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Title: Greenhouse gas emissions associated with urban water
infrastructure: What we have learnt from China's practice

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Abstract

Municipal water and wastewater services have complicated sources of greenhouse gas (GHG) emissions, and quantifying their roles is critical for tackling global environmental challenges. In this study we provide a systematic review of the state-of-the-art on GHG emission characterizations of China's urban water infrastructure with the aim of shedding light on global implications for sustainable development. We started by synthesizing a framework on GHG emissions associated with water and wastewater infrastructure. Then we analyzed the different sources of GHG emissions in drinking water and wastewater treatment systems. In drinking water services, electricity consumption is the largest source of GHG emissions. A particular concern in China is the common use of secondary pumping for high-rise buildings. Optimized pressure management with an efficient pumping system should be prioritized. In wastewater services, non-CO₂ emissions such as methane (CH₄) and nitrous oxide (N₂O) emissions are substantial, but vary greatly depending on regional and technological differences. Further research directions may include GHG inventory development for urban water systems at the plant level, quantifications of GHG emissions from sewer systems, emission reduction measures via water reclamation, renewable energy recovery, energy efficiency improvement, cost-benefit analyses, and characterizations of Scope 3 emissions.

Keywords

urban water systems; greenhouse gas accounting; water-energy nexus; non-CO₂ emissions; life-cycle thinking

1. Introduction

In the past few decades, China has made unprecedented progress in economic development. Along with this growth, the country has also dramatically improved its water infrastructure, expanded its clean water service and wastewater treatment, and improved protection of the water environment. The population able to access tap water from municipal water treatment plants increased from 303 million in 2004 to 503 million in 2018 (+4.7%/year), and the total capacity of wastewater treatment increased from 74 million m³/d in 2004 to 181 million m³/d in 2018 (+10%/year) (NBS, 2018). Such a trend will continue as China's urbanization continues; another ~10% of the total population is expected to move to urban areas by 2030 (United Nations, 2017). In the United Nations' Sustainable Development Goals (SDGs), SDG 6 is dedicated to the water, sanitation, and hygiene (WASH) sector (United Nations, 2015), which aims to achieve universal and equitable access to safe and affordable drinking water and sanitation for all, especially for developing countries. In this regard, China has established a good model and will continue to play an exemplary role in achieving the global SDGs (Z. Xu et al., 2020).

Municipal water and wastewater services have complicated sources of greenhouse gas (GHG) emissions. Although the WASH sector's contributions to national GHG inventories are usually between 1~3% (Hillman & Ramaswami, 2010; Bo Zhang et al., 2015), the absolute magnitude from a big country like China is noticeable. There is also high uncertainty around GHG inventories from wastewater and sludge treatment (Hall, West, Sherman, Lane, & de Haas, 2011; Sweetapple, Fu, & Butler, 2013). Recent studies show there are both synergies and trade-offs between achieving SDG 6 on clean water and sanitation and SDG 13 on climate actions (Q. Zhang et al., 2019; Q. Zhou, Leng, & Huang, 2018). Understanding the complete picture of GHG emissions associated with the WASH sector remains a challenging but fundamental requirement for policymakers and practitioners wanting to respond to the grand challenges in water, energy, and climate change.

79

80 In this study, we provided a systematic review of the state-of-the-art on GHG emission characterizations
81 associated with China's urban water infrastructure and hope to shed light on global implications for
82 sustainable development. We conducted a comprehensive literature search and critically reviewed
83 relevant resources on other countries' cases to compare with results found in China.

84

85 Following this introduction, we present a systems perspective to source GHG emissions accompanying
86 the urban water cycle in Section 2. Recent studies on GHG emissions associated with drinking water
87 systems in China are reviewed in Section 3. In Section 4, we critically evaluate current estimations of
88 GHG emissions related to wastewater treatment systems in China. A brief summary of review highlights
89 is provided in Section 5. GHG reduction opportunities for the water sector are illustrated in Section 6.
90 Conclusions and the outlook for future research are provided in Section 7.

91

92 2. A systems perspective of GHGs associated with urban water infrastructure

93 In this review, urban water infrastructure includes municipal drinking water infrastructure and
94 wastewater infrastructure. The term urban water systems (sometimes an urban water system) refers to an
95 integrated system including water and wastewater infrastructure (Arora, Malano, Davidson, Nelson, &
96 George, 2015; Hering, Waite, Luthy, Drewes, & Sedlak, 2013; Loubet, Roux, Loiseau, & Bellon-
97 Maurel, 2014; Lundin & Morrison, 2002) and links to asset management (Alegre et al., 2013; Leitão et
98 al., 2016; L. Liu & Jensen, 2018). Industrial water treatment facilities, as well as large infrastructure for
99 water conservation and long-distance transport like the South-North water transfer project in China, are
100 not included due to different jurisdictions and scopes. The GHG emissions and other environmental

impacts associated with every stage of an urban water system can be analyzed using a life cycle assessment (LCA) (Lemos, Dias, Gabarrell, & Arroja, 2013).

Figure 1 illustrates the potential GHG emission sources associated with urban water systems with a focus on system-level life-cycle thinking (Q. Zhang, Nakatani, Wang, Chai, & Moriguchi, 2017). In the urban water system, five main sections include water purification, water distribution, water end-use, wastewater collection, and wastewater treatment (including sludge treatment and disposal). The operations and maintenance (O&M) stage of urban water infrastructure is the primary source of both direct and indirect GHG emissions. For example, Scope 1 and 2 emissions (see Sidebar) include emissions from onsite fossil fuel consumption and purchased electricity consumption, and methane (CH₄) and nitrous oxide (N₂O) emissions from wastewater and sludge management. Some studies also consider Scope 3 GHG emissions from the energy consumption embodied in chemical inputs to water and wastewater treatment processes. Besides the O&M stage, the emissions associated with the construction stage and end-of-life stage are theoretically within this accounting framework as part of Scope 3 emissions.

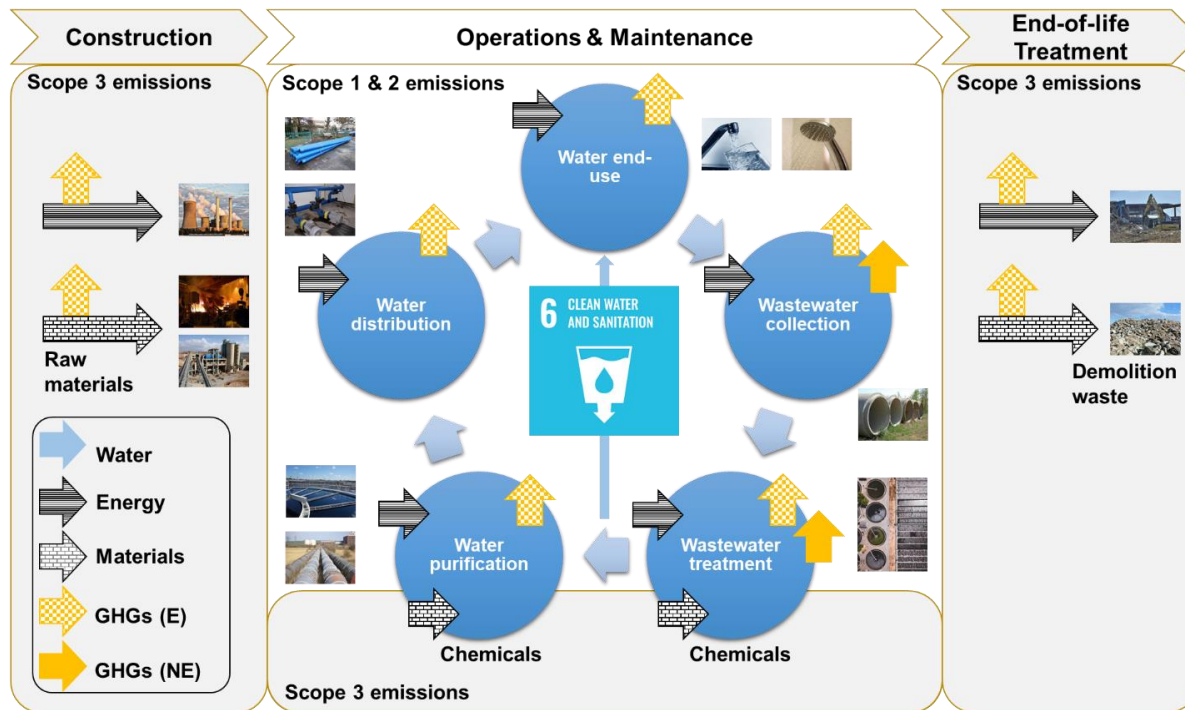


Figure 1 A systematic illustration of GHG sources associated with urban water infrastructure. GHGs (E) represents energy-related GHG emissions. GHGs (NE) represents non-energy-related GHG emissions (mainly CH₄ and N₂O emissions from wastewater systems)

