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Life cycle assessment of antibiotic mycelial residues management in China

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A B S T R A C T

The increasing antibiotic mycelial residues (AMRs) have brought significant threats to our ecosystems and public health. Aiming to quantify pollutants generated from AMRs management, evaluate corresponding environmental impacts, and identify the key factors related with AMRs management, a life cycle assessment approach was employed on analyzing AMRs. In order to improve the accuracy of results, uncertainty analysis was also performed so that a holistic picture of environmental emissions generated from AMRs management is presented. Results show that human toxicity, terrestrial ecotoxicity, marine ecotoxicity, and fossil depletion are the major environmental impacts caused by AMRs management. Uncertainty analysis reveals that the gasification of AMRs is the best option among the four AMRs treatment scenarios due to its large energy recovery capacity, while incineration scenario has the worst environmental performance due to its large pollutants emissions and sodium hydrogen consumption. Results obtained from this study (e.g., environmental impacts, key factors, and potential improvement) could provide valuable insights to policymakers so that the overall environmental impacts from AMRs management can be mitigated.

1. Introduction

Antibiotic drugs have been widely produced and used due to their important functions on medical interventions and treatment [1,2]. The global human consumption of antibiotics had increased by 36% during 2000–2010 [2]. Specifically, China has been the world's largest raw materials producer and exporter for producing antibiotics since 2009 [3]. China produced 0.12 million ton of antibiotics in 2013, of which 0.03 million ton was exported, accounting for 70% of the global market [4]. Such a large amount of production also led to a large volume of antibiotic mycelial residues (AMRs). AMRs have been considered as hazardous wastes in China since 2008 because they contain several toxic substances (for example, antibiotics residues, heavy metals, and metabolic intermediates) [5–8]. Many studies uncovered that antibiotic residues exposed to the natural environment can lead to antibiotic-resistant bacteria [9,10]. The World Health Organization (WHO) reported that antibiotic-resistant bacteria have already emerged and spread among human beings and animals [11]. Under such a circumstance, it is critical to safely manage AMRs.

AMRs generated from bio-fermentation processes for antibiotics production could be considered as a significant source of biomass resources [12]. In the past few years, the AMRs were used as food additives for the livestock or as a type of fertilizer for agricultural purposes [13,14]. But these measures have been forbidden since 2008, when AMRs were listed in China's National Catalogue of Hazardous Wastes. Currently, the most commonly used disposal methods are landfilling and incineration. Unfortunately, these two disposal methods are not being widely accepted due to their high costs and serious secondary environmental problems [13]. Resource recovery efforts have been made to respond such issues and also address energy shortage in recent years [15,16]. Several technologies have been developed and applied since AMRs are significant sources of biomass. These technologies can help improve biogas production via exploring optimum conditions for the anaerobic digestion (AD) of AMRs, including hydrothermal pretreatment [5,12], alkaline pretreatment [17,18], the combination of hydrothermal and alkaline [19], and the potential biogas production under the co-digestion of AMRs with food wastes [20,21].

LCA is an efficient method for quantifying the environmental impacts of materials and energy flows in processes' life cycles [22].

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As one of the best environmental management tools [23,24], LCA has been considered an effective method for evaluating the environmental impacts related with alternative solid waste management options [25,26] and has been widely employed in the United Kingdom [27,28], Spain [29], Italy [30,31], Japan [32], Iran [33], Australia [34], the United States [35], Canada [36], and Brazil [37]. Especially, LCA studies on various wastes have been conducted in China, such as municipal solid wastes (MSW) [38,39], food wastes [40], electronic wastes [41], industrial hazardous wastes (IHW) [42], and sewage sludge [43–45]. For example, one LCA study on IHW landfill and incineration was conducted to quantity environmental impacts generated from one IHW landfill and incineration site, identify key factors contributing to the environmental impacts, and provide suggestions for reducing related environmental impacts [42]. However, to the best of our knowledge, no specific LCA studies on AMRs management have been performed. Also, there are few studies on evaluating the environmental impacts generated from different AMRs management options. Under such a circumstance, it is critical to conduct life cycle assessment (LCA) on different AMRs management options so that a holistic picture of environmental performance generated from AMRs management can be obtained. In addition, uncertainty analysis is conducted so that results can be convincible and accurate for policy making. The whole paper is organized as below. After this introduction section, Section 2 details research methods and data sources. Section 3 shows research results and Section 4 discusses policy implications. Finally, Section 5 draws research conclusions.

2. Material and methods

2.1. Functional unit and system boundary of life cycle assessment

Defining a functional unit is central to LCA. A functional unit provides a quantified reference for both inputs and outputs of one system [46]. In this study, one ton of dry AMRs (DAMRs) was selected as the functional unit.

Four scenarios of AMRs management, namely, hazardous waste incineration (S-1), gasification (S-2), anaerobic digestion (AD) with energy recovery plus landfill (S-3), and AD with energy recovery plus incineration (S-4), were set up and investigated in this study. In all the four scenarios, the common processes include raw materials and energy production, road transport, and direct emissions (for example, particulates, nitrogen oxides, and mercury). Energy recovery of biogas from AMRs via AD process was considered in S-3 and S-4. For S-1 and S-2, fly ash was disposed of as a type of hazardous waste to landfill. For S-3, sanitary landfill for the final disposal of biogas residue was considered after the AD process. For S-4, incineration of biogas residue was considered after the AD process. In addition, the wastewater was directly reused after wastewater treatment process for S-1 and the AD process. Fig. 1 shows the system boundary of all the four scenarios.

2.2. Life cycle impact assessment

Life cycle impact assessment (LCIA) results were evaluated based on ReCiPe method [47,48], which is a commonly used framework for LCA. ReCiPe uses an impact mechanism with a global scope and considers a broad set of 18 midpoint impact categories. These midpoint impact categories include climate change, ozone depletion, terrestrial acidification, freshwater eutrophication, marine eutrophication, human toxicity, photochemical oxidant formation, particulate matter formation, terrestrial ecotoxicity, freshwater ecotoxicity, marine ecotoxicity, ionising radiation, agricultural land occupation, urban land occupation, natural land transformation, water depletion, metal depletion, and fossil depletion. Detailed information and complete characterization factors of the ReCiPe method are available on the website from the Institute of Environmental Science at Leiden University, the Netherlands [49]. Normalization was conducted for comparison among

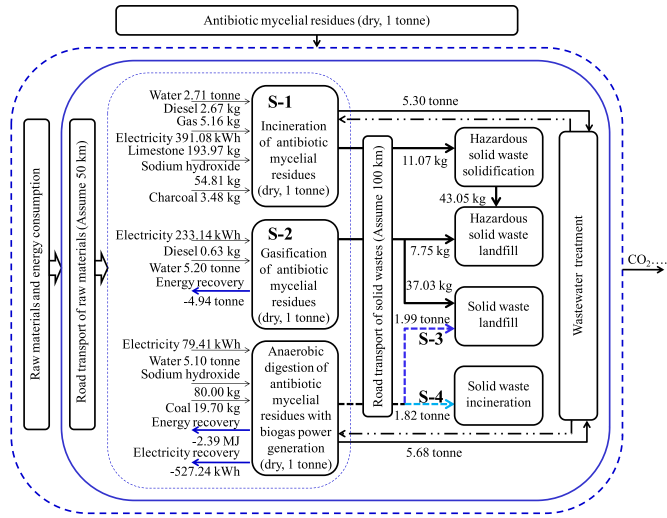

Fig. 1. System boundary.

the midpoint impact categories and to analyze the respective share of each impact category to the overall impact [50].

