LIFE CYCLE ENVIRONMENTAL ASSESSMENT ON THE URBAN WASTEWATER SYSTEM CONSIDERING TECHNOLOGICAL OPTIONS AND SPATIAL STRATEGIES

by

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SYMBOLS AND ABBREVIATIONS

AAO: Anaerobic Anoxic Oxic

AD: Anaerobic Digestion

ANAMMOX: Anaerobic Ammonia Oxidation

AOB: Ammonia-Oxidizing Bacteria

AR: Assessment Report published by IPCC

BOD: Biochemical Oxygen Demand

CH_4: Methane

CO_2: Carbon Dioxide

CO_2-eq.: CO_2 Equivalent

COD: Chemical Oxygen Demand

CWWT: Centralized Wastewater Treatment System

DWWT: Decentralized Wastewater Treatment System

FU: Functional Unit

GHG: Greenhouse Gas

GWP: Global Warming Potential

HRT: Hydraulic Retention Time

IPCC: Intergovernmental Panel on Climate Change

LCA: Life Cycle Assessment

LCI: Life Cycle Inventory

MLVSS: Mixed Liquor Volatile Suspended Solid

N_2O: Nitrous Oxide

NPR: Non-potable Reuse

OD: Oxidation Ditch

PAC: Poly Aluminum Chloride

PAM: Polyacrylamide Flocculant

RWTPs: Reclaimed Water Treatment Plants

RWU: Reclaimed Water Use

SBR: Sequencing Batch Reactor

SDGs: Sustainable Development Goals

ST: Sludge Treatment

STPs: Sludge Treatment Plants

TN: Total Nitrogen

UWS: Urban Wastewater System

WWT: Wastewater Treatment

WWTPs: Wastewater Treatment Plants

ABSTRACT

Since the mid-19th century, when industrialization began, the Earth's temperature has risen and the trend towards global warming has continued. It is a widely acknowledged fact that the main cause of global climate change is the excessive emission of GHGs into the atmosphere from human activities, which are six gases including carbon dioxide (CO_2), nitrous oxide (N_2O) and methane (CH_4). This warming has caused glaciers at the poles to melt, resulting in a rise in sea levels and, indirectly, in abnormal rainstorms, droughts and increased desertification. Climate change is a common challenge for all mankind, and it has become a common task for all of us. Governments have made commitments to reduce emissions, and developing countries are under more pressure from social, economic, and environmental factors. The situation in China is that there are still deficiencies in many areas due to the late start, such as basic theoretical research, the establishment of a database of relevant information, public awareness and knowledge, and the development of engineering technologies.

As a result of urbanization, cities are growing larger and more people are living in them, consuming more resources (e.g., fresh water and energy) and producing more municipal waste (e.g., wastewater and solid waste). It will cause significant harm to the urban environment when municipal waste is not managed properly. The urban wastewater systems (UWS) in this study include waste treatment facilities (e.g., wastewater treatment plants and sludge treatment plants), transport systems (e.g., water supply pipelines, sewers, and sludge transport), and resource recycling systems (e.g., reclaimed water use and sludge recycling). It plays an essential role in achieving sustainable development by reducing environmental pollutants and increasing resource recycling. However, conventional wastewater and waste treatment processes consume large amounts of fresh water, energy, and chemicals as well as release GHGs to atmosphere, which are the main sources of GHG emissions. In developed countries, energy consumptions of the UWS occupied 3% of the total energy consumption in the society. The CO_2 emissions, N_2O emissions, and CH_4 emissions from the UWS account for 4%, 3%, and 5% of total emissions, respectively. In China, GHG emissions from the UWS contributed 1.3%-4% of total society's GHG emissions.

In the context of the global response to climate change, there has been a trend to convert WWTPs into energy supply plants, as the effluent is rich in renewable resources that can be recycled (e.g. heat, reclaimed water, nitrogen, phosphorus recovered from sludge). Practical experiences from countries include the Strass WWTP in Austria, NEWs in the Netherlands, NEWater in Singapore, and the New Concept WWTP in Yixing, China.

A systematic and comprehensive evaluation of the GHG emissions from the UWS is essential to work for its sustainability. However, there has not been a comprehensive evaluation system for quantitative GHG emissions from the UWS due to some reasons such as complex generation mechanisms, a wide range of technological options, and the many industries and sectors involved. On the other hand, centralized systems widely apply in cities because of the economic scale. However, in developing countries, urbanization can result in WWTPs require expansion after they begin operation, technically upgraded, and relocated further away from the city center. There are few previous studies on the balance between economics of scale and decentralization, quantifying the environmental impact of both and solving the problem of choosing between them, especially for the scenario of community-scale wastewater treatment integrating reclaimed water use.

Life Cycle Assessment (LCA) provides a theoretical framework for quantitative GHG emissions estimation and optimization of the UWS due to its advantages of systematization, quantification, standardization, and universality. Therefore, this study improved a GHG evaluation system with a basic framework of LCA for the UWS (see as Chapter 3), which takes wastewater as the study target. Its boundary starts when the wastewater enters the collection system (urban sewer) and ends when potential users use the wastewater-based reclaimed water, covering the sludge treatment and final disposal. The GHG estimation model developed in this study considers direct GHG emissions, building material consumption, chemical consumption, and energy consumption for transport, civil engineering, and process operations, as well as analyses the emission reduction potential of technical options (see Chapter 4) and spatial strategies (see Chapter 5).

In previous studies, technology selection mostly took an economic perspective to obtain the minimum economic cost; in addition, wastewater and sludge treatment technologies tended to choose separately, with few evaluations of municipal wastewater systems integrated from wastewater, sludge, and resource recycling systems. Chapter 4 analyzed nine alternative wastewater and sludge treatment scenarios to evaluate the GHG emissions from technological treatment options. The results show that:

1) Direct emissions and indirect emissions caused by electricity consumption are key contributors of GHG emissions from UWS.

2) Total GHG emissions in nine scenarios ranged from 58-127 kt CO_2-eq. per year. The SBR-Incineration scenario has an advantage in terms of low GHG emissions, while AAO-Composting is the scenario that results in maximum emissions.

3) The direct N_2O emissions and emissions caused by electricity consumption are the primary

GHG emissions sources, and the sum of the contributions of the two sources exceeds 70% in all scenarios. In addition, the results highlighted that not considering direct fossil CO_2 emissions may cause deviations in the estimation of GHG emissions.

Previous studies of environmental assessment of WWTPs with different implementation scales have mainly focused on energy consumption to examine the correlation between energy intensity and implementation scale. This paper compares the environmental loads of an urban community-scale WWT integrating RW use in the case of two spatial strategies (decentralized system and centralized system). The CWWT is more environmentally and economically advantageous because of the scale effect; however, it is more expensive to build and maintain, and its planning involves more sectors of interest. The DWWT is another option for treating wastewater and reusing effluent of the communities, with the advantage of not requiring a pipeline system, even though it is generally not considered to have economies of scale. Chapter 5 analyses GHG emissions in a community-scale case under scenarios with different spatial strategies, where the wastewater generated requires treatment and implementation of on-site reuse; the results show that:

1) CWWT consumes only 20% of the electricity of DWWT in its operation phase but consumes 14 times more chemicals and 158 times more freshwater than DWWT.

2) Pipeline system supporting CWWT contributes 65% of total GHG emissions during the construction phase.

3) The critical distance (minimum distance for selecting DWWT) is 56 km when applying 300 mm internal diameter reinforced concrete pipes (RCP) and shortened in scenarios where thicker RCPs are used and replaced with prestressed concrete cylinder pipes.

Key words: *GHG emissions, LCA, LCI, Decentralized wastewater treatment, Community-scale reclaimed water facilities, Low carbon scenario analysis, Critical distance, Technological options, Spatial strategies*

1. INTRODUCTION

1.1 Background

1.1.1 Introduction of urban wastewater system (UWS)

The urban wastewater system (UWS) was defined as shown in Figure 1.1 in this study. It is an integrated municipal waste management system and an essential artificial component of the urban water cycle. It consists of three sub-systems, the wastewater system, the sludge system, and the resource recovery system. The wastewater treatment plants (WWTPs) and the sludge treatment plants (STPs) are connected to the three sub-systems by a pipeline network and road transport, serving as the urban infrastructure for water, solids, and energy conversion.

(1) Wastewater system

The wastewater system consists of collecting, transporting, and treating wastewater from households and industries to avoid pollution of the environment.

The collection and transport of wastewater (also called urban sewer system) is defined as transporting wastewater through the pipeline network and into a WWTP.

Treating wastewater involves removing pollutants (for domestic wastewater is nutrients such as organic matter, nitrogen, and phosphorus) from wastewater using physical, chemical, and biological methods. Generally, the treated effluent from WWTPs discharged into receiving water (such as rivers, lakes, and oceans); however, the effluent discharge can pollute the receiving waters even if it meets discharge standards. The WWTPs have to meet stricter discharge standards for reducing the environmental loads on the receiving waters, which means more energy consumption without breaking the technological bottleneck. The other option is flowing into a reclaimed water treatment plant (RWTPs) as an input source. The effluent from RWTPs can use for urban greening, river recharge, and landscape water supplement. Effluent reuse is a better strategy for managing municipal wastewater than direct discharge because of avoids polluting loads on the receiving waters and mitigates water resource scarcity.

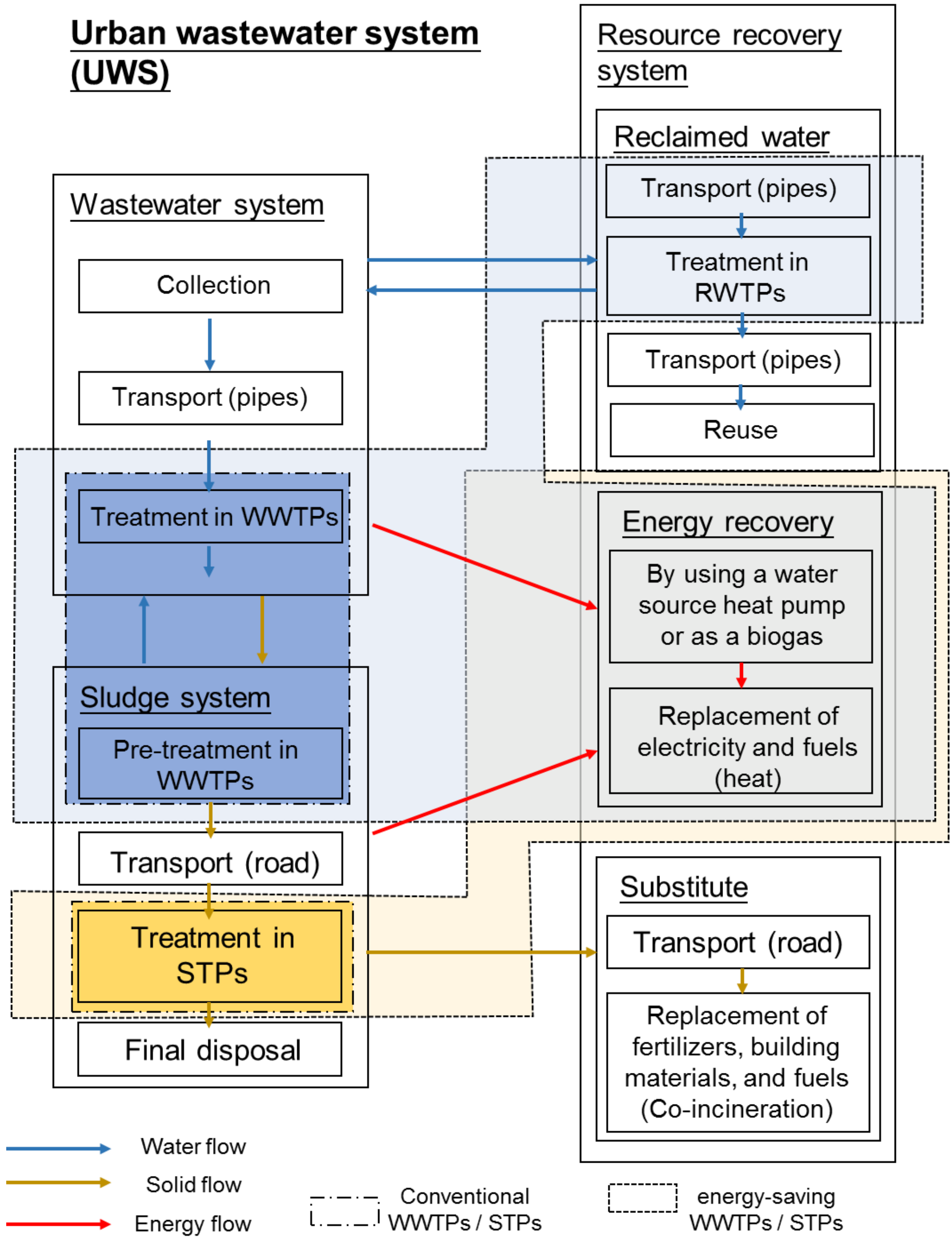
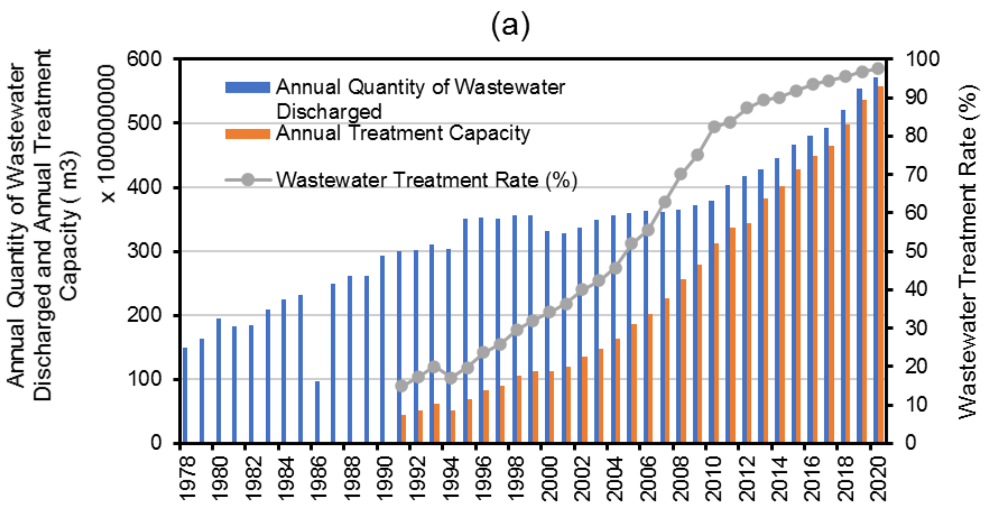
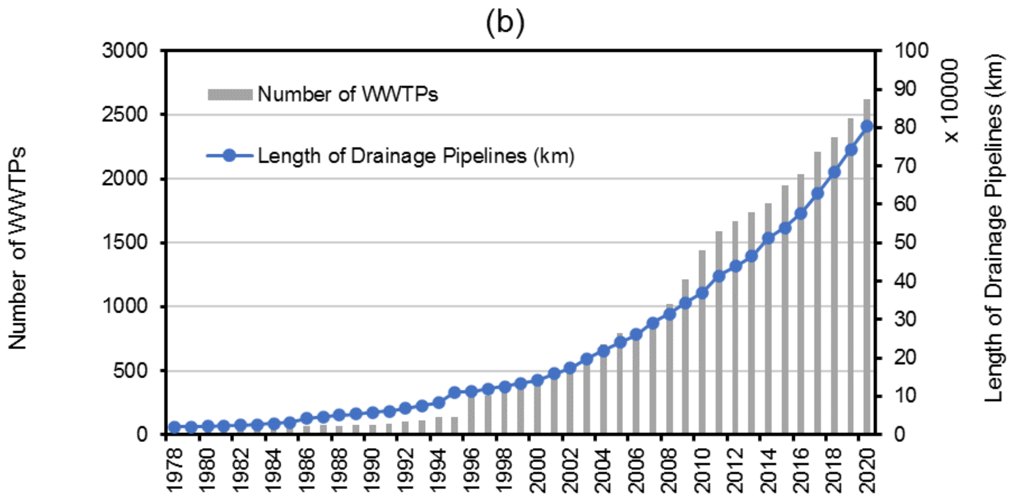


Figure 1.1 The urban wastewater system (UWS)





Data source: MOHURD (2011-2020)

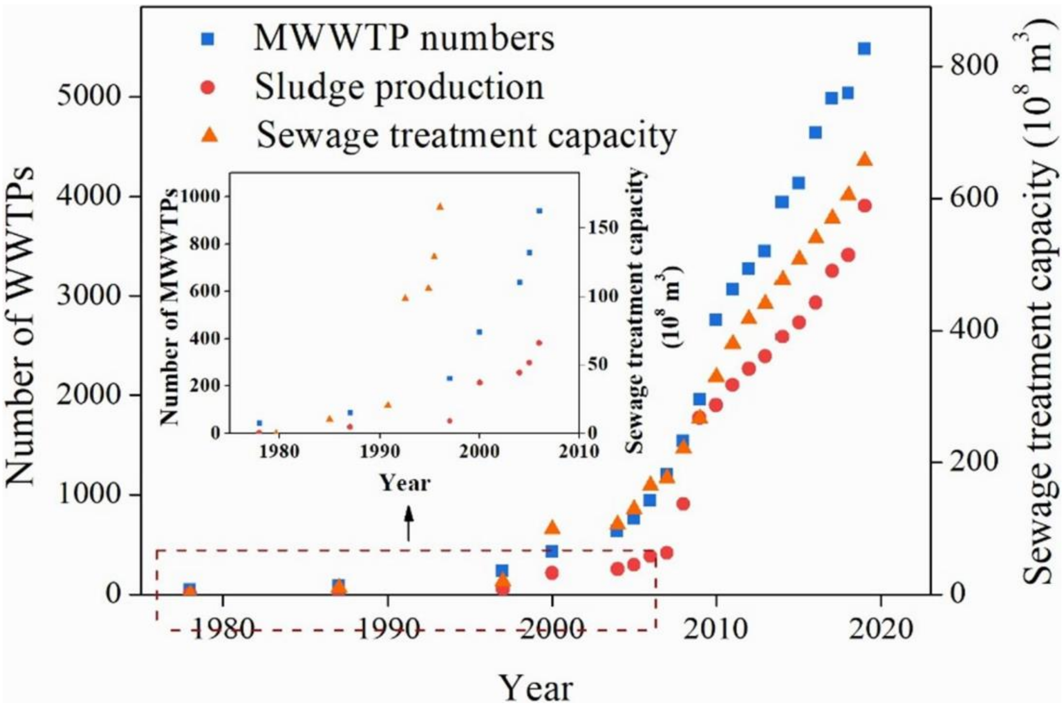
Figure 1.2 Current status of urban wastewater management (a) and infrastructure (WWTPs and sewer system) construction (b) in China during the period of 1978–2017

Figure 1.2 presents the rapid development of the wastewater treatment industry during the period 1978-2020, along with China's urbanization: Figure 1.2 (a) shows that the quantity of wastewater generated increased from 14.9 billion m3 per year in 1978 to 57.1 billion m3 per year in 2020, and the treatment rate increased from 15% (1991) to 98% (2020); Figure 1.2 (b) shows that there are 2,618 urban WWTPs with drainage pipelines of 803,000 km by 2020, compared to only 37 urban WWTPs with 20,000 km of drainage pipelines in 1978.