Historical studies on China's city-level GHG inventories do not often focus on urban water systems. However, some studies in other countries shine light on this subject. Mahgoub et al. (2010) conduct a life-cycle assessment of the urban water system in one city in Egypt and indicate that a decentralized wastewater treatment system can help reduce the total energy consumption. A systematic evaluation of energy consumption during the urban water cycle and its environmental impacts in Norway reveals critical processes, e.g., pumping, aeration, anaerobic digestion, and pipelines rehabilitation, *etc.* (Venkatesh & Brattebø, 2011). Other evidence indicates that energy consumption from water end-users (e.g. household) can be more significant than water supply or wastewater treatment processes (Rothausen & Conway, 2011). Frijns (2012) proposes a standard framework of the carbon footprint for

the water sector, mentioning that the accounting boundary should include CO₂ from energy consumption, direct CH₄ and N₂O emissions in processes, and indirect CO₂ emissions from production of the chemicals used in treatment processes. His work shows that GHG emissions in the wastewater subsystem are much more significant than in the water supply subsystem in the case of the Netherlands, mainly due to CH₄ and N₂O emissions from wastewater treatment. However, it fails to include indirect CO₂ emissions from materials used in construction because of its limited contribution over the whole life-cycle. A comprehensive LCA of the urban wastewater system in France reports significant environmental impacts from the construction of sewer infrastructure (Risch, Gutierrez, Roux, Boutin, & Corominas, 2015). Other LCA studies show considerable GHG emissions from the development of water infrastructure (pipelines or treatment facilities) (Petit-Boix et al., 2014; Piratla, Ariaratnam, & Cohen, 2011; Venkatesh, Hammervold, & Brattebø, 2009). In recent years, studies on China have reported energy use in water supply (Smith, Liu, & Chang, 2016; Smith, Liu, Liu, et al., 2016), CH₄ emissions from wastewater treatment (Jinhe Wang et al., 2011; X. Zhao et al., 2019), emissions from sludge treatment (B. Liu, Wei, Zhang, & Bi, 2013; C. Xu, Chen, & Hong, 2014) and LCA case studies for a municipal water treatment plant (WTP) or a municipal wastewater treatment plant (WWTP) (Y. Li, Luo, Huang, Wang, & Zhang, 2013; Q. H. Zhang, Wang, Xiong, Chen, & Cao, 2010). However, there is no comprehensive study that provides a system-level understanding of the GHG emissions associated with China's water infrastructure.

There is an emerging need to estimate non-CO₂ GHG emissions from the urban water cycle, mainly from wastewater treatment. There are several recent publications on measurement and technology-based estimations of CH₄ emissions (Cakir & Stenstrom, 2005; Foley, Yuan, & Lant, 2009; Guisasola, de Haas, Keller, & Yuan, 2008; Shahabadi, Yerushalmi, & Haghighat, 2009; Jinhe Wang et al., 2011) and N₂O emissions in the wastewater treatment processes (Foley, De Haas, Yuan, & Lant, 2010; Law, Ye, Pan, & Yuan, 2012). They estimate the potential generation of CH₄ in the wastewater collection system,

which is ignored under some GHG accounting protocols, and conclude that the dissolved CH₄ in a rising main is significant (Foley et al., 2009; Guisasola, Sharma, Keller, & Yuan, 2009). Treatment plants that achieve high levels of nitrogen removal are found to emit less N₂O, indicating that no compromise is required between high water quality and lower N₂O emissions (Law et al., 2012). However, another study suggests a trade-off between nutrient removal and GHG emissions in WWTPs (Soda et al., 2013). An extreme case is that N₂O emissions accounted for three quarters of a plant's carbon footprint in one WWTP in the Netherlands, according to 16 months of measurement (Daelman, van Voorthuizen, Van Dongen, Volcke, & Van Loosdrecht, 2013). These studies demonstrate that non-CO₂ emissions from the wastewater system are substantial, but there is considerable uncertainty and variety depending on the regional context and technologies used (Mannina et al., 2016).

China's urbanization has been driven by decades of rural-to-urban migration (K. H. Zhang & Song, 2003). Water resources have become one of the controlling factors in urbanization due to water scarcity and pollution (C. Bao & Fang, 2012). China's 'new-normal' urbanization plan makes it harder for residents to settle down in larger cities and is cautious when providing new land for urban expansion in mega-cities (X.-R. Wang, Hui, Choguill, & Jia, 2015). The urbanization level in China is estimated to reach ~70% in 2030 (Shi et al., 2012). More construction of urban water infrastructure and associated water services provision are expected during the continuing process of urbanization in China.

Urbanization significantly enhances GHG emissions associated with the water cycle (X. Wang et al., 2018). The sponge city initiative has greatly improved the city's capability of dealing with stormwater and water body pollution. A recent study for Shanghai showed that the life-cycle (30-year) GHG emissions of sponge city projects can be offset by those enhanced carbon sinks after 19 years (Lin et al., 2018). However, China's growth is disproportionate. In the urban water market, there is inequality among different types of cities (Tong et al., 2016). Generally speaking, smaller and less affluent cities have lower levels of wastewater treatment and water supply coverage. Water utilities in small towns

perform poorly in critical variables, including water supply reliability, water quality, and financial performance, as well as having inadequate wastewater treatment (Browder et al., 2007; Y. Jiang, 2015; Q. H. Zhang et al., 2016).

3. Drinking water systems

The process of supplying water typically involves four steps: (1) extracting water from the source, (2) treating water to the desired quality, (3) transferring water through an underground water distribution system, and (4) lifting water within a user's building. Scope 1 emissions within water supply include diesel use during sludge trucking and diesel use for backup generators, Scope 2 emissions result from grid electricity use, and Scope 3 emissions occur during the construction and manufacturing of infrastructure, equipment, and chemicals required during water supply and during the use of treated water after it arrives at the user.

Greenhouse gas emissions for water supply in China and a number of other countries, regions and cities are presented in Table 1 for comparison. Key factors determining the emissions intensity of water supply are (1) electricity use and (2) the energy mix used to generate electricity (Smith & Liu, 2019). The emissions intensity of water supply in China is higher than three areas listed in Table 1 – Oslo, Toronto and New Zealand – which is mainly due to China's high emission factor for electricity generation compared to Norway, Canada and New Zealand (Smith & Liu, 2019; Smith, Liu, & Chang, 2016). These three countries rely greatly on hydropower, wind power or nuclear power, whereas China relies mostly on coal (Smith & Liu, 2019; Smith, Liu, & Chang, 2016).

203 Shifting from emissions-intensive electricity sources is crucial to reducing emissions in China's water
 204 supply sector (Smith, Liu, & Chang, 2016) and China is in the process of becoming less coal-dependent
 205 (Yuan, Behrens, Tukker, & Rodrigues, 2018). The percentage of non-fossil fuel energy in primary
 206 energy consumption increased from 3.4% in 1978 to 14.3% in 2018. The percentage of coal
 207 consumption decreased from 70.7% in 1978 to 59.0% in 2018, in particular due to the increase in the
 208 share of renewables over the last 10 years (Zheng et al., 2020).

209 Table 1 Total (GHG_{water}), per capita ($GHG_{per\ cap}$) and per unit ($GHG_{per\ unit}$) GHG emissions for urban
 210 water supply, GHG emissions for urban water supply as a percentage of total national GHG emissions
 211 for electricity/heat ($GHG\%$), and emission factor (EF) for countries, regions and cities represented in the
 212 literature. Table source: Smith, Liu, and Chang (2016) © 2015 by Yale University.

Region	$GHG_{per\ cap}$	$GHG_{per\ unit}$	GHG_{water}	$GHG\%$	EF
Author and Date	(kg CO ₂ -eq/cap/yr)	(kg CO ₂ -eq/m ³)	(Mt CO ₂ -eq)	(%)	(kg CO ₂ -eq/kWh)
China	24.5	0.213	7.63	0.18 ^a	0.737
Smith et al. 2016					
India				3-16% 16 cities	
Miller et al. 2013				<3% for 23 cities ^b	
Australia	92.1 ^c	0.721 ^c	1.11 ^c	0.46 ^d	0.908 ^e
Kenway et al. 2008					
UK		0.34 ^f	2 ^g	0.82 ^h	
DEFRA 2008					
Water UK 2010					
USAⁱ	76.9 ^j	0.311 ^j	18.6 ^j	0.69 ^k	0.608 ^l
Hutson 2004					
EPRI 2002					
Selected areas^m	7.32 ⁿ	0.0596 ⁿ			0.209 ^o
New Zealand					
Kneppers et al. 2009					
California		1.093 ^p			
Stokes and Horvath 2009					

Toronto, Canada	23.5 ^g	0.117	0.173 ^r
Racoviceanu et al. 2007			
Oslo, Norway		0.07-0.08	
Venkatesh & Brattebo 2011			
Durban, South Africa		0.481 ^s	
Friedrich et al. 2009			

^aUsing China's 2011 CO₂ emissions for electricity/heat from WRI (2014).

^bIncludes wastewater and is given as % of city-wide emissions.

