three case cities in this study gave us an overview of RWRP in 72 cities worldwide (Fig. 4). Our analysis finds that there are no significant differences in the RP of RO and RM among the cities in this overview based on each city’s income levels. Previous studies (Hoornweg and Bhada-Tate, 2012; Vergara and Tchobanoglous, 2012) reported that more RM was generated in wealthy regions and more RO were generated in economically underdeveloped regions. With that in mind, we classified the 72 cities into three groups based on per capita annual GDP: 1) lower than \$10,000, 2) 10,000 to less than \$20,000, and 3) \$20,000 or more. The criteria for grouping and the case cities selection of this study are the same (see Table 1). No direct relation was found between the RW stream generation and income levels. Moreover, after switching views to RWRP between Chinese cities and foreign cities, a higher proportion of RO was observed in China. This is due to Chinese dietary habits and preference for unpackaged fruit and vegetables.

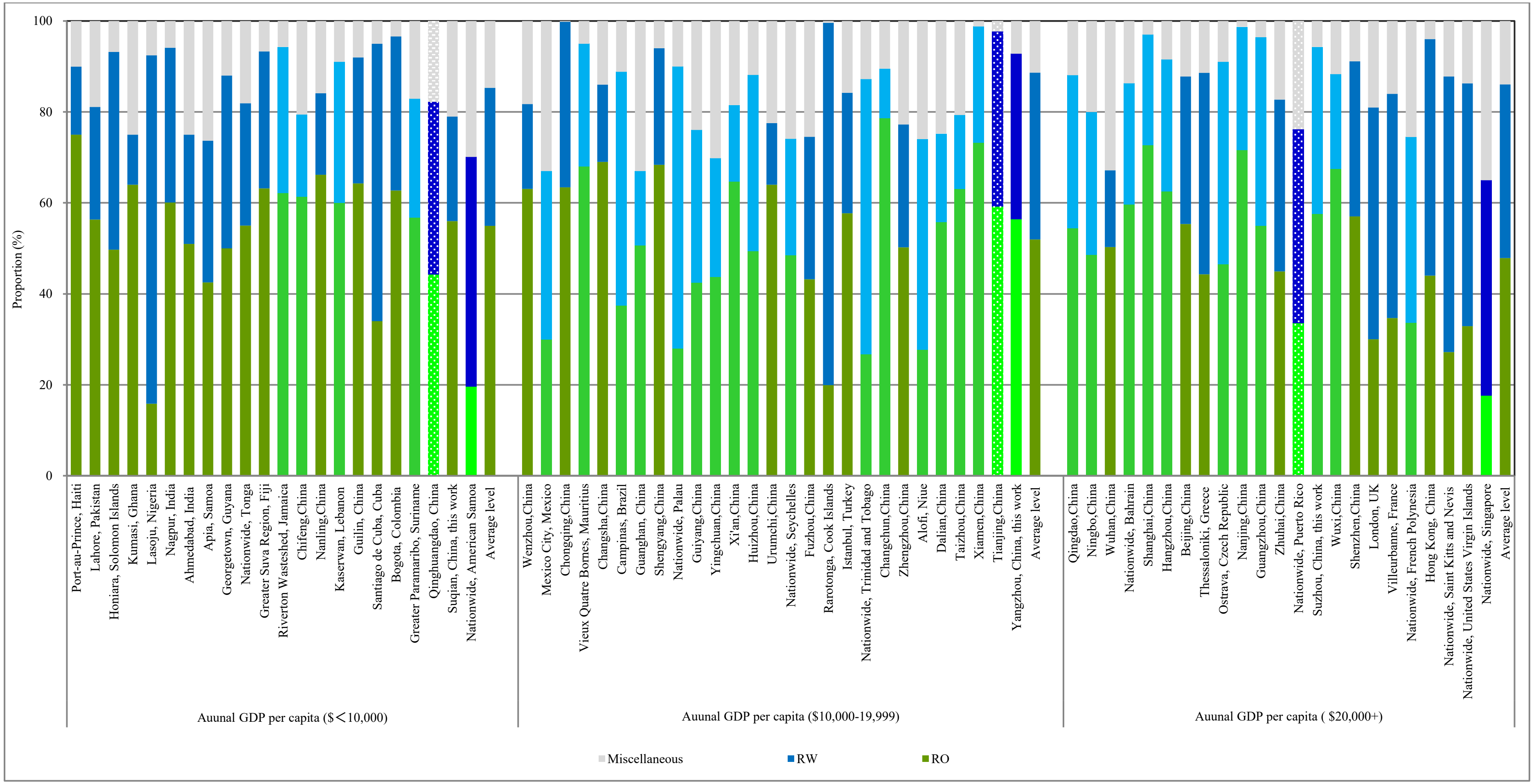


Fig. 4. RWRP in 72 cities throughout the world.

Note: Data sources for the references are provided in Table S7.

At the end of the 72-city international comparison, a notable lack of obvious differences in RWRP was recorded. However, significant fluctuations were noted in individual cities. For example, the highest RO proportion (75.0%) was in the low-income city of Port-au-Prince, Haiti (Vázquez-Rowe et al., 2015), and the highest RM (79.7%) was in the middle-income city of Rarotonga, Cook Islands (Mohee et al., 2015).

