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Development, current state and future trends of sludge management in China: Based on exploratory data and CO_2-equivaient emissions analysis

Liangliang Wei[a,*], Fengyi Zhu[a], Qiaoyang Li[a], Chonghua Xue[b], Xinhui Xia[a], Hang Yu[a], Qingliang Zhao[a], Junqiu Jiang[a], Shunwen Bai[a,*]

[a] State Key Laboratory of Urban Water Resources and Environment (SKLUWRE); School of Environment, Harbin Institute of Technology, Harbin 150090, China
[b] Center of Science & Technology of Construction of the Ministry of Housing and Urban Rural Development, China

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ABSTRACT

This study statistically reported the current state of sludge treatment/disposal in China from the aspects of sources, technical routes, geographical distribution, and development by using observational data after 1978. By the end of 2019, 5476 municipal wastewater treatment plants were operating in China, leading to an annual sludge productivity of 39.04 million tons (80% water content). Overall, 29.3% of the sludge in China was disposed via land application, followed by incineration (26.7%) and sanitary landfills (20.1%). Incineration, compost, thermal hydrolysis and anerobic digestion were the mainstream technologies for sludge treatment in China, with capacities of 27,122, 11,250, 8342 and 6944 t/d in 2019, respectively. Incineration and drying were preferentially constructed in East China. In contrast, sludge compost was most frequently used in Northeast China (46.5%), East China (22.4%) and Central China (12.8%), while anaerobic digestion in East China, North China and Central China. The capacities of sludge facilities exhibited a sharp increase in 2009–2019, with an overall greenhouse gas emissions in China in 2019 reached 108.18×10^8 kg CO_2-equivaient emissions, and the four main technical routes contributed as: incineration (45.11%) > sanitary landfills (23.04%) > land utilization (17.64%) > building materials (14.21%). Challenges and existing problems of sludge disposal in China, including high CO_2 emissions, unbalanced regional development, low stabilization and land utilization levels, were discussed. Finally, suggestions regarding potential technical and administrative measures in China, and sustainable sludge management for developing countries, were also given.

1. Introduction

Waste activated sludge (WAS) is the solid waste produced during the biological treatment of wastewater (Wei et al., 2020), especially from municipal wastewater treatment plants (WWTPs). In China, the continuous construction of sewage drainage systems and wastewater treatment facilities has resulted in an increase in the treatment of domestic wastewater and thus a dramatic growth in WAS production (Sun et al., 2016; Wang et al., 2018). According to the publication of Zhang et al. (2016), more than six million tons of WAS (dry basis) was generated in China. Theoretically, as much as 85–95% of the heavy metals, phosphorus, pathogenic microorganisms, 40–50% of the organic components, and 20–30% of total nitrogen within the influent of WWTPs could finally emerge as WAS after biological treatment (Tiravanti et al., 1997).

Large amounts of wastewater-borne pollutants, especially pathogens (Qin et al., 2020), heavy metals (Wei et al., 2019; Zheng et al., 2020) and other viruses (Lizasoain et al., 2018), have been frequently detected in WAS; finally the inappropriate handling of these sludges severely threatens human health and causes serious environmental issues (Fijalkowski et al., 2017; Wei et al., 2019). However, WAS also contains a high content of valuable nutrients, such as nitrogen, phosphorus, and organics, which can be disposed of as fertilizer and biogas after undergoing the appropriate treatments (Wainaina et al., 2020). Thus, the comprehensive disposal of WAS is an important aspect of WWTPs operation and municipal waste management (Lu et al., 2019), and attracted increasing interest from researchers worldwide.

Currently, incineration, land application, sanitary landfills, and building materials production are the most widely used technical routes of sludge processing in China (Wang et al., 2010; Chen et al., 2012, 2019; Lu et al., 2019). Meanwhile, treatment approaches, such as thickening, conditioning, dewatering, digestion, composting, and drying, are widely used for sludge volume reduction, nutrient recycling and energy recovery purposes (Yang et al., 2015). China has a vast

* Corresponding authors.
E-mail address: weill333@163.com (L. Wei).

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territory, and the sludge treatment and disposal techniques thus have been adopted varied widely among different regions. For instance, the recent publication of Zhang et al. (2016) reported that composting is the predominant technology applied in South China, while incineration for the East China. In contrast, landfilling is the prominently adopted approach in the remaining regions (Lu et al., 2019). In past decades, Chinese government has enforced stricter laws and management in wastewater and has made remarkable achievements (Li et al., 2012; Qu et al., 2019). However the recent publication of Lu et al. (2019) still recognized sludge disposal as an important remaining gap for WWTPs operational optimization.

Besides pathogen inactivation, odor control, and toxic compounds removal (Kelessidis and Stasinakis, 2012), the proper sludge treatment/disposal could also achieve a significant decrease in greenhouse gas (GHG) emissions. As highlighted by Peters and Rowley (2009), the technical lever of a specific region significantly affects GHG emissions. For example, the incineration with mechanical drying of Kempele, a WWTP in Finland, generated 2307 tons CO_2-equivaient (CO_2-eq) GHG emissions (Piippo et al., 2018). The GHG emissions results of 128 WWTPs in Greece revealed that the emitted CH_4 was approximately 9668 kg CH_4 per day, and the predominant source was related to landfills disposal of sludge (7973 kg CH_4 per day) (Koutsou et al., 2018). Although many works have previously focused on how to improve sludge disposal efficiency in China (Yang et al., 2015; Xiao et al., 2018; Lu et al., 2019), to the best of our knowledge, there is still needing a clear view of the development, current state and trend prediction of sludge treatment and disposal in China based on regional discrepancies, historical management data and CO_2-eq emissions analysis.

Here, changes in China's municipal sludge treatment and disposal techniques and their contributions were studied using data collected after 1978, especially in the period of 2009–2019. Spatial and temporal distributions of the aerobic composting, anaerobic sludge digestion, incineration, deep dewatering, drying and thermal hydrolysis projects were emphatically analyzed. Then, regional contributions of sludge treatment/disposal to GHG emissions were discussed and compared, and the potential impact were evaluated. Finally, future perspectives of sludge disposal in China and suggestions on sustainable sludge management for developing countries were deduced.

2. Data sources of sludge management

2.1. Data sources

The data on the WWTPs distribution and sludge production of different provinces were obtained from statistical data and official websites of the Ministry of Housing and Urban-Rural Development (MOHURD), and most were from the yearbook (MOHURD, 2019). Constructed project information about anaerobic digestion, aerobic composting, incineration, thermal hydrolysis, drying, deep dewatering, and projects was obtained from the statistical data of the Urban Water Management Office of MOHURD and gathered data.

2.2. Calculation of CO_2-eq emissions

Direct and indirect GHG emissions (including CO_2, CH_4 and N_2O), caused by sludge disposal, were calculated and finally converted into carbon dioxide equivalents. According to Chai et al. (2015), the global warming potential (GWP) was 1 for CO_2, 25 for CH_4, and 298 for N_2O. GWPs calculation was divided into five different technical routes, including anaerobic digestion + land application (Route 1), aerobic compost + land application (Route 2), sanitary landfills (Route 3), sludge incineration (Route 4) and building materials production (Route 5). According to research reported by Murray et al. (2008), Liu et al. (2013) and Niu et al. (2013), the total CO_2-eq of the abovementioned five technical routes could be calculated as:

Route 1: $$TotalCO_{2-eq} = CO_{2-eq\ of\ CH_4\ Emission} + CO_{2Emission} - CO_{2-avoided} \quad (1)$$

Route 2: $$TotalCO_{2-eq} = CO_{2-eq1} + CO_{2-electricity} + CO_{2-coal} + CO_{2-avoided} \quad (2)$$

Route 3: $$TotalCO_{2-eq} = CO_{2-electricity} + CO_{2-diesel} + CO_{2-CH_4} + CO_{2-biogenic} \quad (3)$$

Route 4: $$TotalCO_{2-eq} = CO_{2-electricity} + CO_{2-coal} + CO_{2-CH_4,N_2O} \quad (4)$$

Route 5: $$TotalCO_{2-eq} = CO_{2-electricity} + CO_{2-coal} + CO_{2-CH_4,N_2O} \quad (5)$$

where $CO_{2\text{-}eq\ of\ CH4\ Emission}$ is the CH_4 generated during anaerobic digestion, as well as the CH_4 produced from the land application of organic residues (kg). $CO_{2\ Emission}$ includes the CO_2 produced from CH_4 combustion (defined as $CO_{2\text{-}CH4}$, in kg), biogenic CO_2 (defined as $CO_{2\text{-}biogenic}$, in kg) produced from organic residue land applications, landfills and digestion tanks. The avoided CO_2 (defined as $CO_{2\text{-}avoided}$, in kg) is attributed to biogas utilization, electricity generation, digested/composed sludge land utilization and heat conservation. $CO_{2\text{-}eq1}$ represents the equivalent CO_2 emitted during sludge aerobic compost, and $CO_{2\text{-}electricity}$ and $CO_{2\text{-}coal}$ refer to the equivalent CO_2 emissions of electricity and coal consumed during aerobic compost, respectively. $CO_{2\text{-}diesel}$ refers to the equivalent CO_2 emissions of electricity and diesel use from sludge landfills; $CO_{2\text{-}CH4}$ is the equivalent CO_2 emissions of CH_4 emissions from sludge landfills; $CO_{2\text{-}CH4,N2O}$ is the equivalent CO_2 emissions of CH_4 and N_2O emissions from sludge incineration and building material production.