^cCalculated based on total emissions, energy for water supply and population served for 2006/2007 table 1 p6 Kenway et al. (2008).

^dCalculated using Kenway et al. (2008) and average CO₂ emissions for electricity/heat for Australia between 2006/2007 from WRI (2014).

^e∑ GHG emissions for energy-related sources across 6 cities/ ∑ total energy demands from table 1 p6 Kenway et al. (2008).

^fSourced from Water UK (2010).

^gFor yr 2005-2006. Read from figure 12 p68 DEFRA (2008) (get, treat, distribute water + leakage).

^hCalculated using DEFRA (2008) and average CO₂ emissions for electricity/heat for the UK between 2005/2006 from WRI (2014).

ⁱEIA GHG emissions data are only available in terms of CO₂ not CO₂-eq.

^jCalculated using estimated water withdrawal for and population served by "public water supply" in 2000 (table 5 p14 Hutson (2004)), total electricity for "public water supply" in 2000 (table 1.2 p1.5 EPRI (2002)) and emissions factor 0.608 kg CO₂-eq/kWh.

"Public supply" refers to water withdrawn by public and private water suppliers that furnish water to at least 25 people or have a minimum of 15 connections.

Hutson (2004, 13).

^kCalculated using 18.6 Mt and total 2000 CO₂ emissions for electricity/heat from WRI (2014).

^lCalculated using electricity generation (3.8x10¹² kWh: EIA (2013)) and CO₂ emissions (2.3x10⁹t: EIA (2012)) from the US electric power sector.

^mKneppers et al. (2009) reviews four water systems with different characteristics representing just over 8% of the national population.

ⁿCalculation based on table 7 p20 Kneppers et al.(2009).

^o∑ GHG emissions/ ∑ total energy from table 7 p20 Kneppers et al. (2009).

^pPrediction: Assumes California imports all water

^q136 kWh/cap*0.173 kg CO₂-eq/kWh from Racoviceanu et al. (2007).

^rUnits converted from 48.1 t CO₂-eq/TJ for electricity generated for Ontario in 1997 from Racoviceanu et al. (2007).

^sFull LCA most likely used, including dam emissions.

213

214 3.1 Scope 1 & 2 GHG emissions

215 This section focuses primarily on Scope 2 emissions from grid electricity use, which are the main type of
216 emissions associated with water supply in China and for which data are more accessible. Drinking water
217 treatment plants do produce some Scope 1 fugitive emissions due to sludge from sedimentation tanks
218 and use of ozonation for disinfection, but this is difficult to quantify and low compared to wastewater
219 treatment plants. Most GHG emissions from drinking water treatment plants are from the use of grid
220 electricity (Smith, Liu, & Chang, 2016). Continuing reliance on coal for electricity generation in China

means Scope 2 emissions remain important, though the energy mix in China is expected to be greener over time (Q. Liu, Lei, Xu, & Yuan, 2018).

3.1.1 Emissions from sourcing water

Water sourcing is not the leading emitter for most urban water supply systems. Surface water accounts for more than 80% of freshwater withdrawals in China (Food and Agriculture Organization, 2019), and surface water extraction tends not to be emissions-intensive unless it is long-distance (Wakeel, Chen, Hayat, Alsaedi, & Ahmad, 2016). Long-distance transfer and agricultural water use are not within the scope of this study. One major southern city, Changzhou, is an example of the contribution sourcing surface water can make to overall energy use for water supply when water users are located near the source. Only 3% of energy for water supply in Changzhou is used for locally sourcing water (Y. Zhou, Zhang, Wang, & Bi, 2013). Li et al. (2016) estimate the energy intensity of water lifting from rivers and lakes to be 0.20 kWh/m³ based on a summary of the literature from China. Emissions are generally higher for sourcing groundwater (Rothausen & Conway, 2011). Energy use varies greatly based on the depth from which groundwater must be sourced (Smith & Liu, 2019), but the average energy intensity across provinces of China (including rural areas) is estimated to be 0.37 kWh/m³ (X. Li et al., 2016).

Emissions associated with sourcing water are mainly for pumping and fall into Scope 1 if pumps are run using diesel or Scope 2 if pumps are run using grid electricity. Diesel pumps are used for around 25% of groundwater wells in rural areas of China (Jinxia Wang et al., 2012) and are rarer in urban areas, although they may be used in cases where private bores fall within city boundaries. The difference between diesel and electric pumps is largely due to (1) pump efficiency and (2) emission factor, with

both higher for electric pumps. In China, the efficiency of diesel pumps (~15%) is estimated to be much lower than that of electric pumps (40%, less 15% due to transmission and distribution of electricity) (Jinxia Wang et al., 2012). The emission factor of diesel was around one-third of the emission factor of electricity generated in China at that time.

China's move towards the Sponge City concept has increased the focus on the collection and management of rainwater as an opportunity to supplement other water sources (H. Wang, Mei, Liu, & Shao, 2018). Research in other countries suggests that energy use for rainwater harvesting in buildings tends to be higher than for centralized water supply because rainwater is often stored in basement-level tanks and pumped up to users (Vieira, Beal, Ghisi, & Stewart, 2014). A comprehensive review of energy intensity data for rainwater harvesting systems around the world by Vieira et al. (2014) finds these systems use three times more energy than a centralized water supply system. However, for people living in high-rise buildings that already need onsite pumping (just under 10% of China's population), rainwater harvesting may not increase emissions associated with water supply (Smith & Liu, 2019). This could be an interesting area for comparative research in the future.

3.1.2 Emissions from water treatment

Conventional treatment of groundwater and locally sourced or transferred surface water is not a major source of emissions. In China, treatment generally involves coagulation, sedimentation, filtration, and disinfection, which are processes that do not require much energy (Can et al., 2013; X. Zhang et al., 2016). For example, Changzhou uses 1% of the total energy for centralized water supply on water treatment (Y. Zhou et al., 2013). Energy use for conventional water treatment processes in China can be found in Smith, Liu, Liu, and Guo (2018). The primary source of emissions is filtration due to electricity

use for filter scouring and backwashing. Disinfection can also be a high energy user, but chlorine disinfection is most commonly used in China and is a low-energy process (Ye, Wang, Yang, Wei, & E, 2009). Some big cities have implemented more advanced pre-treatment or post-treatment processes, such as ozonation, activated carbon filtration, etc. These processes will increase overall energy consumption, but limited data points are available so far from China. According to a reference from the United States, water chlorination/de-chlorination consumes 2×10^{-5} – 5×10^{-4} kWh/m³, while energy use for ozonation is between 0.03 and 0.15 kWh/m³ (Plappally & Lienhard V, 2012). We do not find any first-hand information in the refereed literature about electricity intensity or carbon intensity of drinking water treatment from China's operating water treatment plants. All the current studies in China are based on aggregated energy use data from statistics. The global median values of energy use for surface water and ground water treatment are found to be 0.22 and 0.25 kWh/m³, respectively (Y. Liu et al., 2016). China's aggregated energy intensity of water treatment is between the global median and the 75th percentile.

Seawater desalination for water supply emits more GHGs than conventional drinking water treatment. The two main forms of desalination in China are reverse osmosis (RO) and multi-effect distillation (MED), the former of which accounts for around 70% of the total desalination capacity (State Oceanic Administration, 2018). Other technologies such as multi-stage flash (MSF) and electrodialysis (ED) are less common in China. Emissions for RO are due to the use of electricity to create high pressure, with the exact amount depending on the salt concentration of the feed. As shown in Table 2, electricity use for RO is influenced by the age and size of the plant and can range between 3.3 and 5.2 kWh/m³ in China (Smith, Liu, Liu, et al., 2018). MED involves heating seawater to form water vapor, which condenses to form pure water. Energy use is independent of salt concentration, but MED is more energy-intensive than RO (6-7 kWh/m³ vs 4.5 kWh/m³) (J. P. Chen, Wang, Yang, & Zheng, 2011; Smith, Liu, Liu, et al., 2018). Furthermore, this technology uses both electricity and thermal energy. GHG emissions are high because thermal energy from burning coal is the main energy source; most low-temperature MED plants

in China are co-located with coal-fired power stations to utilize waste heat for operation. By comparison, RO only uses grid electricity, which is generated using a combination of fossil fuels and renewable sources (e.g., hydro) in China. In 2016, Shandong Province had roughly 1.6 times larger seawater desalination capacity than Hebei Province, but its GHG emissions from seawater desalination are only 47.4% of GHG emissions from Hebei's desalination plants. This difference is because Shandong Province only uses RO plants, whereas Hebei Province depends largely on MED plants (65% of its total desalination capacity) (Jia, Klemeš, Varbanov, & Wan Alwi, 2019).

Table 2 Electricity use by seawater reverse osmosis (SWRO) and low-temperature multi-effect distillation (LT-MED) plants in China. Table source: Smith, Liu, Liu, et al. (2018) © 2018 Elsevier Ltd.