Another example of high RM is in the low-income city is Ilorin, Nigeria (Ibikunle et al., 2020). This is not surprising. Haiti is a low income country with a per capita annual GDP of $730.3. Surprisingly, it had the highest RO, which is typically linked to lower purchase power. Margallo et al. (2019) ultimately connects a higher RO proportion with more food purchases, and Haiti is also a family-farm reliant country, which has benefited from USAID (USAID, 2020), making RO a prominent part of their MSW. Cook Islands is located in the central-southern Pacific Ocean and is comprised of 15 islands. In this country, tourism and offshore finances are very developed, which explains the high import rates of consumer products, accompanied by high levels of packaging waste. The plastic, glass and metal components are 22.1%, 23.5%, and 33.5%, respectively (Mohee et al., 2015). Nigeria has a higher RM of 76.6%, with plastic and nylon accounting for 57.7%. The nylon is imported from the sale of sachet water packaged in polyethylene sachets, also known as cellophane/nylon bags. It is a low cost way to capture water in the perennial high temperatures above 30 C near Niger (Ibikunle et al., 2020).

To provide a more comprehensive understanding of RWRP, further estimations and comparisons were carried out. Distribution curve ranges were characterized; then, 10,000 simulations were set. Table 2 presents the results of three case studies versus 69 other cities in the world, and 35 Chinese cities versus 37 foreign cities, as well as previous studies' results. RO proportions of Chinese cities were on a similar level ranging from 56.0 to 59.3% of the total MSW. Foreign cities had an RO proportion that was inversely proportional to the economic development level. Resultant RO proportions of this study agree with the World Bank's reports (World Bank, 2012) and Hoornweg and Bhada-Tata (2012), but the RO proportions were higher than the global components (Margallo et al., 2019). Vergara and Tchobanoglous (2012) agreed that RM is positively correlated with income level. Several factors support this finding: First, factors, such as economic situation, dietary habits, residential lifestyles, consumption structure, geographical location, energy usage, local customs and culture influence the RW stream generation (Chen et al., 2010; Gu et al, 2017a; Margallo et al., 2019; Vergara and Tchobanoglous, 2012). Second, the definition of a city differs from country to country. Thus, the definition of a city in China is broader than in some other countries (Wang et al., 2012). As a result, many tourist attractions, industrial parks, and even rural suburbs are included within the city limits of different cities in China. Different urban areas generate different RW streams. Third, RW data include several cities, sources, and authors as well as data covering longer time spans. Although the data represents results after 2000.

Table 2. RWRP in this and previous studies

Income level	Cities	RO	RM	Others
	Case cities versus other cities			
	Suzhou	57.6	36.7	5.7
High	Other 20 cities in the world	47.9	38.2	13.9
	Yangzhou	56.4	36.4	7.2
Middle	Other 26 cities in the world	51.9	32.0	16.1
	Suqian	53.2	32.5	14.3
Low	Other 23 cities in the world	55.0	30.4	14.7
	Chinese cities versus foreign cities			
	Chinese cities	57.0	32.1	10.8
High	Other cities in the world	36.0	46.1	17.9
	Chinese cities	59.3	25.7	16.6
Middle	Other cities in the world	42.7	41.7	15.6
	Chinese cities	58.4	24.2	17.4
Low	Other cities in the world	53.9	32.1	14.0
	The World Bank, (2005)			
China		57.0	21.0	22.0
	Ma et al., (2020)			
China		53.7	34.9	11.4
	Margallo et al., (2019)			
Global in 2016		44.0	38.0	18.0

3.4. Factors impacting recyclable waste (RW) stream generation

Modeling results show that the RW stream generation was positively correlated with these factors of household consumption structure (HCS), urban resident population, and economic development (Table 3), in which, HCS plays an important role. The HCS in this study is presented by the eight different ICE items (see the note in Table 3).

In Suzhou, the linear regression between two variables (RW generation and GDP) is statistically significant (t_{stat} = 5.444, *sig.* = 0.002) and has an R^2 of 0.832. In Yangzhou and Suqian, the driving factors of ICE were significantly associated with their RW, and the ICE items were the same in the two cities as shown in the statistical test results in Yangzhou (t_{stat} = 4.711, *sig.* = 0.002, and R^2 = 0.760) and in Suqian (t_{stat} = 5.949, *sig.* = 0.001, and R^2 = 0.855), respectively. Take RO as another example: In Suzhou, the linear regression of RO and urban resident population is statistically significant (t_{stat} = 7.884, *sig.* = 0.000) with a good fit (R^2 = 0.899); in Yangzhou, the linear regression of RO and food items is statistically significant (t_{stat} = 7.110, *sig.* = 0.000) with a good fit (R^2 = 0.592); in Suqian, the linear regression of RO and food items is statistically significant (t_{stat} = 10.262, *sig.* = 0.000) with a good fit (R^2 = 0.968).