Detailed information on the calculation of each equation and the explanation of the parameters can be found in the Supplementary information, and the calculated equivalent CO_2 ($CO_{2\text{-}eq}$) of Route 1, Route 2, Route 3, Route 4 and Route 5 is 960, 731, 1587, 2341 and 1240 kg $CO_{2\text{-}eq}$/t DM, respectively.

3. Development of municipal WWTPs and sludge production in China

3.1. Overall status of WWTPs and sludge production

Statistical data from MOHURD revealed that remarkable progress was made in China from 2009 to 2019, as proven by the significant increase in numbers of WWTPs, from 1199 to 5476 (Fig. 1 and Fig. S2). By the end of 2019, the overall treatment capacity of those WWTPs in China was 1.80×10^8 m^3/d, and the production was 39.04 million tons of 80% water content (W_c) sludge. The estimated production of WAS (tons, 80% W_c) in China in 1987 was approximately 2.47×10^5, and it

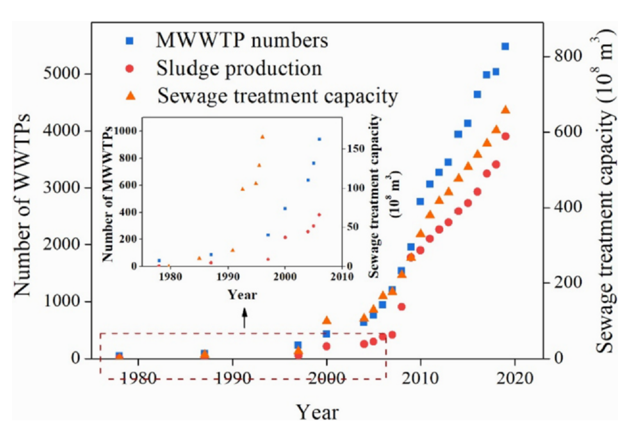

Fig. 1. WWTPs number, wastewater treatment capacity and sludge production in China during the period of 1978–2019 (Sludge production (10^4 tons, 80% W_c)).

Table 1
Production and growth rate of sludge in different provinces of China in 2009–2019.

Province	2009	2011	2013	2015	2017	2019	Annual averagegrowth rate
Shandong	1.99	2.12	2.36	2.89	3.44	3.86	6.88%
Zhejiang	1.61	2.21	2.45	2.67	3.57	3.58	8.36%
Guangdong	0.39	2.47	2.70	2.99	3.14	3.56	24.87%
Jiangsu	1.48	2.03	2.28	2.46	2.77	3.30	8.33%
Henan	0.95	1.27	1.53	1.68	1.99	2.38	9.68%
Hebei	0.77	1.35	1.67	1.76	1.95	2.03	10.14%
Beijing	1.02	1.12	1.17	1.06	1.15	1.77	5.66%
Sichuan	0.31	0.45	0.64	0.93	1.18	1.58	17.65%
Liaoning	0.65	0.84	0.87	1.04	1.10	1.47	8.44%
Shaanxi	0.47	0.43	0.59	0.96	1.10	1.44	11.79%
Shanghai	1.00	1.14	0.95	0.91	1.18	1.31	2.72%
Hubei	0.37	0.51	0.62	0.73	1.11	1.16	12.19%
Anhui	0.38	0.63	0.79	0.87	0.97	1.11	11.46%
Hunan	0.26	0.52	0.63	0.66	0.77	1.02	14.54%
Shanxi	0.23	0.37	0.46	0.55	0.74	0.99	15.57%
Chongqing	0.30	0.41	0.50	0.62	0.68	0.98	12.74%
Fujian	0.30	0.45	0.49	0.54	0.56	0.78	10.04%
Tianjin	0.29	0.32	0.29	0.37	0.55	0.76	10.15%
Inner Mongolia	0.16	0.21	0.34	0.46	0.60	0.75	16.53%
Jilin	0.15	0.24	0.31	0.33	0.40	0.74	17.37%
Heilongjiang	0.20	0.30	0.38	0.45	0.60	0.70	13.34%
Gansu	0.08	0.16	0.22	0.34	0.46	0.62	22.82%
Yunnan	0.25	0.33	0.38	0.48	0.54	0.57	8.73%
Xinjiang	0.15	0.22	0.28	0.31	0.48	0.56	14.09%
Guangxi	0.20	0.28	0.28	0.37	0.43	0.51	9.70%
Guizhou	0.05	0.15	0.22	0.28	0.27	0.44	23.11%
Jiangxi	0.10	0.25	0.28	0.30	0.34	0.42	16.01%
Ningxia	0.11	0.13	0.11	0.14	0.21	0.33	11.15%
Hainan	0.02	0.09	0.06	0.07	0.11	0.16	21.34%
Qinghai	0.03	0.05	0.04	0.08	0.10	0.14	14.93%
Tibet	0	0	0	0.01	0.01	0.02	–

[a] Sludge production (million tons).
[b] 80% W_c sludge.

gradually increased to 2.13×10^6 in 2000 and further to 2.96×10^6, 1.92×10^7, 2.14×10^7, 2.43×10^7, 2.73×10^7 and 3.25×10^7 tons in 2005, 2009, 2011, 2013, 2015 and 2017, respectively. Overall, the production of WAS in China exhibited a rapid expansion during 2005–2010, with an annual average growth rate of 44.9% (7.3% for 2010–2019).

3.2. Yearly production variation of different provinces

分省污泥总量

In 2019, Shandong Province had the highest WAS productivity of 3.86×10^6 tons (80% W_c) among all 31 provinces (Table 1), accounting for 9.86% of the overall sludge productivity in China. Zhejiang ranked second (contributed 9.19%), followed by Guangdong (9.08%), Jiangsu (8.43%), Henan (6.11%) and Hebei (5.22%). Correspondingly, the following 10 provinces/cities of Beijing, Sichuan, Liaoning, Shaanxi, Shanghai, Hubei, Anhui, Hunan, Shanxi and Chongqing accounted for 32.78% totally (individually accounting for 2.50–4.51%), while that of the other 15 provinces contributed only 19.33%. Over the period of 2009 to 2019, the highest annual average growth of sludge production was observed for Guangdong (24.87%), Guizhou (23.11%), Gansu (22.82%), Hainan (21.34%) and Sichuan (17.65%), clearly revealing an imbalanced regional development of WWTPs before 2009. Above results could be also proven by relatively lower annual average growth rate of sludge production of Beijing and Shanghai (< 6%), although they were among the top 10 sludge producers in China in 2019.

增长率

人均量

Per capita sludge production (in kg/y; 80% W_c) of China in 2019 was 28.05, as shown in Fig. 2A. Regarding the different provinces, Beijing, Tianjin and Zhejiang were ranked the top 3 among all provinces, with sludge production of 81.9, 70.4, 62.5 kg/y (80% W_c) per capita, respectively; closely related to the relatively high urbanization rates in those regions and the remarkable construction of public sewerage systems (Song et al., 2018). Per capita sludge production in provinces of Shanghai, Ningxia, Jiangsu, Shandong, Shaanxi, Liaoning, Chongqing, Guangdong, and Inner Mongolia, ranging from 29.44 to 53.98 kg/y, were higher than that in other part of China. Overall, the 15 northern provinces of China exhibited an 18.6% higher per capita sludge production (in kg/y) than the southern provinces (30.65 *vs* 25.84).