Based on Taylor series expansion, uncertainty analysis was performed [51]. Eq. (1) lists the detailed calculation process.

$$(\ln GSD_O^2)^2 = S_1^2(\ln GSD_1^2)^2 + S_2^2(\ln GSD_2^2)^2 + \ldots S_n^2(\ln GSD_n^2)^2 \tag{1}$$

where Si, GSD_i^2, and $GSDo^2$ are the model sensitivity for each input parameter (i), the geometric variation coefficient for each individual input, and the final geometric variation coefficient, respectively. GSD^2 is defined as the 2.5th and 97.5th percentiles, namely, with a 95% confidence interval of a probability distribution around the median μ.

The geometric standard deviation for the ratio of A/B ($GSD^2_{A/B}$) is calculated according to Eq. (2) so that scenario A and scenario B can be compared.

$$(\ln GSD_{A/B})^2 = \sum_{i}^{l} S_{Ai}^2(\ln GSD_{Axi})^2 + \sum_{j=l+1}^{m} S_{Bj}^2(\ln GSD_{Bxj})^2 + \sum_{k=m+1}^{n}(S_{Ak} - S_{Bk})^2(\ln GSD_{Xk})^2 \tag{2}$$

where S_{Ai}, S_{Bj}, GSD_{Ai}, and GSD_{Bj} represent the sensitivity and the geometric standard deviation of individual processes x_i and x_j for the scenario A and scenario B, respectively. Meanwhile, S_{Ak} and S_{Bk} are the sensitivity of the common parameters k for the scenario A and scenario B, respectively. GSD_{Xk} is the geometric standard deviation of common parameter k for both scenarios. Detailed information on applying the explicit analytical method can be obtained from Hong [52] and Hong et al. [53].

2.3. Data sources

Four AMRs management scenarios were proposed and investigated in this study. For S-1 and S-2, on-site data (for example, energy and materials consumption, direct emissions, and the amounts of wastewater and solid wastes) were obtained based on annual monitoring data. For S-3 and S-4, report data of North China Pharmaceutical Group Co., Ltd. [54] were used. The biogas yield and sodium hydrogen consumption during the AD process were taken from Tian [55], which was performed based on the AMRs obtained from North China Pharmaceutical Group Co., Ltd. In addition, energy consumption, energy recovery capacity, and direct emissions of AD process were obtained from Xu et al. [40]. Chinese data on solid waste landfill and incineration [38], coal-based electricity generation [56], road transport [57], and wastewater treatment [40] are used in this study. Relevant background data (e.g., infrastructure, hazardous waste solidification,

Table 1
Primary characterization factors of the AMRs considered in this study.

Characteristics	Unit	Value
Water content	%	65.48
Calorific value (Qb, ar)	MJ/kg	5.62
C (Car)	%	16.96
N (Nar)	%	2.07
H (Oar)	%	1.10
O (Oar)	%	9.04
Volatile Solid/Total Solid (VS/TS)	%	94.78

hazardous waste landfill, and diesel production) from Europe [58] are used because of the lack of such information in China. Table 1 presents the characteristics of the AMRs used in this study.

2.4. Life cycle inventory

Table 2 provides the life cycle inventory (LCI) results of the aforementioned four scenarios. All energy and materials consumption, direct emissions, waste production, and energy recovery were based on the same functional unit. CO_2 emissions from the incineration or utilization of AMRs were omitted because AMRs are considered a type of biogenic source.

3. Results

3.1. LCIA results

Table 3 shows the LCIA midpoint results based on ReCiPe method for all the four scenarios. S-1 has the highest impact on most impact categories, except freshwater eutrophication, terrestrial ecotoxicity, and water depletion. S-3 has the highest impact on freshwater eutrophication, while S-4 has the highest impact on terrestrial ecotoxicity and water depletion. Compared with S-4, S-3 has lower values in all impact categories except freshwater eutrophication. Meanwhile, S-2 has the lowest environmental impact on all impact categories except terrestrial acidification, photochemical oxidant formation, and parti-

Table 2
Life cycle inventory results of each scenario. Values are presented per functional unit.

		Unit	S-1 (Hazardous waste incineration)	S-2 (Gasification)	S-3 (AD+Landfill)	S-4 (AD+Incineration)
Resources consumption	Water	ton	2.71	5.20	5.10	5.10
	Electricity consumption	kWh	391.08	233.14	79.41	79.41
	Gas	kg	5.16	–	–	–
	Diesel	kg	2.67	0.63	–	–
	Charcoal	kg	3.48	–	–	–
	Limestone	kg	193.97	–	125.36	–
	Sodium hydroxide	kg	54.81	–	80.00	80.00
	Electricity recovery	kWh	–	–	−527.24	−527.24
	Steam recovery	MJ	–	-1.72×10^4	−2.39	−2.39
	Coal	kg	–	–	19.70	19.70
Emissions to air	Mercury	g	0.41	3.59×10^{-3}	–	–
	Cadmium	g	–	2.90×10^{-2}	–	–
	Thallium	g	–	0.29	–	–
	Antimony	g	8.31×10^{-3}	0.29	–	–
	Arsenic	g	3.24	1.74×10^{-4}	–	–
	Lead	g	8.72	0.14	–	–
	Chromium	g	0.87	0.23	–	–
	Cobalt	g	–	0.14	–	–
	Copper	g	0.32	0.14	–	–
	Magnesium	g	0.41	0.23	–	–
	Nickel	g	0.64	0.17	–	–
	Tin	g	1.27	–	–	–
	Particulates	g	1.14×10^3	49.25	–	–
	Carbon monoxide	g	501.62	492.51	–	–
	Sulfur dioxide	g	5.24×10^3	78.80	–	–
	Nitrogen oxides	g	5.85×10^3	789.98	–	–
	Dioxins	g	3.97×10^{-6}	2.03×10^{-6}	–	–
	Hydrogen fluoride	g	1.74	–	–	–
	Hydrogen chloride	g	10.66	–	–	–
	Nitrogen	g	–	–	1.57×10^3	1.57×10^3
	Hydrogen	g	–		165.35	165.35
	Hydrogen sulfide	g	–	–	281.71	281.71
Emissions to soil	Chromium	mg	4.23	–	–	–
	Lead	mg	2.78	–	–	–
	Cadmium	mg	0.70	–	–	–
	Mercury	mg	5.79×10^{-2}	–	–	–
	Copper	mg	0.84	–	–	–
	Zinc	mg	1.71	–	–	–
	Barium	mg	31.89	–	–	–
	Nickel	mg	19.00	–	–	–
	Fluorine	mg	8.04×10^4	–	–	–
	Arsenic	mg	1.95×10^{-2}	–	–	–
Waste	Wastewater	ton	5.30	–	5.68	5.68
	Hazardous waste to solidification	kg	11.07		–	–
	Hazardous waste to landfill	kg	43.05	7.75	–	–
	Solid waste to landfill	kg	–	37.03	1.99×10^3	–
	Solid waste to incineration	ton	–	–	–	1.82

Table 3
LCIA midpoint results of each scenario. Values are presented for one functional unit.