All RW stream generation models are shown in Table 3, where HCS affected the RW stream generation and was well verified in the case cities. The results of this study agree with some previous studies (Gu et al., 2017a; Li et al., 2011; Liu et al., 2019), indicating that RW is an important physical component of MSW coming from urban residents' consumption. However, the per capita annual GDP factor was found in Suzhou, and the urban resident population factor was found in Yangzhou. This could explain the HCS's impact on RW stream generation, which was counteracted by rapid economic development and urban population growth. Suzhou is a typical developed city, bordering Shanghai, with an annual average growth rate of 10.7% in 2019 (according to a local official), which is faster than China's overall growth rate (6.1%). Factors resulting from our study in Suzhou agree with our preliminary finding (Gu et al., 2018) wherein the per capita annual GDP is not only a variable used to explain the MSW generation, but also a better indicator of change than the size of the residential population. Some talent introduction policies were carried out in Yangzhou. With more and more people crowding into the city, the corresponding MSW and RW stream generation are changing. More details policy implications are available in Section 4.2.

Table 3. Linear regression analysis for RW stream generation in case cities.

Cities	Variables	Coefficients (unstandardized) Coefficients	Standard Error	*t-stat*	*Sig.*	90% CI		R^2	AdR^2	Durbin–Watson
Suzhou	RW									
	Constant	-843.362	176.609	-4.775	0.003	-1186.545	-500.178	0.832	0.804	2.974
	GDP	1.221	0.224	5.444	0.002	0.785	1.657			
	RO									
	Constant	-8.454	3.514	-2.606	0.037	-15.111	-1.797	0.899	0.884	2.299
	GDP	0.008	0.001	7.884	0.000	0.006	0.010			
	RM									
	Constant	39.553	5.266	7.510	0.000	29.319	49.787	0.883	0.863	2.178
	HOUS	0.005	0.001	6.722	0.001	0.004	0.007			
	HRVRM									
	Constant	18.056	5.290	3.413	0.014	7.777	28.335	0.856	0.831	2.612
	ICE (HOUS+Education+RECE)	0.003	0.001	5.961	0.001	0.002	0.004			
	LRVRM									
	Constant	-74.736	8.144	-9.177	0.000	-90.165	-59.307	0.951	0.945	1.353
	GDP	0.336	0.029	11.718	0.000	0.282	0.391			
	ICE (Food)	0.001	0.000	3.419	0.011	0.000	0.002			
	Traditional RW									
	Constant	69.224	3.778	18.322	0.000	62.066	76.382	0.849	0.824	2.568
	ICE (HOUS+clothing+Education+RECE)	0.012	0.002	5.812	0.001	0.008	0.016			
Yangzhou	RW									
	Constant	-25.363	9.415	-2.694	0.031	-43.200	-7.526	0.760	0.726	2.015
	ICE (Clothing+FUNI+HLTH+OTHR)	0.008	0.002	4.711	0.002	0.005	0.011			
	RO									
	Constant	64.772	2.213	29.269	0.000	60.580	68.965	0.592	0.534	2.311
	ICE (Food)	0.067	0.000	7.110	0.000	0.048	0.086			
	RM									
	Constant	26.167	0.599	43.707	0.000	25.032	27.301	0.793	0.763	1.611
	Urban resident population	0.055	0.000	5.178	0.001	0.036	0.075			
	HRVRM									
	Constant	-2.234	0.914	-2.443	0.035	-3.967	-0.502	0.713	0.672	1.510
	ICE (HOUS+Education+RECE)	0.002	0.000	4.173	0.004	0.001	0.003			
	LRVRM									
	Constant	-203.956	59.087	-3.452	0.014	-318.773	-89.139	0.710	0.662	1.074
	ICE (Food+OTHR)	0.288	0.075	3.835	0.009	0.142	0.434			

	Traditional RW									
	Constant	-29.309	8.572	-3.419	0.011	-45.549	-13.069	0.830	0.806	1.868
	ICE (Food+Clothing+FUNI+Education+RECE+HLTH+OTHR)	0.003	0.001	5.847	0.001	0.002	0.004			
Suqian	RW									
	Constant	-1416.152	269.700	-5.251	0.002	-1940.228	-892.076	0.855	0.831	2.678
	ICE (Clothing+FUNI+HLTH+OTHR)	2.038	0.343	5.949	0.001	1.373	2.704			
	RO									
	Constant	-15.960	3.008	-5.306	0.001	-21.659	-10.261	0.968	0.938	1.923
	ICE (Food)	0.006	0.001	10.262	0.000	0.005	0.007			
	RM									
	Constant	-2.809	1.096	-2.562	0.037	-4.886	-0.732	0.854	0.834	2.396
	ICE (Clothing+FUNI+HLTH+OTHR)	0.002	0.000	6.407	0.000	0.001	0.003			
	HRVRM									
	Constant	-52.300	6.534	-8.004	0.000	-64.679	-39.921	0.947	0.939	1.201
	ICE (HOUS+Education+RECE)	0.269	0.024	11.180	0.000	0.224	0.315			
	LRVRM									
	Constant	-4.263	0.982	-4.342	0.003	-6.123	-2.403	0.874	0.857	2.345
	ICE (Food)	0.001	0.000	6.984	0.000	0.001	0.002			
	Traditional RW									
	Constant	-13.071	2.958	-4.419	0.003	-18.676	-7.467	0.950	0.943	2.431
	ICE (Food+Clothing+FUNI+Education+RECE+HLTH+OTHR)	0.003	0.000	11.528	0.000	0.003	0.004			

Note: HOUS represents "household appliances," RECE represents "cultural and recreation services," FUNI represents "furniture articles and services," HLTH represents "medicine and medical services," OTHR represents "miscellaneous commodities and services." The ICE used the Yuan (RMB), the GDP per capita is 10,000 Yuan (RMB) and the urban resident population consists of approximately 10,000 people (population). Generally, the monetary unit does not affect RW stream generation, and more attention is paid to the factors, factors' plus or minus, and factors' intensity.