(2) Sludge system

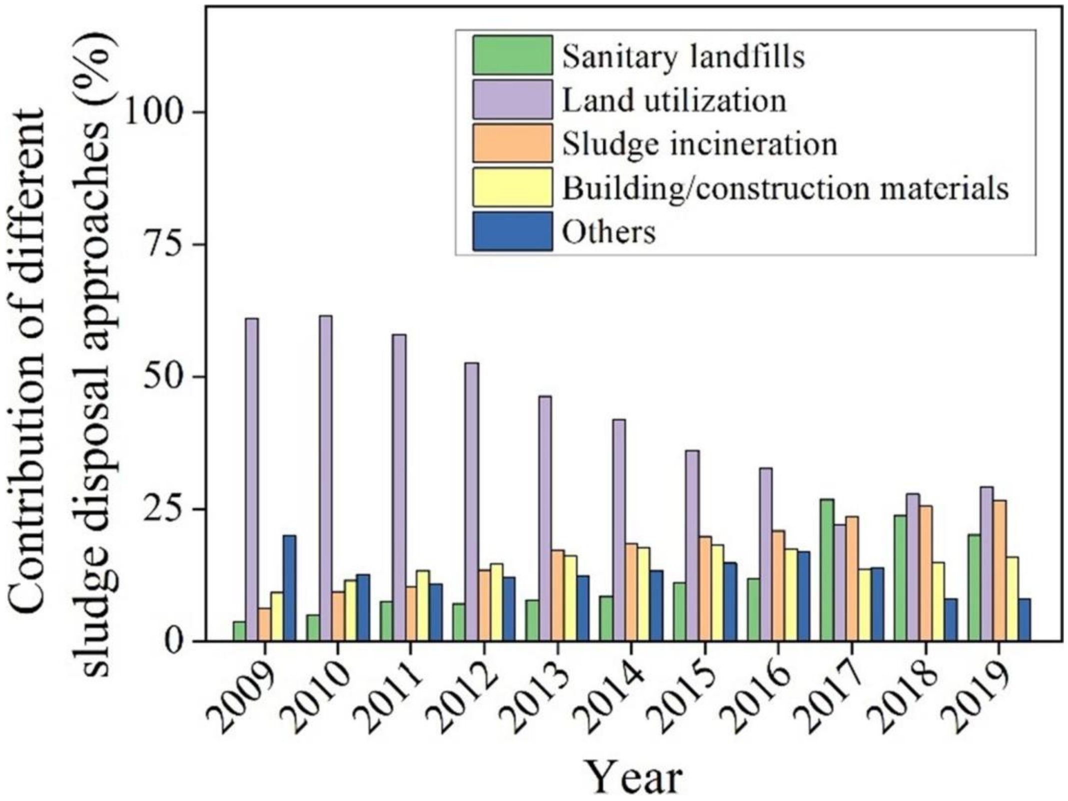
The sludge system contains collecting, transporting, and treating sludge. The sludge is a type of biomass waste and an inevitable product of the biological treatment process in WWTPs (Figure 1.1). Similar evidence presents in Figure 1.2, where the increasing trend of sludge production and the number of WWTPs are generally consistent. As is usually the case in China, raw sludge from WWTPs is pre-treated (concentrated and dewatered) on-site to a water content of less than 80% and then transported by truck or pipeline to the STPs for treatment (such as incineration, composting, and landfills) and final disposal.

The estimated production of sludge (80% water content) was approximately 0.25 million tons in 1987, and it increased to 2.13 million tons in 2000. Meanwhile, the annual average growth rate was 44.9% for 2005–2010 and 7.3% for 2010–2019 (Liangliang Wei et al., 2020).



Source: Liangliang Wei et al. (2020)

Figure 1.3 WWTPs number, wastewater treatment capacity and sludge production in China during the period of 1978–2019 (Sludge production (104 tons, 80% w.c.))



Source: Liangliang Wei et al. (2020)

Figure 1.4 Contribution of different sludge disposal routes in China during 2009–2019.

Figure 1.4 shows that China's main technological routes for sludge treatment and disposal include sanitary landfills, incineration, building material production, and land utilization (included directly and after composting as fertilizer). By the end of 2019, it is estimated that 29.3% (39.04 million tons, 80% w.c.) of the sludge disposed of via land utilization, followed by incineration (26.7%), sanitary landfills (20.1%), building material utilization (15.9%), and others (8.0%). (Liangliang Wei et al., 2020)

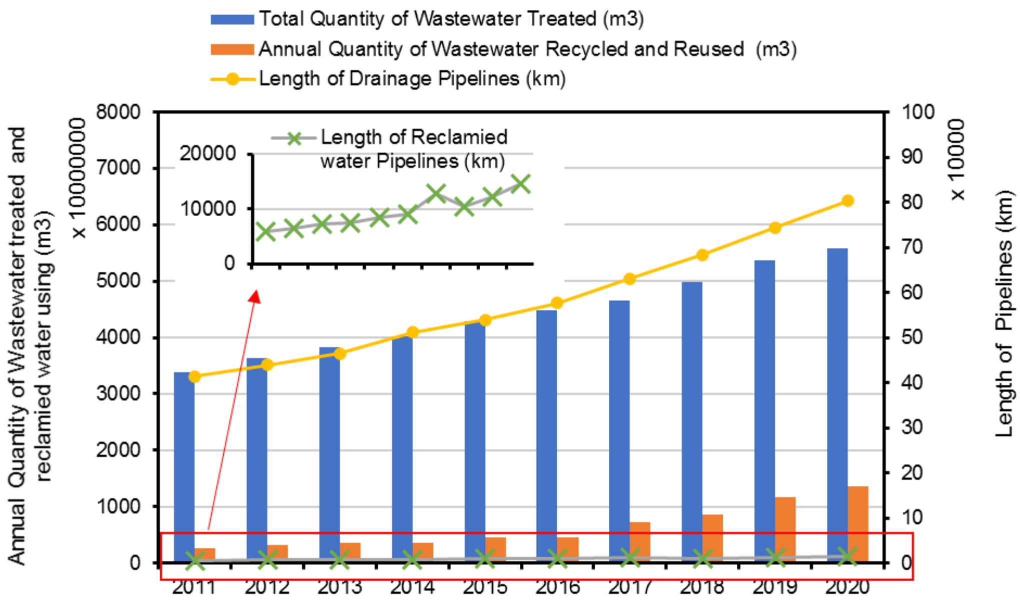
(3) Resource recovery system

It includes Non-potable Reuse (NPR), energy recovery, and sludge recycling as a substitute.

i) Urban wastewater mainly comes from a domestic source is a valuable reusable resource because it is rich in nutrients (carbon, nitrogen, and phosphorus). NPR of treated effluent applied for toilet flushing, agricultural irrigation, and river recharge.

Since the 1980s, the effluent from centralized WWTPs as reclaimed water has been applied to toilet flushing, river recharge, and green irrigation in some cities in Japan (e.g., Fukuoka City). However, reclaimed water use is still limited for various reasons. As of 2016, reclaimed water consumption (210 million m^3/year) represented only 1.3% of the total wastewater production. Only 8% of WWTPs have reclaimed water facilities in Japan (Haruka, T. et al., 2020). Reclaimed water use started late in China, and reclaimed water consumption (7.13 billion m^3/year) accounted for only 15% of the total wastewater production through 2017. The length of the reclaimed water pipeline is 13,000 km, which is only 2% of the total length of the wastewater pipeline (MOHURD, 2018). Recently, NDRC (2021) presented that a goal to increase the reclaimed water use ratio by more than 25%, up to 35% in the Beijing-Tianjin-Hebei region. Meanwhile, Tianjin Municipal Water Bureau (2020) published that the goal for Tianjin is more than 62% by 2030.

As shown in Figure 1.5, reclaimed water use increased from 268 million m^3/year in 2011 to 1,354 million m^3/year in 2020, and the length of reclaimed water pipes increased from 5,851 km in 2011 to 14,630 km in 2020. Compared to wastewater treatment infrastructure, recycled water infrastructure remains uncompleted. The length of reclaimed water is less than 2% (by 2020) of the sewer pipeline; meanwhile, the production capacity of reclaimed water is less than 25% (by 2020) of the wastewater treatment capacity.



Data source: MOHURD (2011-2020)

Figure 1.5 Reclaimed water use (m^3/a) and length of reclaimed water pipeline (km) vs. wastewater treatment (m^3/a) and length of sewer pipeline (km) in China during the period of 2011-2020

ii) Water source heat pumps can recover heat from the wastewater treatment process, and the methane gas (CH_4) produced during anaerobic digestion (AD) can also provide an alternative to fuel.

Table 1.1 Application of anaerobic digestion (AD) projects in new WWTPs in China during the period 1985-2014

	1985-1994	1995-2004	2005-2014	Total
New WWTPs	96	569	1099	1764
New AD	10	18	31	59
AD/WWTPs	10.4%	3.2%	2.8%	3.3%

Data source: Zhao Lejun (2016)

However, Table 1.1 shows that less than 5% of new WWTPs operated AD in China between 1985 and 2014. In addition, the proportion decreased from 10% in 1985-1994 to 3% in 2005-2014. The reasons for this may be that the low organic matter content of the sludge from WWTPs, high construction costs of AD projects, lack of attention from planners, and the high requirement for the operator with professional knowledge.

iii) The product of sludge recycling is a substitute for fertilizer from composting and for building materials from co-incineration.

Figure 1.4 shows that the proportion of sanitary landfills and sludge incineration increased rapidly between 2009 and 2019. The proportion of construction material and land utilization (alternative fertilizer) increased slowly or even decreased. The contribution of land utilization gradually decreased from 61% in 2009 to 22% in 2017 (Figure 1.4). However, the slight increase to 29.3% in 2019 results from the fact that both direct land utilization (random landfills) and land utilization after composting instead of fertilizer are included in the data of land utilization. The method of producing construction materials in China accounted for only 9% of disposed of sludge in 2009, gradually increasing to 15% in 2012 and reaching a maximum of 18% in 2015.

1.1.2 GHG emissions from the UWS

As shown in Figure 1.6, GHG from UWS can be classified as direct emissions, indirect emissions, and carbon offsets.

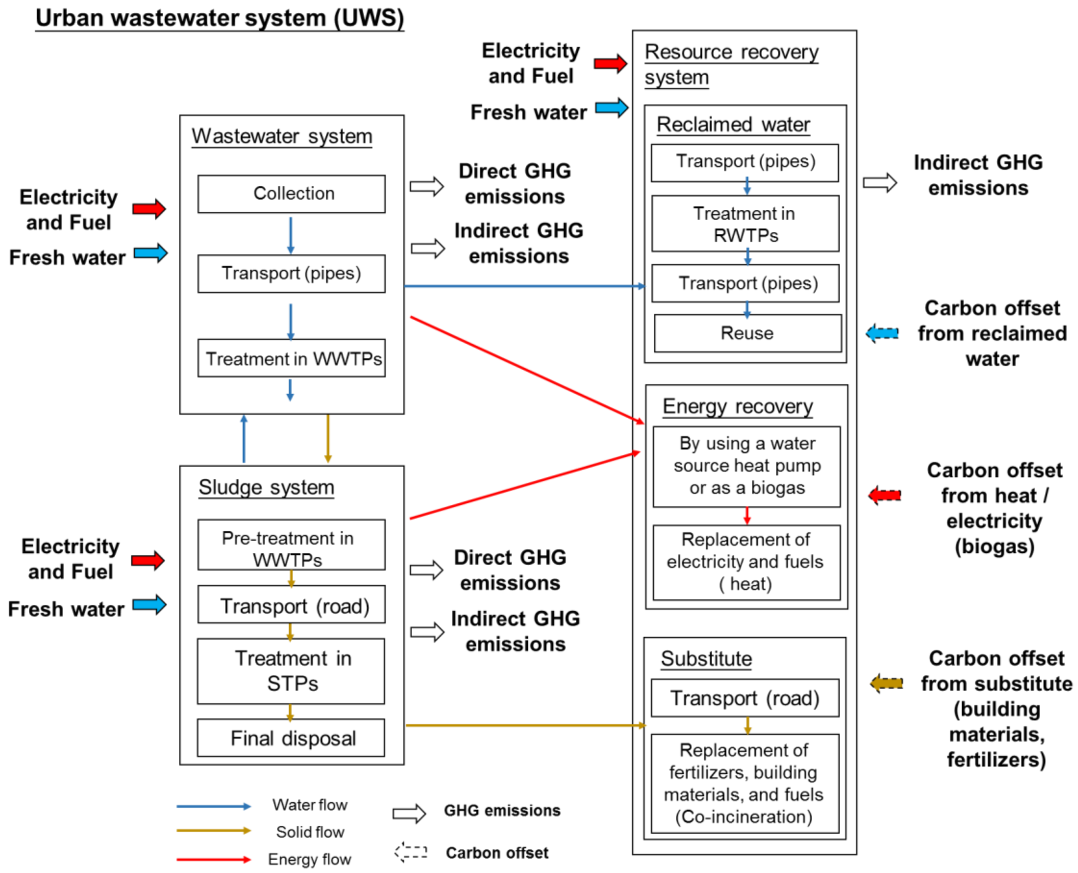


Figure 1.6 GHG emissions from the UWS

(1) Direct GHG emissions

Direct emissions come from wastewater treatment, sludge treatment, reclaimed water production, and composting. As shown in Table 1.2, the direct sources of GHG emissions from UWS are wastewater biological treatment, sludge treatment, and sludge disposal. The wastewater biological treatment process emits CO_2, CH_4, and N_2O; the sludge treatment process emits CH_4 and N_2O; and the sludge disposal process emits CH_4 and N_2O.

Table 1.2 Emissions sources and types of GHG from the UWS

			CO2	CH4	N2O
Direct emission					
	Wastewater system				
		Wastewater Treatment	*○	○	○
	Sludge system				
		Sludge Treatment			
		Incineration	—	—	○
		Composting (aerobic fermentation)/Aerobic Digestion	—	○	○
		Disposal			
		landfill (direct, after composting)	—	○	○
		Recycling (as fertilizers)	—	—	○
Indirect emission					
	Energy consumption of electricity and fuel				
		Purchased electricity	○	—	—
		Fuel consumption:			
		Gasoline, diesel, etc.	○	Δ	Δ
		LPG, LNG, etc.	○	Δ	Δ
		Transport:			
		Gasoline, diesel, etc.	○	Δ	Δ
	Freshwater consumption		○	—	—
	Chemicals consumption		○	—	—
Carbon offsets					
	Reclamied water use		○	—	—
	Substitutive fertilizer		○	—	—
	Biogas use		○	—	—

○ Included in the GHG emissions calculation
— not included in GHG emissions calculation
Δ May not be the same due to differences in upstream companies
* Fossil CO2

i) Direct CO_2 emissions from wastewater treatment

The principle of biological wastewater treatment is that the organic matter in wastewater is synthesized into new microbial cells through the metabolic reactions of microorganisms and is present in the form of residual sludge, which is then separated from the sludge by sedimentation, thereby purifying the water. Under aerobic conditions, organic matter is oxidized by microorganisms to produce CO_2 and H_2O. The IPCC classifies organic matter in wastewater as a biogenic origin; However, previous studies reported that 4-14% (Law et al., 2013) of organic matter comes from fossil carbon (such as detergents, cosmetics, and chemicals) and that fossil carbon is metabolized in the same pathway as fossil carbon during treatment of wastewater and sludge. Similar findings pointed to a 28% and 25% contribution of fossil carbon in the influent and effluent, respectively (Tseng et al., 2016 and Griffith et al., 2009). Schneider et al. (2015) estimated the contribution of fossil carbon to the overall wastewater treatment industry at 11-15%.

ii) Direct CH_4 emissions from wastewater treatment

Figure 1.7 (a) shows the two-stage theory of anaerobic digestion. The CH_4 is produced in the anaerobic process of wastewater treatment, where the organic matter is converted to organic acids and then methanogenic bacteria break down organic acids into CH_4, CO_2, and H_2O.

iii) Direct N_2O emissions from wastewater treatment

The N_2O emissions occur during the nitrogen removal process of wastewater treatment, including nitrification (Figure 1.7 (b): process 1-2), denitrification (Figure 1.7(b): process 3-6), and anaerobic ammonia oxidation (ANAMMOX) (Figure 1.7(b): process 8).

Hydroxylamine (NH_2OH) is an intermediate product in the nitrification when ammonia (NH_4^+) is oxidized to nitrite (NO_3^-) by ammonia-oxidizing bacteria (AOB), and it is readily oxidized to produce N_2O (Kampschreur M. J. et al., 2009 and Ma B. et al., 2016). There is consensus that N_2O is an inevitable intermediate product of the denitrification process and that its emission intensity is influenced by the dissolved oxygen concentration, C/N value, NO_2^- concentration, and pH value (Jeffrey F. et al., 2009, Maite P. et al., 2014, and Theoni M. M. et al., 2017). The mechanism of N_2O formation is complex, influenced by a variety of external factors, and involves a variety of denitrifying bacteria, and therefore has been a hot topic of research of wastewater treatment engineering. The current consensus is that because the global warming potential of N_2O is roughly 300 times that of CO_2, its release is a major source of direct GHG emissions from wastewater treatment processes.

iv) Direct CH_4 emissions from sludge treatment

It arises from the anaerobic decomposition of organic matter in sludge and is mainly generated from sludge landfills, poorly managed sludge anaerobic digestion tanks, and aerobic fermentation plants.

v) Direct N_2O emissions from sludge treatment

It arises from the aerobic fermentation of sludge, anaerobic digestion, and emissions from the soil after land use.

(2) Indirect GHG emissions

Indirect emissions resulted from energy consumption of electricity and fuel, freshwater consumption, and chemical consumption.