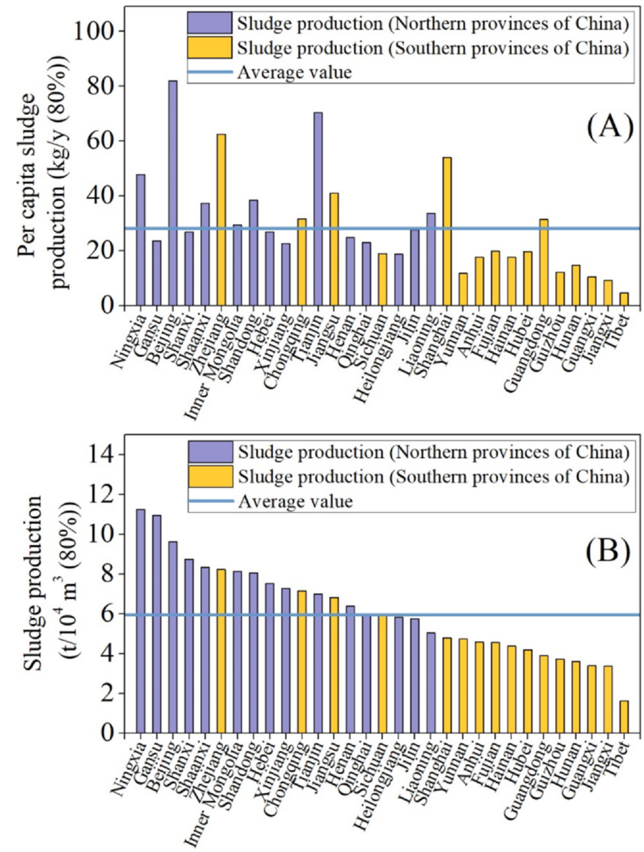

Fig. 2. Sludge production of per capita (A) and per 10,000 m^3 wastewater treated (B) of different provinces of China in 2019.

SP per WWT

Sludge production of per 10,000 m^3 wastewater treated of China in 2019 was 5.94 tons (80% W_c) (Fig. 2B), and the top 10 provinces in terms of this statistic declined as: Ningxia (11.24) > Gansu (10.94) > Beijing (9.62) > Shaanxi (8.74) > Zhejiang (8.22) > Inner Mongolia (8.13) > Shandong (8.05) > Hebei (7.52) > Xinjiang (7.27) > Chongqing (7.15). All of the above-mentioned provinces are distributed in northern China, except for Zhejiang and Chongqing. It is interesting to note that the sludge production among different provinces exhibited a distribution trend, in that those provinces in the northern regions of China (15 provinces) were much higher than those of the 16 provinces distributed in the southern part of China (7.36 *vs* 5.07). Generally, relatively lower pollutant loadings of inflows of WWTPs in southern provinces of China, related to the high groundwater level, long rainy season and high groundwater infiltration (Xie et al., 2012; Sun et al., 2016), might be the main reasons for this pattern.

北方高于南方

3.3. Effect of regional characteristics and development on sludge production

因素

Regional urbanization rate, gross domestic product (GDP) and population size significantly affected WAS production; this similar to the observation of Xu et al. (2019), who found a strong correlation among population, GDP, and domestic sludge production in urban agglomerations. Pearson correlation analysis were used to analyze the effects of per capita GDP and the urbanization rate on per capita sludge

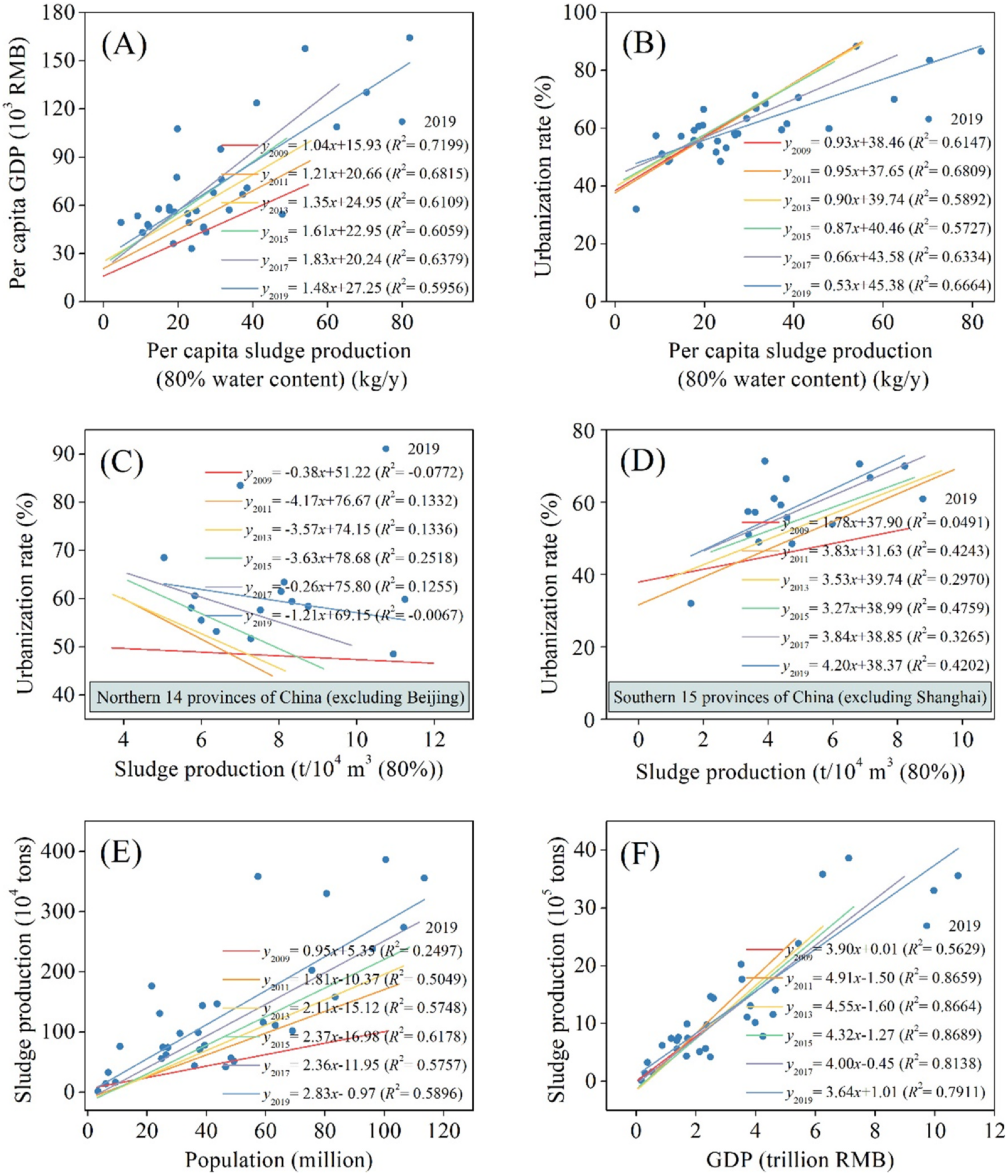

Fig. 3. Linear correlation analyzing between sludge production and per capita GDP (A, F), urbanization rate (B, C, D) and population (E) of different provinces in China from 2009 to 2019.

production (in supplementary materials of Table S3), the obtained correlation coefficients in 31 provinces are higher than 0.75 (with $p < 0.01$) and indicated a strong positive correlation (Fig. 3(A) and (B)). The higher urbanization rate and per capita GDP are, the higher per capita sludge production is. Moreover, it is interesting to note that a negative correlation between urbanization rate and sludge production (in tons/10^4 m^3 wastewater treated) was observed for the northern 14 provinces of China in 2019 (excluding Beijing city), while it was positive for the 15 southern provinces (excluding Shanghai). The main reason might be that the primary industry dominated urbanization in the northern regions of China (agglomerated population and industry), whereas the tertiary industry played a major role in the southern regions (rapidly developing with scattered individuals and industry) (Wang et al., 2016). This trend in the 15 southern provinces was increasingly apparent during the period from 2009 to 2019, as evidenced by Pearson correlation (Table S3). In addition, a stronger correlation between the regional GDP and sludge production to that of population size (0.893 *vs* 0.777) demonstrated that economic development and investment in wastewater facilities exhibited a more significant effect on sludge production in China. For example, the population of Jiangsu Province was almost the same as that of Sichuan (80.51 million *vs* 83.41 million), whereas the production of WAS was approximately 109% higher.