4. Policy implications

Policy implications of this study are valuable for other cities nationwide, and the results of this study represent high-, middle- and low-income cities in China, which can be compared to other cities around the world. Based on understanding the RWRP systematic multiple longitudinal field tracking survey results, three policy implementations are recommended: 1) initiating flexible and well planned recycling strategies, 2) developing a composting product distribution market as well as 3) incorporating economic policies and demographic policies as follows.

First, flexible and well planned recycling strategies should be implemented to accommodate localized RM generation characterization. The results show that significant spatial disparities of RWRP were observed between inter-city and within intra-city (Fig. 3). A major contributor of RW in the case study cities is RVRM. Suzhou has more HRV-plastics (14.7%) and HRV-paper (9.0%) in new districts. Yangzhou has more LRV-plastics (10.0%) and HRV-paper (6.8%) in new districts. Additionally, more RVRM is always generated during weekends and before Lunar New Year's events, which are represented in our existing investigation results (Gu et al., (2015; 2018)). Additionally, more RO (63.0%) were found in the old districts of Suzhou, and more RO (59.2%) were found in the new districts of Yangzhou (Fig. 3). Therefore, recycling policies should be more flexible to accommodate well planned management. For instance, the professional recovery equipment for HRV-plastics and HRV-paper should increase in Suzhou, especially in new districts. Sustainable development officials of Yangzhou and Suqian should focus more attention on finding ways to reduce LRVRM to promote recycling and transform or reverse other negative effects of waste prevention. However, there is no significant difference between RO and RM in the intra-city of Suqian. It is inevitable that more and more RVRM will generate H-L-RV paper and H-L-RV plastics, along with the development of the economy and improved living standards.

Second, a regulated market should be established to promote compost product distribution. RO contributes more than 55.0% to the overall MSW in Suzhou, Yangzhou, and Suqian, as noted in this study (Figs. 2 and 3). RO is a reliable and constant recycling source in Chinese cities (Gu et al., (2015; 2017b; 2018)), which break down the MSW. The RO will remain stable due to Chinese dietary habits, which has been proven consistent throughout the country's history. Compost application is seen as a measure that brings nutrients from urban compost products back to the soil that supports the harvest in the rural areas and makes composting possible (Ardolino et al., 2020; Matter et al., 2015). Thus, current composting technology should be developed for market composted fertilizer. RO represents one of the main challenges of a sustainable society and/or city development in China (Guo et al., 2019). Unfortunately, composting is lagging behind in China, especially composting product sales (Wei et al., 2017). To avoid the dissemination of low-quality compost products in the market, a regulated market must be established. Relevant departments should strictly check the quality of composting products. Fiscal incentives should progressively encourage the development of the recycling industry. Moreover, a comprehensive multi-objective green supply chain should be established (Iqbal et al., 2020; Paul and Bussemaker, 2020; Xu et al., 2017).

Third, the economic policy and demographic policy shall be intricately linked and integrated into RW recycling and management. RW stream generations in the three case study cities were driven by HCS; moreover, RW stream generation in Suzhou and Yangzhou was influenced by urban resident population and per capita annual GDP (Table 3). China has endorsed further development of economic revitalization and urbanization in the next decade. However, the sudden onset of COVID-19 in 2020 caused a two-month shutdown of all non-essential businesses. The Chinese government moved quickly to contain the economic fallout, and economic activity has started to rebound. The rapid development of China's economy is an inevitable trend. At the same time, the Chinese urban population increased from 43.4% (556.9 million population) in 2005 to 60.1% (839.5 million population) in 2018 (CNBS, 1981-2019), and will increase continuously based on a conservative forecast of 70.0% (1,008.8 million population) in 2030. The 1,008.8 million urban population is calculated, according to the estimation of 1441.2 million Chinese people in 2030 with at least a 10% increase in urban population (Wang et al., 2012). In 2018, the urban population in Suzhou, Yangzhou, and Suqian accounted for 76.1% (8.2 million population), 67.1% (3.1 million population), and 60.0% (2.9 million population), respectively (JBSC, 2019). Jiangsu is a typical developed province on China's eastern coast with an urban population of 68.7% in 2018, which is higher than the national average level (SBSC, 2019; NBSC, 1981-2019). In this context, our three case study cities are expected to substantially increase undergoing efficient economic development and their urban populations. Moreover, Suzhou is an attractive city with 130.3 million tourists in 2018 (SBSC, 2019). Yangzhou has promulgated a talent introduction system, and Suqian is promoting enterprise investment. Therefore, more urban people, more consumption, and more diversity will affect these cities' RW stream generation. Appropriate policy changes aimed at controlling urban population are unlikely to occur in the next decade in the background of economic development and urbanization. But some policies aimed at achieving a reasonable population density are feasible to deal with the excessive rapidly growing urban population's need to recycle. For instance, Suzhou could implement a residence certificate system that would strengthen the floating population registration and an overall service management system for its citizens. Yangzhou could raise the threshold of talent recruitments; Suqian could access cleaner production and eco-environmental enterprises, making Suqian more environmentally friendly to enter (Wu et al., 2020). We suggested that economic policy and demographic policy should be intricately linked and integrated into RW recycling and management. However, there are 293 prefecture-level cities in China (CNBS, 2020). The three case study cities represent high-, middle- and low-income levels, which are typical when considering the microcosm of Chinese cities, especially, the southeast coastal area cities. Furthermore, in 2050, almost 66% of the people (approximately 6.4 billion) live in urban areas (United Nations Department of Economic and Social Affairs, 2018); thus, the topics of economic policy, demographic policy, and recycling policy will merit attention in the near future.