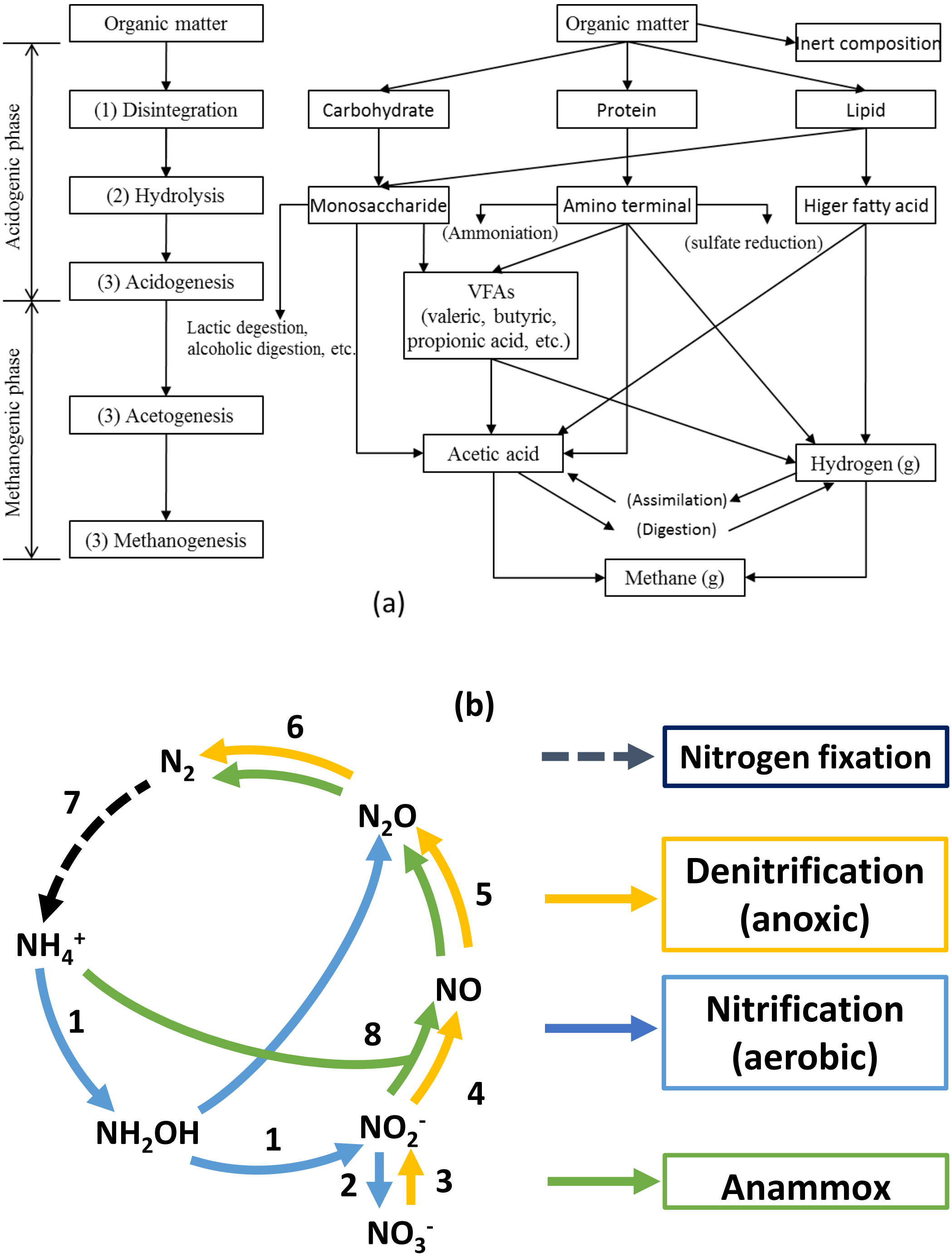
i) During the construction phase, electricity, fuel, and construction materials are consumed, and their production, transportation, and use generate GHG emissions, which are defined as indirect GHG emissions during the construction phase.

ii) During the operational phase, resources (such as electricity, fuel, and chemicals) are inevitably consumed to keep the plants running (such as equipment operation, technology requirements, and transport), of which the production, transportation and use cause GHG emissions. The main sources of electricity consumption are the lift pump, aeration, and sludge return pump; the main sources of chemical consumption are external carbon sources, flocculants, and sludge thickener.

iii) During the demolition phase, GHG emissions come from energy consumption, the treatment and disposal of construction waste, and transportation.

(3) Carbon offset

Carbon offsets are processes where products/energy generated from waste treatment can offset the GHG emitted from producing products/energy. It contains wastewater-based recycled water replacing freshwater, sludge recycling products replacing fertilizer/building materials, and heat recovery replacing energy consumption.



Source：(a) Noike T. et al. (2009); (b) modified from Chai C. (2017) and Lu J. (2017)

Figure 1.7 Principles of CH_4 and N_2O generation in wastewater treatment：(a) Two-stage theory of anaerobic digestion and (b) Nitrogen removal during wastewater treatment (not include nitrogen fixation)

1.1.3 Status of global climate change and challenges of the UWS

(1) The theory and evidence of climate change

i) The evidence of global warming

Figure 1.8 (c and d) provides the evidence of global warming caused by atmospheric GHG has increased since the industrial revolution. The other evidence includes global temperature rise (a), warming ocean (a), and sea level rise (b). All evidence indicted climate system is warming, the reason is emitted GHG is too much.

ii) Reasons for climate change

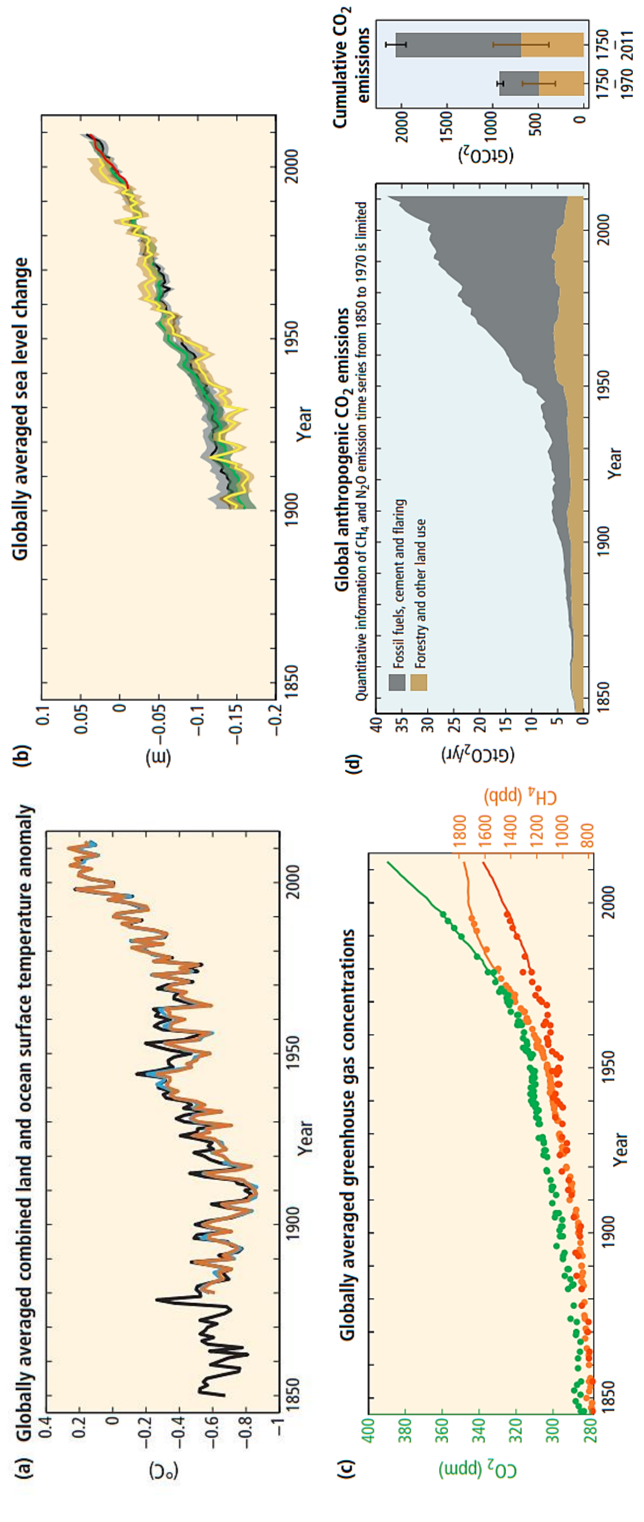
The reasons for climate change may be divided into natural and anthropogenic factors. The former includes solar activity, volcanic activity, and changes within the climate system; the latter includes increases in atmospheric GHG concentrations caused by human burning of fossil fuels and deforestation, changes in atmospheric aerosol concentrations, land changes, and changes in land cover.

The IPCC's Assessment Report (AR), which concluded that excessive GHG emissions from human activities are the main cause of global climate change, has increased in probability from 66% (3rd AR, 2002) to 90% (4th AR, 2007) to a very high probability of 95% (5th AR, 2013).

iii) The current state of global GHG emissions

The Netherlands Environmental Assessment Agency (PBL, 2020) reports that total global GHG emissions have increased by an average of 1.4% per year since 2010. A record high was reached in 2019, with total emissions (excluding land use change) reaching 52.4 billion tons of carbon dioxide equivalent, a result 44% and 59% higher than in 2000 and 1990 respectively. Global per capita GHG emissions reached 6.8 tons of carbon dioxide equivalent.

Sources of GHGs, mainly carbon dioxide emissions from industrial processes such as fossil fuel incineration and cement production, account for 72.6% of total global GHG emissions (2010-2019). Methane (CH_4) and nitrous oxide (N_2O) account for about 19.0% and 5.5% of emissions respectively, and a further 2.9% of emissions come from fluorinated gases such as hydrofluorocarbons (HFCs), perfluorocarbons (PFCs) and sulfur hexafluoride (SF6).



Source: IPCC (2014) SPM

Figure 1.8 Globally averaged combined land and ocean surface temperature anomaly change (a), global average sea level change (b), global average GHG concentration change (c), and global anthropogenic CO_2 emissions change (d) from 1850 to 2005

The International Energy Agency (IEA, 2019) reports that the contribution of CO_2 emissions from fossil fuel incineration is 43.8%, 34.6% and 21.6% from coal, oil and natural gas respectively. The results easily show that the burning of coal with the same calorific value emits approximately twice as much CO_2 as natural gas.

Electricity and heat, transport, and industry are the largest contributors to global CO_2 emissions, together accounting for around 85%.

The Emissions Gap Report (UNEP, 2020), the top six emitters (regions) together account for 62.5% of total global GHG emissions (excluding land use change) over the period 2010-2019, in descending order of contribution from China (26%), the US (13%), the EU-27 and the UK (8.6%), India (6.6%), and Russia (4.8%). Russia (4.8%), and Japan (2.8%). Global per capita emissions in 2019 are about 6.8 tons, with the US three times higher than the world average, while India is about 60% lower compared to the world average.

iv) Status of China's GHG emissions

The Netherlands Environmental Assessment Agency (PBL, 2020) reports that China's GHG emissions reached 14 billion tons of carbon dioxide equivalent, or about 9.7 tons of carbon dioxide equivalent per capita, with total emissions accounting for about 27% of total GHG emissions (excluding land use change) in the Golden Globe. From 2010 to 2019, China's total GHG emissions will grow at an average annual rate of about 2.3%, which is higher than the global average. since 2010, China's total GHG emissions have increased by about 24%, including a 26% increase in CO_2.

In 2019, carbon dioxide emissions accounted for 82.6% of China's total GHG emissions, about 10 percentage points above the global average. 11.6% of emissions, in addition to carbon dioxide, originated from methane, and about 3% and 2.8% from nitrous oxide and fluorinated gas emissions.

The IEA (2019) reports that coal incineration is the most important source of CO_2 emissions from fossil fuel incineration, with coal, oil, and natural gas incineration accounting for 80%, 14%, and 6% of carbon emissions, respectively (2018).

By sector, the electricity and heating account for about half of carbon emissions and industry for 28%, with a combined total of nearly 80%, in addition to transport and residential use, which are also important areas of CO_2 emissions.

(2) SDGs, Carbon Peaking and Carbon Neutrality Goals

i) Carbon Peaking

Broadly speaking, carbon peaking refers to a point in time when carbon dioxide emissions stop growing and peak, after which they gradually fall back. According to the World Resources Institute, carbon peaking is a process whereby carbon emissions first plateau and can fluctuate within a certain range before entering a steady decline.

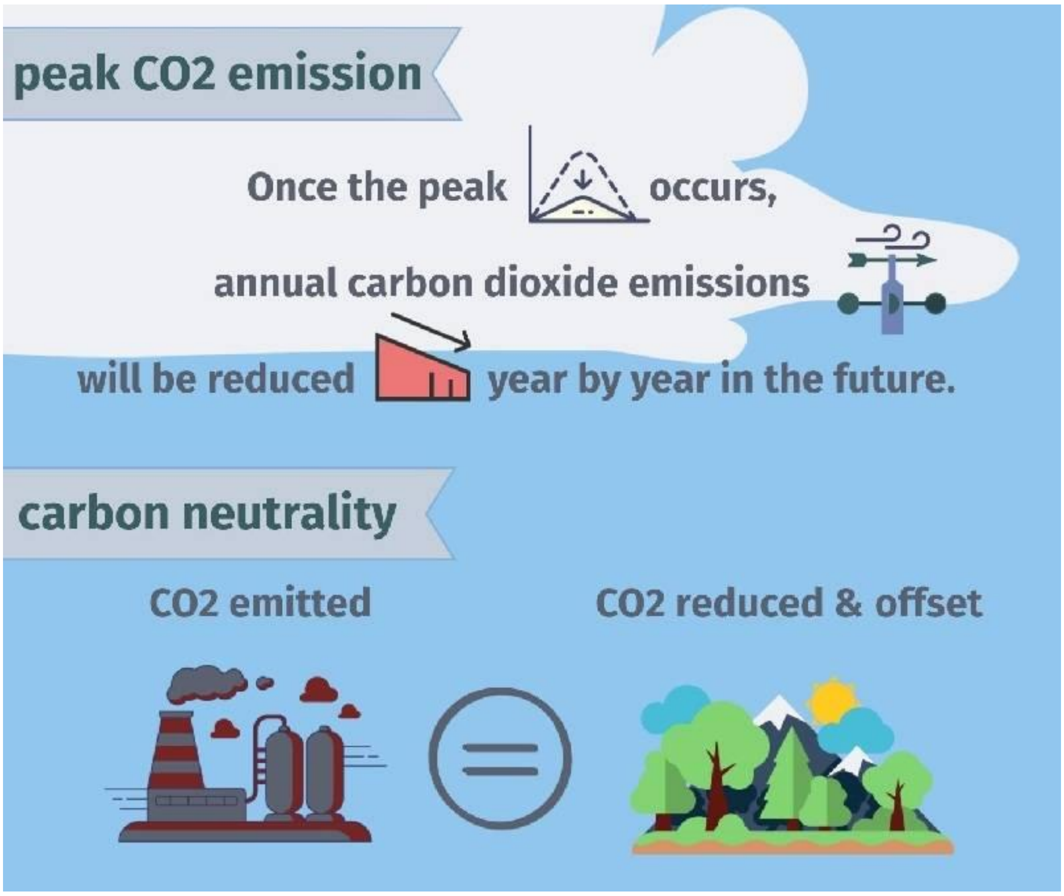
ii) Carbon Neutrality

They were also known as carbon offsetting. It is the process by which an enterprise, group, or individual measures the total amount of GHG emissions produced directly or indirectly within a certain period of time and offsets its own carbon dioxide emissions by planting trees, saving energy, and reducing emissions, in order to achieve "zero" carbon dioxide emissions. Carbon neutrality as shown in Figure 1.9, which means taking offsetting measures to balance the total amount of carbon emitted to achieve zero emissions, is one of the modern efforts to reduce global warming. People calculate the amount of carbon dioxide they produce directly or indirectly through their daily activities and offset the corresponding amount in the atmosphere by planting trees or buying carbon credits from third parties.

iii) SDGs

The Sustainable Development Goals (SDGs), also known as the Global Goals, were adopted by all United Nations Member States in 2015 as a universal call to action to end poverty, protect the planet and ensure that all people enjoy peace and prosperity by 2030. The 17 SDGs are integrated—that is, they recognize that action in one area will affect outcomes in others, and that development must balance social, economic, and environmental sustainability.

The 17 SDGs are: (1) No Poverty, (2) Zero Hunger, (3) Good Health and Well-being, (4) Quality Education, (5) Gender Equality, (6) Clean Water and Sanitation, (7) Affordable and Clean Energy, (8) Decent Work and Economic Growth, (9) Industry, Innovation and Infrastructure, (10) Reducing Inequality, (11) Sustainable Cities and Communities, (12) Responsible Consumption and Production, (13) Climate Action, (14) Life Below Water, (15) Life On Land, (16) Peace, Justice, and Strong Institutions, (17) Partnerships for the Goals.

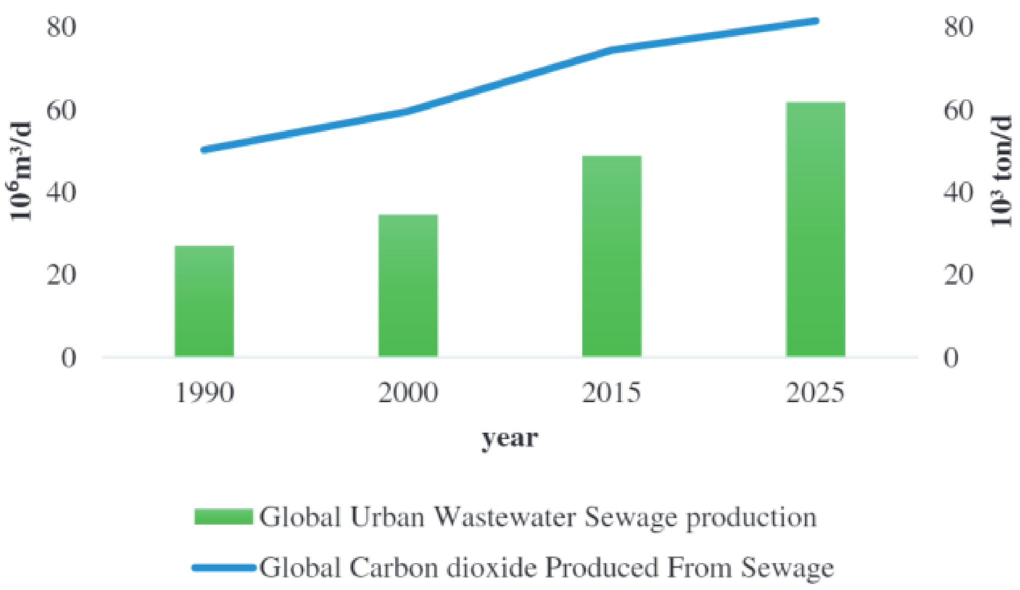


Source: https://www.sohu.com/a/439736200_162758

Figure 1.9 Carbon Peaking and Carbon Neutrality

(3) Status of GHG emissions from UWS

In developed countries, energy consumptions, CO_2 emissions, N_2O emissions, and CH_4 emissions from WWTPs account for 3% (Mo W. et al., 2012), 4% (Martin W. et al., 2008), 3% (Kampschreur M. J. et al., 2009), and 5% (El Fadel M. et al., 2001) of total consumptions/emissions, respectively.