Categories	Unit	Value			
		S-1	S-2	S-3	S-4
Climate change	kg CO_2 eq	558.74	−250.99	−232.16	−217.72
Ozone depletion	kg CFC−11 eq	7.90×10^{-6}	-2.76×10^{-6}	8.42×10^{-7}	2.21×10^{-6}
Terrestrial acidification	kg SO_2 eq	10.72	−0.67	−0.72	1.69
Freshwater eutrophication	kg P eq	2.78×10^{-3}	-1.27×10^{-3}	6.15×10^{-3}	5.66×10^{-3}
Marine eutrophication	kg N eq	0.31	-1.03×10^{-2}	0.10	0.15
Human toxicity	kg 1,4-DB eq	558.77	8.85	33.67	130.18
Photochemical oxidant formation	kg NMVOC	8.57	−0.30	−0.54	1.35
Particulate matter formation	kg PM_{10} eq	3.57	−0.19	−0.25	0.48
Terrestrial ecotoxicity	kg 1,4-DB eq	1.25	6.86×10^{-3}	1.72	1.77
Freshwater ecotoxicity	kg 1,4-DB eq	0.10	-1.72×10^{-2}	4.23×10^{-2}	5.32×10^{-2}
Marine ecotoxicity	kg 1,4-DB eq	0.99	4.04×10^{-2}	0.65	0.72
Ionising radiation	kBq U_{235} eq	4.97	−3.32	1.67	2.21
Agricultural land occupation	m^2a	16.56	−0.32	1.19	4.09
Urban land occupation	m^2a	3.92	−1.94	-4.43×10^{-2}	0.90
Natural land transformation	m^2	3.89×10^{-2}	-1.55×10^{-2}	2.96×10^{-3}	1.08×10^{-2}
Water depletion	m^3	6.13	3.73	4.93	7.61
Metal depletion	kg Fe eq	6.45	−4.01	0.83	1.90
Fossil depletion	kg oil eq	156.67	−922.78	−49.61	−9.53

culate matter formation, in which S-3 has the lowest values for these categories.

Fig. 2 depicts the normalized midpoint results for all the four scenarios. In order to better identify the key factors, the contributions of dominant processes to four scenarios are illustrated in Fig. 3. For S-1, the impact on human toxicity contributes the most to the overall environmental impact. Impacts in the forms of terrestrial acidification, particulate matter formation, terrestrial ecotoxicity, marine ecotoxicity, and fossil depletion categories are also important, whereas other impacts are negligible. These impacts mainly come from the direct emissions and the consumption of sodium hydrogen and electricity during hazardous waste incineration (shown in Fig. 3). Therefore, it is crucial to decrease direct air emissions, sodium hydrogen and electricity consumption.

For S-2, fossil depletion contributes the most to the overall environmental impact, while other impact categories are insignificant. Such a result is mainly due to the large amount of steam recovery generated from S-2 (shown in Fig. 3). Thus, it is important to improve energy recovery capacity.

For S-3 and S-4, impacts on human toxicity, terrestrial ecotoxicity, and marine ecotoxicity are the major parts of the total environmental impact, while the rest impact categories are insignificant. These impacts mainly come from sodium hydrogen consumption and energy recovery of biogas via AD process. In addition, the incineration of

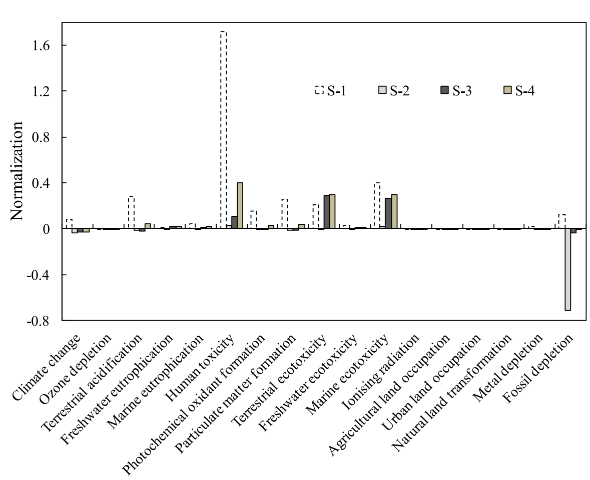

Fig. 2. Normalized results for all the four scenarios.

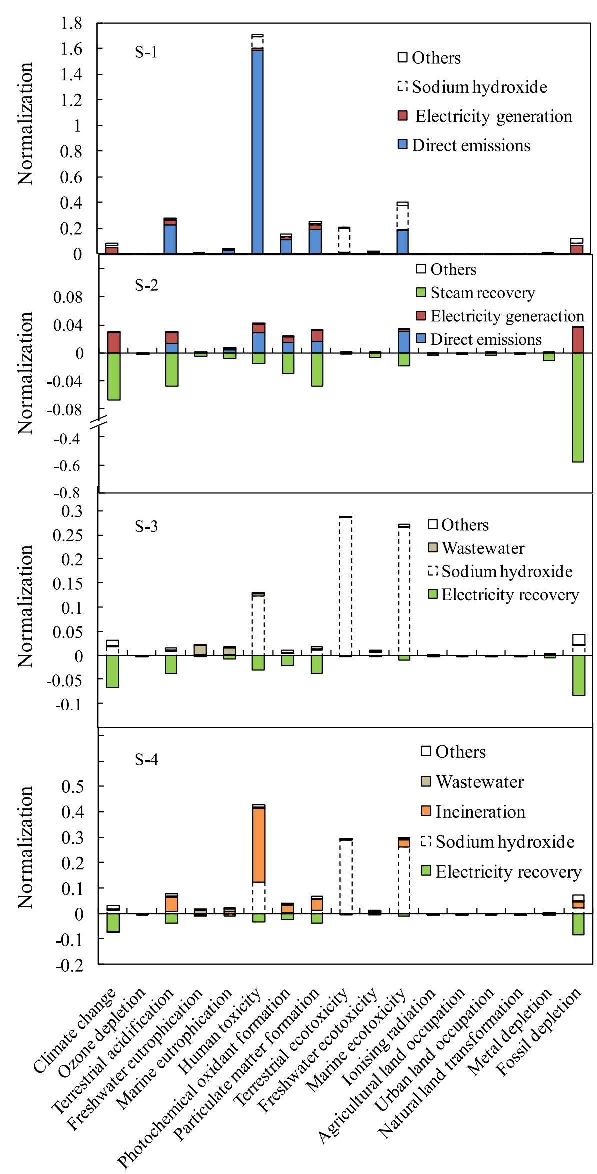

Fig. 3. Contributions of dominant processes.

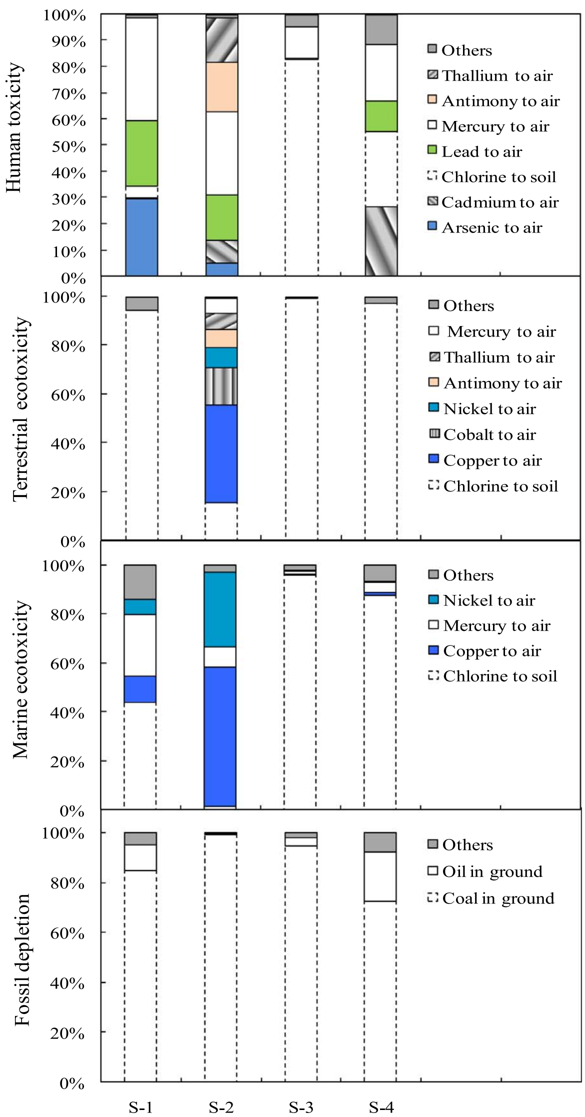

Fig. 4. Contributions of dominant substances to key impact categories a) human toxicity, b) terrestrial ecotoxicity, c) marine ecotoxicity, d) fossil depletion.

biogas residues provides additional contribution to the overall impact for S-4. Therefore, improving biogas production potential and decreasing sodium hydrogen consumption is helpful to reduce the environmental impacts of both S-3 and S-4.