5. Conclusions

ZW cities are springing up all over the world to abate the increasing and uncontrollable MSW. Recycling is an essential practice for a ZW city to be constructed successfully. Most RW information is currently unreliable. Quantifying and recognizing the local RWRP are essential to building a strong foundation for future ZW cities. A systematic field tracking survey covering multiple longitudinal case cities was implemented in Suzhou, Yangzhou, and Suqian, which represent high-, middle- and low-income cities, respectively in China, between 2016 and 2019.

Results show that the case cities with different income levels have different dynamic changes in RWRP. Suzhou showed a slight increase in HRV-paper; Yangzhou had a slight increase in LRV-paper, and Suqian had a huge increase in NRV paper and plastic. Moreover, RO was a constant in the three cities. An important finding is that spatial disparities of RMRP existed between inter-city and within intra-city. For instance, Suzhou had more HRV-plastics and HRV-paper in its new districts and more RO in its old districts; Yangzhou had more LRV-plastics and HRV-paper in its old districts, more RO in its new districts. However, there was no significant difference between RO and RM in the intra-city of Suqian. Additionally, the HCS plays an important role in RW steam generation. Furthermore, other factors, such as economic development, urban resident population, consumption preferences, local customs, culture, and residential lifestyles also influence RW stream generation. Finally, three policy implications (flexible and well planned recycling strategies, developing a composting products distribution market, and incorporating an economic policy and demographic policy) are proposed.

The dynamic changes and disparities of RWRP were reasonably quantified, and the key factors influencing the RW stream generation were recognized in the three case cities, along with their limitations. RW data came from our field survey covering the period from 2016 to 2019, as well as our Suzhou's survey initiated in 2011. We plan to carry out more multiple longitudinal case surveys in more cities, including internationally representative cities around the world. Although models were established, the future forecast of RWRP and RW stream generation was not determined due to incomplete data. This investigation represents the beginning of our quest for reliable RW data that can serve research, planning, business development, and environmental management. Our ultimate goal is to establish an integrated MSW management system that can ensure the successful implementation of a national recycling strategy to promote ZW cities construction. The ultimate goal is to develop a model for MSW management system that can serve as a model for all countries.

Acknowledgments

This work was supported by the National Natural Science Foundation of China (Project No. 71603227 and Project No. 71974166). We also appreciate the reviewers for their valuable comments and suggestions on improving the quality of this paper.

References

Ardolino, F., Colaleo, G., Arena, U., 2020. The cleaner option for energy production from a municipal solid biowaste. J. Clean. Prod. 266, 121908. https://doi.org/10.1016/j.jclepro.2020.121908.

Ayodele, T.R., Alao, M.A., Ogunjuyigbe, A.S.O., 2018. Recyclable resources from municipal solid waste: Assessment of its energy, economic and environmental benefits in Nigeria. Resour. Conserv. Recycl. 134, 165–173. https://doi.org /10.1016/j.resconrec.2018.03.017.

Barrett, J., Scott, K., 2012. Link between climate change mitigation and resource efficiency: a UK case study. Global Environ. Change 22 (1) 299–307. https://doi.org/10.1016/j.gloenvcha.2011.11.003.

Brooks, A.L., Wang, S., Jambeck, J.R., 2018. The Chinese import ban and its impact on global plastic waste trade. Sci. Adv. 4, 1–8. https://doi.org/10.1126/sciadv.aat0131.

C40 Cities Climate Leadership UK. 2018. August 28, 2018 press release entitled "23 Global Cities and Regions Advance Towards Zero Waste," available at https://www.c40.org/press_releases/global-cities-and-regions-advance-towards-zero-waste with complete list of signatories at https://www.c40.org/other/zero-waste-declarationavailable and original declaration.

C40 Cities Climate Leadership Group, U.S. 2019. Advancing Towards Zero Waste Declaration, at https://www.c40.org/press_releases/global-cities-and-regions-advance-towards-zero-wte. and with complete list of signatories at https://www.c40.org/other/zero-waste-declaration.

Chang, N.B., Davila, E., 2008. Municipal solid waste characterizations and management strategies for the Lower Rio Grande Valley, Texas. Waste Manag. 28, 776–794. https://doi.org/10.1016/j.wasman.2007.04.002.