Source: Thi Kieu Loan Nguyen et al., 2019

Figure 1.10 Global CO_2 emissions from sewage

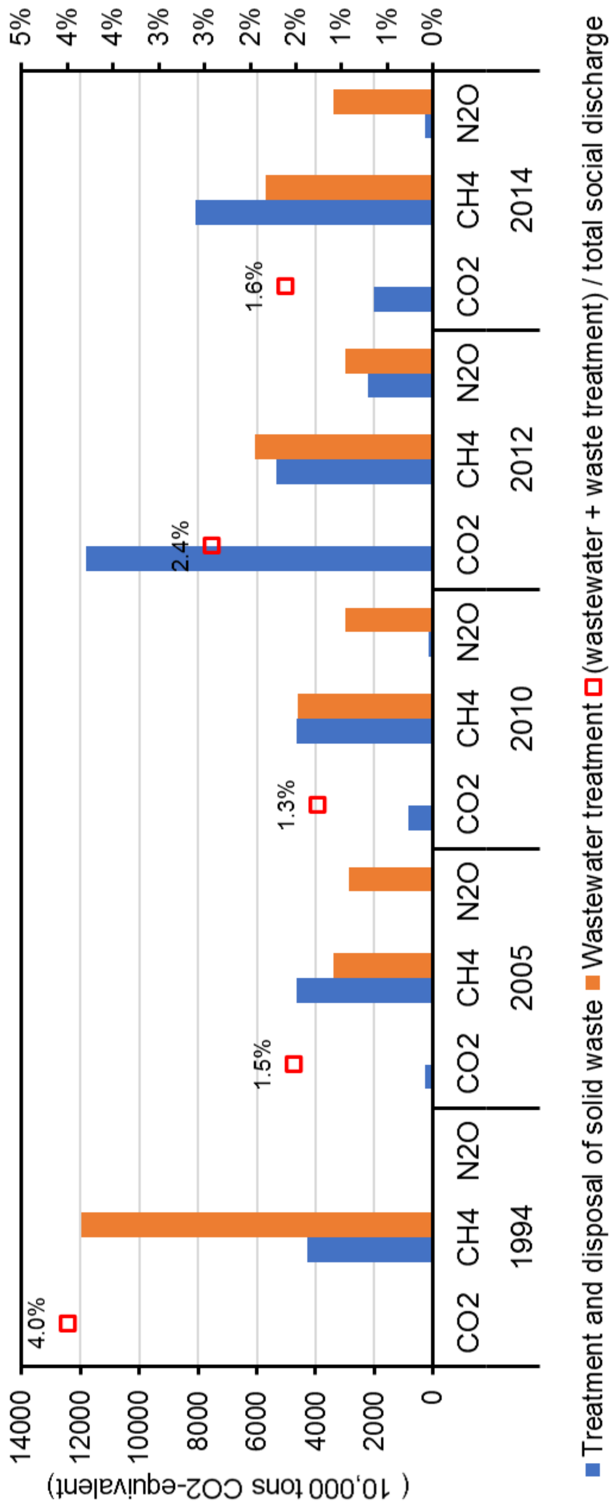
In China, the GHG emissions inventory published by NDRC and MEE on five occasions (see Table 1.3), which contain data for 1994, 2005, 2010, 2012 and 2014, report that China's waste treatment industry (wastewater treatment and waste treatment & disposal) accounts for 1.6% (195 million tons CO_2-equivalent in 2014) of total society's GHG emissions, with the maximum value was 4% in 1994 and the minimum value was 1.3% in 2005 (see Figure 1.11).

In addition, only CH4 was accounted for in the 1994 inventory, CO_2 and N_2O were accounted for in 2005 and 2010 respectively, however, all of CO_2 from wastewater treatment was ignored as a biological origin.

Table 1.3 GHG emissions inventory from wastewater and waste treatment in China

Year of inventory	GHG types	Total social emissions (excluding land-use change and forestry				Data source	Publisher, year
			Treatment and disposal of solid waste	Wastewater treatment	(wastewater + waste treatment) / total social		
			* (10,000 tons CO2-equivalent)				
1994	CO2	307300	-	-	0.040	Initial National Communications on Climate Change	NDRC, 2004
	CH4	72009	4263	11949			
	N2O	26350	-	-			
2005	CO2	597557	266	-	0.015	2nd National Communications on Climate Change	NDRC, 2012
	CH4	93282	4628	3402			
	N2O	39370	-	2883			
2010	CO2	870700	845		0.013	3th National Communications on Climate Change	MEE, 2018
	CH4	116300	4635	4607			
	N2O	54700	155	2976			
2012	CO2	989300	11804	-	0.024	First Biennial Update Report on Climate Change	NDRC, 2016
	CH4	117422	5315	6073			
	N2O	63829	2201	3007			
2014	CO2	1027500	2006	-	0.016	2nd Biennial Update Report on Climate Change	MEE, 2018
	CH4	116113	8068	5714			
	N2O	60977	279	3410			

NDRC: National Development and Reform Commission of the People's Republic of China
MEE: Ministry of Ecology and Environment the People's Republic of China
*: Global Warming Potential (GWP) using values on a 100-year time scale (IPCC, 2nd AR)



Data source: Table 1.3

Figure 1.11 GHG emissions contribution from wastewater and waste treatment during 1994-2014 in China

(4) Challenges of urban wastewater systems (UWS) in China

i) As shown in Figure 1.12, the treatment efficiency of WWTPs is generally low and the influent COD loading is unstable (Q.H. Zhang et al., 2016).

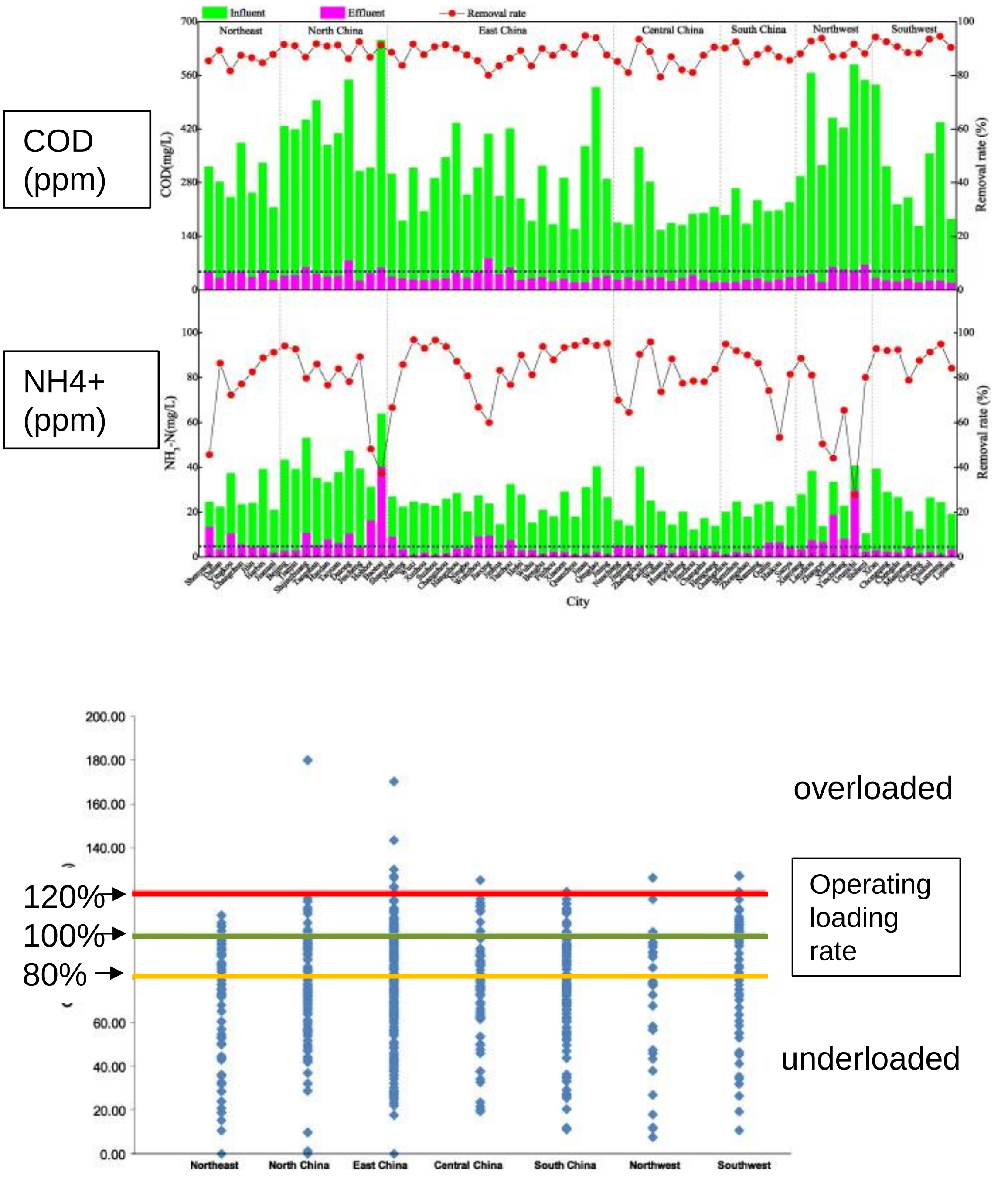
ii) As shown in Figure 1.12, it is common for wastewater treatment plants to operate at overload or underload, with only 30% of them operating at 80%-120%. (Q.H. Zhang et al., 2016)

ii) As shown in Table 1.1, less than 5% of new WWTPs operated AD projects.

iii) WWTPs commonly used solutions to increase treatment efficiency may result in additional GHG emissions. High energy and chemicals consumption, and the widespread use of increased aeration and added chemicals (more PAC for phosphorus removal, more carbon sources for nitrogen removal) to meet increasingly stringent discharge standards, reducing water pollution while there is an increase in GHGs (disguised pollutant transfer).

iv) The pipe network system, with severe phenomena such as dripping and leakage, and the low collection rate of separated rainwater and sewage, resulting in low organic content in the inlet of the wastewater plant, which is not conducive to biological treatment for nitrogen and phosphorus removal and necessitates the addition of other organic matter as a carbon source.

Overall, WWTPs are faced with the dual challenge of reducing pollutants and reducing GHG emissions. The discharge standards had to be raised to reduce the environmental load from the effluent. On the other hand, the pressure for sustainable development requires them to consider carbon emissions and energy consumption.



Source: Q.H. Zhang et al. (2016)

Figure 1.12 Pollutant loads and operating loads of WWTPs in China

1.1.4 LCA of urban wastewater systems (UWS)

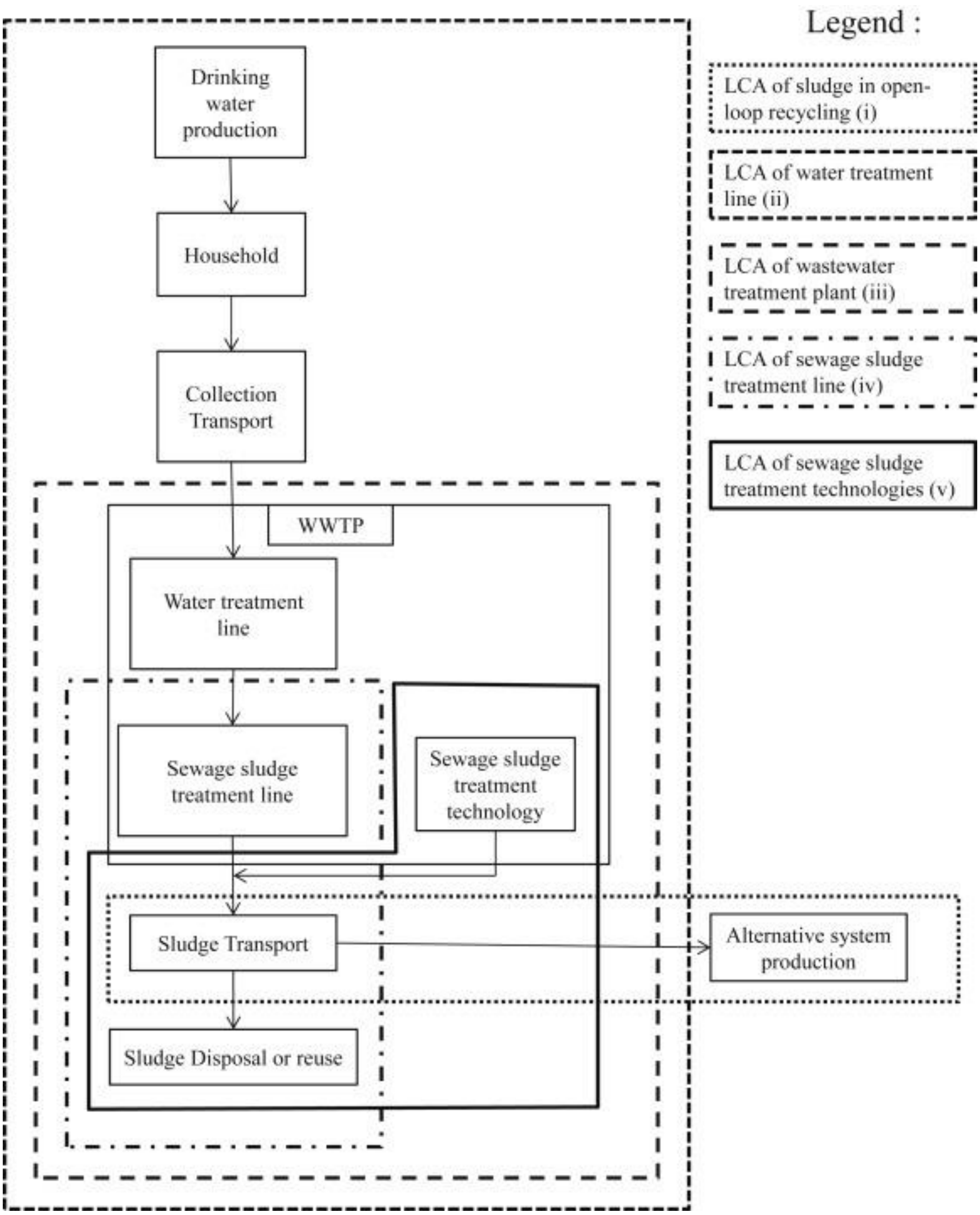
The evaluation for UWS is primarily based on technical and economic analysis to ensure the output quality of WWTPs. This evaluation method is mainly aimed at achieving water quality standards, considering the cost and benefit of different treatment processes from an economic perspective, and analyzing the economic rationality of the treatment process. However, in the face of development challenges of pollutant reduction, energy conservation, and emission reduction, this emphasis on the evaluation of processing technology performance will highlight its shortcomings; thus, a systematic environmental impact analysis should be established. With the development of urbanization in China, energy consumption and GHG emissions of UWS will become an important aspect of growth. As wastewater treatment is gradually moving towards sustainable development, it is necessary to systematically consider GHG emissions throughout the entire process of UWS. The LCA can provide a systematic research framework for energy conservation and GHG emission reduction of UWS.

In previous studies, the application of LCA in wastewater treatment was mostly to evaluate the environmental impact (EI) of WWTPs or wastewater treatment processes and to compare the wastewater treatment processes. Mahgoub et al. (2010) evaluated the EI including CO_2 emissions from an urban water system in Egypt by using the LCA approach. Rodriguez RM et al. (2016) used the LCA method to compare heterogeneous and homogenous Fenton processes for the treatment of pharmaceutical wastewater. Frijins (2012) mentioned that the accounting boundary should include direct and indirect CO_2 from energy consumption, direct CH4 and N_2O emissions from treatment processes, and indirect CO_2 emissions from production of chemicals used in relevant processes.

(1) Evaluation boundaries

The system boundaries in most previous studies of LCA for wastewater treatment processes ignore the construction and demolition phases, which they consider to be less influential than the operational phase (Ali Hussein Sabeen et al., 2018). This conclusion is because urban infrastructure typically has a service life of 30-50 years and ignores technological updates and modifications to WWTPs. However, China is in a phase of rapid economic and social development. Many WWTPs within 20 years have undergone technological improvements (to meet more stringent emission standards), expansions (to meet increased treatment demand), and relocations due to urban expansion (to move away from city centers). In addition, the construction phase of decentralized systems cannot be ignored due to economic scale effects. When LCA is used for process comparison, discussing only the operational phase is fine.

However, when assessing a single WWTP, the lack of comparison between the construction and demolition phases will make the results much less accurate. (LI Shuang et al., 2020)



Source: Marilys Pradel et al., 2016

Figure 1.13 The system boundaries of UWS

In addition, many of the previous LCA studies in China did not include the environmental impacts caused by the sludge treatment and disposal process. A few have only calculated sludge transport and solid waste discharge, while the environmental impacts caused by the sludge treatment and disposal have not been effectively assessed. (LI Shuang et al., 2020) As an inevitable product of biological treatment of wastewater, the environmental impact of sludge overall UWS during its treatment and disposal cannot be ignored. It is bound to have an impact on the environment if sludge is not well treated and disposed.

(2) Functional unit

Functional units are characteristic quantities used to identify evaluation objectives, the main purpose of which is to enable comparison between different evaluation objects with different input and output data. The functional units selected therefore need to meet a certain level of comparability in order to ensure that objective comparisons can be made in different study systems (e.g. centralized and decentralized treatment), with different treatment processes and water volume scales.

i) for wastewater treatment

Most previous studies have used volume equivalents, such as m^3/d, m^3/year, or Total wastewater treated in LCA (m^3). However, treatment efficiency varies from case to case depending on the quality of the influent water; the environmental impact per unit volume of effluent treated by different wastewater treatment system for sludge treatment stems is clearly systematically different in nature when the same effluent discharge standards are met. Therefore, the population equivalent (PE) is often used as a functional unit, generally using the daily load of BOD_5 per capita (60 g BOD_5/d in developed countries; 40 g BOD_5/d in developing countries) and the wastewater generation per capita (China: 100-200 L/d) as a conversion factor between population equivalent and volume equivalent.

ii) for sludge treatment

Most previous studies chose mass as FU for sludge treatment (e.g., treating 1 ton of dry sludge), while others have chosen volume-based FU or PE (the amount of sludge generated in a specific time by one individual) (Liu et al., 2013, De et al., 2008, Foley et al., 2008, Peng et al., 2013 and Beibei et al., 2013).

(3) LCI

It includes direct emissions from the wastewater and sludge treatment process (e.g. CO_2, CH_4,

N_xO, SO_2, etc.) as well as indirect environmental emissions from the consumption of energy and materials invested in the treatment process. However, the database in China is not yet detailed and complete, especially as the official database is not far from databases such as Ecoinvent and ELCD.

(4) Selection of evaluation indicators

The Global Warming Potential (GWP) are considered in all LCA applications and fall into a generic impact category; moreover, GWP production is largely determined by pollutant removal and energy consumption efficiency, and the extent to which it affects global climate change is broadly consistent and does not require a region-specific factor.

Table 1.4 Global Warming Potential (GWP) values for GHGs covered by emissions from UWS

Common name	Chemical formula	GWP values for 100-year time horizon		
		Second Assessment Report (SAR)	Fourth Assessment Report (AR4)	Fifth Assessment Report (AR5)
Carbon dioxide	CO_2	1	1	1
Methane	CH_4	21	25	28
Nitrous oxide	N_2O	310	298	265

1.2 Significance and objectives of the study

1.2.1 Significance

There is a consensus that excessive GHG emissions from human activities are the leading cause of global warming. Governments are taking measures to combat climate change. The verification of GHG emissions and the compilation of accurate GHG emission inventories is a priority. It can provide data to support the formulation of mitigation policies, urban development planning, and the achievement of sustainable development. The Chinese government has also released national-level GHG emission inventories. Still, there is a lack of industry-level studies of emission mechanisms, emission factors, and inventories, particularly concerning urban wastewater systems.