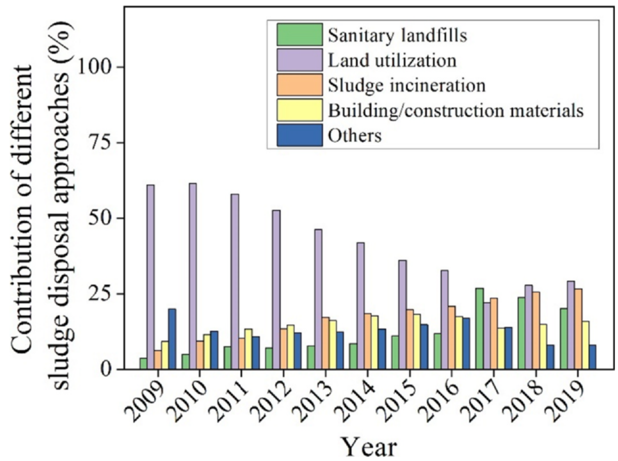

Fig. 4. Contribution of different sludge disposal routes in China during 2009–2019.

4. Spatial/temporal trends of sludge management techniques in China

4.1. Sludge management in China and comparisons with European regions

4.1.1. Sludge management in China

Currently, the predominant technical routes of sludge management mainly include sanitary landfills, incineration, building materials production, land application and others (Yang et al., 2015), and percentage contribution variation of the abovementioned five approaches in China during the period of 2009–2019 was analyzed in Fig. 4. By the end of 2019, it was estimated that approximately 29.3% of the sludge produced in China (39.04 million tons, 80% W_c) was disposed via land application, followed by incineration (26.7%) and sanitary landfills (20.1%), whereas the amounts disposed via building materials utilization (15.9%) and others (8.0%) were much lower.

Land application was the mainstay technical route of sludge disposal in China. Specifically, the contribution gradually declined from 60.9% in 2009 to 52.6% in 2012 and further to 21.9% in 2017 (Fig. 4). The above results clearly revealed a dramatic change in sludge disposal strategy in China from random stacking to strategic land application, which could be further proven by the slightly increasing of land application to 29.3% in 2019. Although Elmi et al. (2020) noted that the phytotoxicity of heavy metals in sludge limits its long-term land utilization, the digested/composed sludge with the majority of metals stabilized and pathogens inactivated could be applied for land disposal to improve soil quality and recycle nutrients (Fytili and Zabaniotou, 2008b; Chen et al., 2012).

The percentage contribution of sludge incineration increased from 6.2% in 2009 to 17.3% in 2012 and further to 26.7% in 2019, exhibiting an average annual growth rate of 29.1% in the period of 2009–2013 and 7.5% in 2014–2019. Sludge incineration completely eliminates viruses and pathogens and converts sludge organics into CO_2 (Chen et al., 2012; Odegaard et al., 2014), might be the main reason why sludge incineration is recommended as the best practice for sludge disposal in regions with limited available space.

As illustrated in Fig. 4, the sanitary landfills contributed 3.71% of the disposed sludge in 2009 in China, which gradually increased to 11.9% in 2016 (annual growth rate of 18.1%). Several studies have revealed that the main drawbacks of sludge landfills are leachate pollution, massive space needs, and GHG emissions (Chen et al., 2012; Vashistha et al., 2019); thus, stricter regulations and laws have been implemented for potential environmental risk control (Zhao et al., 2019). However, the sharp increase in sludge sanitary landfills to 26.8% in 2017 in China was closely related to the emergency treatment of sludge for feedback from national environmental protection supervision (Wang et al., 2019c). The percentage contribution of sanitary landfills was 20.1% in 2019, partially related to sludge management in small towns with limited technical/financial resources (Chen et al., 2012).

Statistical data revealed that the approach of building material production contributed only 9.2% of the disposed sludge in China in 2009, which gradually increased to 14.8% in 2012 and reached its maximum value (18.2%) in 2015. However, a slight decline in the contribution, to 13.7% in 2017 and 15.9% in 2019, was related to China's regional coal consumption control strategy (Liu et al., 2016a). In addition, the significant decline in the contribution of other approaches in the past decade (from 19.9% in 2009 to 8.0% in 2019) revealed that the Chinese government achieved great progress in the safety and standardized disposal of WAS.

4.1.2. Sludge management in European 重点阅读

Land utilization (direct or after composting) and incineration were the two main practices for sludge management in European, quite similar to that of China; moreover, the work of Kelessidis and Stasinakis (2012) stated that this trend would be more significant in the near future. The municipal WAS production and management in Japan and 31 European countries, including Germany, France, United Kingdom, Poland, Spain, Netherlands and etc., are summarized and analyzed in Table S4. EK1

In overall, German, United Kingdom, Spain, France and Poland, are the top five countries in sludge production among European countries, with a productivity (dry basis, in thousand tons) of was 1.80 (in 2015), 1.76 (in 2009), 1.21 (in 2009), 1.17 (in 2017), 0.59 (in 2017), respectively. The production of sludge in China is significantly higher than those European countries, for instance the sludge production of China in 2017 was 32.5 million tons (80% water content), about 2.63-fold and 4.57-fold higher than that of German and France. Sludge production trend in European countries during the period of 2007–2017 was quite discrepant, in which the production of sludge in Malta exhibited the highest expansion rate (annual average growth rate of 37.63%), and Croatia was the lowest (−6.29%); whereas majority countries under a constant level.

Attitudes on sludge management in different European countries varied widely (see in Fig. 5). In brief, the incineration was the predominant practices for sludge treatment in those countries of like Belgium, Netherlands and Germany, because the agricultural utilization of sludge is strictly restrained by regulations (Mininni et al., 2015). The recent work of Fytili and Zabaniotou (2008a) demonstrated that the sludge incineration remains as the most attractive disposal method, currently in Europe, and in this context there will be an increase in the role of incineration in the long run. By contrast, majority European countries have a more balanced sludge treatment strategies, being agricultural utilization in the range of 20–40% whereas incineration also plays an important role, such as France, Croatia, Sweden, Austria and etc. The study of Stoll and Parameswaran (1996) stated that the incineration can serve as first priority options for sludge disposal in EU after their multi-criteria analysis, despite the direct agricultural utilization of municipal sludge exhibited the least cost as compared with sanitary landfill and incineration, and they believed that the selection of an option for sludge disposal cannot be based only on the cost competitiveness.

The approach of compost and other application also been widely applied in those typical agricultural countries of Slovakia, Hungary, Luxembourg, Lithuania, Estonia and etc., with a percentage contribution of 40–60% of the sludge disposal. On the other hand, landfill was still the predominant approaches in Malta, Croatia and Turkey, which accounted as much as 42–100% of the total sludge generated. It should be pointed out that differed from the fast increasing in sludge production in China, yearly increasing rate of the sludge production in different EU countries were varied widely, due to their enforcement in law/regulation of sludge treatment/disposal and huge investment on sludge management in some European countries.

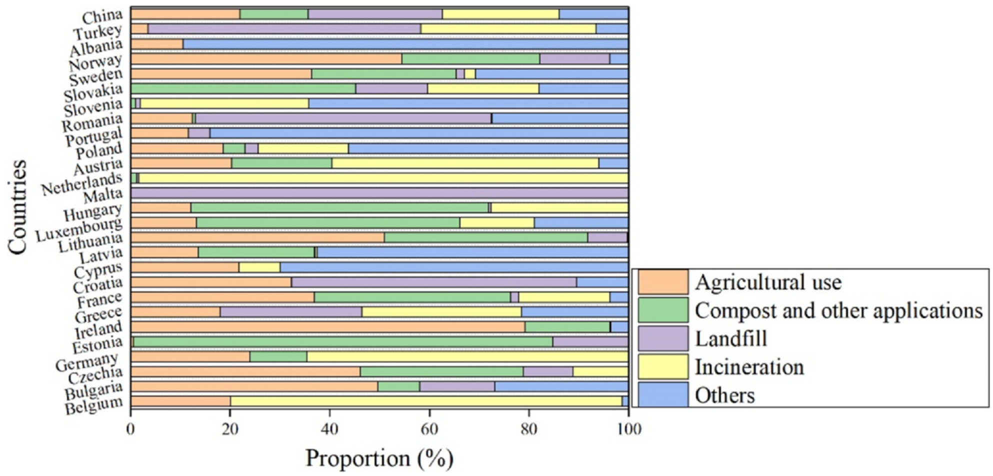

Fig. 5. Comparison of sludge disposal routes of China, Japan and European countries in 2017.

4.2. *Geographic distribution of typical sludge treatment techniques in China*

The construction of anaerobic digestion, aerobic composting, incineration, drying, deep-dewatering, and thermal hydrolysis plants required high financial investment (Yang et al., 2015), thus related information of those projects constructed in China were analyzed here. Specifically, data from seven administrative regions of China, including North China, South China, Southwest China, Central China, East China, Northwest China, and Northeast China, were compared and the results are listed in Fig. 6.