Plant name	Technique	Capacity (m ³ /day)	Direct electricity use (kWh/m ³)	Steam use (kJ/m ³)	Electrical equivalent for thermal energy (kWh/m ³) ^d	Total equivalent energy use (kWh/m ³)
Hebei Caofeidian ^{a,b}	LT-MED	5.0x10 ⁴	2.11	2.69x10 ⁵	5.85	7.96
Tianjin Beijiang ^a	LT-MED	1.0x10 ⁵	1.45	2.51x10 ⁵	5.46	6.91
Qingdao Baifa ^a	SWRO	1.0x10 ⁵	4.40	n/a	n/a	4.40
Shandong Changdao County ^c	SWRO	1000	4.5	n/a	n/a	4.5

^aY. Li, Xiong, Zhang, Wang, and Wang (2016); ^bValues for this plant include transfer to Beijing (270 km), which increases energy use; ^cL. Zhang, Xie, Chen, and Gao (2005) ^dElectrical equivalent is the electrical energy that cannot be produced in a turbine because the given quantity of steam was extracted (Encyclopedia of Desalination and Water Resources, 2017). We use the conversion provided by Encyclopedia of Desalination and Water Resources (2017), which estimates 46 000 kJ of steam produces 1 kWh of electricity, and assumes that pressure in the condenser of a large commercial steam turbine is kept at 0.1 bara, seawater temperature is 35°C and steam extraction pressure is 0.5 bara (loss is 258 kJ/kg steam).

China has a common practice of wastewater reclamation (Z. Chen, Wu, Wu, & Hu, 2016) but uses a comparatively small amount of desalinated seawater, although the use of both types of water has increased significantly over the past two decades. Tianjin is one of the main producers of desalinated water, but desalination only accounts for 1.2% of water supplied to Tianjin; by comparison, reclaimed wastewater accounts for between 5% and 20% or more in major northern Chinese cities (Y. Li et al., 2016). Note that reclaimed wastewater is not yet for potable use in China's cities, but mainly for landscape and industrial use (Zhu & Dou, 2018).

308

309 **3.1.3 Emissions for water distribution**

310 Water distribution is often the main source of emissions in the water supply process, particularly if the
311 distribution of water in high-rise buildings is considered (Smith, Liu, Liu, et al., 2018). The topography
312 of a city and the percentage of people living in high-rise buildings are two of the main factors
313 determining energy use for water distribution (Q. Zhang et al., 2019).

314

315 There are only a few city-level studies in China that show the importance of water distribution in energy-
316 related emissions. In Beijing, water distribution accounts for 63% of total emissions of centralized water
317 supply (Smith, Liu, Liu, Liu, & Wu, 2017). Another study by He et al. (2019) estimates the energy use
318 for water distribution in Beijing to be 0.44 kWh/m³, which is 54% of the total energy use for sourcing,
319 treatment and centralized distribution. In Changzhou, centralized water distribution accounts for 96% (Y.
320 Zhou et al., 2013), and it accounts for 44%, according to one case study in Taipei (Cheng, 2002).

321

322 Pressure management is a countermeasure against water leakage during distribution, which has clear co-
323 benefits of energy savings and GHG reductions (Q. Xu, Chen, Ma, Blanckaert, & Wan, 2014; Y. Zhou et
324 al., 2013). Energy is used during distribution to meet water quality objectives, to provide sufficient
325 pressure to satisfy consumers, and ensure fire hydrants are functional. Many countries recommend a
326 minimum pressure be used in the distribution system for these reasons, and a selection of these
327 regulations and guidelines is shown in Figure 2. For example, the minimum recommended pressure is 28
328 m in China, which delivers to the sixth story and under. Other examples are 15 m for Australia and 10 m
329 for Spain.

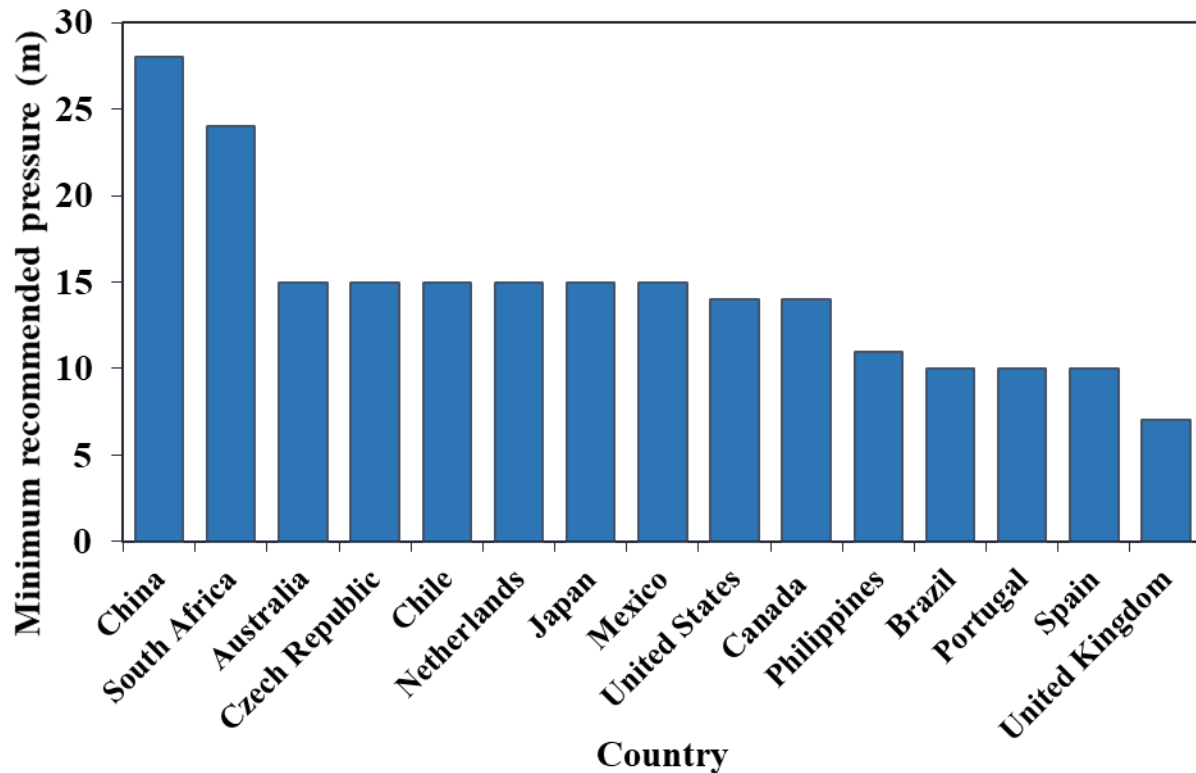


Figure 2 Examples of minimum pressure guidelines for countries around the world. Data source: Smith and Liu (2021).

Onsite pumping for households not serviced by the minimum pressure can be a significant contributor to emissions for water supply, but this can be reduced by using an efficient pumping system. As an example, energy use for high-rise pumping is estimated to account for 43% of total electricity use for water supply in a major Chinese city where 9% of the population live on the seventh story or above (Smith et al., 2017). The most common high-rise pumping systems in China are the booster pump and break tank system and the entirely pressurized booster system. Both have basement tanks to store water, and variable speed pumps to lift water to upper levels, but the tanks are different – one is a large break tank that holds water at atmospheric pressure, and the other is a small pressurized tank. The former system uses 0.019 kWh to lift one cubic meter of water one meter, and the latter arrangement uses less energy – 0.010 kWh/m³·m – because pressure provided in the water distribution system is retained (Smith et al., 2017).

3.2 Scope 3 GHG emissions

Scope 3 emissions data for China's water supply are challenging to obtain and do not contribute significantly to overall emissions according to both bottom-up and top-down estimations.

3.2.1 Evidence from bottom-up estimations

We found the only city-level study in China that included Scope 3 greenhouse gas emissions associated with water supply was a study comparing the carbon footprint of different water supply infrastructure options in Ningbo (Wu, Mao, & Zeng, 2015). The study investigates four potential sites for a water treatment facility. All four sites would use ferrous chemicals for coagulation, and polymer chemicals for sludge dewatering. Three sites would use chlorine disinfection, and the fourth would use ozone disinfection. Scope 3 emissions included in Wu et al. (2015) were those associated with chemicals and fuels. Although Wu et al. (2015) showed that the incorporation of fuel-related Scope 3 emissions is useful for assessing the impact of a plant's location on overall emissions, other factors were more influential in deciding plant location (e.g., elevation with respect to source and consumers). Scope 2 emissions from electricity consumption and Scope 1 emissions from ozone disinfection (in the case where this was used) were the main contributors to the carbon footprint of water supply infrastructure options.