The UWS involves wastewater and waste management, urban infrastructure development and planning, and recycling industries. Traditionally, planners and policymakers have focused on maximizing economic efficiency and lacking consideration of environmental impacts when making technology choices and locating facilities. Secondly, the sectors are independent and lack careful consideration and planning of the entire system.

1.2.2 Objectives

The work of this study is as follows：

(1) To clarify the sources of GHG emissions from municipal wastewater systems, accurately account for and compile an inventory of GHG emissions, analyze the potential for reducing emissions from municipal wastewater systems, and consider technological options.

(2) A complete LCA covering the construction, operation, and demolition phases, involving a wastewater unit, a sludge unit, and a resource recovery unit, considering technological options and site locations, was completed to compare the environmental impacts of the two management strategies and to highlight their respective benefits and limitations. A detailed LCI was compiled through a field investigation of two WWTPs with different implementation scales, which complements the industry-level LCI data from China. A case of a community-scale wastewater treatment facility integrating reclaimed water use was analyzed. In addition, a distance-based optimization model of decentralized systems was developed to quantify the distance between decentralized and centralized systems for urban hybrid applications, providing relevant environmental information for planners, researchers, and policymakers.

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2. A REVIEW OF PREVIOUS STUDIES AND METHODOLOGIES

2.1 Estimation methods of GHGs emission from UWS

2.1.1 IPCC procedure (emission factor method)

The IPCC Guidelines for National Greenhouse Gas Inventories are a standard international methodology for evaluating GHG emissions at the national and sectoral levels (IPCC, 1996, 2000, and 2006).

The 2006 Guidelines for National Greenhouse Gas Inventories (IPCC, 2006) were published in response to the need for individual countries to report on their emissions. The methodology in the Inventory Guidelines is based on activity data and emission factors, also known as the emission factor approach. The emission factors for the different sources in various sectors are determined based on relevant professional databases, and scientific studies validate all emission factors.

Depending on how the data is obtained, the methodology is divided into two different accounting methods: 'top-down' and 'bottom-up'. The 'top-down' approach is based on national statistics to develop emission inventories. Depending on the circumstances, default emission factors are used, with appropriate modifications to the emission factors. The 'bottom-up' approach, based on technical processes, requires a higher degree of data accuracy. This approach requires country-specific emission factors or data on production processes and plant levels for assessment.

$$E = \sum A_i \times EF_i$$

where

E: Greenhouse gas emissions.

A_i: Activity level data for sources of GHG emissions (such as energy consumption, material consumption, and freshwater consumption).

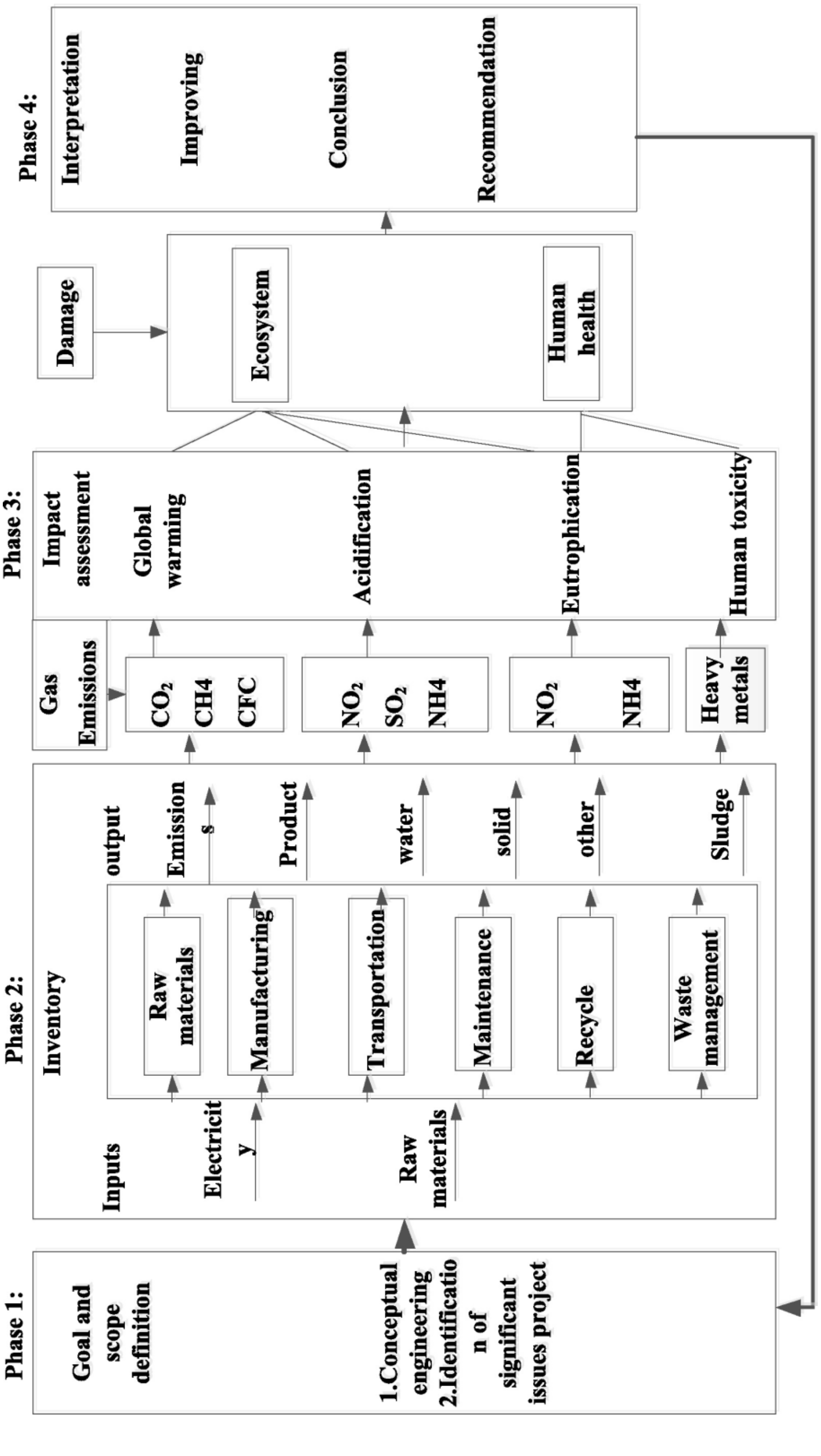
EF_i: Greenhouse gas emission factors (such as GHG emissions per unit of energy consumed).

2.1.2 LCA procedure

LCA, as defined by the Society of Environmental Toxicology and Chemistry (SETAC, 1993), is a systematic approach to evaluating the environmental impact of a product or process by identifying and quantifying the material and energy flows and pollutant emissions throughout the life cycle of the system, to identify opportunities for improvement. The LCA covers the entire life cycle from raw material extraction, processing/production, transport/marketing, use to waste disposal.

In 1997, the International Organization for Standardization (ISO), in its international standard for LCA (ISO 14040), stated that the 'products' assessed by LCA can be generally manufactured product systems or service systems or service products provided by the service industry. A product system's continuous and interlinked phases are referred to the life cycle, either from raw materials extraction to final disposal of a product (e.g. Cradle-to-Grave) or a selection of Cradle-to-Gate depending on need for a study.

The LCA procedure is widely used in GHG emissions studies at the enterprise level or at the process technology level. The methodology has been widely used in GHG assessment standards and corporate carbon disclosure projects, such as Greenhouse Gas Protocol Product Life Cycle Accounting and Reporting Standard (WRI, 2011) and the Specification for the Assessment of Life Cycle Greenhouse Gas Emissions from Goods and Services and Guidance for their Use (PAS2050) produced by the BSI (BSI, 2008).



Source: Ali Hussein Sabeen et al., (2018)

Figure 2.1 The framework for LCA

The research framework for LCA is the definition of objectives and scope, inventory analysis and impact evaluation. Firstly, the reasons and intentions of the LCA study are clarified, i.e. the objectives of the study are defined; secondly, the system boundaries, functional units, and data requirements of the product system under study are clearly described in detail, i.e. the scope is defined; this is followed by inventory analysis, which is the process of analyzing and constructing an inventory of the input and output data in the system in detail; finally, the results of the inventory analysis are used to evaluate the environmental impacts of the product during the various stages of its life cycle. For example, the transformation of inventory data into specific impact categories (e.g. climate change) and indicator parameters (e.g. kilograms of carbon dioxide equivalent) to facilitate the understanding of the environmental impact of a product's life cycle is essentially a process of qualitative or quantitative ranking of the inventory analysis results, which is the core of LCA and the most difficult part.

2.1.3 Measurement method

Most studies on GHG emissions from wastewater treatment processes have been based on empirical emission values or experimental data, while studies on comprehensive continuous emission monitoring of actual operating wastewater treatment facilities are relatively rare. As most WWTPs are not yet equipped with GHG monitoring equipment (continuous emission monitoring system, CEMC) and as most GHG emissions from wastewater treatment processes are fugitive, it is difficult to monitor them continuously. Therefore, based on the mechanism of microbial action in the wastewater treatment process to produce GHG emissions, some research teams have set up multiple sampling points within the plant to obtain more accurate first-hand data.

Foley et al. (2010) measured N_2O emissions from seven Australian wastewater treatment plants using biological denitrification based on the law of conservation of mass using sampling methods.

2.1.4 Kinetic model simulation methods

Wastewater treatment kinetic models are used to calculate the GHGs such as CO_2, N_2O and CH_4 produced during the treatment of wastewater by simulating the kinetics of biological treatment.

The classical Activated Sludge/Anaerobic Digester model (AS/AD) can calculate CO_2 emissions from wastewater treatment processes with some accuracy. The sources of CO_2 include endogenous microbial respiration, oxidation of organic matter and anaerobic digestion of organic matter.

The Anaerobic Digestion Model No. 1 (ADM1) can be used to calculate CH_4 emissions from anaerobic digestion of sludge to produce biogas, while CH4 emissions from anaerobic treatment of wastewater are largely disregarded.

The N_2O emissions from wastewater treatment processes occur mainly during biological denitrification, where N_2O is an intermediate product of incomplete denitrification during the nitrification and denitrification of wastewater. As an extension of ASM #1, the Activated Sludge Model for Nitrogen (ASMN) is currently a more accurate kinetic model for N_2O production, which embeds the four steps of denitrification: NO_3^- to NO_2^- to NO to N_2O to N_2.

G. Rodriguez-Garcia et al. (2012) introduced a Direct Emission Estimation Model (DEEM), a model for estimating CO_2 and N_2O emissions from WWTP, using the ASMN. The model has the advantage of simplicity and applicability and is more suitable for life cycle assessment and carbon footprint studies. In their analysis of a full life cycle GHG emissions inventory for an A/O process wastewater treatment plant in Spain, they found that direct emissions of N_2O were eight times higher than indirect GHG emissions due to electricity consumption.

2.1.5 Summary of the comparison of the method options

This section introduces four methods of accounting for GHG emissions in wastewater treatment systems:

i) the IPCC procedure is mostly used for calculations at the national and regional levels, and emission factors need to be corrected according to the specific situation of the country and region where they are located.

ii) the LCA procedure covers the whole process from raw material extraction to final disposal, and can show hidden carbon emissions well, but the inventory data demand is large and data acquisition

Life Cycle Assessment (LCA) has the following advantages: systematic (covering multiple life cycle stages to avoid transferring environmental problems between them), quantitative (including indicators for various types of environmental impacts to avoid transferring environmental problems between them), standardized (uniform international standards), and universal (applicable to the environmental assessment of all products and services, providing environmental data for various technical, regulatory or policy decisions).

iii) The results of the measurement method are the closest to the real emission data, but they are costly and require a certain level of expertise of the measurement personnel, and few wastewater plants have installed the corresponding monitoring instruments.

iv) the modelling method can estimate GHG emissions well, but due to the complexity of the biological treatment mechanism and the need to make corrections for specific operational parameters, it is difficult to apply in a wide range of practical applications and is mostly used to explain the mechanisms of GHGs production.

2.2 Factors affecting the GHG emissions from UWS

2.2.1 Technological options

Yang Qin (2012) analyzed the carbon footprint of four common small and medium-sized urban wastewater treatment processes (AO, AAO, oxidation ditch and SBR) using life cycle assessment. The types of GHGs studied include CO_2, CH_4 and N_2O from the wastewater treatment process, as well as indirect emissions from operational energy and pharmaceutical consumption.

Pan et al. (2011) estimated and evaluated the GHG emissions of vertical submerged artificial wetlands and centralized urban wastewater treatment plants in Changzhou City based on the life cycle assessment method and the IPCC inventory calculation model. The scope of the evaluation included wastewater collection, wastewater treatment and discharge, and sludge disposal; the construction and demolition of wastewater treatment plants were not considered. The GHGs accounted for included, CH_4 emissions from the wastewater and sludge treatment processes, N_2O emissions from the nitrification and denitrification process of the effluent in the receiving water body (N_2O emissions from the wastewater treatment process were neglected), and indirect CO_2 emissions from the energy consumption in the wastewater sludge treatment process.

Cakir et al. (2005) used a wastewater treatment model to compare the characteristics of three aerobic treatment processes (conventional activated sludge, delayed aeration activated sludge and high load activated sludge) and one anaerobic treatment process, UASB, in terms of GHG emissions during the treatment of domestic wastewater, with all four treatment processes using anaerobic digestion of the residual sludge and recycling of CH_4, with the treated effluent meeting All four treatment processes use anaerobic digestion of the residual sludge and recycle CH_4, with the effluent meeting secondary treatment requirements (BOD $\leqslant$ 30 mg/L). The study showed that GHG emissions from the aerobic process were positively correlated with sludge age and influent BOD.

Li Sha et al. (2012) analyzed the mechanism of N_2O emissions from nitrification and denitrification from the perspective of microbial action mechanism and analyzed the N_2O emissions and related influencing factors of several typical wastewater treatment processes (especially the AAO process).

Wang Jinhe (2011) conducted a one-year field sampling, in-situ water quality monitoring and laboratory analysis of three urban wastewater treatment plants in Jinan using the AAO treatment process and three other urban wastewater treatment plants using typical nutrient removal BNR processes (pre-anaerobic-oxidation ditch process, pre-anaerobic-AAO process and inverted AAO process, respectively), focusing on N_2O and CH_4 emission levels, emission patterns and influencing factors were studied. Field sampling was divided into gas sample collection using floating gas flux hoods and gas sampling bags, dissolved gas sample collection using the upper space method and water quality sample collection, and the collected gas and dissolved gas samples were later analyzed in the laboratory using gas chromatography. It was found that the AAO process had an N_2O emission factor of 0.12-0.20% (N_2O-N/TN remove) and the inverted AAO process had the lowest GHG emissions. Subsequently, Wang Jinhe et al., (2012) investigated the release flux of N_2O from an SBR treatment process at a wastewater treatment plant in Qingdao using the same research and analysis method and obtained an emission factor of 1.1% (N_2O-N/TN remove) for N_2O from this process.

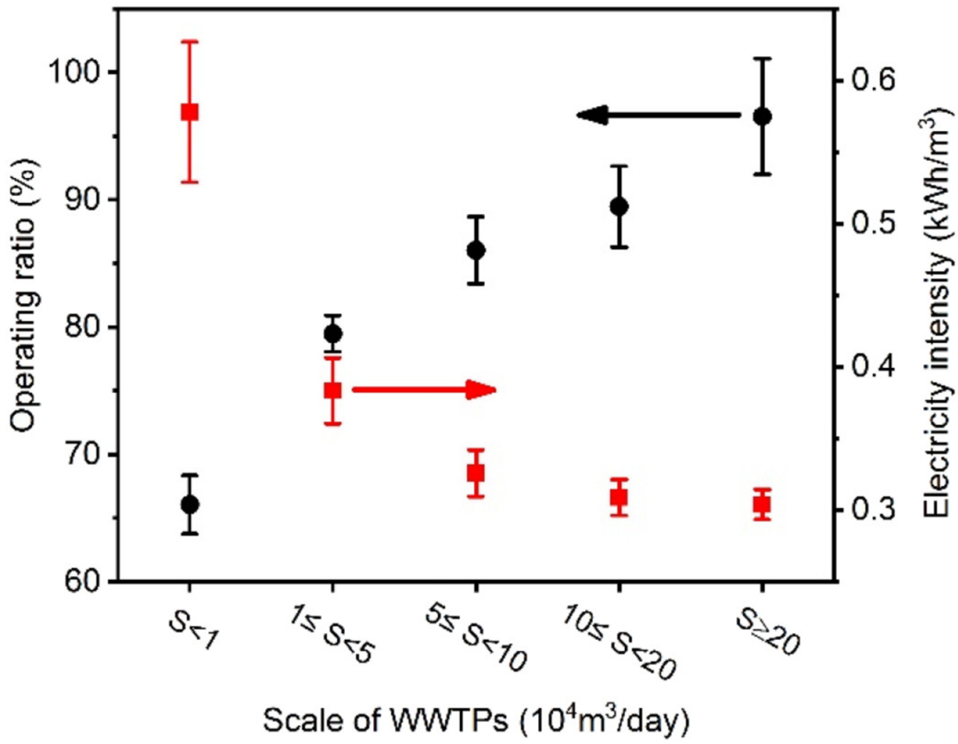
2.2.2 Implementation Scale

(1) Economies of scale

The widespread use of CWWT in cities is a result of economies of scale, meaning that large scale of operation of WWTPs benefits the minimization of costs (Mingjie X. et al., 2019).

Similar findings were reported the energy intensity decreases as the larger scale (Pablo K.C., 2016). Some researchers (G. De Feo et al., 2017 and Tamar, O. et al., 2016) shown that small-scale DWWT have advantages over CWWT because of operating low-energy technologies (e.g. constructed wetlands), but they did consider the urban application of DWWT integrated with Reclaimed Water use on a large community scale and the influence of scale on the same technology.

Previous environmental assessment studies of WWTPs with different implementation scales have mostly focused on energy consumption, with the aim of examining the correlation between energy intensity and implementation scale.



Source: Jia-Yuan Lu et al., (2019)

Figure 2.2 Operating ratio and electricity intensities of different-sized WWTPs in China

(2) Decentralized wastewater treatment systems (DWWT)

Sven Eggimann et al. (2015) used Sustainable Network Infrastructure Planning (SNIP), a two-step techno-economic heuristic modelling approach based on shortest path-finding and hierarchical-agglomerative clustering algorithms, to determine the optimal degree of concentration of WWTPs. The results show that the optimum degree of centralization is influenced by the terrain complexity and the settlement dispersion, which decreases as two factors increase. In addition, settlement dispersion is the largest influencing factor.

Sven Eggimann et al. (2016a) analyzed the total cost of a hybrid wastewater management system for different scenarios of connection rate (CR=0, 40, 60, 70, and 100%) and found the optimal CR using the state of Glarus as a case study. The results shown that the optimal CR depends on organizational and institutional arrangements rather than on maximizing economic benefits.