In addition, although background data (for example, infrastructure, hazardous waste solidification, hazardous waste landfill, and diesel production) from Europe [58] were used in this study, impacts from these processes are insignificant, due to the minimal contributions of the aforementioned processes to the overall environmental impact under all the four scenarios.

Fig. 4 depicts the contributions of significant substances to the identified key impact categories (i.e., human toxicity, terrestrial ecotoxicity, marine ecotoxicity, and fossil depletion). For S-1 and S-2, the diffusion of mercury, lead, and arsenic to the air is the significant contributor to human toxicity. In addition, the diffusion of antimony, thallium, and cadmium into the air provides additional contributions for S-2.

For S-1, the dominant contributor to terrestrial ecotoxicity is the chloride discharged to the soil, while the chloride discharged to the soil and mercury diffused to the air contribute to marine ecotoxicity. For S-2, copper and cobalt diffused to the air, and chloride discharged to the soil are the significant contributions to terrestrial ecotoxicity. Similarly, copper and nickel diffused to the air are the dominant contributions to marine ecotoxicity.

For S-3 and S-4, chlorine discharged to the soil from sodium hydrogen production is the dominant contribution to human toxicity, terrestrial ecotoxicity, and marine ecotoxicity. Cadmium, mercury, and lead diffused to the air are mainly generated from incineration of biogas residue and provide additional contributions to human toxicity under S-4. For all the four scenarios, coal mainly used for energy generation has the dominant contribution to fossil depletion.

Table 4
Uncertainty analysis results for the four scenarios.

	$P_{(S\text{-}2\geq S\text{-}1)}$	$P_{(S\text{-}2\geq S\text{-}3)}$	$P_{(S\text{-}2\geq S\text{-}4)}$	$P_{(S\text{-}1\geq S\text{-}4)}$	$P_{(S\text{-}3\geq S\text{-}1)}$	$P_{(S\text{-}3\geq S\text{-}4)}$
Human toxicity	0%	0.2%	0%	98.9%	0%	0.4%
Marine ecotoxicity	0%	0%	0%	83.1%	9.3%	35.9%
Terrestrial ecotoxicity	0%	0%	0%	12.4%	84.1%	42.3%
Fossil depletion	0%	0%	0%	100%	0%	21.6%

3.2. Uncertainty analysis

LCI data were obtained from different AMRs management sites and may lead to significant disparity due to regional differences. In order to provide more accurate results, uncertainty analysis is performed. Table 4 shows that the probability of S-2≥S-1, S-2≥S-3, S-2≥S-4, S-1≥S-4, S-3≥S-1, and S-3≥S-4 in human toxicity category was 0%, 0.2%, 0%, 98.9%, 0%, and 0.4%, respectively. In other words, these results indicate that S-1 leads the highest environmental impact in the form of human toxicity, followed by S-4; S-2 has the lowest environmental impact. Simulation results for other impact categories are listed in Table 4.

By combining the uncertainty analysis results (shown in Table 4) with the normalization results (shown in Section 3.1), it is clear that S-2 leads to the lowest environmental impact among the four scenarios. This can be explained by the huge energy recovery capacity of S-2. S-1 leads to the highest environmental impact, followed by S-4, indicating that gasification is the best approach among the four AMRs management methods, while incineration of AMRs is the worst choice because of its higher environmental impacts.

3.3. Sensitivity analysis

3.3.1. Main contributors to environmental impacts

Sensitivity analysis is a systematic approach for estimating the effects of selected data on the outcome of one study [46]. Fig. 5 presents the sensitivity analysis results of the aforementioned dominant contributors to the associated impact categories. A 5% decrease of sodium hydrogen consumption under S-1 could lead to approximately 0.25%, 4.73%, 2.32%, and 0.66% environmental benefit to human toxicity, terrestrial ecotoxicity, marine ecotoxicity, and fossil depletion, respectively, while an increase of 5% steam recovery under S-2 could result in 1.33%, 1.48%, 1.72%, and 4.76% reductions in human toxicity, terrestrial ecotoxicity, marine ecotoxicity, and fossil depletion, respectively. Similar simulation results for the other contributors and scenarios are shown in Fig. 5.

For S-1, the decrease of direct emissions provides the highest environmental benefit to human toxicity, while the decrease of sodium hydrogen consumption contributes the most to mitigating terrestrial ecotoxicity. Both direct emissions and sodium hydrogen consumption provide significant contributions to marine ecotoxicity.

For S-2, the decrease of direct emissions provides the highest environmental benefit to human toxicity, terrestrial ecotoxicity, and marine ecotoxicity, while the decrease of sodium hydrogen consumption provides the highest contribution to mitigating the overall impact for S-3.

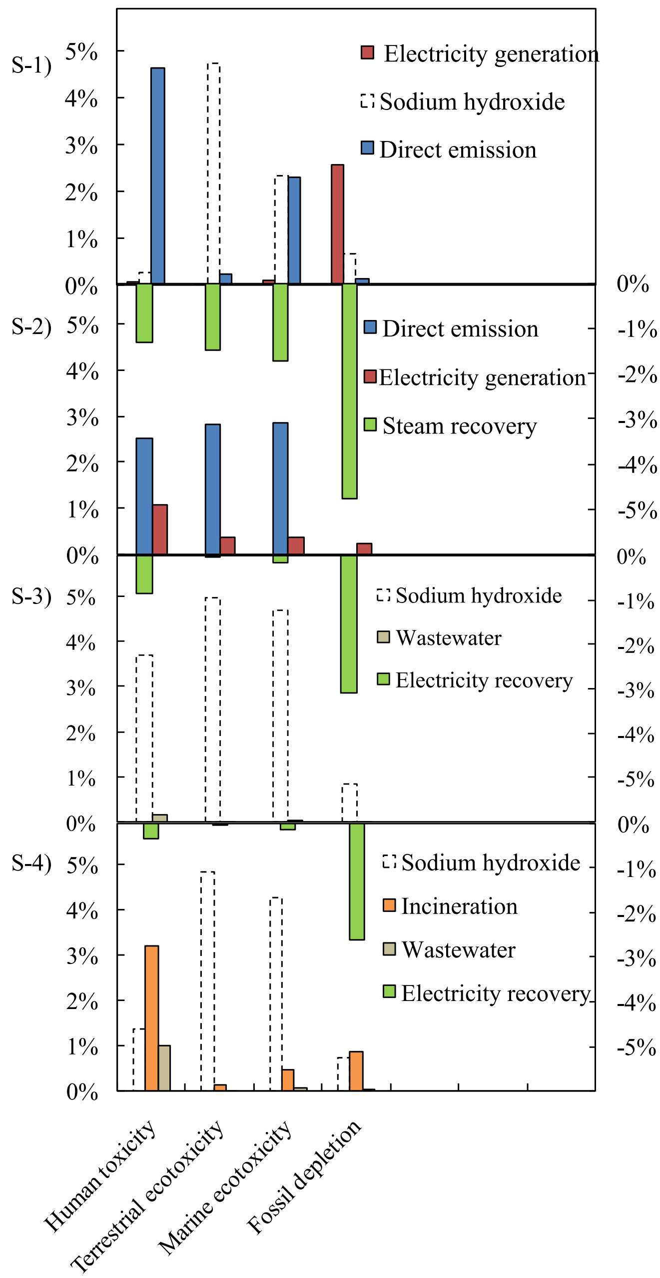

Fig. 5. Sensitivity analysis of dominant processes for four scenarios.