Chen, H., Jing, L., Teng, Y., Wang, J., 2018. Characterization of antibiotics in a large-scale river system of China: Occurrence pattern, spatiotemporal distribution and environmental risks. Sci. Total Environ. 618, 409–418. https://doi.org/10.1016/j.scitotenv.2017.11.054.

Chen, X., Geng, Y., Fujita, T., 2010. An overview of municipal solid waste management in China. Waste Manag. 30, 716–724. https://doi.org/10.1016/j.wasman.2009.10.011.

China National Bureau of Statistics (CNBS), 1981-2020. China Statistical Yearbook. China Statistical Press, Beijing (in Chinese).

Fernández-González, J.M., Grindlay, A.L., Serrano-Bernardo, F., Rodríguez-Rojas, M.I., Zamorano, M., 2017. Economic and environmental review of Waste-to-Energy systems for municipal solid waste management in medium and small municipalities. Waste Management, 67, 360–374. https://doi.org/10.1016/j.wasman.2017.05.003.

Fudala-Ksiazek, S., Pierpaoli, M., Kulbat, E., Luczkiewicz, A., 2016. A modern solid waste management strategy–the generation of new by-products. Waste Management 49, 516–529. https://doi.org/10.1016/j.wasman.2016.01.022.

Geng, Y., Sarkis J., Ulgiati S, Z.P., 2013. Measuring China's Circular Economy. Science (80). 339, 1526–1527. https://doi.org/ 10.1126/science.1227059.

Geyer, R., Jambeck, J.R., Law, K.L., 2017. Production, use, and fate of all plastics ever made. Sci. Adv. 3 (7), e1700782. https://doi.org/10.1126/sciadv.1700782.

Ghanimeh, S., Gómez-Sanabria, A., Tsydenova, N., Štrbová, K., Iossifidou, M., Kumar, A., 2019. Two-level comparison of waste management systems in low-, middle-, and high-Income cities. Environ. Eng. Sci. 36, 1281–1295. https://doi.org/10.1089/ees.2019.0047.

Gu, B., Fujiwara, T., Jia, R., Duan, R., Gu, A., 2017a. Methodological aspects of modeling household solid waste generation in Japan: Evidence from Okayama and Otsu cities. Waste Manag. Res. 35, 1237–1246. https://doi.org/10.1177/0734242X17738338.

Gu, B., Jiang, S., Wang, H., Wang, Z., Jia, R., Yang, J., He, S., Cheng, R., 2017b. Characterization, quantification and management of China's municipal solid waste in spatiotemporal distributions: A review. Waste Manag. 61, 67–77. https://doi.org/10.1016/j.wasman.2016.11.039.

Gu, B., Li, Y., Jin, D., Yi, S., Gu, A., Bu, X., Zhou, H., He, S., Cheng, R., Jia, R., 2018. Quantizing, recognizing, and characterizing the recycling potential of recyclable waste in China: A field tracking study of Suzhou. J. Clean. Prod. 201, 948–957. https://doi.org/10.1016/j.jclepro.2018.08.085.

Gu, B., Wang, H., Chen, Z., Jiang, S., Zhu, W., Liu, M., Chen, Y., Wu, Y., He, S., Cheng, R., Yang, J., Bi, J., 2015. Characterization, quantification and management of household solid waste: A case study in China. Resour. Conserv. Recycl. 98, 67–75. https://doi.org/10.1016/j.resconrec.2015.03.001.

Gu, B., Zhu, W., Wang, H., Zhang, R., Liu, M., Chen, Y., Wu, Y., Yang, X., He, S., Cheng, R., Yang, J., Bi, J., 2014. Household hazardous waste quantification, characterization and management in China's cities: A case study of Suzhou. Waste Manag. 34, 2414–2423. https://doi.org/10.1016/j.wasman.2014.06.002.

Guo, H., Zhao, Y., Damgaard, A., Wang, Q., Lu, W., Wang, H., Christensen, T.H., 2019. Material flow analysis of alternative biorefinery systems for managing Chinese food waste. Resour. Conserv. Recycl. 149, 197–209. https://doi.org/10.1016/j.resconrec.2019.05.010.

Hering, J.G., 2012. An end to waste? Science (80). 337, 623. https://doi.org/10.1126/science.1227092.

Hoornweg, Dan., Bhada-Tata, Perinaz., 2012. What a Waste: A Global Review of Solid Waste Management, Urban Development Series Knowledge Paper #15, Produced by the World Bank's Urban Development and Local Government Unit of the Sustainable

Development Network, © World Bank, 2012.

Huang, C.L., Yu, C.P., Lin, T., Ye, Z., 2016. Water conservation significance of municipal solid waste management: A case of Xiamen in China. J. Clean. Prod. 129, 693–703. https://doi.org/10.1016/j.jclepro.2016.03.062.

Ibikunle, R.A., Titiladunayo, I.F. Lukmanc, A.F. Dahunsi, S.O., Akeju, E.A., 2020. Municipal solid waste sampling, quantification and seasonal characterization for power evaluation: Energy potential and statistical modeling. Fuel. 277, 118122. https://doi.org/10.1016/j.fuel.2020.118122.