From a life-cycle perspective, manufacturing and transporting chemicals used in drinking water treatment are responsible for some greenhouse gas emissions. It is difficult to find empirical data on this topic in China in the literature, so a brief review of literature for other countries is provided here. A study by Racoviceanu et al. (2007) on total energy use and GHG emissions for a conventional water treatment

plant in Toronto found that chemical manufacturing and transportation contributed 8% to total greenhouse emissions. Similarly, emissions related to chemical use in the Dutch water sector (including both drinking water and wastewater utilities) accounted for less than 8% of the total carbon footprint (Frijns, 2012). However, those studies did not include the chemicals and other materials used in pipeline construction and maintenance. Studies in Norway suggested that using polyurethane to replace epoxy resin as the material used for pipeline rehabilitation can significantly reduce GHG emissions and other environmental impacts (Venkatesh & Brattebø, 2012; Venkatesh, Brattebø, Sægrov, Behzadian, & Kapelan, 2017).

Another study by Stokes and Horvath (2011) on a conventional water treatment plant estimates 36% of GHG emissions can be attributed to materials, of which 73% is due to chemical production. Chemicals used were ammonia, polymer, caustic soda, hydrofluosilicic acid, sodium hypochlorite, polyaluminum chloride, sodium bisulfite, and alum. They compared this plant to results from two other plants, also in California, for which energy use contributed a much greater percent to GHG emissions and concluded that the large contribution of material production to overall GHG emissions (17% higher than one of the other case study plants) was due to (1) lower energy use by the plant in absolute terms and (2) more comprehensive inclusion of costs associated with constructing and maintaining the system. The same authors found in a previous study (Stokes & Horvath, 2009) on alternative water sources – imported water, desalination, and recycled water – that chemicals accounted for 4-18% of total energy consumption, and the contribution to GHG emissions was similar. Sodium hydroxide and sodium hypochlorite were the biggest contributors to energy consumption. Transport of chemicals accounted for 1% or less.

The contribution of chemicals and materials to life-cycle GHG emissions for the water sector depends on technology and treatment requirements, study scope, and geographic and social factors (Venkatesh, Chan, & Brattebø, 2014). Based on current evidence, we conclude that Scope 2 emissions from electricity use account for the majority of GHG emissions from China's drinking water system.

3.2.2 Evidence from top-down estimations

Top-down estimations of Scope 3 GHG emissions associated with the urban water supply system can be significant. Some upstream activities outside direct suppliers of water utilities are difficult to capture through a bottom-up approach. Using environmentally extended input-output models, Zhang et al. (2017) reveal that Scope 3 emissions contribute up to 17% of total carbon dioxide emissions associated with the operation and maintenance of water utilities, mainly from upstream electricity and heat generation (21%), water management (9%), manufacture of metal products (15%) and plastics (6%), and other chemical inputs (4%). Emissions from cement and lime production (27%) and steel rolling (22%) are the primary sources of Scope 3 emissions for the construction of urban water infrastructure (Q. Zhang et al., 2017). The dataset they used was the Chinese 2007 Input-Output Table with 135 industrial sectors, which provided China's most detailed sector information at that time. Similar insights can be found in a series of studies on the water-energy nexus at the city level (Fang & Chen, 2017; J. Gao et al., 2019; Xian Li et al., 2019). For example, Beijing's construction sector was found to be a high-intensive node for embodied water and carbon (Meng et al., 2019). Mining, mineral processes, and chemical manufacturing are the key sectors for the water-energy nexus in China (Fan, Kong, Zhang, & Wang, 2019).

4. Wastewater treatment systems

Wastewater treatment is one of the major sectors of GHG emissions under the Intergovernmental Panel on Climate Change (IPCC) guideline (Kampschreur, Temmink, Kleerebezem, Jetten, & van Loosdrecht, 2009; Mannina et al., 2018). Increasing concerns have been raised about the generation of GHGs from the rapid expansion of WWTPs in China (Zhao et al. 2019), where the volume of treated municipal wastewater increased ten times between 2001 and 2014 (Society of Chinese Urban Water Supply and Drainage, 2015). Three main GHGs can be generated due to wastewater treatment. These include CH₄ and N₂O released in the sewer systems, generated in wastewater treatment plants, and discharged into receiving water bodies (Scope 1 emissions), and CO₂ and other GHG emissions generated through energy and materials consumption (Scope 2 & 3 emissions).

4.1 Scope 1 GHG emissions from wastewater treatment systems

This section presents various sources of direct CH₄ and N₂O emissions from wastewater treatment systems, including sewer systems, urban WWTPs, sludge treatment/disposal, and receiving water bodies.

4.1.1 Nitrous oxide emissions from WWTPs

Nitrous oxide (N₂O) is produced in WWTPs during nitrogen removal processes, including both nitrification and denitrification reactions. In wastewater treatment processes, N₂O is emitted predominantly in aerated zones (Kampschreur et al., 2009). Globally, N₂O emissions from wastewater treatment were estimated to increase by 13% during the period 2005-2020 (Law et al., 2012). In China, the latest national GHG report showed that N₂O emissions from wastewater treatment were 110 gigagrams (Gg) in 2014, amounting to 5.6% of national N₂O emissions (National Development and Reform Commission, 2018). Measuring onsite N₂O emissions from full-scale WWTPs is complicated

and lacks standardized methods (Law et al., 2012). For example, real-time N₂O emissions at full-scale treatment plants have been found to vary between 0 and 14% of incoming N (Kampschreur et al., 2009).

Several case studies reported onsite monitoring of N₂O emissions in WWTPs. A case study in Beijing's WWTPs found that N₂O was the largest source of GHG among all GHG emissions (CH₄, N₂O, and indirect emissions of CO₂), accounting for 43.5% and 55.6% of total GHG emissions in anoxic/oxic (A/O) and sequencing batch reactor (SBR) WWTPs, respectively (Z. Bao, Sun, & Sun, 2016). Table 3 shows several monitoring results of N₂O emission factors from onsite measurement for the most popular treatment processes in China. In this table, SBR shows the highest emission factor, followed by oxidation ditch, anoxic/anaerobic/oxic (A₂O), and A/O. We found that the emission factors fall into a similar order of magnitude when compared with the results derived from several other countries. These studies provided valuable information on N₂O emissions from WWTPs but were based on limited sample sizes. More lab-scale and full-scale investigations are needed to increase understanding of N₂O emissions and reduction potential for different treatment processes.

Low dissolved oxygen (DO) concentration, increased nitrite concentrations during both nitrification and denitrification stages, a low COD/N ratio in the denitrification stage, as well as low pH levels were found to be the main factors leading to N₂O emissions in WWTPs (Kampschreur et al., 2009; S. Wang, Zhao, & Huang, 2019). Li et al. (2017) investigated N₂O emissions in a constructed wetland in Jinan city of China and found that the C/N ratio of influent wastewater significantly affected N₂O emissions. These findings indicate that considerable potential exists in reducing N₂O emissions in China. For example, Chen et al. (2014) found that the N₂O emissions were reduced by 42% through canceling the anaerobic phase and extending the idle phase in an SBR treatment process in Shanghai.

4.1.2 Nitrous oxide emissions from sewers and receiving water bodies

Recent studies have shown significant N₂O emitted from gravity sewer systems (Fries, Schiffman, Shuster, & Townsend-Small, 2018). Hydraulic turbulence during high flow periods is a strong driver for N₂O production from gravity sewer systems (Short et al., 2014). However, further case studies on N₂O emissions in sewer systems worldwide, including in China, are still sparse. In terms of receiving water bodies from WWTPs, case studies in both China and Canada have found that rivers close to urban WWTPs had significantly higher N₂O emissions than rivers receiving natural and agricultural runoff (Rosamond, Thuss, & Schiff, 2012; Jianing Wang, Chen, Yan, Wang, & Yang, 2015). Rosamond et al. (2012) indicated that dissolved inorganic nitrogen discharged from WWTPs has no linear relationship with N₂O emissions in rivers. He et al. (2017) further revealed that nitrogen substrates and DO could be good predictors of the N₂O emissions of urban rivers. They also suggested a revision of the default emission factors of N₂O in the IPCC approach in China's context.

Table 3 Emission factors of N₂O derived from onsite measurement for different wastewater treatment processes in China and some other countries.