Sven Eggimann et al. (2016b) found highly non-linear economies of density for distributed wastewater systems. Low densities in sparsely populated regions thus result in higher costs for both centralized and decentralized system.

(3) Spatial optimization

Mingjie Xu et al. (2019) used life cycle cost analysis to examine conventional wastewater treatment and resource-directed systems for different population sizes. The economic performance and feasibility of a resource-directed system, an innovative sustainable sanitation facility based on vacuum pipe technology and source separation, is assessed. Parameters for the spatial distribution of households are introduced in the model, innovatively seeking a balance between economies of scale and decentralization. The results show that source separation systems have positive environmental and social effects due to lower energy consumption and sewerage costs, outperforming current systems in terms of life-cycle costs. Furthermore, settlement density is a key factor in collection costs, but the optimal collection size is not fixed and is determined by local market conditions.

Xu Zhiqiang et al. (2007) used the critical distance as the basis for determining the economics of centralized or decentralized wastewater treatment and reuse based on a cost-effectiveness function, which provides a theoretical basis for the siting of reuse water reclamation plants. The results show that a centralized wastewater reclamation and reuse plant in the study area can be considered within a radius of 5 km. In comparison, a decentralized system should be used in areas beyond 5 km to provide higher economic efficiency.

(4) spatial optimization of decentralized systems for NPR

Olga Kavvada et al. (2018) developed a generalized model aimed at minimizing economic and environmental indicators (economic costs, energy intensity, and GHG emissions) by considering relevant site-specific conditions to determine the optimal size of NPR (for toilet flushing) systems. The model focused on large buildings and involved influencing factors such as the location of the building (geographical elevation information) as well as the population and size characteristics of the building. The result indicated that decentralized systems are usually more efficient at larger scales (population size) because they benefit from economies of scale.

2.3 Innovations of this study and the framework of this dissertation

2.3.1 Innovations

(1) Improving a life cycle GHGs assessment system considering technological options

The evaluation method of UWS is primarily based on technical and economic analysis to ensure the output quality of WWTPs. This evaluation method is mainly aimed at achieving water quality standards, considering the cost and benefit of different wastewater treatment processes from an economic perspective, and analyzing the economic rationality of the treatment process. However, in the face of development challenges of pollutant reduction, energy conservation, and emission reduction, this emphasis on the evaluation of processing technology performance will highlight its shortcomings; thus, a systematic environmental impact analysis should be established. With the development of urbanization in China, energy consumption and GHG emissions of UWS will become an important aspect of growth.

IPCC does not consider direct CO_2 emissions from wastewater treatment, and NCSC does not consider direct CO_2 emissions from the wastewater treatment and direct N_2O emissions from sludge treatment. Overall, it is necessary to develop a systematic and comprehensive GHG evaluation system for UWS (including wastewater system, sludge system, and resource recovery system). It covers the whole life cycle (construction, operation, and decommissioning phases) and examines the three types of GHGs from direct and indirect emissions (including direct CO_2 from WWT).

Therefore, the following have been conducted in this study:

1) A GHG estimation model basing on LCA procedure was constructed, and the research objects were CH_4, N_2O, and CO_2 that were produced by the UWS. The estimation model of the GHG emissions was summarized and improved in the UWS considering technological options.

2) The GHG emission source from UWS was analyzed, and the level and key links of environmental loads generated by different technological options were identified. This helps to understand and compare the environmental impacts and provides suggestions for the sustainability.

3) The GHG emission characteristics of nine scenarios of different technological options were analyzed, and the environmental impacts caused by energy consumption and chemicals consumption were studied. Consequently, the wastewater-sludge treatment process under low carbonization and low environment impact were proposed.

(2) Modeling location optimization of UWS for low environmental loads

Previous studies did consider the urban application of DWWT integrated with reclaimed water use on a large community scale and the influence of scale on the same technology. In the other hand, the situation of how to choose between DWWT and CWWT or the location optimization for hybrid applications of both must be considered. This study uses LCA to evaluate the environmental loads (e.g. GWP) of a large community-based wastewater treatment system integrated with Reclaimed Water use, comparing operating in DWWT or CWWT. The life cycle inventories (LCIs) of two WWTPs were examined, as DWWT of 1,000 m^3/d and as DWWT of 500,000 m^3/d. The LCIs cover energy consumption, materials consumption, and transport during the construction, operation, and demolition phases. This study provided a quantitative analysis of optimizing location of hybrid applications of decentralized and centralized in cities through defining the critical distance. This study is intended to provide researchers, managers, and decision makers with information on the environmental loads of centralized and decentralized wastewater management strategies.

2.3.2 Framework

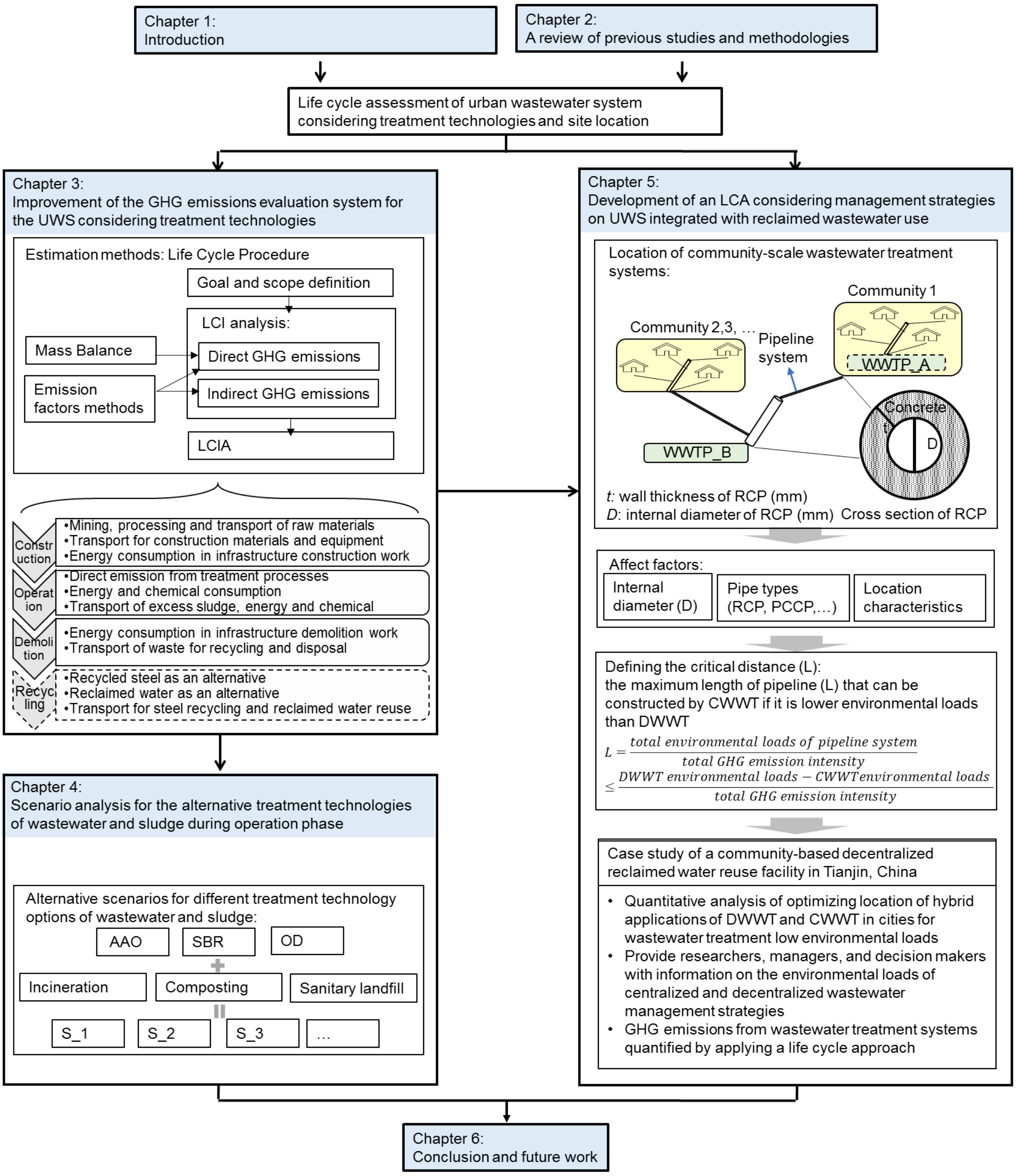


Figure 2.3 Framework of this dissertation

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3. IMPROVEMENT OF THE GHG EMISSIONS EVALUATION SYSTEM FOR THE UWS CONSIDERING TREATMENT TECHNOLOGIES

3.1 Construction of a life cycle GHG emissions model for UWS

3.1.1 Goal and scope definition

(1) Goal:

i) Analysis of levels and characteristics of GHG emission from conventional municipal wastewater systems (excluding resource recovery).

This study constructed a GHG emissions model covering direct and indirect sources, examined nine scenarios consisting of three conventional wastewater treatment technologies and three typical sludge treatment/disposal processes, and analyzed the factors affecting the GHG emissions.

ii) Analysis of GHG emissions and abatement potential of municipal wastewater systems with integrated resource recovery.

Using the constructed GHG emissions model, the GHG emissions of wastewater reclamation and reuse, sludge anaerobic digestion and biogas recycling, and land use of sludge compost instead of fertilizer are studied, and the GHG reduction potential and influencing factors of municipal wastewater systems through the resource recovery route are analyzed.

iii) Comparative analysis of the levels and characteristics of GHG emissions from municipal wastewater systems under scenarios applying different management strategies (decentralized and centralized).

The critical distance is defined, which takes the environmental load (GHG emissions) as a constraint, and it can provide a basis for selecting a management strategy for community municipal wastewater systems that integrate recycled water use at the scale of an urban community. Using the critical distance as a basis for determination can provide environmental information for the planning and siting of wastewater treatment and recycled water reuse in new communities (new campuses).

(2) Scope:

The scope of GHG emissions from the UWS can be determined by the type of emission source as：

i) Direct emission sources include three main types of GHG emissions, namely CO_2, CH_4, and N_2O, of which CO_2 is produced during the aerobic biological treatment of wastewater and sludge, CH_4 during anaerobic treatment, and N_2O emissions during denitrification.

ii) Indirect emission sources are GHG emissions caused by the consumption of energy and chemicals in the treatment process, which include the consumption of energy and resources during the construction and demolition of wastewater/sludge collection and transportation and treatment facilities; the consumption of energy and pharmaceuticals during the collection and transportation of wastewater/sludge and during the treatment of wastewater and sludge.

3.1.2 System boundaries and function unit

Figure 3.1 shows the system boundaries of this study. In DWWT (b+c), wastewater flows directly into the wastewater treatment system (b), and treated water is used on site. In CWWT (a+b+c+a'), wastewater is collected and transported through the wastewater pipeline system (a) into the wastewater treatment system (b), and the treated water is transported by the reclaimed water pipeline system (a') to a user. The excess sludge generated from both systems is transported after dewatering to 80% water content into the sludge treatment system (c) for incineration and landfills disposal.

The functional unit (FU) is defined as the population equivalent (PE) in one year, that is, 1 FU = 1 PE·a. According to CUWA (2016), the volume of treated wastewater per PE per day was 0.154 m^3. Service life was defined as 20 years (Xiaodi H., 2019)

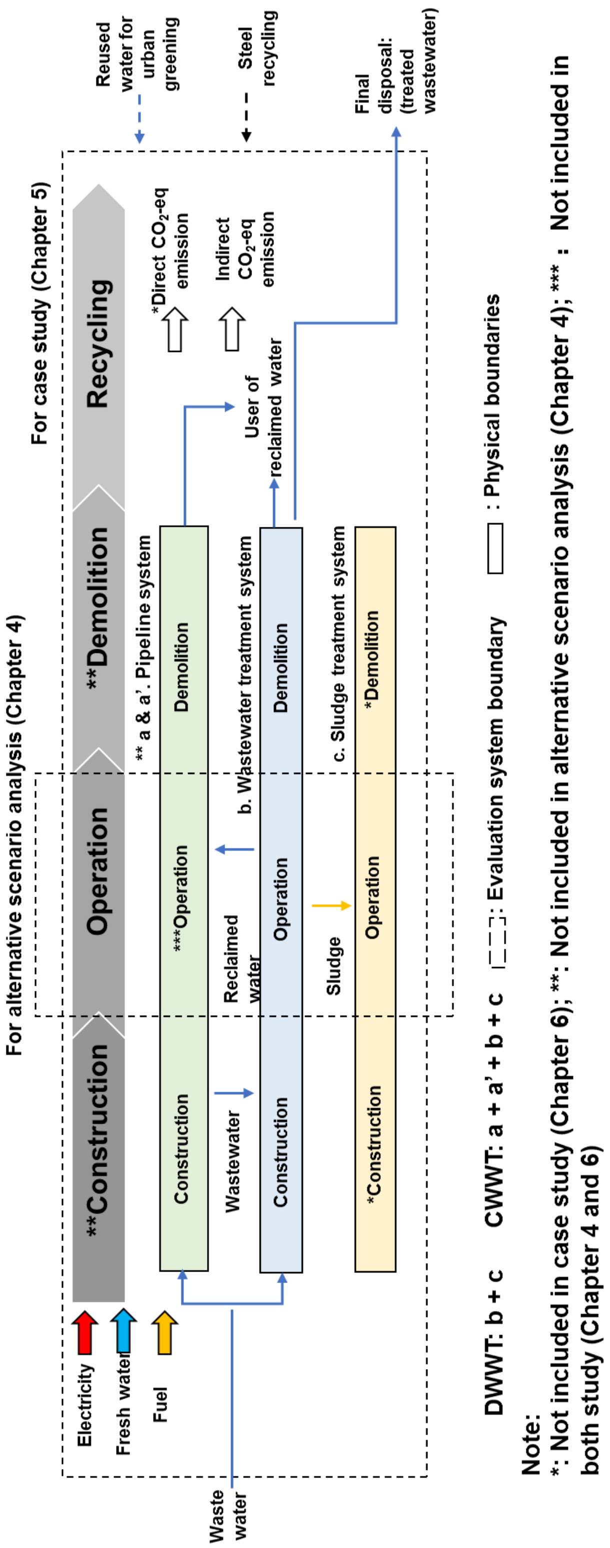


Figure 3.1 System boundaries for GHG emissions evaluation system in this study (covering Chapters 4 and 5)

3.1.3 LCI analysis and Life Cycle Impact Assessment (LCIA)

i) The process of analyzing and creating an inventory of the input and output data in a research system is known as inventory analysis. Quantitative accounting of different types and sources according to the accounting methodology for GHG emissions from UWS (see details in Chapter 3.2).

When quantifying the different GHG emissions, it is common to use GWP to convert the different GHG emissions into kilograms of CO_2 equivalent (CO_2-eq). This study evaluated three GHGs released by UWS: CO_2, N_2O, and CH_4, whose global warming potential (GWP) as CO_2-eq on a 100-year timescale are 1, 25, and 298, respectively (IPCC, 2007).

ii) The GHG emission evaluation indicators selected in this study include:

direct GHG emissions from wastewater treatment, indirect GHG emissions due to energy consumption of wastewater treatment, and indirect GHG emissions due to energy consumption of wastewater treatment. emissions from wastewater treatment, indirect GHG emissions from wastewater treatment chemicals consumption and GHG emissions from sludge treatment and disposal.

E_{CWWT}, E_{DWWT}, E_{WWTP_A}, E_{WWTP_B}, E_{sludge}, E'_{sludge}, $E_{pipeline}$, E_{pipe_WW} and E_{pipe_RW}: total GHG emissions from CWWT, DWWT, WWTP_A, WWTP_B, DWWT sludge treatment system, CWWT sludge treatment system, pipeline system, wastewater pipeline, and reclaimed water pipeline, respectively, $kgCO_2$-eq/FU;

E_{MC}, E_{MT} and E_{WT}: GHG emissions from construction materials consumption, materials transport and waste transport, respectively, $kgCO_2$-eq;

$E_{CO2/N2O,wastewater}$ E_{WWT}^{CO2}, E_{WWT}^{N2O}: direct CO_2 and N_2O emission from wastewater treatment, $kgCO_2$-eq/d;

$E_{indirect,wastewater}$ $E_{WWT}^{ind.}$, $E_{WWT}^{dir..}$: indirect, direct GHG emissions from wastewater treatment, $kgCO_2$-eq/d;

$E_{GHG,wastewater}$: GHG emissions from wastewater treatment, $kgCO_2$-eq/d;

$E_{GHG,land.}$, $E_{GHG,comp.}$, and $E_{GHG,comb.}$: GHGs emission rate from sludge landfills, composting, and incineration treatment process, $kgCO_2$-eq/d;

3.2 Life Cycle Inventory (LCI) analysis

3.2.1 Collection of data during construction phase

(1) pipeline system

i) construction materials

The estimated consumption of construction materials for the two types of pipes involved in this study (RCP and PCCP) are described in Equation 3.1a and 3.1b.

$$\boldsymbol{MC_{RCP}} = \sum_i \boldsymbol{A} \times \boldsymbol{L} \times \boldsymbol{\rho_i} \times \boldsymbol{P_i} = \sum_i [\boldsymbol{\pi}(\boldsymbol{D}/2 + \boldsymbol{t})^2 - \boldsymbol{\pi}(\boldsymbol{D}/2)^2] \times \boldsymbol{L} \times \boldsymbol{\rho_i} \times \boldsymbol{P_i} \quad \textbf{(eq. 3.1a)}$$

$$\boldsymbol{MC_{PCCP}} = \sum_i \boldsymbol{L} \times \boldsymbol{W_i} \times \boldsymbol{P_i} \quad \textbf{(eq. 3.1b)}$$

Where, MC_{RCP} and MC_{PCCP} are total materials consumption of pipeline system using RCP and PCCP (tons or m^3); A is area of the cross-section of the pipe (m^2); L is total length of pipeline system (m); ρ_i is density of building material i (t/m^3); P_i is the proportion of material i in pipes (100% concrete for RCP assumed in this study); W_i is weight per meter of PCCP (t/m); D is internal diameter of pipes (m); and t is wall thickness of pipes (m).