Treatment capacities of six typical technologies exhibited rapid growth during the whole statistical period, especially during 2009–2019. In 2019, incineration and aerobic sludge composting were the most widely used technologies in sludge treatment in China, reaching an overall treatment capacity of 27,122 and 11,250 t/d (W_c, 80%), respectively. In addition, thermal hydrolysis, as an effective approach for sludge pretreatment, been widely constructed in China recently and the treatment capacity increasing to 8342 t/d in 2019. Detailed information on construction and geographic distributions of different sludge treatment projects in China can be found below.

4.2.1. *Anaerobic digestion*

The overall treatment capacity (W_c, 80%) of all anaerobic sludge digestion projects in China in 2000 was 386 t/d (Fig. 6A), it rapidly increased to 2814 t/d during the period of 2000–2010 (sharply increased since 2008) and further to 6944 t/d in 2019. Specifically, those anaerobic sludge digestion plants were widely distributed, in which East China and North China contributed as much as 27.1% and 18.7% of the total treatment capacity, respectively, followed by Central China (18.2%) and Northwest China (16.1%), while that of South China, Southwest China and Northeast China was quite low (totally 19.9%).

Overall, anaerobic sludge digestion is the most energy-efficient means to capture energy from biosolids compared with other technologies. In the EU, approximately 50% of WWTPs equipped with anaerobic digestion facilities (Werle and Wilk, 2010); besides large amount of energy extracts, land application of those digested sludge also saves fertilizer consumption. For example, recent work of Mayer et al. (2019) reported that the recycling of digested sludge (60 million tons WAS) in the EU would save approximately one million tons of nitrogen and 2×10^7 tons of organic carbon (Paes et al., 2019). Theoretically, up to 2.68 million tons of WAS (W_c, 80%) were digested in China by the end of 2019, and an estimated generation of 1.32×10^8 m^3 CH_4 and another 1.47 million tons fertilizer.

4.2.2. *Sludge compost*

As illustrated in Fig. 6B, sludge compost was one of the most widely used technical routes for sludge treatment in China, exhibiting explosive growth during the past 10 years. Treatment capacity of the sludge compost plants built in China in 2009 was approximately 300 t/d (W_c, 80%), and it dramatically increased to 8140 t/d in 2015 and further to 11,250 t/d in 2019. Similarly, sludge composting has been used in 25 out of 27 countries in Europe, which accounted for as much as 42% of total sludge production in 1995 (3 million Mg of dry matter in suspend solids), and it quickly increased to 59% in 2010 (Collivignarelli et al., 2019), finally being used for agricultural utilization.

Currently, sludge composting plants have been widely constructed in Northeast China, with a maximum sludge treatment capacity of 5202 t/d in 2019, contributing as much as 46.5% of the total composted sludge in China. This was followed by East China (2510 t/d), Central China (1430 t/d) and North China (862 t/d), whereas the amount in Northwest China, South China and Southwest China were quite lower (10.7% in total). Wainaina et al. (2020) noted that a maximum reduction of 65% of the initial volume of sludge would be reached after sludge composting. Combined with the test results from the calculation of CO_2-eq emissions reduction during organic waste composting (Varma et al., 2018), we can deduce that a reduction as high as 0.55 million tons of CO_2-eq emissions would be achieved in China in 2019 via sludge composting, exhibited economic feasibility in carbon markets.

4.2.3. *Incineration*

Incineration, achieved an overall treatment capacity of 27,122 t/d (80% W_c) in China by the end of 2019, with the highest sludge treatment capacity among all six selected technical routes. East China contributed as much as 70.3% of the total quality of incinerated sludge in China, followed by North China (7.5%), Southwest China (6.8%) and South China (5.9%), whereas less was obtained in Northeast China, Northwest China and Central China. Generally, sludge incineration has been widely used in developed areas/cities (fast economic growth and sufficient financial support) with land scarcity issues (Chen et al., 2012), such as the cities of Xiamen, Xuzhou, Jiaxing, and Wuhu for WWTPs sludge treatment (Zhang et al., 2016; Lu et al., 2019). Theoretically incineration of dry sludge, characterized by a calorific value of 12–20 MJ/kg (11.7–15.8 MJ/kg for lignite), would effectively achieve the recovery of heat to produce steam (Menendez et al., 2002; Samolada and Zabaniotou, 2014), and exhibited the highest environmental benefit in all categories, except for human toxicity (Xu et al., 2014).

The incinerated sludge amount in China in 2005 was approximately 172 t/d, which gradually increased to 3752 t/d in 2010 and further to 13,222 t/d in 2015. The average annual increasing rate of sludge

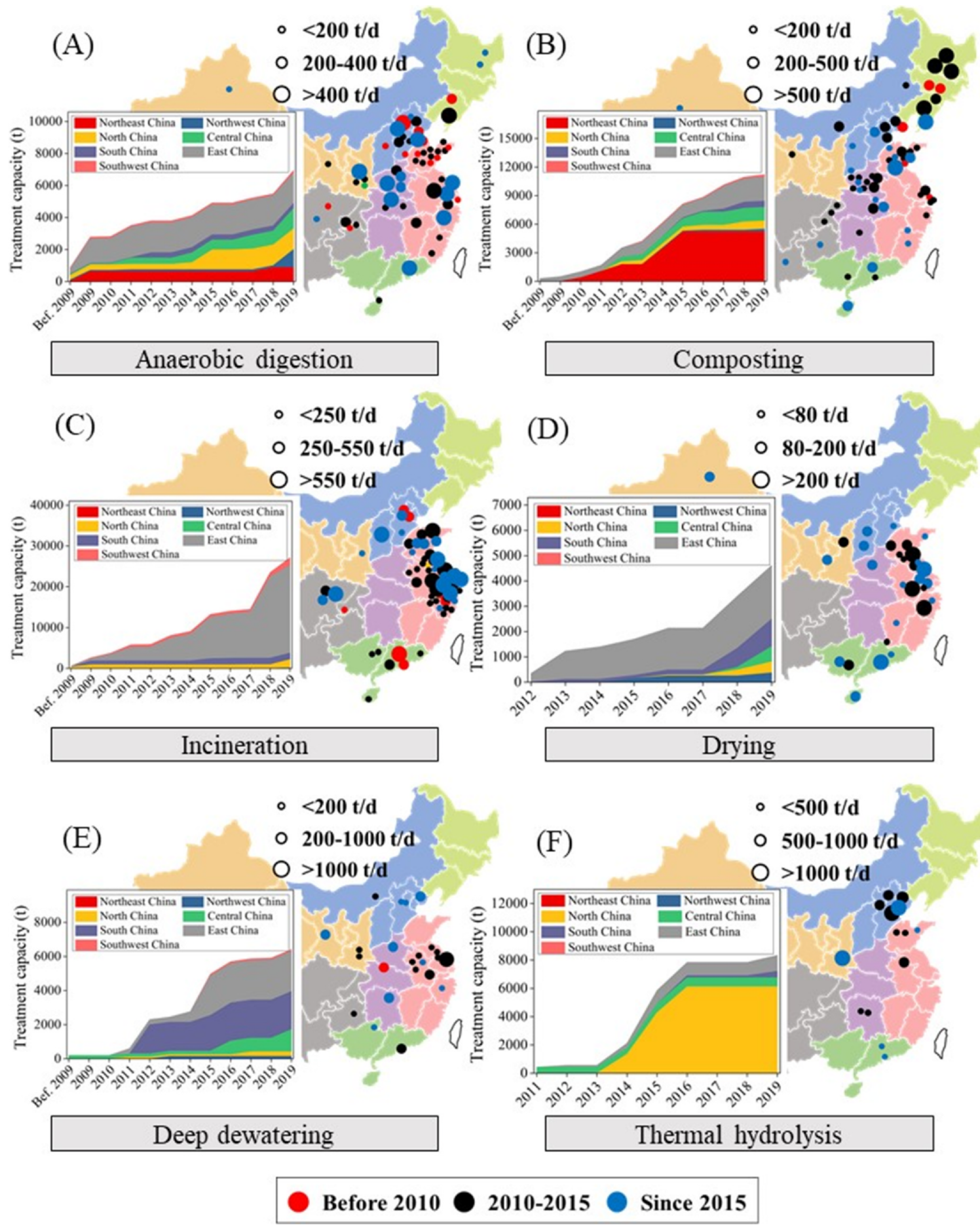

Fig. 6. Geographic distribution and treatment capacities of the projects of anaerobic sludge digestion (A), composting (B), incineration (C), drying (D), deep dewatering (E), and thermal hydrolysis (F) in different geographic regions of China.

incineration in different periods declined as follows: 2005–2010 (85.3%) > 2010–2015 (28.7%) > 2015–2019 (19.7%) and the highest was in 2005–2010. A similar observation of sludge incineration was observed in the EU during the periods of 1999–2005, which almost doubled from 11% to 21% (Kelessidis and Stasinakis, 2012). However, investing and construction of sludge incineration can be restrained due to the high energy consumption and air pollution problems raised from incineration (Xiao et al., 2018).