Treatment process	Location	Emission factors	References	Scale
A ₂ O	Beijing	0.0195 (N ₂ O/TN influent)	(S. Sun et al., 2013)	Full-scale
A ₂ O	Beijing	0.4–1.5 (0.8) (g N ₂ O/kg TN removed)	(Yan, Li, & Liu, 2014)	Full-scale
A ₂ O	Jinan	1-1.3 (g N ₂ O/kg TN removed)	(Jinhe Wang et al., 2011)	Full-scale
A/O	Beijing	0.016±0.0089 (N ₂ O/TN influent)	(S. Sun et al., 2017)	Full-scale
Oxidation Ditch	Beijing	0.8–8.8 (3.6) (g N ₂ O/kg TN removed)	(Yan et al., 2014)	Full-scale
A ₂ O	Shanghai	0.095–3.44 (1.29 average) (gN ₂ O/kg TN influent)	(Y. Wang et al., 2016)	Full-scale

SBR	Beijing	6.52 (N ₂ O/TN influent)	(S. Sun et al., 2013)	Full-scale
SBR	Switzerland	1.0 (gN ₂ O/kg TN influent)	(Gruber et al., 2020)	Full-scale
A ₂ O	South Korea	1.605 (gN ₂ O/kg TN influent)	(Hwang, Bang, & Zoh, 2016)	Full-scale

4.1.3 Methane emissions from WWTPs

Methane (CH₄) is formed when organic matter decomposes in anaerobic conditions during wastewater or sludge treatment (Mannina et al., 2018). Although CH₄ is the second most important GHG from anthropogenic sources, there are fewer studies on CH₄ emissions in WWTPs than N₂O emissions (X. Zhao et al., 2019). Globally, CH₄ emissions from the wastewater treatment sector account for 5-7% of anthropogenic methane emissions (Saunio et al., 2020; US EPA, 2012). According to the Second Biennial Update Report on Climate Change of the People's Republic of China, CH₄ emissions from wastewater treatment were 2721 Gg in 2014, amounting to 4.9% of China's total CH₄ emissions.

Both plant-level and region-level studies can be found in CH₄ emission studies in China. Studies at a regional level are usually based on a mass balance approach derived from the *2006 IPCC Guidelines for National Greenhouse Gas Inventories* (IPCC, 2006). Researchers have evaluated CH₄ emissions at the regional level (Du et al., 2018; Ma et al., 2015) and city level (X. Zhao et al., 2019), and these estimations showed similar results to total national emissions (Table 4). Zhao et al. (2019) found that CH₄ emissions from municipal WWTPs for China's 229 prefectural-level cities were over three times that of the municipal WWTPs in the United States in 2016. Although the IPCC approach has considerable uncertainty (Law et al., 2012), it is still a suitable way to quantify CH₄ of WWTPs at a regional level compared to the onsite monitoring approach (Zhao et al. 2019).

Case studies of CH₄ emissions at the plant level for China are rare and not comparable with the results from regional studies. Specific studies found that GHG emissions from vertical subsurface flow constructed wetlands could be much lower than conventional WWTPs (Pan, Zhu, & Ye, 2011). Wang et al. (2011) investigated CH₄ emissions from an A₂O WWTP in China. However, based on the emission factor (kg CH₄/kg COD) derived from their studies, the scaled-up national CH₄ emissions were only 6.2 Gg, much smaller than results from the IPCC approach. Local GHG inventories require higher accuracy (Meyer, Deventer, Zhao, & Klemm, 2019). Thus, it is urgent to carry out more monitoring work with different treatment processes, scales, and geophysical locations to provide more accurate CH₄ emission inventories for municipalities. Besides, different sludge treatment and disposal techniques could be the predominant source of CH₄ generation. Wei et al. (2020) summarized five routes of sludge treatment and disposal, which are anaerobic digestion and land application (route 1), compost and land application (route 2), sanitary landfills (route 3), sludge incineration (route 4), and building materials production (route 5). They found that incineration contributed the most to CH₄ and CO₂ emissions (45.11%), followed by sanitary landfills (23.04%), land utilization (17.64%), and building materials (14.21%).

Table 4 China's methane emissions from WWTPs using the IPCC approach

Year	Municipal CH ₄ emissions Unit: Gg	Industrial CH ₄ emissions Unit: Gg	References
2010	611	1624	(Ma et al., 2015)
2000	245	1104	(Du et al., 2018)
2014	1271	2159	(Du et al., 2018)
2014	1170	-	(X. Zhao et al., 2019)

Many factors can influence the onsite emissions of CH₄. Methane emissions are directly correlated with the degradable organics present in wastewater, but they are also tightly related to the treatment conditions and processes. Zhao et al. (2019) found that cities in China with a higher gross domestic product (GDP), household food consumption expenditure, or household consumption expenditure produced wastewater with more degradable organics, thus generating more CH₄ emissions. Mechanical

aeration results in high DO concentrations in wastewater, which can inhibit methane formation. However, it also leads to the air stripping of dissolved methane from wastewater. As a result, DO concentrations are positively correlated with CH₄ flux from the aerated grit chambers and oxic tanks but not significantly relevant in anaerobic tanks (Jinhe Wang et al., 2011).

4.1.4 Methane emissions from sewers and receiving water bodies

The supersaturated CH₄ in rising mains of sewers can drive CH₄ emissions to the atmosphere. Monitoring results proved that a large amount of CH₄ emitted from sewers might be comparable to that from WWTPs (Guisasola et al., 2008). Higher CH₄ concentrations in sewers are highly related to long hydraulic residence time and larger pipe area-to-volume ratio (Foley et al., 2009). Three-year monitoring on 37 km-long sewer systems in Xi'an of China reports much higher onsite CH₄ emissions from the sewer system than previous estimations (Jin, Gu, Shi, & Yang, 2019), indicating the importance of examining CH₄ emissions from sewer systems in China. Researchers have found CH₄ dissolved in effluent increased the CH₄ emissions in rivers (Brigham et al., 2019). However, the volume of CH₄ emissions with discharged effluent is relatively insignificant, only accounting for a small share of the total CH₄ emitted in the drainage system (Alshboul, Encinas-Fernández, Hofmann, & Lorke, 2016).

One of the critical differences between China's urban wastewater systems and those of other countries is the inclusion in China of a pretreatment system (Liao, Hu, & Roker, 2015). The widely distributed septic tanks to pretreat household wastewater represent such a system, and these are huge sources of GHG emissions that are currently ignored. Hao et al. (2017) estimated that CH₄ emissions in China's urban septic tanks are at the same level as the GHG emissions from the WWTPs. It is important to include the pretreatment system in future studies to monitor and evaluate GHG emissions from wastewater treatment. Evaluating the cost and benefit of pretreatment system removal should consider GHG

generation potential in septic tanks and other pretreatment facilities, which is essential to mitigating GHG emissions further.

4.2 Scope 2 & 3 GHG emissions

The CO₂ emissions from energy consumption required for WWTPs operation are the major indirect source of GHG emissions from wastewater treatment (Daelman et al., 2013). From 2007 to 2015, the total amount of wastewater treated in China increased from 17.0 billion m³ to 46.7 billion m³, and the electricity consumed increased from 3.9 billion kWh to 14.0 billion kWh (Smith, Liu, Liu, et al., 2018). Among the multiple stages of the wastewater treatment process, secondary treatment and tertiary treatment (for wastewater reclamation) are the most energy-consuming stages, followed by pretreatment, primary treatment, and sludge treatment (Q. H. Zhang et al., 2010). A comprehensive investigation of energy consumption in China's WWTPs suggests that energy use of different secondary treatment technologies depends heavily on the capacity of the plant, which is defined as the volume of treated water per day (Y. He et al., 2019). The research found that SBR with energy consumption efficiency ranging from 0.128-0.424 kWh/m³ is more efficient for small-scale WWTPs (<10×10⁴ m³/d), oxidation ditch (0.126-0.434 kWh/m³) is more efficient for medium-scale WWTPs (10-20×10⁴ m³/d), and A₂O (A/O) (0.141-0.473) kWh/m³ is more efficient for large-scale WWTPs (>20×10⁴ m³/d) (Y. He et al., 2019). The energy use of tertiary treatment technologies for wastewater reclamation varies with the specific use of the reclaimed water and the associated water quality standards in China (Lyu, Chen, Zhang, Fan, & Jiao, 2016). In general, pressure-driven membrane processes, such as RO, nanofiltration (NF), and ultrafiltration (UF), are more energy-intensive than conventional processes such as coagulation, disinfection, and activated carbon adsorption (Plappally & Lienhard V, 2012). Based on data from 1079 WWTPs across China, Zeng et al. (2017) estimated that GHG emissions from WWTPs could decrease by 32.2% if all plants worked efficiently. Niu et al. (2019) established an energy consumption model for WWTPs in China using data from a national pollution source census (which

covers 3035 WWTPs supplying 85% of the total amount of wastewater treated in China in 2010). They found that WWTPs in China are more energy-intensive than their international counterparts. Low inlet COD concentrations and low utilization rates are the root causes of the abnormally high energy consumption of small-scale WWTPs in China (Niu et al., 2019).

GHG emissions can be induced by energy embodied in the upstream construction, production of input materials for operation, or downstream demolition phase of WWTPs (Beibei Zhang, Ariaratnam, Huang, & Zhang, 2014). Indirect emissions from WWTPs can also be substantial. For example, Bao et al. (2016) compared the direct and indirect GHG emissions of two WWTPs with A/O and SBR treatment processes in Beijing, China. They found that indirect GHG emissions due to the consumption of electricity and chemicals (flocculants) accounted for 50.8% and 30.2% in A/O and SBR WWTPs, respectively. On a national scale, up to 3.3×10^6 Gg of construction materials may be needed to build up urban water and wastewater infrastructure to meet the increasing demand by 2050 (T. Wang, Shi, Zhang, Qian, & Hashimoto, 2018).