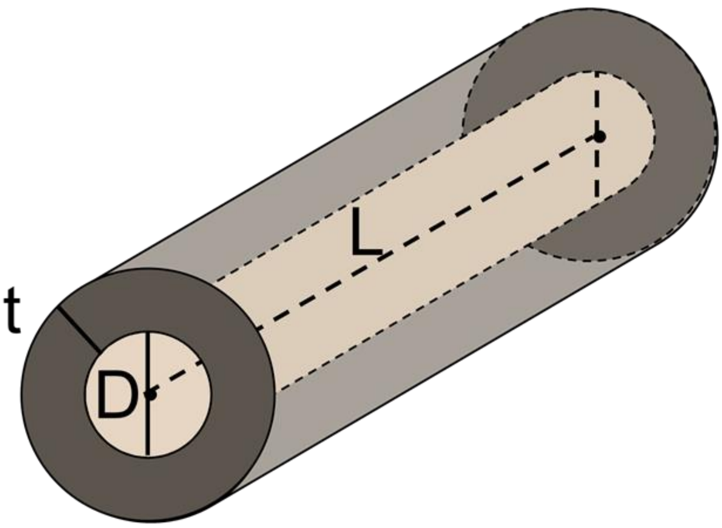


Figure 3.2 View of pipe cross section

Table 3.1Calculation parameters for PCCP (P_i and W_i)

		Diameter (D), m			
		1.676	1.829	2.134	2.743
*Proportion of materials (*Pi*)	Steel	2.93%	2.84%	2.81%	2.22%
	Cement	18.87%	18.12%	17.11%	13.72%
	Concrete	75.96%	76.90%	75.91%	81.01%
	Steel wire	2.23%	2.15%	4.17%	3.04%
**Weight per meter (*Wi*), t/m		2.09	2.41	3.13	4.97

* : obtained from Lalit Chilana et al., (2016)

** : obtained from GB/T 19685-2017

Table 3.2 Calculation parameters for RCP (*D* and *t*)

Diameter (D), m	Thickness of pipes (t), m	Area of the cross-section of the pipe (A), m^2
0.20	0.03	0.02
0.30	0.03	0.03
0.40	0.04	0.06
0.50	0.05	0.09
0.60	0.06	0.12
0.80	0.08	0.22
1.00	0.10	0.35
1.10	0.11	0.42
1.20	0.12	0.50
1.40	0.14	0.68
1.60	0.16	0.88
1.80	0.18	1.12
2.20	0.22	1.67
2.40	0.23	1.90
2.80	0.26	2.45
3.00	0.28	2.83
3.20	0.29	3.18
3.50	0.32	3.84

ii) construction work

GHG emissions during the construction operation of pipeline systems were calculated according to the empirical formulae for concrete pipes (250 mm ⩽ D ⩽ 1200 mm) in JSTT (2018).

Table 3.3 GHG emissions from construction work of sewer

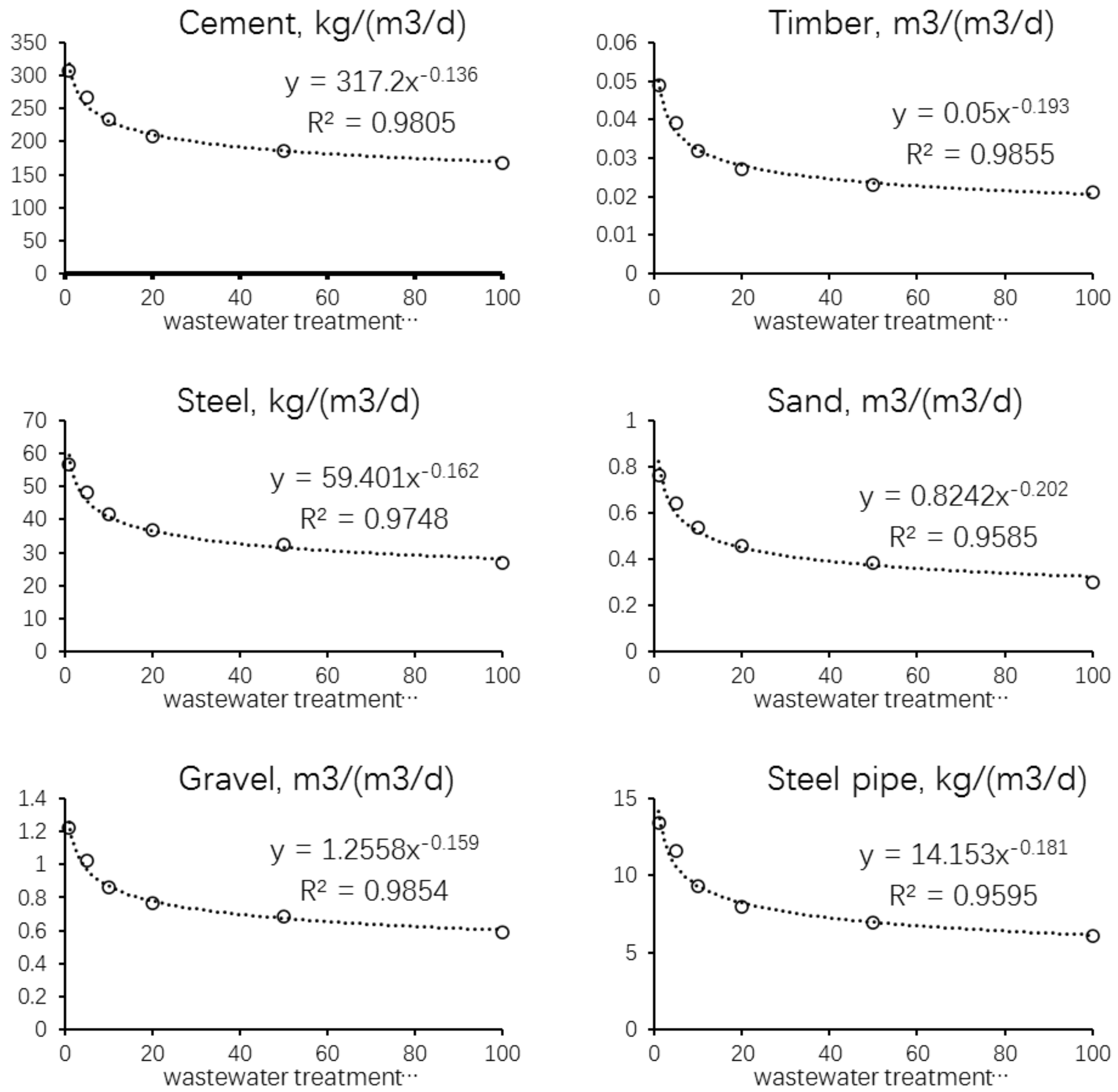
Construction methods		Pipe type	Empirical equations	Unit	**Range of variables	Coefficient of determination (R^2)
Excavation		PVC pipes	$y = -0.00008x^2 + 0.12621x + 39.12643$	Kg CO_2/m	150≤ D ≤600	0.9209
		RCP	$y = 0.00002x^2 + 0.03733x + 64.39136$		250≤ D ≤1200	0.9722
Pipe jacking	Small diameter	*Low load-bearing	$y = 0.0002x^2 - 0.0687x + 37.0390$		150≤ D ≤450	0.9451
		*High load-bearing	$y = 0.0001x^2 - 0.0145x + 29.4953$		250≤ D ≤600	0.9941
	Large diameter	RCP	$y = 0.00001x^2 + 0.1686x + 19.7026$		800≤ D ≤3000	0.9737
Shaft construction		Steel sheet pile	$y = 3547.5Ln(x) - 3411.1$	Kg CO_2	$(2.0m \times 2.0m)$ $4.5 \le De \le 19$	0.9841
			$y = -24.758x^2 + 1316.2x - 1052.3$		$(5.3m \times 2.9m)$ $4.5 \le De \le 19$	0.9852
			$y = -32.616x^2 + 1775.5x - 1514.7$		$(7.2m \times 3.6m)$ $4.5 \le De \le 19$	0.9864
			$y = -37.949x^2 + 2114.8x - 1844.4$		$(8.2m \times 4.4m)$ $4.5 \le De \le 19$	0.9875

RCP: reinforced concrete pipes
PVC: polyvinyl chloride
* Low load-bearing contains Rigid PVC pipes, etc.; High load-bearing contains RCPs, ductile iron pipes, ceramic pipes, etc.
** D: diameter of pipes, mm; De.: digging depth, m; and (Length × Width).

Source: JSTT (2018)

(2) WWTPs

i) construction materials consumption



Source: drawn and obtained from Wanxin Hou, etc., (2014)

Figure 3.3 Material intensity for structure WWTPs in different scales

This study considered the consumption of six construction materials in the construction of WWTPs, which are cement, timber, steel, sand, gravel, and metal (/steel) pipe. Hou et al. (2014) reported the consumption of these six materials for WWTPs with different implementation scale (10-1,000 thousand m^3/d). By collating the data, we drew Figure 3.3 and found a correlation between six material intensities and implementation scale.

ii) energy consumption from WWTPs construction

As shown in Table 3.4, the energy consumption of WWTPs construction come from civil works (excavation) and diesel/electricity consumption from WWT unit construction. This is expressed uniformly in units of kilograms of diesel, as shown in Table 3.4, with energy intensities of 1 and 2 kg for the construction of civil engineering and WWT units respectively.

Table 3.4 energy consumption of excavation work per machine-team

		* Quota for machine-team	* Energy consumption	Energy consumption	Energy consumption
		machine-team / 1000 m^3	kg-diesel / machine-team	kg-diesel / m^3	kg-diesel / m^3
Excavation work	Crawler-type single bucket mechanical excavators (1.5 m^3)	1.881	72.7	0.137	0.147
	Crawler bulldozers	0.188	56.5	0.011	
** Energy consumption from construction work of WWTPs is 1843070 kJ/m^2 (43.212 kg-disel/m^2).					

*: Zhejiang Standard & Cost (2018) and RISN (2015)

**: Zhang Q.H., (2010)

(3) sludge treatment unit

Not covered in this study.

(4) fuel consumption from transport

i) National average transport distance (km)

As shown in Table 3.2, two modes of transport are considered for construction materials, including railway and road, and chemicals, etc. transported by road only. The equipment during demolition phase was assumed that the steel can be recycled and transported 100 km by trunk.

ii) Energy consumption for transport

As shown in Table 3.3.

Table 3.5 National average transport distance

	National average transport distance (km)	
	*by train	**by trunk
Construction Materials		
Steel	1108	257
Cement	375	75
Gravel	312	74
Sand	312	74
Bricks	500	181
Metal pipe	500	181
Timber	1122	239
**Chemicals		
NaAc		260
NaClO		260
PAC		260
PAM		260
Fertilizer		161
***Waste		
Sludge		100
for recycling		100
Equipment		
**during construction		317
***during demolition		100

source:

* China Statistical Yearbook (2014);

** Ministry of Transport of China (2013)

*** Assumed in this study

Table 3.6 Energy consumption intensities of different transport modes

	Energy consumption for transport		
	* (g coal-eq./tkm)	** (g diesel/tkm)	*** (kgCO2/tkm)
Railway transport	3.7	2.5393	0.0079
Road transport	80.7	55.3840	0.1716
Inland waterway	6.8	4.6668	0.0145

* : adopted from GB/T50878-2013

** : calculated by 1.4571 t diesel/t coal-eq.

*** : calculated by 3.0981 kg CO2/ t diesel

3.2.2 Collection of data during operation phase

(1) pipeline system

Not covered in this study.

(2) WWT unit _ estimation of direct CO_2 emission

i) The direct CO2 emissions from aerobic oxidation of organic matter

In the biotreatment process, organic matter is oxidized by microorganisms (biomass) under aerobic conditions to produce CO_2. In this study, the organic matter was represented by $C_{10}H_{19}O_3N$ (Rittmann and McCarty, 2001), and the oxidation process of $C_{10}H_{19}O_3N$ can be described as:

$$2C_{10}H_{19}O_3N + 25O_2 \rightarrow 20CO_2 + 16H_2O + 2NH_3 \quad \text{(eq. 3.1)}$$

The ratio between O_2 and CO_2 is then $(25 \times 32)/(20 \times 44)$ which is $1/1.1$. Thus, a conversion factor is that 1.1 kg CO_2 every one kg oxygen (O_2) is produced (eq. 3.2).

$$\mathbf{E_{CO2,ae}} = \mathbf{1.1} \times \mathbf{E_{O2,ae}} \quad \text{(eq. 3.2)}$$

The total O_2 consumption in the aerobic process is used for the $C_{10}H_{19}O_3N$ oxidation and the growth of microorganisms (eq. 3.3) (Monteith HD et al., 2005).

$$\mathbf{E_{O2,ae}} = \frac{\mathbf{Q \times \Delta BOD}}{\mathbf{f_1}} - \mathbf{1.42} \times \mathbf{X_{aerobic}} \quad \text{(eq. 3.3)}$$

In which,

$$\mathbf{X_{aerobic}} = \mathbf{Y} \times \mathbf{Q} \times \mathbf{\Delta BOD} \quad \text{(eq. 3.4)}$$

$$\mathbf{\Delta BOD} = \mathbf{BOD_{inf}} - \mathbf{BOD_{eff}} \quad \text{(eq. 3.5a)}$$

When running data (input and output of BOD) is not available, it can also be equal to eq. 3.5b. The removal rate of BOD_5 (η_{BOD}) for a given technology (e.g. AAO) is fixed within a certain range, which can use technical manual / standard recommendations, statistical data of field investigation or empirical values from professional engineer. In addition, COD is a more common WWTPs' monitoring item compared to BOD_5 and can be converted according to eq. 3.5c for cases where BOD_5 data is not available.

$$\mathbf{\Delta BOD} = \frac{\eta_{\mathbf{BOD}}}{\mathbf{1} - \eta_{\mathbf{BOD}}} \times \mathbf{BOD_{eff}} \quad \text{(eq. 3.5b)}$$

$$\frac{\boldsymbol{BOD}}{\boldsymbol{COD}} = \boldsymbol{f_2} \quad \text{(eq. 3.5c)}$$

Thus, substituting equations 3.3, 3.4 and 3.5a (or 3.5b) into 3.2 gives CO_2 emissions from aerobic oxidation of organic matter equal to:

$$\mathbf{E_{CO2,ae}} = \mathbf{1.1} \times \left(\frac{\mathbf{1}}{\boldsymbol{f_1}} - \mathbf{1.42} \times \boldsymbol{Y}\right) \times \frac{\boldsymbol{\eta_{BOD}}}{\mathbf{1} - \boldsymbol{\eta_{BOD}}} \times \boldsymbol{BOD_{eff}} \times \boldsymbol{Q} \quad \text{(eq. 3.6)}$$

Where

$E_{CO2,ae}$: the CO_2 emissions rate from aerobic oxidation of organic matter, $kgCO_2$ / d;

$E_{O2,ae}$: the O_2 consumption rate from aerobic oxidation of organic matter, kgO_2 / d;

Q: the average daily flow, m^3 / d;

ΔBOD: the amount of BOD removal in biotreatment, kgBOD / m^3;

BOD_{eff}: the effluent BOD, kgBOD / m^3;

f_1: the ratio of BOD_5 and BOD_u;

f_2: the ratio of BOD_5 and COD;

η_{BOD}: the removal rate of BOD, %;

Y: cell-yield coefficient, kgVSS / kgBOD; and

$X_{aerobic}$: the net biomass produced per day, kgVSS / d.

ii) The CO_2 emissions from biomass endogenous decay

The biomass can be represented by the formula $C_5H_7O_2N$ (Rittmann and McCarty, 2001), the chemical reaction of biomass endogenous decay was described by:

$$\boldsymbol{C_5H_7O_2N + 5O_2 \rightarrow 5CO_2 + 2H_2O + NH_3} \quad \textbf{(eq. 3.7)}$$

The relationship reveals that 5 moles of CO_2 are released for every mole of biomass decay. The gram molecular weights of the biomass ($C_5H_7O_2N$) and CO_2 are 113 and 44 respectively. A conversion factor is that 1.947 kg CO_2 every one kg biomass decayed endogenously (Monteith HD et al., 2005). The CO_2 emissions arising from endogenous decay can be estimated from eq. 3.8 and eq. 3.9:

$$\mathbf{E_{CO2,de} = 1.947 \times X_{decay}} \quad \textbf{(eq. 3.8)}$$

$$\mathbf{X_{decay} = Q \times HRT \times MLVSS \times k_d} \quad \textbf{(eq. 3.9)}$$

Thus, the CO_2 emissions from endogenous decay equal to:

$$\mathbf{E_{CO2,de} = 1.947 \times k_d \times HRT \times MLVSS \times Q} \quad \textbf{(eq. 3.10)}$$

Where

$E_{CO2,de}$: the CO_2 emissions rate from endogenous decay, $kgCO_2$ / d;

X_{decay}: the biomass decay per day, kgVSS / d;

HRT: the hydraulic retention time, day;

MLVSS: the concentration of mixed liquid volatile suspended solids, kg / m^3; and

k_d: the endogenous decay coefficient, d^{-1}.

iii) The CO_2 emissions from nitrogen removal

The biological nitrogen removal process includes nitrification and denitrification, and most of the nitrogen source pollutants are present as the form of ammonia (NH_4^+) in the wastewater. The nitrification is that the conversion process from NH_4^+ to nitrate (NO_3^-), and the denitrification is that the conversion process from NO_3^- to nitrite (NO_2^-). The nitrification process and the denitrification process can be described by eq. 3.11a and eq. 3.11b, respectively.

Nitrification:

$$\boldsymbol{20CO_2 + 14NH_4^+ \rightarrow 10NO_3^- + 4C_5H_7O_2N + 24H^+ + 2H_2O} \quad \textbf{(eq. 3.11a)}$$

Denitrification:

$$\boldsymbol{C_{10}H_{19}O_3N + 0.5HCO_3^- + 0.5NH_4^+ + 4.8NO_3^- + 4.8H^+ \rightarrow 26C_5H_7O_2N + 40CO_2 + 2.4N_2 + 7.9H_2O} \quad \textbf{(eq. 3.11b)}$$

As described in eq. 3.11a, the CO_2 is absorbed and fixed during the nitrification. However, as described in eq. 3.11b, the CO_2 produced during denitrification (eq. 3.11b) is not calculated because organic matter ($C_{10}H_{19}O_3N$) is oxidized as an electron donor during denitrification. Therefore, the calculation of the CO_2 produced is already included in the calculation for organic matter ($C_{10}H_{19}O_3N$) oxidation (eq. 3.1).

The relationship (eq. 3.11a) reveals that 20 moles of CO_2 are consumed 14 moles of ammonium ion (NH_4^+). The gram molecular weights of the $C_5H_7O_2N$, CO_2 and N are 113, 44 and 14, respectively. A conversion factor is that 4.49 kg CO_2 every one kg oxidized nitrogen (shown as eq. 3.12). The CO_2 emissions arising from nitrification can be estimated from eq. 3.11, eq. 3.12 and eq. 3.13:

$$\mathbf{E_{CO2,N} = 4.49 \times S_{N,nitrified}} \quad \textbf{(eq. 3.12)}$$

$$\mathbf{S_{N,nitrified} = Q \times \Delta N - X_{N,biomass}} \quad \textbf{(eq. 3.13)}$$

$$\boldsymbol{\Delta N = N_{inf} - N_{eff}} \quad \textbf{(eq. 3.14a)}$$

$$\boldsymbol{\Delta N = \frac{\eta_N}{1-\eta_N} \times N_{eff}} \quad \textbf{(eq. 3.14b)}$$

Thus,

$$\mathbf{E_{CO2,N}} = \boldsymbol{4.49\left(\frac{\eta_N}{1-\eta_N} \times N_{eff} \times Q - X_{N,biomass}\right)} \quad \textbf{(eq. 3.15)}$$

Where

$E_{CO2,N}$: the CO_2 fixation rate from nitrification, kgCO_2 / d;

$S_{N,nitrified}$: the ammonia is nitrified per day, kgN /d;

ΔN: the amount of nitrogen removal in biotreatment, kgN / m^3;

N_{eff}: the effluent nitrogen, kgN / m^3;

η_N: the removal rate of nitrogen, %; and

$X_{N,biomass}$: the amount of nitrogen in the biomass, kgN / kgVSS.