For S-4, incineration of biogas residue provides the highest contribution to mitigating human toxicity, while the reduction of sodium hydrogen consumption can mitigate most terrestrial ecotoxicity and marine ecotoxicity. In addition, decreasing the consumption of electricity can minimize the fossil depletion for S-1, while the increase of energy recovery can minimize the fossil depletion for the other scenarios.

Sensitivity analysis results reconfirm that decreasing direct air emissions, reducing the consumption of sodium hydrogen and electricity, improving energy recovery capacity are useful to reduce the environmental impacts of AMRs management.

3.3.2. Sensitivity of energy recovery

Energy recovery capacity from 1 t of AMRs may vary due to several factors, such as methane content of biogas, efficiency production efficiency, and the utilization efficiency of biogas. It is clear that increasing energy recovery capacity can lead to the largest environ-

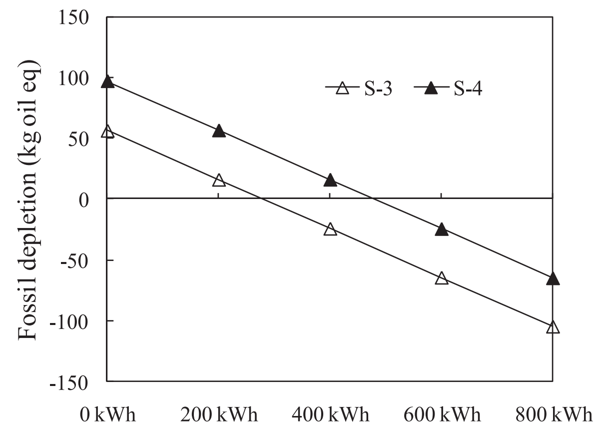

Fig. 6. Sensitivity analysis of energy recovery for fossil depletion category.

mental benefit to reduce fossil depletion for S-3 and S-4 (shown in 3.1). In order to further compare the results from these two scenarios, sensitivity analysis of electricity generated from AD based methane is performed for S-3 and S-4.

A linear relation between electricity recovery capacity and potential environmental burden in fossil depletion is shown in Fig. 6, in which a 100 kwh/t-DAMRs increase of electricity recovery could lead to the decrease of 1.76 kg 1,4-DB equivalent, 5.31×10^{-4} kg 1,4-DB equivalent, 3.97×10^{-3} kg 1,4-DB equivalent, and 20.16 kg oil equivalent in human toxicity, terrestrial ecotoxicity, marine ecotoxicity, and fossil depletion, respectively. If the electricity recovery capacity is increased to 600 kWh/t-DAMRs, fossil depletion will be reduced to −64.29 kg oil equivalent for S-3 and −24.20 kg oil equivalent for S-4. Similar results can also be found for the other impact categories.

4. Discussions

This study presents the LCIA results from four different AMRs management scenarios. It shows that environmental impacts under different AMRs management approaches are different. Uncertainty analysis was also performed in order to improve the accuracy of our results.

Among the four scenarios, S-2 leads to the lowest environmental impacts on all impact categories except terrestrial acidification, photochemical oxidant formation, and particulate matter formation, due to the huge amount of energy recovery generated from S-2 (shown in Fig. 3).

S-1 leads to the largest environmental impact on most impact categories except freshwater eutrophication, terrestrial ecotoxicity, and water depletion, due to direct pollutants emissions and the consumption of sodium hydrogen (shown in Fig. 3).

Both S-3 and S-4 lead to relatively high environmental impact in the forms of human toxicity, terrestrial ecotoxicity, and marine ecotoxicity because of the emission of chlorine to the soil from sodium hydrogen consumption process (shown in Fig. 3). Moreover, emissions of cadmium, mercury, and lead to the air from biogas residue incineration also contribute to human toxicity under the S-4 scenario. Therefore, resource utilization on AMRs is highly recommended instead of incineration, while gasification of AMRs is the best method compared with other three management methods due to its lowest environmental impact.

Specially, the impact on human toxicity contributes the most to the overall environmental impacts for S-1 (Section 3.1). This result is consistent with previous study [42], in which the environmental burden of IHW incineration was evaluated. In addition, the reduction of fossil depletion of incineration in this study (156.67 kg oil/t-incineration) is close to previous reported studies (54.1–249.9 kg oil/t-incineration) [42,58]. The environmental impact on global warming in this study (558.74 kg CO_2 eq) is lower than that of previous study (1.77–2.68 t CO_2 eq) [42,58]. This difference is caused by the fact that CO_2 emissions during incineration process is omitted in this study given that AMRs is a type of biogenic source (aforementioned in Section 2.4).

Since AMRs were listed in China's National Catalogue of Hazardous Wastes, the most popular management methods have changed to hazardous waste landfill or incineration. However, both methods are facing serious problems. For example, the limited disposal capacity of current hazardous landfills is far away from the soaring demands [59]. Also, there are more public concerns on dioxin released from incineration plants. With increasing environmental awareness, most residents do not want to expand the scales of existing hazardous wastes landfills and incinerators or construct the new ones [59]. Academically, Ai et al. [60] demonstrated that both incineration and landfill are not effective methods for AMRs. After the Chinese government issued the policies for accelerating the waste utilization [15,16], research and development (R & D) efforts have been widely supported for seeking innovative technologies on AMRs utilization. In this regard, several studies [3,61,62] on the characteristics of AMRs have been conducted, providing a solid foundation for the utilization of AMRs.

For instance, the AD technology has been proven to be an effective method for AMRs treatment [5,12,17,19]. However, the biogas yields of AMRs are influenced by various factors, such as temperature, pH, and types of substrate. Related studies are summarized in Table 5. The biogas yields normally range from 249.6 mL/g-VS to 446 mL/g-VS under different operation conditions (shown in Table 5). According to the reference [54], the biogas yield can achieve 450 mL/g-DAMRs by using actual operation data. The biogas yield in this study was 435 mL/g-DAMRs with a methane content of 65%, similar to other studies.

Co-digestion of lignocellulosic biomass with protein-rich wastes could mitigate the adverse impact generated from ammonia and volatile fatty acids during the AD process [63]. Several published studies also indicate that co-digestion of AMRs with other materials has higher methane production capacity than the single digestion [20,21]. For example, Gao found that co-digestion of food waste and penicillin bacterial residue (C/N=25:1) had a methane yield of 0.38 L/gVS d, presenting 11.76% increase compared with digesting penicillin bacterial residue alone [20]. Yin also found that the input of food waste could improve the methane yield of cephalosporin bacterial residue [21]. Therefore, it is critical to determine the appropriate feed stocks for co-digestion in order to improve methane yield.