5. Challenges of accounting life-cycle GHG emissions from the water sector

Based on our in-depth review, the hotspots of China's current research related to GHG emissions from the water sector are qualitatively summarized in Table 5. The sub-area with high-quality literature refers to multiple studies on similar topics found in China with comparable results or over the years. In contrast, the sub-area with low-quality literature means no or single study is found in China with large potential to improve. Although there are extensive studies from China focusing on energy use for drinking water systems and non-CO₂ emissions from wastewater treatment, some challenges – either to methods or data availability – are identified in this review.

589

Table 5 Research hotspots regarding GHG emissions from China's water sector

DRINKING WATER SYSTEMS			WASTEWATER SYSTEMS	
	Plant level	City/region level	Plant level	City/region level
GHG_{SCOPE 1}	Low	Medium	High	High
GHG_{SCOPE 2}	Low	High	High	Medium
GHG_{SCOPE 3}	Low	Low	Low	Low

590 Note: 1) High: Multiple studies on similar topics found in China with comparable results or over the years; 2)
 591 Medium: Sparse studies found with limited representativeness or implications for China; 3) Low: No study or only
 592 a single study found in China with large potential to improve.

593

594 5.1 Plant-level case study of the drinking water system in China

595 We fail to find any plant-level case study of a centralized drinking water system in China with
 596 transparent raw data of energy or chemical use. Only one study in Ningbo of China discusses the impact
 597 of GHG emissions on the planning choices of municipal water infrastructure (Wu et al., 2015). Although
 598 many studies on the energy-water nexus report GHG emissions using aggregated intensities or emission
 599 factors, aggregated data are less accurate than specific data when discussing technological optimization
 600 that could reduce a water utility's carbon footprint (Molinos-Senante & Sala-Garrido, 2017). Given
 601 China's context, it is very difficult to get public operational data for specific water treatment plants. This
 602 might be one reason for lacking case studies. However, we urge more of China's researchers in drinking
 603 water treatment to add the GHG mitigation lens as one of the functional objectives in their technological
 604 advancement. Many studies on wastewater systems in China estimate GHG emissions on a lab or full
 605 scale to advocate a carbon-neutral agenda (Qu et al., 2019), whereas researchers in drinking water
 606 treatment may feel less inclined to focus on this topic as most of the GHG emissions from mainstream
 607 treatment processes come from electricity use. However, some solutions discussed in Section 6.2 shed
 608 light on future studies for China to approach a low-carbon roadmap in the drinking water system.

609

Quantifying the impact of urban form transitions on GHG emission accounting in the water sector is another challenge. China's ongoing urbanization is a lasting factor in shaping the future water market. The implementation of urban policies, water policies, and energy policies in China's cities cannot be independent. Zhang et al. (2019) examine the water sector's complex response to Chinese urban policies taking resource efficiency into account. A hypothetical grid-city model was developed to connect technical parameters in urban water infrastructure systems to socioeconomic changes, such as population growth, housing blocks, and water end-use. The results show that pipeline construction dominates material use and locks in significant GHG emissions in the water sector. The most efficient urbanization scenario can reduce GHG emissions from water infrastructure by 60%, compared to urban sprawl with residency restriction in large cities. Zhao et al. (2020) introduces more technical parameters of pumping systems into model development and confirms that urban forms and population distribution significantly impact energy used by the water supply system. More plant-level case studies are expected to bring those insights into practice.

5.2 Scope 3 emission accounting

As shown in Table 5, few studies work on scope 3 emission accounting in drinking water and wastewater systems for China. Zhang et al. (2017) estimated these emissions for China between 2006 and 2012 using top-down approaches and aggregated emission factors. More studies from a comprehensive life-cycle assessment at the plant level are needed to validate those results. Lam and van der Hoek (2020) point out many water-related "wider opportunities" can also contribute to GHG emissions abatement for cities. They use Amsterdam as a case study to estimate marginal GHG abatement costs, including both in-utility and wider opportunities. Promising cost-effective opportunities to reduce Scope 3 emissions of the water sector include recovery, supply, and storage of thermal energy and the biogas upgrading process in drinking water treatment (Lam & van der Hoek, 2020). A proper Scope 3 emission accounting and accreditation system will encourage water utilities to extend their roles

beyond water and wastewater services providers.

5.3 Accounting for interactions in the wastewater system

Previous studies in China have separately monitored and quantified the GHG emissions for different components of wastewater systems, e.g., sewer system, urban WWTPs, sludge disposal system, and receiving water bodies. However, focusing on reducing one component of GHG emissions will not optimize the whole system (Mannina et al., 2018). There is still a lack of studies accounting for and comparing the GHG emissions from the whole system with different treatment processes, sewer, and river conditions at a city level. Also, the interaction mechanisms between different components need to be further investigated. The operation of an upstream component might increase the GHG emissions in downstream components. For example, the low ratio of C/N in the influent water from the sewer system will lead to a higher generation of N_2O during wastewater treatment (Liao et al., 2015). The contribution of WWTPs to N_2O emissions in the receiving water bodies is poorly understood, which might be the main cause of a large discrepancy between the global N_2O budget and atmospheric measurement (Rosamond et al., 2012). Specifically, the cumulative effect of wastewater discharge into the sediment of water bodies, which might lead to a large amount of N_2O emissions, still lacks investigation.

Moreover, some key issues need to be resolved for China to estimate methane emissions from wastewater systems accurately. First, the trade-offs between CH_4 and N_2O emissions need to be further investigated along with the whole drainage system. For example, nitrite can substantially inhibit CH_4 production (G. Jiang, Gutierrez, Sharma, & Yuan, 2010), but it might increase N_2O emissions (Kampschreur et al., 2009). Second, drainage system construction is also found to contribute significantly to GHG emissions (Nguyen et al., 2020; Q. Zhang et al., 2019). The newly developed urban drainage system should consider GHG emissions from both construction and rehabilitation phases in accordance with a life-cycle perspective.

6. Prospects on China's GHG reduction for urban water infrastructure

Water and wastewater utilities have made tremendous progress in increasing energy efficiency, recovering resources, and developing new integrated water management solutions. The water sector has significant potential to decarbonize, not only from a technological point of view but from practical, operational and policy-making aspects (Qu et al., 2019; Zodrow et al., 2017). New technologies have enabled utilities to reduce energy use and integrate with renewable energy, recover renewable energy and value-added products, and accomplish advanced treatment goals to improve efficiency and standards (Babovic, Babovic, & Mijic, 2018; Mihelcic et al., 2017). In the meantime, environmental policies and regulations have evolved to incorporate more measures to improve energy efficiency and reduce GHG emissions, which drives innovation and implementation (Rauch et al., 2017). Other factors include socioeconomic and behavioral changes toward sustainable living and circular economy, which provide the industry with broader opportunities to decarbonize and be more sustainable (Ren, 2019). While this study cannot cover all aspects of best practice, we highlight two promising pathways.

6.1 Wastewater energy and resource recovery

WWTPs are among the largest energy consumers in many cities and towns. Studies estimated that wastewater treatment could account for up to 3% of global electricity if treated with conventional technologies. On the other hand, the chemical energy embedded in wastewater is estimated to be more than 9 times that required to treat the wastewater (W.-W. Li, Yu, & Rittmann, 2015). Technologies such as anaerobic digestion (AD), anaerobic membrane bioreactors (AnMBR), and microbial electrochemical technology (MET) have been tested to replace energy-intensive aerobic treatment and recover renewable energy in the form of biogas, hydrogen, and electricity. Among them, biogas from AD has been implemented in many large scale WWTPs and the captured biogas is used for heat recovery combined

with electricity production or upgraded to vehicle fuel and household gas. According to a recent study, China may achieve a net energy production of 3.41 billion kWh annually from AD-based wastewater treatment processes (Huang, Lu, & Li, 2020). A recent review summarized different processes and highlighted alternative wastewater treatment pathways for CO₂ capture and utilization, making WWTPs a potential contributor to negative carbon emissions (Lu et al., 2018).

Biogas to replace fossil fuel use is the primary way in which utilities reduce emissions (Murray, Galik, & Vegh, 2017). Renewable energy generation can lead to long-term cost savings for plants. For example, Marselisborg WWTP in Denmark produces 40% more electricity than the plant requires through cogeneration without the need for extra organic waste (State of Green, 2020). Sweden and Norway are among the European countries that support the use of biogas for fuel through government funding and infrastructure, like biogas filling stations for public buses and biogas upgrading facilities. In Strasbourg of France, the environmental services provider, Suez, teamed with the city's natural gas distributor to inject biogas from wastewater into the gas network and replace natural gas use by 5000 households (Suez, 2020). The availability of European Union funding, existing gas demand and infrastructure, and the easing of regulations controlling gas injections into the network were vital enabling factors. In order to meet China's requirements to improve WWTP discharge standards and achieve sustainable water management, the adoption of advanced treatment and resource utilization from wastewater (e.g., water, organic matter, nutrients) are highly recommended (Y. Sun et al., 2016).