Iqbal, M.W., Kang, Y., Jeon, H. W., 2020. Zero waste strategy for green supply chain management with minimization of energy consumption. J. Clean. Prod. 245, 118827. https://doi.org/10.1016/j.jclepro.2019.118827.

Jang, Y.C., Lee, G., Kwon, Y., Lim, J.H., Jeong, J.H., 2020. Recycling and management practices of plastic packaging waste towards a circular economy in South Korea. Resour. Conserv. Recycl.158, 104798. https://doi.org/10.1016/j.resconrec.2020.104798.

Jiangsu Bureau of Statistics of China (JBSC), 2019. Jiangsu Statistical Yearbook. Jiangsu Statistical Press, Jiangsu (in Chinese).

Kaza, Silpa; Yao, Lisa C.; Bhada-Tata, Perinaz; Van Woerden, Frank. 2018. What a Waste 2.0 : A Global Snapshot of Solid Waste Management to 2050. Urban Development;. Washington, DC: World Bank. © World Bank. https://openknowledge.worldbank.org/handle/10986/30317 License: CC BY 3.0 IGO."

Khandelwal, H., Dhar, H., Thalla, A.K., Kumar, S., 2019. Application of life cycle assessment in municipal solid waste management: A worldwide critical review. J. Clean. Prod. 209, 630–654. https://doi.org/10.1016/j.jclepro.2018.10.233.

Li, Z., Fu, H., Qu, X., 2011. Estimating municipal solid waste generation by different activities and various resident groups: A case study of Beijing. Sci. Total Environ. 409, 4406–4414. https://doi.org/10.1016/j.scitotenv.2011.07.018.

Liu, G., Agostinho, F., Duan, H., Song, G., Wang, X., Giannetti, B.F., Santagata, R., Casazza, M., Lega, M., 2020. Environmental impacts characterization of packaging waste generated by urban food delivery services. A big-data analysis in Jing-Jin-Ji region (China). Waste Manag. 117, 157–169. https://doi.org/10.1016/j.wasman.2020.07.028.

Liu, J., Li, Q., Gu, W., Wang, C., 2019. The impact of consumption patterns on the generation of municipal solid waste in China: Evidences from provincial data. Int. J. Environ. Res. Public Health 16, 1–19. https://doi.org/10.3390/ijerph16101717.

Ma, S., Zhou, C., Chi, C., Liu, Y., Yang, G., 2020. Estimating Physical Composition of Municipal Solid Waste in China by Applying Artificial Neural Network Method. Environ. Sci. Technol. 2020, 54, 9609−9617. https://dx.doi.org/10.1021/acs.est.0c01802.

Maghmoumi, A., Marashi, F., Houshfar, E., 2020. Environmental and economic assessment of sustainable municipal solid waste management strategies in Iran. Sustainable Cities and Society. 59, 102161. https://doi.org/10.1016/j.scs.2020.102161.

Margallo, M., Ziegler-Rodriguez, K., Vázquez-Rowe, I., Aldaco, R., Irabien, Á., Kahhat, R., 2019. Enhancing waste management strategies in Latin America under a holistic environmental assessment perspective: A review for policy support. Sci. Total Environ.

689, 1255–1275. https://doi.org/10.1016/j.scitotenv.2019.06.393.

Matter, A., Ahsan, M., Marbach, M., Zurbrügg, C., 2015. Impacts of policy and market incentives for solid waste recycling in Dhaka, Bangladesh. Waste Manag. 39, 321–328. https://doi.org/10.1016/j.wasman.2015.01.032.

Mohee, R., Mauthoor, S., Bundhoo, Z.M.A., Somaroo, G., Soobhany, N., Gunasee, S., 2015. Current status of solid waste management in small island developing states: A review. Waste Manag. 43, 539–549. https://doi.org/10.1016/j.wasman.2015.06.012.

National Bureau of Statistics of China (NBSC), 1981-2019. China Statistical Yearbook. China Statistics Press, Beijing (in Chinese).

Organization for Economic Co-operation and Development (OECD), 2018. Environment Data: Compendium 2018. http://www.oecd.org.

Ozcan, H.K., Guvenc, S.Y., Guvenc, L., Demir, G., 2016. Municipal solid waste characterization according to different income levels: A case study. Sustain. 8. https://doi.org/10.3390/su8101044.

Paes, M.X., de Medeiros, G.A., Mancini, S.D., Gasol, C., Pons, J.R., Durany, X.G., 2020. Transition towards eco-efficiency in municipal solid waste management to reduce GHG emissions: The case of Brazil. J. Clean. Prod. 263, 121370. https://doi.org/10.1016/j.jclepro.2020.121370.

Paul, M., Bussemaker M.J., 2020. A web-based geographic interface system to support decision making for municipal solid waste management in England. J. Clean. Prod. 263, 121461. https://doi.org/10.1016/j.jclepro.2020.121461.

Pietzsch, N., Ribeiro, J.L.D., de Medeiros, J.F., 2017. Benefits, challenges and critical factors of success for Zero Waste: A systematic literature review. Waste Manag. 67, 324–353. https://doi.org/10.1016/j.wasman.2017.05.004.