4.2.4. Sludge drying

The geographic distribution of the sludge drying plants was similar to that of sludge incineration, with a declining treatment capacity as follows: East China (45.9%) > South China (23.5%) > Central China (13.0%) > North China (9.5%) > Northwest China (8.0%) > other regions (0.0%). The sludge drying capacity of China in 2012 was 350 t/d and increased to 4615 t/d in 2019, annually increased 44.5%. As an specific example, it has been successfully used in the largest Chinese WWTP of Shanghai Shidongkou (Wang et al., 2010). Despite significant volume reduction, heating value increases and transportation cost savings have been observed for sludge drying (Chang et al., 2020), the high energy consumption during water vaporizing (3150 kJ/kg water) was main disadvantage of such progress. Thus, emerging energy sources, such as solar energy, have been considered as economic and ecological alternative energies which can be applied for sludge drying (Dichtl et al. (2007). The case studies in Greece and Turkey proven that the application of solar drying and limited liming/solar drying would achieve a better sludge dewatering efficiency, than that using the liming method only (Salihoglu et al., 2007).

4.2.5. Deep-dewatering and thermal hydrolysis

Deep-dewatering technologies, including high-speed centrifuges, belt filter press, plate-frame and air drying, have been widely used in China to meet the stricter solid content requirements (Yang et al., 2015). East China contributed as much as 36.8% of the sludge deep-dewatering approaches in 2019, while South China was the second-ranked region (34.8%), Central China was the third (20.3%), and other regions contributed 8.1% totally. Sludge deep-dewatering plants were substantively constructed during the period of 2010–2015 (treatment capacity increased from 200 t/d to 4970 t/d), exhibiting an average annual increasing rate of 70.8%. By the end of 2019, the capacity of sludge deep-dewatering in China was approximately 6410 t/d. An effective sludge deep dewatering approach should consider both economic feasibility and dewatering efficiency, Zhang et al. (2019a) found that sludge dewatered by hydraulic compression can greatly reduce the sludge volume.

Similarly, thermal sludge hydrolysis has recently exhibited rapid application in China, with the treatment capacities increasing from 2122 t/d in 2014 to 8342 t/d at the end of 2019. Thermal hydrolysis is normally applied prior to sludge digestion to improve the hydrolyzation efficiency, and example of Fdzpolanco et al. (2008) revealed that a 40% increase in energy output would be achieved by thermal hydrolysis. Overall, sludge thermal hydrolysis plants were prominently constructed in North China (73.46%), followed by East China (13.6%), Central China (7.6%) and South China (5.4%). A case study in Athens demonstrated that the application of thermal hydrolysis improved the dewaterability significantly, led to a dewatering efficiency improvement from 22% to 31% (Zikakis et al. (2019). According to the study of Phothilangka et al. (2008), as much as75-80% increasing in biogas production, as well as a 25% operation cost reduction in anerobic digestion after the implementation of full-scale thermal hydrolysis. Therefore, thermal hydrolysis could be developed as a promising alternative route for sludge management, especially for enhancing the efficiency of sludge digestion or deep dewatering (Samolada and Zabaniotou, 2014).

5. Overall status and forecast of CO_2-eq of sludge management in China

5.1. Reginal CO_2-eq of sludge management of China

Regional variations in the CO_2-eq emissions of different sludge disposal routes during the period of 2009–2019 were calculated, and the results are illustrated in Fig. 7. In 2009, the overall GHG emissions (in 10^8 kg CO_2-eq) calculated from sludge incineration, building materials utilization, sanitary landfills and land utilization were 5.17, 1.63, 34.33 and 2.39, respectively (Fig. S6). Specifically, sanitary landfills and incineration were the predominant contributors and accounted for 78.88% and 11.88%, respectively. In 2019, the growth of WAS resulted in a 2.49-fold increase in the total GHG emissions in China (108.18 $\times$ 10^8 kg CO_2-eq). The percentage distribution of the six typical technical routes exhibited a decreasing trend of incineration (45.11%) > sanitary landfills (23.04%) > land utilization (17.64%) > building materials (14.21%), in which the incineration and sanitary landfills of WAS released as much as 48.80 and 24.93 $\times$ 10^8 kg CO_2-eq GHG in 2019, respectively.

East China contributed the highest GHG emissions during past decades, which accounted for 41.35% of the overall emissions in 2009 (17.99 $\times$ 10^8 kg CO_2-eq) and 44.83% in 2019 (48.50 $\times$ 10^8 kg CO_2-eq). The other six regions exhibited a decreasing trend of North China (13.35%) > South China (10.60%) > Central China (9.48%) > Southwest China (8.10%) > Northwest China (7.33%) > Northeast China (6.30%) in 2019. Overall, the percentage contribution of Northeast China, Central China, Southwest China, and South China exhibited a decrease (totally from 43.15% to 34.49%), whereas that of North China and Northwest China increased from 15.50% to 20.68%. In 2009, sanitary landfills caused high GHG emissions in Northeast China and South China; however, landfilling was still the predominant source of GHG emissions in Northeast China in 2019 (contributing 56.16% of released CO_2), whereas incineration for South China (44.72%).

By now, the average GHG emissions in East China was the highest among all seven administrative regions, with an averaged emission rate of 11.78 kg CO_2-eq per capita and 45.80 $\times$ 10^8 kg CO_2-eq total emissions in 2019. In overall, the average GHG emissions in China was slightly lower than that of that of Japan and most European countries (Fig. S5). For instance, the per capita GHG emissions of East China in 2017 was 10.74 kg CO_2-eq, much lower than that of Japan (24.89 kg CO_2-eq per capita) and Netherlands (44.12 kg CO_2-eq per capita). However, with a high population density, the total GHG emissions in China was much higher than other countries. As shown in Fig. S5, up to a 43.74 $\times$ 10^8 kg CO_2-eq emissions released from sludge disposal in East China, were much higher than that of 31.76 $\times$ 10^8 kg CO_2-eq in Germany (the highest in Europe) and 31.53 $\times$ 10^8 kg CO_2-eq in Japan.

Regarding the contribution of individual technical routes of different regions, sludge incineration in East China contributed almost one-third of the overall CO_2-eq emissions of China in 2019 (31.27%). This was followed by sanitary landfills in East China (5.52%), incineration in South China (4.74%), land utilization in North China (4.51%), sanitary landfills in Northwest China (4.50%), and building materials in East China (4.11%), and the remaining proportion were < 4%. East China accounted for 6.5% of the land area of China and 29.33% of the population (Jin et al., 2014); thus, sludge incineration is recommended for the limited land space needed and fewer end-products (Chen and Kuo, 2016). Despite a lower GHG emissions of sanitary landfills, the less efficient gas recovery in landfills in China resulted in direct GHG emissions (Liu et al., 2016b; Koutsou et al., 2018). Similarly, carbon emissions during sludge building material utilization (14.21%) were also higher, partially ascribed to the disposal of sludge via cement manufacturing and other approaches (Vashistha et al., 2019). For the purpose of controlling GHG emissions, sludge incineration and building materials production in East China, as well as sanitary landfills throughout China, should be highly concerned.

5.2. Forecast of CO_2-eq of sludge management

Assuming no variation of sludge production (ton per 10000 m^3 wastewater treated) during the period of 2019–2030, after calculation (seen in Supplementary Materials), the sludge production in China will reach 56.5 million tons (80% moisture content) by the end of 2030. If the percentage distribution of the sludge management approaches stays at the same level to that 2019 (29.3%, 26.7%, 20.1%, 15.9% and 8.0% for land application, incineration, sanitary landfills and building material utilization, and other, respectively), the predicted CO_2-eq would be 170.25 $\times$ 10^8 kg CO_2-eq.