According to Li et al. [64], the power generation efficiency of methane-based electricity (2.66 kWh/m^3) is much less than the design value (3.8–4.2 kWh/m^3). In this study, the energy recovery of AD-based methane is 1.86 kWh/m^3, showing at least a 40% improvement potential for energy recovery. If such a figure can increase to 200 kwh/t-DAMRs, then the environmental benefits of 3.53 kg 1,4-DB equivalent, 1.06×10^{-3} kg 1,4-DB equivalent, 7.94×10^{-3} kg 1,4-DB equivalent, and 40.33 kg oil equivalent to human toxicity, terrestrial ecotoxicity, marine ecotoxicity, and the reduction of fossil depletion will be obtained, respectively. Therefore, in order to reduce the overall environmental impacts of AD with energy recovery process, it is necessary to improve the relevant technology on enhancing methane yield and energy generation efficiency.

Incineration has been broadly applied for hazardous biomaterial treatment in recent years due to its significant contribution to minimize the volume and disinfect wastes [65]. However, the overall disposal capacity of special hazardous waste incinerators is still limited to meet the need of disposing of a huge volume of AMRs in China. Studies on co-combustion of AMRs with other materials have been explored. For example, Du et al. [66] investigated the combustion characteristics and kinetics of co-firing of bio-ferment residue with coal. Jiang et al. [67] investigated the co-combustion of AMRs in municipal solid waste incinerator. Internationally, co-incineration of hazardous waste has been permitted in the EU and the USA [67]. In the updated China's National Catalogue of Hazardous Wastes, fly ash generated from MSW incineration has been removed from the hazardous waste items when some requirements are met [7]. In the proposed standards on pollutants control for municipal solid waste incinerators, co-incineration of AMRs with municipal solid wastes was proposed in 2010 [68]. Although this proposal has not been approved, the co-incineration of AMRs in the high temperature stoves (e.g., municipal solid waste incinerator and cement-kiln incinerator) will be one solution for AMRs treatment with the improvement of related regulations.

Studies on the co-composting of AMRs with sewage also have been investigated [69–73]. However, both Chen et al. [70] and Yang et al. [72] found that co-composting of AMRs with sewage sludge could threaten the environment due to the antibiotic residues and antibiotic resistance genes contained in the composts. Although the composts from AMRs can meet the national standards in terms of nutrition properties, they are dangerous for agriculture application [72]. The China Environment News reported that it is impossible to compost all kinds of AMRs [74]. Consequently, it is not an ideal solution to compost AMRs for large-scale AMRs treatment.

In addition, the management of antibiotics-containing livestock wastes is facing several challenges in China. In the most Chinese rural areas, livestock wastes are directly discharged to the natural environment without any treatments [75,76]. Livestock wastes are significant sources of veterinary antibiotics because most antibiotics are present in urine and feces [1,75]. Moreover, antibiotics have been broadly reported in various environmental compartments, even drinking water system [75,76]. No specific regulations on the treatment of livestock wastes have been issued in China. Wastewater treatment infrastructure is also limited in the rural areas of China [76]. In order to address these concerns, the national government initiated the livestock waste-to-methane projects during the 12th Five-Year Plan (2011–2015). Such efforts are effective since many environmental emissions have been avoided [77]. Therefore, the National Development and Reform Commission (NDRC) of China decided to continue to implement such projects during the 13th Five-Year Plan (2016–2020) so that the resource utilization of livestock wastes can be achieved [78]. Also, Ministry of Agriculture of China released one regulation on promoting the agricultural supply chain reform, in which the avoidance and reduction of antibiotics is actively promoted [79]. In addition, ecological civilization has become one national development strategies since 2013, in which more efforts on improving rural environment will be initiated, such as the construction of more rural wastewater treatment plants and ecological wetlands, the stricter enforcement of related environmental laws and regulations in rural areas, more capacity-

Table 5
Biogas yields under different operation conditions.

Substrate type	Pretreatment	Biogas yield (mL/g-VS)	Methane yield (mL/g-VS)	Methane content (%)	Ref.
Cephalosporin bacterial residue	Hydrothermal (120 °C, pH=7.0, 60 min)	446	290	65.02	[5]
Cephalosporin bacterial residue	Thermal-alkaline (80 °C, pH=12, 60 min)	365	231	63.29	[49]
Penicillin bacterial residue	Thermal-alkaline (70 °C, pH=13, 30 min)	267.5	201.2	75.21	[50]
Penicillin bacterial residue	Thermal-alkaline (100 °C, pH=13, 60 min)	249.6	193.6	77.56	[50]
Cephalosporin bacterial residue	Hydrothermal (120 °C, pH=6.8–7.2, 20 min)	328	227	69.21	[12]

building efforts on improving rural residents' environmental awareness and more research funds to support the application of advanced agricultural technologies (such as organic farms) [80]. With the implementation of these measures, it is expected that the overall environmental impacts from AMRs will be mitigated.

5. Conclusions

China's large production and consumption of AMRs has brought several environmental challenges. In order to solve these challenges, it is critical to identify the most feasible solution so that the overall impact can be reduced. Under such a circumstance, this study aims to quantify the various impacts, identify the key factors, and select the best solution for AMRs management. Four scenarios were set up and assessed by using life cycle analysis. Also, uncertainty analysis was conducted to further validate the research results so that more accurate findings are available for policy making. Results show that the major environmental impacts of AMRs disposal include human toxicity, terrestrial ecotoxicity, marine ecotoxicity, and fossil depletion. From factor identification perspective, direct emissions are the key contributors to the overall environmental burden generated from S-1, while energy recovery is the key contributor for the scenario of S-2. Sodium hydrogen consumption is the major contributor for both scenarios S-3 and S-4. Based upon these findings, several policy recommendations are proposed, including the reduction of direct air pollutants emissions, improving technologies to enhance methane yield and energy recovery efficiency, and decreasing sodium hydrogen consumption. These recommendations should be implemented with the appropriate regulations. Consequently, policy-makers should prepare their own regulations by considering their own realities so that the overall impact from results obtained from this study will provide scientific information on environmental impacts generated from AMRs can be mitigated. During this process, all the stakeholders should work together so that disputes can be solved quickly and experiences and expertise can be shared.