Heat energy recovery and biosolid management are other popular strategies to reduce GHG emissions and sustainably manage waste (H. Gao, Scherson, & Wells, 2014; Mihelcic et al., 2017). In 2010, Vancouver Olympic Village recovered heat from wastewater and heated the athletes' village during the Winter Games (Brenhouse, 2010). Scottish Water captures and returns heat from wastewater to

households before it enters the sewage network (Scottish Water, 2020). They are converting AD biosolids to fertilizers for land application, another beneficial way for GHG emission reduction and revenue generation. This practice can reduce chemical fertilizer use, reduce landfilling, condition soil, and potentially capture carbon (Iocoli, Zabaloy, Pasdevicelli, & Gómez, 2019; Möller & Müller, 2012), most of which are a Scope 3 source of GHG emissions.

In China, the use of biogas from anaerobic digestion has strong prospects. Anaerobic digestion is a highly developed technology that is only applied in a small percentage of Chinese WWTPs, although many Chinese WWTPs are large enough to make anaerobic digestion feasible (Smith, Liu, Liu, et al., 2018). Chinese plants could offset around 50% of their energy use by using the heat and power produced by burning biogas, and there are already numerous examples of plants using a combination of anaerobic digestion and co-generation (H. Li & Feng, 2018). There are successful examples of the sale of methane gas for use as fuel for vehicles; Xiangyang, in central China, is one such city that has achieved this (Fu, Zhong, Jagannathan, & Fang, 2017). However, extra purification is required, and the energy demands of this process need to be considered.

There is potential to recover heat energy from wastewater in northern China, and it is best recovered from wastewater treatment plant effluent (Xiaodi Hao, Li, van Loosdrecht, Jiang, & Liu, 2019). Winter effluent temperatures are 10-16°C in northern China, which is well above ambient air temperature. Heat could be used for heating buildings onsite or nearby, or for heating greenhouses to reduce emissions from burning coal or gas. It is already applied in a number of demonstration plants, but more investigation is needed to understand the potential for emissions reduction (Xiaodi Hao, Wang, et al., 2019).

Land application of anaerobic digestion biosolids is not a common sludge disposal method in China; it is only used to dispose of 2.4% of wastewater sludge produced (Smith, Liu, Hu, Dong, & Wen, 2018; Yang, Zhang, & Wang, 2015). Thus, there is an opportunity for China to increase land application, which would reduce GHG emissions by avoiding the need for synthetic nitrogen fertilizer production (H. Chen, Yan, Ye, Meng, & Zhu, 2012). This would require a review of policies controlling both the sale of sludge-based fertilizer and contaminant standards for land application of sludge.

6.2 Solar power penetration and energy efficiency

Solar panels are used on utility sites for power generation and evaporation control. SA Water – one of the leading utilities of Australia – is seeking to reduce net grid electricity cost to zero with the installation of solar panels and energy storage devices across 80 sites. The 'Zero Cost Energy Future' strategy is also expected to reduce CO₂ emissions by around 90,000 tonnes per year and will stabilize the cost of water for consumers (SA Water, 2020). Solar panels in the water sector can be partly motivated by secondary benefits, such as reduced water evaporation. Japan and Australia are both using floating solar panels in wastewater treatment plants, thus producing power while reducing extra land use, water evaporation, and algal growth. Floating panels can be more efficient due to the cooling effect of the water (Beca, 2015). Solar panels can be applied in China in conjunction with other emissions-reduction strategies listed in Section 6.1. When installed in wastewater treatment plants, they may only provide a small reduction in electricity use compared to the overall energy used by a plant (Xiaodi Hao, Liu, & Huang, 2015); but China is a major producer and user of solar panels for electricity production, so there is scope for offsetting at sites with large available area and solar resources.

Adhering to water quality and effluent standards is a priority for utilities that is not necessarily at odds with energy efficiency. Another Australian wastewater treatment plant replaced surface aerators with

fine bubble aeration to achieve similar effluent quality and less odor for half the energy cost (Abbott, 2020). Aeration is generally the main energy user in Chinese wastewater treatment plants (Smith, Liu, Liu, et al., 2018). The use of diffused aeration instead of surface aeration is an established way to reduce energy use because it provides more complete mixing of oxygen and wastewater. Implementation of diffused aeration in new activated sludge tanks should be a priority, but conversion of existing activated sludge tanks to diffused aeration is also possible.

7. Conclusions

Municipal water and wastewater services have complicated sources of greenhouse gas emissions. In this review, we present a systems perspective to source GHG emissions associated with urban water infrastructure. Based on China's empirical cases and socioeconomic contexts, we highlight the hotspots of GHG emission sources in drinking water systems and wastewater treatment systems, respectively. In drinking water services, electricity consumption is the largest source of GHG emissions. Alternative water sources (e.g., rainwater harvest, desalination) have different energy intensities, leading to varying implications for GHG emissions. Another particular concern in China is the common use of secondary pumping for high-rise buildings. Optimized pressure management with an efficient pumping system should be prioritized. In wastewater services, progress has been made to estimate non-CO₂ GHG emissions based on information collected from different WWTPs. These studies demonstrate that non-CO₂ emissions are substantial in GHG emission sources from the wastewater system, but considerable uncertainty and variety exists, depending on different regional contexts and technologies used. It is recommended that N₂O emission factors from the onsite measurement of different treatment processes in China replace default values in the IPCC approach. Evidence about Scope 3 emissions, especially those beyond the operations and maintenance stage, is still scarce. From a practical viewpoint, harnessing the co-benefits of energy and cost savings when applying new technology or carrying out maintenance can

be an effective way for water utilities to reduce GHG emissions. Some cases include renewable energy recovery, biosolid management, and onsite solar power installation.

GHG in water infrastructure is a new and evolving topic, and there is much work needed to advance the field:

1. A feasible systems accounting method for plant-level/city-level GHG inventories of the urban water cycle. This requires more data and higher accuracy, and onsite measurement is favored. Since LCA studies are based on functional units, a reasonable scaling-up approach is essential to compile a full-scale GHG inventory.

2. GHG reductions from water reclamation. China has reused massive amounts of water for non-potable purposes. However, there is no explicit study on the estimations of GHG reductions from water reclamation, especially from the scope of the urban water cycle.

3. GHG emissions from sewer systems. One recent study in China reported substantial progress on this topic, but only for one city in China. Geographic factors, temperature, precipitation, and the types of sewer pipes may cause emissions to vary significantly.

4. The dynamics of CH₄ and N₂O emissions. The trade-off mechanism between CH₄ and N₂O emissions in wastewater treatment processes needs to be further investigated. In an emission-factor-based GHG accounting method, linear and independent assumptions between different GHG sources are the most common. However, if the microbial community can simultaneously change CH₄ and N₂O production, the accounting method needs to be revised and validated accordingly.

5. Scope 3 emissions embodied in supply chains. Our knowledge about Scope 3 emissions associated with water infrastructure is still insufficient. In an emerging market economy, GHG emissions associated with the expansion of infrastructure can be significant. City-level accounting of Scope 3 emissions

probably faces a lack of both data and human resources. Quantifying Scope 3 emissions involves communication and engagement with stakeholders and the forging of a shared responsibility for climate change mitigation and other sustainability agendas.

6. Technology innovation and system integration. Many emerging technologies hold promise for GHG emission reduction and the recovery of value-added energy and products. Such system-level analyses are needed to identify opportunities and challenges and guide innovation and market translation.

Sidebar

S1 GHG inventory compilation and reporting methods

Anthropogenic greenhouse gas (GHG) emissions have been one of the principal reasons for global warming since the mid-20th century (IPCC, 2014). The *2006 IPCC Guidelines for National Greenhouse Gas Inventories* provide internationally agreed methodologies to estimate national GHG emissions (IPCC, 2006). To develop an inventory of GHG emissions, one should carefully consider types of greenhouse gases, emission sectors (or removal sinks), territory boundary, inventory year, and accounting methods to keep the constructed inventories comparable between countries with a satisfactory quality regarding transparency, completeness, and consistency.

The *GHG Protocol Corporate Accounting and Reporting Standard* initiated by the World Resources Institute (WRI) and the World Business Council for Sustainable Development (WBCSD) provides guidance on GHG accounting for companies or other organizations (WRI & WBCSD, 2004). It introduces the concept of the 'scope' of GHG emissions in inventory compilation, requiring both direct and indirect emissions to be reported in the GHG inventory using this standard. Scope 1 emissions refer to direct GHG emissions occurring from owned or controlled sources by the entity. Scope 2 emissions refer to indirect GHG emissions from purchased or otherwise acquired electricity and heat power consumed by the entity. Scope 3 emissions refer to all other indirect GHG emissions as a consequence of

the entity's activities but occurring from sources not owned or controlled by the entity. This concept of indirect emissions has been adopted in ISO standards (ISO 14064-1/2/3) (Bastianoni, Marchi, Caro, Casprini, & Pulselli, 2014) and is presented in the IPCC's fifth assessment report.

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