It should be pointed out that the management strategies of sludge in China changed significantly in recent years (Lu, 2019). For instance, the newly published Control standards for pollutants in sludges from agricultural use (GB4284-2018), which firstly published in 1984, loosened the restriction of the heavy metals in sludge during agricultural utilization, especially for Zn and Cu. Undoubtedly, percentage contribution of the sludge management for land utilization would increase in near future. Besides, the growing land scarcity in those regions of Beijing-Tianjin-Hebei, Yangtze River Delta, Pearl River Delta and etc., would further lead to an increase of incineration and building materials utilization of WAS, whereas might be a noteworthy declining in sanitary landfill. Therefore, we assumed that the percentage contribution of sludge management for sanitary landfill, land utilization, incineration, building materials utilization and other will be 10%, 35%, 25%, 25% and 5%, respectively. Based on above, we can predict that the overall GHG emissions of sludge management in China, in 2030, would be 155.93 $\times$ 10^8 kg CO_2-eq (8.41% lower than the predicted data under

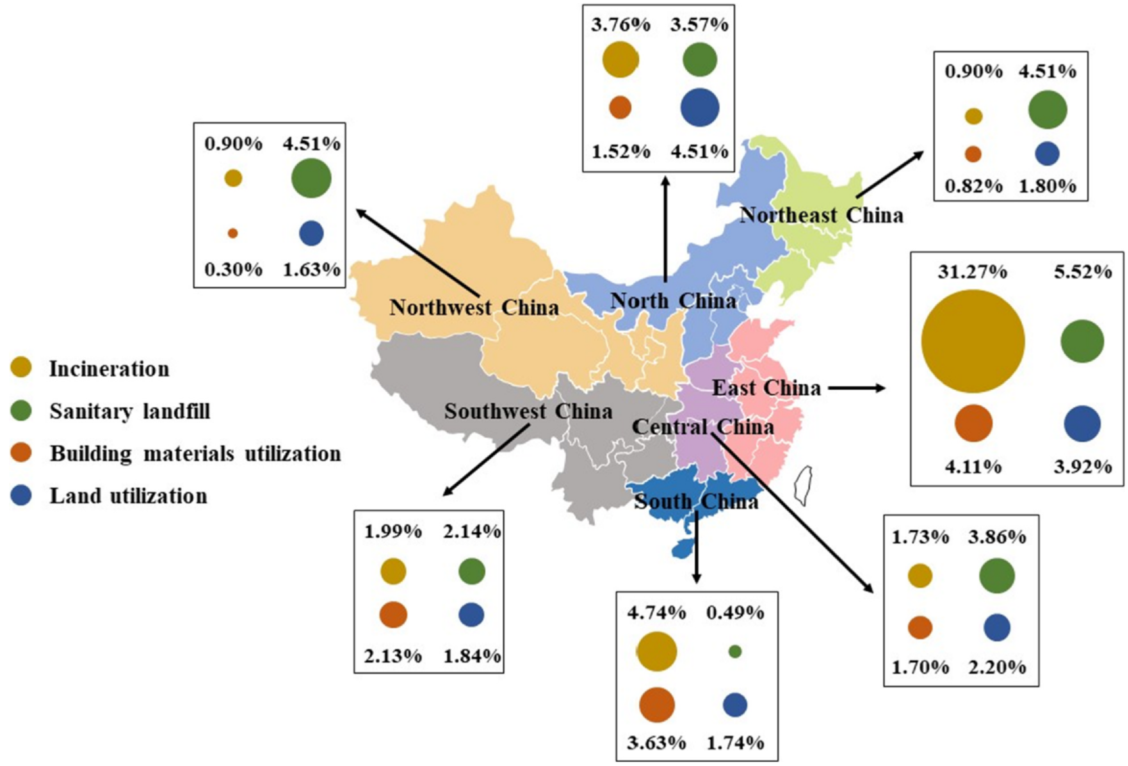

Fig. 7. Reginal percentage contribution of CO_2-eq of different sludge disposal routes in China in 2019.

current situation). In brief, sludge incineration contributed as much as 42.45% of the overall CO_2-eq, followed by building materials production (22.49%), land utilization (18.55%) and sanitary landfill (11.51%), respectively. If percentage contribution of the above mentioned five technical routes reaches 10%, 50%, 20%, 15% and 5%, the overall GHG emissions would decline to 146.40×10^8 kg CO_2-eq instead. Thus, the optimization of the sludge disposal strategies, especially the sludge incineration, would undoubtedly decrease the GHG emission in China.

6. Regional discrepancy and legislation of sludge management in China

Significant differences and unbalanced distributions of sludge treatment techniques were observed among the seven administrative regions of China (Fig. 8), which could be summarized as follows: (1) Regional economic conditions, urbanization rate, financial support and land price would affect the sludge treatment/disposal technical routes chosen. Incineration and drying, which require higher financial investment, energy consumption and less space, are preferentially used in East China and North China. In contrast, aerobic compost was most frequently used in Northeast China and Central China, and anaerobic digestion also in Northeast China. (2) Treatment capacities of sludge facilities exhibited a significant increase in 2009–2019, especially 2011–2015 for sludge compost, 2014–2019 for anaerobic digestion, 2005–2010 for incineration, and 2013–2016 for sludge deep-dewatering and thermal hydrolysis. (3) Incineration contributed as much as 45.1% of the overall CO_2-eq of sludge disposal in China, specifically in East China. (4) Land application of digested/composed WAS as fertilizer in China is still lower than that of the EU (26.5% *vs* 50.0%). (5) Close interlinks between different technical routes and their pretreatments in East China and North China been built, such as thermal hydrolysis and anaerobic digestion. (6) The emerging disposal of sludge led to a short-term increase in landfills in China in 2016–2017. (7) The chemical compositions of sludge should be first considered during the choice of technical routes, especially in terms of those with lower organics, and the abundant existence of metals (Jin et al., 2014).

By now, more than 36 sludge standards have been promulgated in China in recent decades (in Fig. S8), of which two were published in the 1980s, and four in the 1990s. The 2000s and 2010s, especially the 2000s, had significant developments in sludge policies, standards, laws and regulations, and more than 18 and 12 standards/regulations, respectively, were promulgated in China. A detailed information of the policies and standards and etc. could be found in Supplemental Materials.

7. Future perspectives and suggestions

7.1. Challenges associated with sludge treatment and disposal in China

Although remarkable achievements have been achieved in China in sludge management recently, especially since 2008, the current status of sludge treatment and disposal of sludge still needs upgrading, with the challenges summarized as follows.

(1) Regional imbalance of sludge treatment and disposal in China should be of great concern, especially in the regions of Northwest and Southwest China.

(2) There is still a need for a scientific plan for sludge treatment/disposal for efficient recycling, utilization of valuable matter within WAS, and for a mandatory consideration of sludge-to-energy and GHG emissions controlling.

(3) Sludge stabilization and land utilization in China are still low.

(4) The low organic concentration and high sand content of WAS in China should be of great concern.

(5) The lack of compulsory standards for sludge disposal might be the main reason for the poor sludge disposal situation in China (Lu et al., 2019).

(6) Funding for sludge plant construction is still low, in which sludge treatment obtained only 7.58% of the investment in treatment in China in 2011–2015, much lower than that in developed countries (30–50%) (Jin et al., 2014).