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References

[1] Yan C, Yang Y, Zhou J, Liu M, Nie M, Shi H, et al. Antibiotics in the surface water of the Yangtze Estuary: occurrence, distribution and risk assessment. Environ Pollut 2013;175:22–9.
[2] Van Boeckel T, Gandra S, Ashok A, Caudron Q, Grenfell B, Levin S, et al. Global antibiotic consumption 2000 to 2010: an analysis of national pharmaceutical sales data. Lancet Infect Dis 2014;14:742–50.
[3] Guo B, Gong L, Duan E, Liu R, Ren A, Han J, et al. Characteristics of penicillin bacterial residue. J Air Waste Manag Assoc 2012;62:485–8.
[4] CNPIC China. National Pharmaceutical Industry Corporation; 2014Accessed via: ⟨http://www.cnpic.com.cn/353-867-11258.aspx⟩ [Accessed 18 March 2017].
[5] Li C, Zhang G, Zhang Z, Ma D, Wang L, Xu G. Hydrothermal pretreatment for biogas production from anaerobic digestion of antibiotic mycelial residue. Chem Eng J 2015;279:530–7.
[6] MEP, Ministry of Environmental Protection, thePeople's Republic of China. National Catalogue of Hazardous Wastes, 2008. Accessed via: ⟨http://www.gov.cn/flfg/2008-06/17/content_1019136.htm⟩ [Accessed 18 March 2017].
[7] MEP, Ministry of Environmental Protection, thePeople's Republic of China. National Catalogue of Hazardous Wastes; 2016. Accessed via: ⟨http://www.mep.gov.cn/gkml/hbb/bl/201606/t20160621_354852.htm⟩ [Accessed 18 March 2017].
[8] Meng X, Miao Y, Zhu Y, Wang X. Study on the antibiotic bacterial residue for the human health risk assessment. Adv Mater Res 2013;788:476–9.
[9] Pruden A, Joakim Larsson D, Amézquita A, Collignon P, Brandt K, Graham D, et al. Management options for reducing the release of antibiotics and antibiotic resistance genes to the environment. Environ Health Perspect 2013;121:878–85.
[10] Xu W, Zhang G, Li X, Zou S, Li P, Hu Z, et al. Occurrence and elimination of antibiotics at four sewage treatment plants in the Pearl River Delta (PRD), South China. Water Res 2007;41:4526–34.
[11] WHO, World Health Organization. Antimicrobial resistance: Global report on surveillance; 2014. Accessed via: ⟨http://www.who.int/drugresistance/documents/surveillancereport/en/⟩ [Accessed 18 March 2017].
[12] Zhang G, Li C, Ma D, Zhang Z, Xu G. Anaerobic digestion of antibiotic residue in combination with hydrothermal pretreatment for biogas. Bioresour Technol 2015;192:257–65.
[13] Zhong W, Li Z, Yang J, Liu C, Tian B, Wang Y, et al. Effect of thermal–alkaline pretreatment on the anaerobic digestion of streptomycin bacterial residues for methane production. Bioresour Technol 2014;151:436–40.
[14] Zhu X, Yang S, Wang L, Liu Y, Qian F, Yao W, et al. Tracking the conversion of nitrogen during pyrolysis of antibiotic mycelial fermentation residues using XPS and TG-FTIR-MS technology. Environ Pollut 2016;211:20–7.
[15] MEP, Ministry of Environmental Protection, thePeople's Republic of China; 2012. Accessed via: ⟨http://www.zhb.gov.cn/gkml/hbb/gwy/201206/t20120619_231910.htm⟩ [Accessed 18 March 2017].
[16] MEP, Ministry of Environmental Protection, thePeople's Republic of China; 2012. Accessed via: ⟨http://kjs.mep.gov.cn/hjbhbz/bzwb/wrfzjszc/201203/t20120319_224790.htm⟩ [Accessed 18 March 2017].
[17] Zhong W, Li G, Gao Y, Li Z, Geng X, Li Y, et al. Enhanced biogas production from penicillin bacterial residue by thermal-alkaline pretreatment. Biotechnol Biotechnol Equip 2015;29:522–9.
[18] Li Z, Zuo J, Tian B, Yang J, Yu X, Chen P, et al. Thermal-alkaline pretreatment on the decomposition of the streptomycin bacterial residue. Biotechnol Biotechnol Equip 2012;26:2971–5.
[19] Li C, Zhang G, Zhang Z, Ma D, Xu G. Alkaline thermal pretreatment at mild temperatures for biogas production from anaerobic digestion of antibiotic mycelial residue. Bioresour Technol 2016;208:49–57.
[20] Gao Y. Harmless treatment of penicillin bacterial residue by anaerobic digestion [Dissertation for the Master Degree]. Hebei University of Science and Technology; 2015.
[21] Yin Q. Study on anaerobic digestion of cephalosporin bacterial residue [Dissertation for the Master Degree]. Hebei University of Science and Technology; 2014.
[22] ISO 14040, International Organization for Standardization. Environmental management-life cycle assessment-principles and frame work. London: British Standards Institution; 2006.
[23] Arena U, Mastellone M, Perugini F. The environmental performance of alternative solid waste management options: a life cycle assessment study. Chem Eng J 2003;96:207–22.
[24] Hou Q, Mao G, Zhao L, Du H, Zuo J. Mapping the scientific research on life cycle assessment: a bibliometric analysis. Int J Life Cycle Assess 2015;20:541–55.
[25] Liamsanguan C, Gheewala S. LCA: a decision support tool for environmental assessment of MSW management systems. J Environ Manag 2008;87:132–8.
[26] Cherubini F, Bargigli S, Ulgiati S. Life cycle assessment (LCA) of waste management strategies: landfilling, sorting plant and incineration. Energy 2009;34:2116–23.
[27] Evangelisti S, Lettieri P, Borello D, Clift R. Life cycle assessment of energy from waste via anaerobic digestion: a UK case study. Waste Manag 2014;34:226–37.
[28] Tagliaferri C, Evangelisti S, Clift R, Lettieri P, Chapman C, Taylor R. Life cycle assessment of conventional and advanced two-stage energy-from-waste technologies for methane production. J Clean Prod 2016;129:144–58.
[29] Bueno G, Latasa I, Lozano P. Comparative LCA of two approaches with different emphasis on energy or material recovery for a municipal solid waste management system in Gipuzkoa. Renew Sustain Energy Rev 2015;51:449–59.
[30] Ripa M, Fiorentino G, Giani H, Clausen A, Ulgiati S. Refuse recovered biomass fuel from municipal solid waste. A life cycle assessment. Appl Energy 2017;186:211–25.
[31] Ripa M, Fiorentino G, Vacca V, Ulgiati S. The relevance of site-specific data in Life Cycle Assessment (LCA). The case of the municipal solid waste management in the metropolitan city of Naples (Italy). J Clean Prod 2017;142:445–60.
[32] Hong J, Hong J, Otaki M, Jolliet O. Environmental and economic life cycle assessment for sewage sludge treatment processes in Japan. Waste Manag 2009;29:696–703.
[33] Rajaeifar M, Tabatabaei M, Ghanavati H, Khoshnevisan B, Rafiee S. Comparative life cycle assessment of different municipal solid waste management scenarios in Iran. Renew Sustain Energy Rev 2015;51:886–98.
[34] Lundie S, Peters G. Life cycle assessment of food waste management options. J Clean Prod 2005;13:275–86.
[35] Wang H, Wang L, Shahbazi A. Life cycle assessment of fast pyrolysis of municipal solid waste in North Carolina of USA. J Clean Prod 2015;87:511–9.
[36] Assamoi B, Lawryshyn Y. The environmental comparison of landfilling vs. incineration of MSW accounting for waste diversion. Waste Manag 2012;32:1019–30.
[37] Mendes M, Aramaki T, Hanaki K. Comparison of the environmental impact of incineration and landfilling in São Paulo City as determined by LCA. Resour Conserv Recycl 2004;41:47–63.
[38] Hong J, Li X, Cui Z. Life cycle assessment of four municipal solid waste management scenarios in China. Waste Manag 2010;30:2362–9.