Saeed, M.O., Hassan, M.N., Mujeebu, M.A., 2009. Assessment of municipal solid waste generation and recyclable materials potential in Kuala Lumpur, Malaysia. Waste Manag. 29, 2209–2213. https://doi.org/10.1016/j.wasman.2009.02.017.

Shahbazi, S., Wiktorsson, M., Kurdve, M., Jönsson, C., Bjelkemyr, M., 2016. Material efficiency in manufacturing: Swedish evidence on potential, barriers and strategies. J. Clean. Prod. 127, 438–450. https://doi.org/10.1016/j.jclepro.2016.03.143.

Suzhou Bureau of Statistics of China (SBSC), 2019. Suzhou Statistical Yearbook. China Statistical Press, Suzhou (in Chinese).

Tai, J., Zhang, W., Che, Y., Feng, D., 2011. Municipal solid waste source-separated collection in China: A comparative analysis. Waste Manag. 31, 1673–1682. https://doi.org/10.1016/j.wasman.2011.03.014.

Thanh, N.P., Matsui, Y., Fujiwara, T., 2010. Household solid waste generation and characteristic in a Mekong Delta City, Vietnam. J. Environ. Manag. 91, 2307-2321. doi:10.1016/j.jenvman.2010.06.016.

United Nations Department of Economic and Social Affairs Population Division , 2018 Revision of World Urbanization Prospect, 2018.

US Aid. 2020. Economic Growth and Agricultural Development, online USAID report on

Haiti, https://www.usaid.gov/haiti/agriculture-and-food-security.

Vázquez-Rowe, I., Golkowska, K., Lebuf, V., Vaneeckhaute, C., Michels, E., Meers, E., Benetto, E., Koster, D., 2015. Environmental assessment of digestate treatment technologies using LCA methodology. Waste Manag. 43, 442–459. https://doi.org/10.1016/j.wasman.2015.05.007.

Vergara, S.E., Tchobanoglous, G., 2012. Municipal solid waste and the environment: A global perspective, Annual Review of Environment and Resources. https://doi.org/10.1146/annurev-environ-050511-122532.

Wan, C., Shen, G.Q., Choi, S., 2018. Understanding public support for recycling policy: To unveil the political side of influence and implications. Environ. Sci. Policy 82, 30–43. https://doi.org/10.1016/j.envsci.2018.01.005.

Wang, H., Zhang, R., Liu, M., Bi, J., 2012. The carbon emissions of Chinese cities. Atmos. Chem. Phys. 12, 6197–6206. https://doi.org/10.5194/acp-12-6197-2012.

Wei, Y., Li, J., Shi, D., Liu, G., Zhao, Y., Shimaoka, T., 2017. Environmental challenges impeding the composting of biodegradable municipal solid waste: A critical review. Resour. Conserv. Recycl. 122, 51–65. http://dx.doi.org/10.1016/j.resconrec.2017.01.024.

Wu, D., Zhang, C., Lü, F., Shao, L., He, P., 2014. The operation of cost-effective on-site process for the bio-treatment of mixed municipal solid waste in rural areas. Waste Manag. 34, 999–1005. https://doi.org/10.1016/j.wasman.2013.12.002.

Wu, W., Zhang Q., Liang Z., 2020. Environmentally responsible closed-loop supply chain models for joint environmental responsibility investment, recycling and pricing decisions. J. Clean. Prod. 259, 120776. https://doi.org/10.1016/j.jclepro.2020.120776.

World Bank, 2012. Waste Management in China: Issues and Recommendations. In: 9, U.D.W.P. (Ed.), East Asia Infrastructure Department.

World Bank, 2018. What a Waste: An Updated Look into the Future of Solid Waste Management," an online news article published Sept. 20, 2018, available at https://www.worldbank.org/en/news/immersive-story/2018/09/20/what-a-waste-an-updated-look-into-the-future-of-solid-waste-management.

Xiao, S., Dong, H., Geng, Y., Tian, X., Liu, C., Li, H., 2020. Policy impacts on Municipal Solid Waste management in Shanghai: A system dynamics model analysis. J. Clean. Prod. 262, 121366. https://doi.org/10.1016/j.jclepro.2020.121366.

Xu, Z., Elomri, A., Pokharel, S., Zhang, Q., Ming, X.G., Liu, W., 2017. Global reverse supply chain design for solid waste recycling under uncertainties and carbon emission constraint. Waste Manag. 64, 358–370. https://doi.org/10.1016/j.wasman.2017.02.024.

Yangzhou Bureau of Statistics of China (YBSC), 2019. Yangzhou Statistical Yearbook. China Statistical Press, Yangzhou (in Chinese).

Zhang, F., Wang, Y., Ma, X., Wang, Y., Yang, G., Zhu, L., 2019. Evaluation of resources and environmental carrying capacity of 36 large cities in China based on a support-pressure coupling mechanism. Sci. Total Environ. 688, 838–854. https://doi.org/10.1016/j.scitotenv.2019.06.247.

Zheng, L., Song, J., Li, C., Gao, Y., Geng, P., Qu, B., Lin, L., 2014. Preferential policies promote municipal solid waste (MSW) to energy in China: Current status and prospects.

Renew. Sustain. Energy Rev. 36, 135–148. https://doi.org/10.1016/j.rser.2014.04.049.