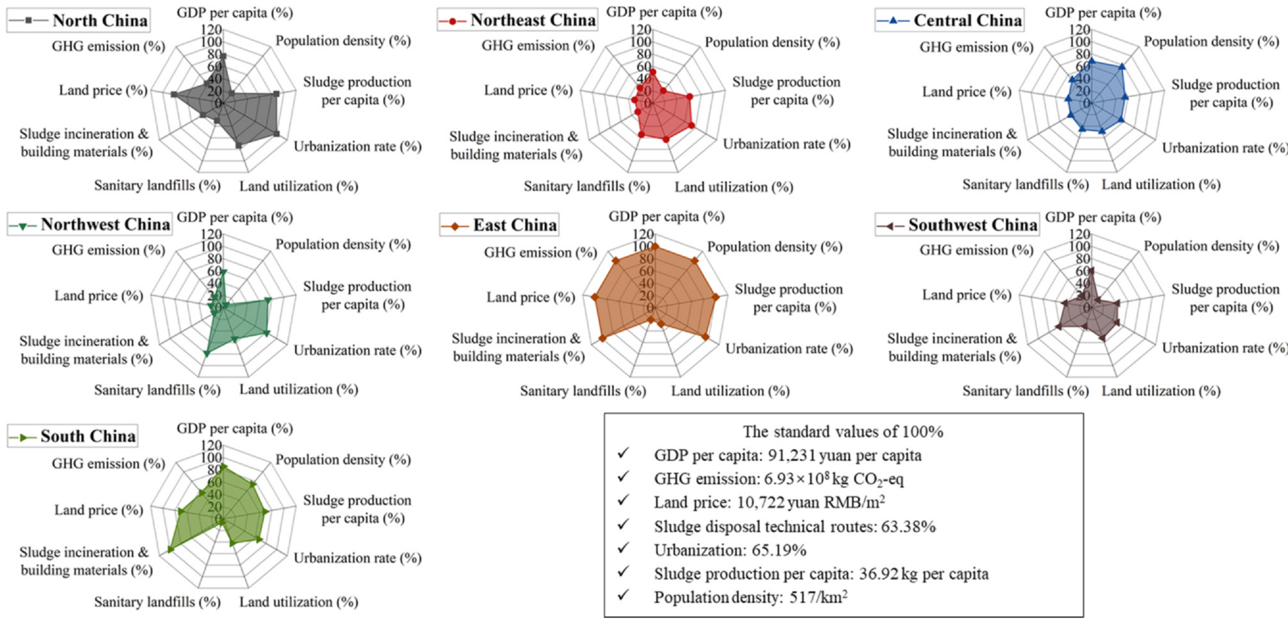

Fig. 8. Interlinks between sludge disposal technical routes (sanitary landfill, building materials production + incineration, land application) to that of reginal characteristics (land price, GHG emission, GDP per capita, population density, urbanization, sludge production per capita).

7.2. Future perspectives on sludge treatment and disposal in China

Based on the statistical data, as well as the discrepancies and challenges analyzed in Sections 6 and 5.1, some actions should be taken from technical and management aspects to improve the overall level of sludge disposal in China.

7.2.1. Technical aspects

First, it is important to enhance sludge nutrient reuse. WAS, which is rich in organics, phosphorus and micronutrients, exhibited high potential for nutrient recovery. Grobelak et al. (2019) noted that thermo-composting would realize both energy and matter recovery, theoretically, an optimal temperature, moisture, pH, aeration conditions, and balancing of nutrients were necessary for higher quality fertilizer production (Agnew and Leonard, 2003; Wainaina et al., 2020). For instance, Wang et al. (2019) stated that elements of N and S would be efficiently preserved via KH_2PO_4 and $FeSO_4$ additives during sludge compost. Gondek et al. (2018) revealed that the addition of composts to soil decreased the contents of the mobile phases of Cu, Zn, Cd, and Pb, especially under a well-proportioned C/N of 22 (Wang et al., 2019).

Second, it is necessary to develop and implement new technologies for disintegrating sludge flocs and enhancing energy conservation. To ensure better end-use of sludge-related products (Kelessidis and Stasinakis, 2012), tremendous emerging innovative technologies, such as hydrothermal pyrolysis, air oxidation, and co-digestion, have been widely studied in case studies (Xiao et al., 2018). For example, Hii et al. (2014) found that the application of wet air oxidation and thermal hydrolysis enhanced sludge digestion, and Huang et al. (2019) established a biological thermal-alkaline synergistic system for enhancing hydrolysis-acidification rate. Similarly, Yang et al. (2019) developed a combined system of post-thermal hydrolysis and centralized recirculation for enhancing anaerobic sludge digestion.

Third, low-carbon operation of sludge technologies. WAS may be recognized as a renewable fuel and be eligible for 'carbon credits'; thus, the low-carbon operation, sustainable management and resource recovery of sludge disposal should be highly considered in China. For example, the recent work of Mayer et al. (2019) reported the digestion of 60 million tons of WAS, and subsequent recycling using digested sludge would save approximately one million tons of nitrogen and 20 million tons of organic carbon. Correspondingly, Hao et al. (2020) stated energy recovery from sludge combustion benefited energy balance of WWTPs; similarly, a case in Italy revealed that electro-dewatering process upgrading would be feasible if sludge disposal followed an incineration route (Zhang et al. (2019b). Forecasting of the Chinese GHG emission of sludge management in 2030 (Section 5.2) also indicated the as much as 14.0% of the CO_2-eq would be achieved once the contribution of the sanitary landfill, land utilization, incineration, building materials utilization and other changed to into 10%, 50%, 20%, 15% and 5% percentage distribution.

Fourth, a new area that demands attention is the emerging contaminants in WAS. Adherences and bindings of emerging contaminants, such as pharmaceutical products, antibiotic resistance genes, hormones, nanomaterials, alkylphenols, radioactive nuclides, and etc. (Mailler et al., 2017; Picó et al., 2017; Yui et al., 2018; Li et al., 2019), onto the WAS would negatively affect their disposal; thus, the behavior and performance of those toxic emerging contaminants should be of high concern.

7.2.2. Management aspects

(1) Strategies for increasing organic content and declining heavy metals in WAS should be taken, especially in terms of perfecting the drainage system.

(2) Compulsory standards for sludge treatment should be promulgated as soon as possible. Moreover, land utilization of digested/composed sludge should be encouraged in regions with large areas of economic crop planting.

(3) Basic information collection on the chemical characteristics of WAS and statistics on the distribution and construction of sludge treatment/disposal plants, as well as the final utilization of those treated sludge, should be strengthened.

(4) A reliable system that involves the proposal procedures, specifications and technical guidelines to guide the rationalization and implementation of sludge treatment and disposal is especially needed.

7.3. Suggestions on sustainable sludge management other developing countries

Sludge management played an important role in sanitation programs via reducing health problems and associated risks. However, rational utilization of sludge is still the biggest challenge for sustainable sludge management in developing countries where sludge disposal takes place unscientifically without proper planning during the past few years (Srivastava et al., 2015). Thus, sludge management processes in those developing countries should be seriously evaluated in terms of economic applicability, GHG emission controlling, as well as their efficiency (case-by-case basis), since there is no universal solution. Generally, the primary sludge management should be chosen based on the consideration of sludge reduction, land requirement and energy requirement (Singh et al., 2017).

The regions with fast economic growth, huge population and land scarcity issues, the model of East China for sludge management should be highly recommended. By contrast, those regions with poor economic conditions, low urbanization rate, it is practically urgent to support the beneficial reuse of sludge rather than promoting sludge reduction. Thus, agricultural land utilization of the proper treated sludge should be highly considered for improving crop yielding in those regions without soil degradation. Besides, those countries with considerable soil degradation must think about sludge applications for soil amending (Stoll and Parameswaran, 1996).

Regulations concerning the sludge management in developing countries can also make the reference from the experience of China, for a purpose of enhancing the sludge treatment/disposal from random stacking to strategic management. It should also be pointed out that the recent work of Jimenez et al. (Jimenez et al., 2004) stated that the sludge legislation in China mainly focused on inorganic chemicals. Thus, other aspects, like health concerns, economic conditions, soil characteristics, technical capabilities, and etc., should not be neglected for those developing counties when they establishing their regulations. We strongly suggested that the government of those developing counties should design sustainable management systems for guiding the rationalization and implementation of sludge management, for a purpose of improvement of sanitation of municipal sludge (Bauerfeld et al., 2008).

8. Conclusion

Based on the descriptions and discussions above, the following key conclusions can be drawn:

(1) In total, 39.04 million tons sludge (W_c, 80%) was produced in 2019, in which 29.3%, 26.7%, 20.1% and 15.9% of sludge was disposed via land application, incineration, sanitary landfills and building material utilization, respectively.

(2) The average sludge (80% W_c) production, per capita, of China was 28.1 kg/y in 2019, which was 5.9 tons per 10,000 m^3 wastewater treated. The higher the urbanization rate and per capita GDP were, the higher the per capita sludge production was. Urbanization rate increasing of the northern 14 Chinese provinces (excluding Beijing) led to a significant decline in sludge production (in tons/10^4 m^3 wastewater treated).

(3) Regional economic conditions, urbanization rate, financial support and land price would affect the choice of sludge treatment technical routes, in which the technics of incineration, thermal hydrolysis, and drying were preferentially used in East China and North China, whereas aerobic compost in Northeast China (contributed 46.5%) and East China, and anaerobic sludge digestion was also in Northeast China.

(4) The overall GHG emissions in China in 2019 reached 108.2×10^8 kg CO_2-eq, and incineration in East China contributed 31.3% of the overall GHG emission. The increasing the land utilization and declining of the incineration of sludge would be beneficial for the GHG emissions controlling.

(5) From the government level, forceful administrative orders should be set to improve the sludge management in China. Besides, we strongly suggested that the government of those developing counties should design sustainable management systems for guiding the rationalization and implementation of sludge management.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary material

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envint.2020.106093.

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