[39] Hong J, Chen Y, Wang M, Ye L, Qi C, Yuan H, et al. Intensification of municipal solid waste disposal in China. Renew Sustain Energy Rev 2017;69:168–76.
[40] Xu C, Shi W, Hong J, Zhang F, Chen W. Life cycle assessment of food waste-based biogas generation. Renew Sustain Energy Rev 2015;49:169–77.
[41] Hong J, Shi W, Wang Y, Chen W, Li X. Life cycle assessment of electronic waste treatment. Waste Manag 2015;38:357–65.
[42] Hong J, Han X, Chen Y, Wang M, Ye L, Qi C, et al. Life cycle environmental assessment of industrial hazardous waste incineration and landfilling in China. Int J Life Cycle Assess 2016. http://dx.doi.org/10.1007/s11367-016-1228-0, [Accessed 18 March 2017].
[43] Hong J, Xu C, Hong J, Tan X, Chen W. Life cycle assessment of sewage sludge co-incineration in a coal-based power station. Waste Manag 2013;33:1843–52.
[44] Hong J, Li X. Environmental assessment of sewage sludge as secondary raw material in cement production–a case study in China. Waste Manag 2011;31:1364–71.
[45] Xu C, Chen W, Hong J. Life-cycle environmental and economic assessment of sewage sludge treatment in China. J Clean Prod 2014;67:79–87.
[46] ISO 14044, International Organization for Standardization. Environmental management-Life cycle assessment-Requirements and guidelines. London: British Standards Institution; 2006.
[47] De Schryver A, Brakkee K, Goedkoop M, Huijbregts M. Characterization factors for global warming in life cycle assessment based on damages to humans and ecosystems. Environ Sci Technol 2009;43:1689–95.
[48] Goedkoop M, Heijungs R, Huijbregts M, De Schryver A, Struijs J, Van Zelm R Recipe is a life cycle impact assessment method which comprises harmonized category indicators at the mid-point and the end-point level. Report I: Characterisation. Accessed via: ⟨http://www.lcia-recipe.net⟩ [Accessed 18 March 2017].
[49] ReCiPe. Institute of Environmental Sciences. Accessed via: ⟨http://www.cml.leiden.edu/research/industrialecology/researchprojects/finished/recipe.html⟩ [Accessed 18 March 2017].
[50] Sleeswijk A, van Oers L, Guinée J, Struijs J, Huijbregts M. Normalisation in product life cycle assessment: an LCA of the global and European economic systems in the year 2000. Sci Total Environ 2008;390:227–40.
[51] Hong J, Shaked S, Rosenbaum R, Jolliet O. Analytical uncertainty propagation in life cycle inventory and impact assessment: application to an automobile front panel. Int J Life Cycle Assess 2010;15:499–510.
[52] Hong J. Uncertainty propagation in life cycle assessment of biodiesel versus diesel-global warming and non-renewable energy. Bioresour Technol 2012;113:3–7.
[53] Hong J, Zhang Y, Xu X, Li X. Life cycle assessment of corn-and cassava-based ethylene production. Biomass Bioenergy 2014;67:304–11.
[54] Environmental Protection Department of Hebei Province, the People's Republic of China, 2013. Accessed via: ⟨http://www.hb12369.net/qyfw/zljsfb/gfzljs/201310/t20131010_39134.html⟩ [Accessed 18 March 2017].
[55] Tian B. Study on anaerobic digestion treatment technology of streptomycin bacterial residue [Dissertation for the Master Degree]. Hebei University of Science andTechnology; 2012.
[56] Cui X, Hong J, Gao M. Environmental impact assessment of three coal-based electricity generation scenarios in China. Energy 2012;45:952–9.
[57] Chen W, Hong J, Xu C. Pollutants generated by cement production in China, their impacts, and the potential for environmental improvement. J Clean Prod 2015;103:61–9.
[58] Ecoinvent centre. Swiss centre for life cycle inventories. Accessed via: ⟨http://www.ecoinvent.org/database/⟩ [Accessed 18 March 2017].
[59] Wang Q, Huang Q, Yan D, Li L. Current status and suggestions on hazardous waste management in China. J Environ Eng Technol 2013;3:1–5.
[60] Ai H, Shi P, Wang H, Sun D. Analysis and evaluation of treatment and disposal technology of antibiotics bacterial residue. Chin J Environ Eng 2016;10:906–14.
[61] Zhou B, Gao Q, Wang H, Li Z, Wang D. Research and utilization of penicillin bacterial residue and oxytetracycline bacterial residue and analysis of the characteristics. Hebei J Ind Sci Technol 2011;28:291–4.
[62] Yuan L, Wang M, Wang P, Liu H. Physical and chemical properties of cephalosporin bacterial residue. J Harbin Univ Commer (Nat Sci Ed) 2015;31(691–693):703.
[63] Yang L, Xu F, Ge X, Li Y. Challenges and strategies for solid-state anaerobic digestion of lignocellulosic biomass. Renew Sustain Energy Rev 2015;44:824–34.
[64] Li J, Ying Z, Wu F, Hong D. Electricity production and analysis of influence factors of domestic generator units. China Coalbed Methane 2011;8:39–41.
[65] Yang S, Zhu X, Wang J, Jin X, Liu Y, Qian F, et al. Combustion of hazardous biological waste derived from the fermentation of antibiotics using TG–FTIR and Py–GC/MS techniques. Bioresour Technol 2015;193:156–63.
[66] Du Y, Jiang X, Ma X, Liu X, Lv G, Jin Y, et al. Evaluation of cofiring bioferment residue with coal at different proportions: combustion characteristics and kinetics. Energy Fuels 2013;27:6295–303.
[67] Jiang X, Feng Y, Lv G, Du Y, Qin D, Li X, et al. Bioferment residue: TG-FTIR study and cocombustion in a MSW incineration plant. Environ Sci Technol 2012;46:13539–44.
[68] MEP, Ministry of Environmental Protection, thePeople's Republic of China. Standard for pollution control on the municipal solid waste incineration: Public consultation draft; 2010. Accessed via: ⟨http://www.zhb.gov.cn/gkml/hbb/bgth/201011/t20101125_197974.htm⟩ [Accessed 18 March 2017].
[69] Chen Z, Zhang S, Wen Q, Zheng J. Effect of aeration rate on composting of penicillin mycelial dreg. J Environ Sci 2015;37:172–8.
[70] Chen Z, Wang Y, Wen Q, Zhang S, Yang L. Feasibility study of recycling cephalosporin C fermentation dregs using co-composting process with activated sludge as co-substrate. Environ Technol 2016;37:2222–30.
[71] Zhang S, Chen Z, Wen Q, Yang L, Wang W, Zheng J. Effectiveness of bulking agents for co-composting penicillin mycelial dreg (PMD) and sewage sludge in pilot-scale system. Environ Sci Pollut Res 2016;23:1362–70.
[72] Yang L, Zhang S, Chen Z, Wen Q, Wang Y. Maturity and security assessment of pilot-scale aerobic co-composting of penicillin fermentation dregs (PFDs) with sewage sludge. Bioresour Technol 2016;204:185–91.
[73] Gao J, Huang J, Zhang L. Cooperative fermentation for aerobic co-composting of edible fungi residue and sewage sludge. China Water Wastewater 2016;32:127–30.
[74] China Environment News; 2016. Accessed via: ⟨http://news.cenews.com.cn/html/2016-12/13/node_7.htm⟩ [Accessed 18 March 2017].
[75] Zhou L, Ying G, Zhao J, Yang J, Wang L, Yang B, et al. Trends in the occurrence of human and veterinary antibiotics in the sediments of the Yellow River, Hai River and Liao River in northern China. Environ Pollut 2011;159:1877–85.
[76] Zhang Q, Ying G, Pan C, Liu Y, Zhao J. Comprehensive evaluation of antibiotics emission and fate in the river basins of China: source analysis, multimedia modeling, and linkage to bacterial resistance. Environ Sci Technol 2015;49:6772–82.
[77] NDRC, National Development and Reform Commission, the People's Republic of China; 2016. Accessed via: ⟨http://zys.ndrc.gov.cn/xwfb/201602/t20160219_774944.html⟩ [Accessed 18 March 2017].
[78] NDRC, National Development and Reform Commission, the People's Republic of China; 2017. Accessed via: ⟨http://njs.ndrc.gov.cn/gzdt/201702/t20170210_837550.html⟩ [Accessed 18 March 2017].
[79] Ministry of Agriculture, NDRC, the People's Republic of China; 2017. Accessed via: ⟨http://www.moa.gov.cn/govpublic/BGT/201702/t20170206_5468139.htm⟩ [Accessed 18 March 2017].
[80] The State Council, the People's Republic of China; 2016. Accessed via: ⟨http://www.gov.cn/zhengce/content/2016-12/05/content_5143290.htm⟩ [Accessed 18 March 2017].