According eq. 3.6, eq. 3.10, and eq. 3.15, the estimation of CO_2 generation from wastewater treatment process can be described by eq. 3.16:

$$\mathbf{E_{WWT,dir.CO2}} = \mathbf{E_{CO2,ae}} + \mathbf{E_{CO2,de}} - \mathbf{E_{CO2,N}} = \boldsymbol{\left\{\left[1.1 \times \frac{\eta_{BOD}}{1-\eta_{BOD}} \times BOD_{eff} \times Q \times \left(\frac{1}{f_1} - 1.42\,Y\right)\right] + \left[1.947 \times Q \times HRT \times MLVSS \times k_d\right] - \left[4.49\left(\frac{\eta_N}{1-\eta_N} \times N_{eff} \times Q - X_{N,biomass}\right)\right]\right\} \times GWP_{CO2}} \quad \textbf{(eq. 3.16)}$$

Hence, two values of direct CO2 emission intensity can be obtained:

$$\boldsymbol{EF_{WWT,dir.CO2}} = \mathbf{E_{WWT,dir.CO2}}/\boldsymbol{Q} \quad \textbf{(eq. 3.17a)}$$

Or

$$\boldsymbol{EF'_{WWT,dir.CO2}} = \mathbf{E_{WWT,dir.CO2}}/(\mathbf{\Delta BOD \times Q}) \quad \textbf{(eq. 3.17b)}$$

where

$E_{WWT,\,dir.\,CO2}$: the CO_2 direct emission rate from wastewater treatment process, kg CO_2-eq. /d;

$EF_{WWT,dir.CO2}$: direct CO_2 emission intensity from wastewater treatment basing volume of treating wastewater, kg CO_2-eq./m^3;

$EF'_{WWT,dir.CO2}$: direct CO_2 emission intensity from wastewater treatment basing BOD removal, kg CO_2-eq./kg BOD.

(3) WWT unit _ estimation of direct N_2O and CH_4 emission

i) methods

The global warming potentials (GWPs) (over 100 years) of CH_4 and N_2O are 25 times and 298 times that of CO_2, respectively (IPCC AR4 2007). The emissions of CH_4 and N_2O were converted into carbon dioxide equivalents (CO_2-eq.) by GWP to estimate GHG emissions.

The N_2O emission during the sewage treatment occurred in the biological nitrogen removal process that mainly consisted of nitrification process and denitrification process. Previous study found that N_2O was not only generated as an intermediate product in the denitrification process but was also generated as a by-product in the nitrification process (Kampschreur M. J. et al., 2009 and Ma B. et al., 2016). The mechanism of N_2O production as an intermediate product and by-product is complicated, as it is affected by enzyme inactivation, accumulation of NO_2^-, and reaction condition (such as pH, DO, and C/N) (Jeffrey Foley et al., 2009, Maite Pijuan et al., 2014, and Theoni Maria Massara et al., 2017). The N_2O emission from wastewater treatment was calculated, as shown in eq. 3.18.

$$\mathbf{E_{WWT,dir.N2O}} = \mathbf{Q} \times \mathbf{\Delta N} \times \boldsymbol{EF_{WWT,dir.N2O}} \times \mathbf{GWP_{N2O}} \quad \textbf{(eq. 3.18)}$$

It is commonly assumed that CH_4 is produced under anaerobic conditions, only in anaerobic digesters, and is usually collected. However, previous studies have noted that CH_4 is produced during transport of wastewater within the pipeline and is released to the atmosphere with the aeration process. The CH_4 emission from wastewater treatment was calculated, as shown in eq. 3.19.

$$\mathbf{E_{WWT,dir.CH4}} = \mathbf{Q} \times \mathbf{\Delta COD} \times \boldsymbol{EF_{WWT,dir.CH4}} \times \mathbf{GWP_{CH4}} \quad \textbf{(eq. 3.19)}$$

where

$E_{WWT,dir.N2O}$: the N_2O direct emission rate from wastewater treatment process, kg CO_2-eq. /d;

$E_{WWT,dir.CH4}$: the CH_4 direct emission rate from wastewater treatment process, kg CO_2-eq. /d;

$EF_{WWT,dir.N2O}$: direct N_2O emission intensity from wastewater treatment, kg N_2O/kg N_removal;

$EF_{WWT,dir.CH4}$: direct CH_4 emission intensity from wastewater treatment, kg CH_4/kg COD_removal.

ii) emission factors (EF)

Table 3.7 Summary of the direct emission factor from previous studies

GHGs	value of EF	unit	Ref.
CO_2	0.2-0.5	kg CO_2/kg COD	Junxin, 2014
	1.375	kg CO_2/kg BODu	Cakir & Stenstrom, 2005
	0.6875	kg CO_2/kg BODu	Cakir & Stenstrom, 2005
CH_4	0.001-0.004	kg CH_4/kg COD	Junxin, 2014
	0.4	m^3 CH_4/kg COD_rmv	Tchobanoglous, 2003
	0.25	kg CH_4/kg BODu	Cakir & Stenstrom, 2005
	0.104	kg CH_4/kg COD_ana	Sahely et al., 2006
	0.36-0.41	kg CH_4/kg BOD5_ana	Sahely et al., 2006
N_2O	0.001-0.0009	kg N_2O/kg TN	Junxin, 2014
	0.01	kg N_2O/kg N_inf.	Foley&Lant , 2008
	0.006-0.253	kg N_2O/kg N_denitrified	Foley et al., 2010

Source: Chai (2017)

iii) Wastewater quality analysis of WWTPs

Wastewater quality analysis includes BOD, COD, ammonia nitrogen (NH_4^+-N), total nitrogen (TN), total phosphorus (TP) and total suspended solids (SS), all of which are routinely monitored in WWTPs. The Table 3.8 shows the monthly average survey results for Jinnan WWTP for the period January to December 2015.

Table 3.8 Wastewater quality analysis result from Jinnan WWTP

	Flow rate per day	inf. consentration						eff. consentration					
		COD	BOD	NH4+	SS	TN	TP	COD	BOD	NH4+	SS	TN	TP
	m3/d	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L
January	460000	537	233	50	394	61	6.0	36	6	1.1	5.0	11.1	0.3
February	447000	389	147	41	170	45	4.3	33	4	0.6	4.0	12.3	0.2
March	471000	398	158	42	194	45	4.0	37	5	0.5	5.0	12.2	0.2
April	516000	461	183	43	195	46	4.3	27	5	1.0	4.0	8.4	0.1
May	525000	391	164	41	190	44	4.2	33	5	0.7	5.0	12.4	0.1
June	495000	353	141	42	225	45	3.8	31	5	0.3	6.0	13.0	0.1
July	514000	357	143	43	183	46	3.8	28	4	0.6	6.0	11.4	0.2
August	499000	367	148	38	189	41	4.2	29	5	0.8	6.0	12.1	0.2
September	519000	279	119	31	180	37	3.8	23	4	0.2	6.0	11.5	0.2
October	490000	329	134	37	197	41	3.8	25	5	0.3	7.0	11.7	0.3
November	526000	383	153	39	201	43	4.5	29	5	0.3	7.0	12.4	0.2
December	524000	394	156	38	231	43	6.3	31	5	0.4	7.0	12.3	0.2
ave.	49.5	386.5	156.6	40.3	212.4	44.7	4.4	30.1	4.8	0.6	5.7	11.7	0.2

Source: EIA Report of Jinnan WWTP (2015)

(4) WWT unit _ estimation of indirect GHG emission

i) electricity, chemicals, and water consumption

The Table 3.9 shows the electricity consumption from equipment nameplate information for the Nankai WWTP, the equipment list is in Appendix Table S-1; the Table 3.10 shows the resource consumption for the two WWTPs. In addition, the equipment list for the Jinan WWTP is in Appendix Table S-2.

Table 3.9 Electricity consumption from the equipment nameplate of the Nankai WWTP

	Design power kw	Actual power kw	Ratio
Primary treatment	9.15	7.65	0.076
Secondary treatment	48.5	44.1	0.440
Advanced treatment	61.45	39.95	0.399
Sludge pre-treatment	8.85	8.42	0.084
Total	127.95	100.12	1.000

Table 3.10 Resource consumption of Nankai WWTP and Jinnan WWTP

		Nankai WWTP	Jinnan WWTP
Fresh water	t/a	500	2883
Reclaimed water	t/a		101470000
Electricity	kWh/a	6568	71060000
Chemicals			
NaAc	t/a		916
NaClO	t/a	1	
PAC	t/a	3	247
PAM	t/a	0	86

iv) transport

Referring to Table 3.5, the transport distance for chemical is taken to be 260 km by trunk, the transport distance for sludge is assumed to be 100 km in this study, and the energy consumption of water transported through pipelines is not considered.

(5) sludge treatment unit

The sludge is dewatered and thickened at the WWTPs, which we define as the sludge pre-treatment, and the electricity and chemical (PAM) consumption are shown in Table 3.9 and

Table 3.10 respectively.

3.2.3 Collection of data during demolition phase

The demolition phase consists of GHG emissions from demolition works, waste transport, and waste disposal. Steel can be recycled to reduce raw material production, and the recycling rate can reach 0.38 t/t and can save 60% of energy during production (Xiaodi Hao et al., 2019).

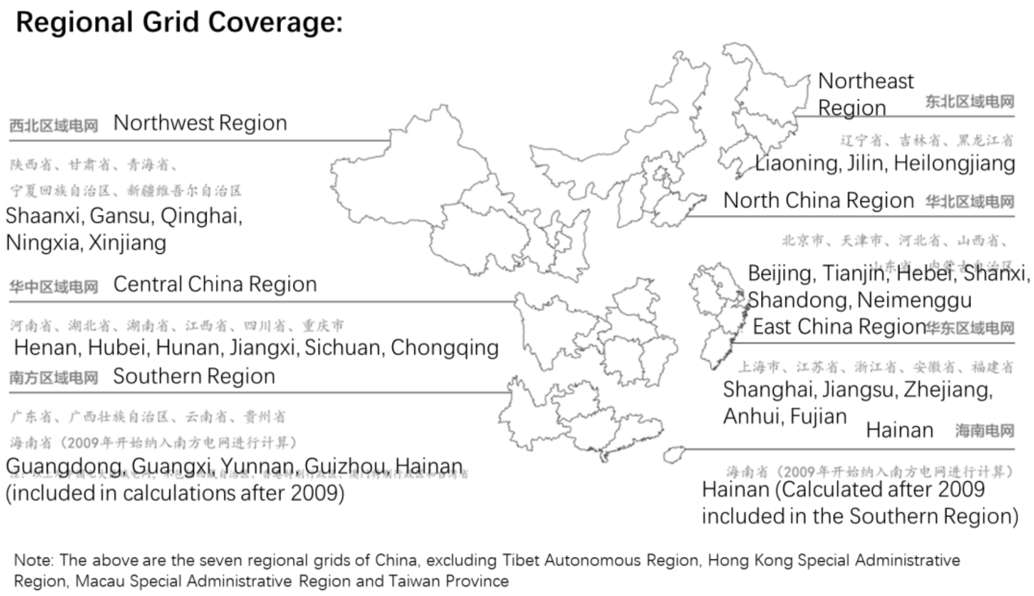
3.2.4 Sources of GHGs emission factor

(1) electricity

China's grid boundaries are uniformly divided into North China, Northeast China, East China, Central China, Northwest China, and South China regional grids. The geographical area included in the above grid boundaries is shown in Figure 3.4. Table 3.11 indicates the emission factors of each regional grid from 2010 to 2012.

Table 3.11 Average CO_2 emission factor (EF) for regional grids in China (kg CO_2/kWh)

	2010	2011	2012
North China Region	0.8845	0.8967	0.8843
Northeast Region	0.8045	0.8189	0.7769
East China Region	0.7182	0.7129	0.7035
Central China Region	0.5676	0.5955	0.5257
Northwest Region	0.6958	0.686	0.6671
Southern Region	0.596	0.5748	0.5271



Source: Song Ranping, et al. (2013)

Figure 3.4 Regional grid coverage in China

(2) fuel

The fuel used in this study is diesel, and its emission factor can be calculated according to Table 3.12, with a value of 3.1 kg CO_2/kg.

Table 3.12 The CO_2 emission factor (EF) of diesel in China (kg CO_2/kg)

* Carbon Emission factor of diesel		** Low-level heating value	CO2 Emission factor of diesel
tC/TJ	tCO2/J	J/kg	kgCO2/kg
1.98E+01	7.26E-11	4.27E+07	3.10E+00
*: China Greenhouse Gas Inventory (2005)		**: China Energy Statistics Yearbook (2008)	

(3) chemicals

The chemicals used in this study and the sources of their emission factors are shown in Table 3.13.

Table 3.13 The CO_2 emission factor (EF) of chemicals consumption (kg CO_2/kg)

Item	Unit	Value	
Chemicals			
NaAc	kgCO2-eq/kg	1.5702	City of Winnipeg (2012)
PAC	kgCO2-eq/kg	0.0227	Chai et al. (2015)
NaClO	kgCO2-eq/kg	0.92	City of Winnipeg (2012)
O3 (10%, liquid)	kgCO2-eq/kg	8.01	City of Winnipeg (2012)
PAM	kgCO2-eq/kg	1.5	Chai et al. (2015)

(4) construction materials, sludge treatment, and alternatives

The construction materials, sludge treatment, and alternatives used in this study and the sources of their emission factors are shown in Table 3.14.

Table 3.14 The CO_2 emission factor (EF) of construction materials, sludge treatment, and alternatives (kg CO_2/kg)

Item	Unit	Value	Source
Materials			
Steel	kgCO2-eq/t	2600	Tao (2015)
Cement	kgCO2-eq/t	730	Tao (2015)
Gravel	kgCO2-eq/t	2	Tao (2015)
Sand	kgCO2-eq/t	2	Tao (2015)
Concrete	kgCO2-eq/m3	350	Tao (2015)
Fresh water	kgCO2-eq/t	0.3	Tao (2015)
Steel pipe	kgCO2-eq/t	15100	Qian et al. (2019)
Timber	kgCO2-eq/t	0.025806	Qian et al. (2019)
Waste			
Sludge incineration (80% w.c.)	kgCO2-eq/t	1318.47	Beibei et al. (2013)
Production			
Steel recycling	kgCO2-eq/t	1560	Xiaodi et al. (2019)
Reclaimed water	kgCO2-eq/t	0.3	Xiaodi et al. (2019)

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4. SCENARIO ANALYSIS FOR THE ALTERNATIVE TREATMENT TECHNOLOGIES OF WASTEWATER AND SLUDGE DURING OPERATION PHASE

4.1 Boundary definition

The wastewater-sludge treatment system receives domestic wastewater as well as discharge treated wastewater and sludge. The wastewater-sludge treatment is a complex reaction system involved a series of biological treatment. In its Second Assessment Report (1997), the IPCC considers that the carbon in BOD converts only into CH_4, whereas in the Fourth Assessment Report by IPCC (2007), CO_2 generated from biomass decay is not considered a part of GHG emissions. In the case of GHG emission accounting, some studies state that electric energy consumption should be counted as a part of the energy sector rather than wastewater-sludge treatment system. The influent BOD converts into CO_2 and biomass, whereas CH_4 is generated only during sludge anaerobic digestion. Direct GHG emissions are generated by the treatment of wastewater and sludge. Indirect GHG emissions are generated by the consumption of chemicals and electric energy in the treatment process. The evaluation boundaries are shown in Figure 4.1, wherein the tetragonal broken line refers to boundary of GHG emissions, tetragonal solid line refers to the treatment process, and the oval solid line refers to materials coming in/getting out the boundary.

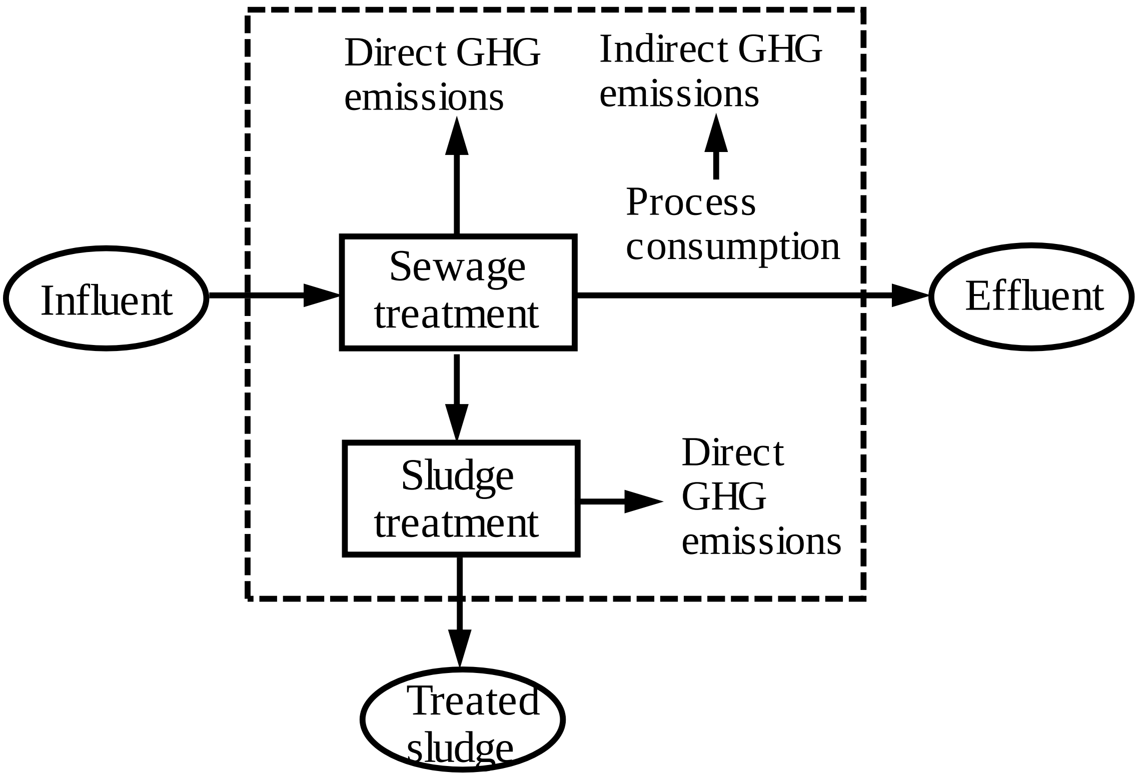

Figure 4.1: The boundary of GHG emissions from wastewater and sludge unit during operation phase for alternative scenario analysis

4.2 Alternative scenario introduction

4.2.1 Selection of wastewater treatment process

According to the List of National Urban Wastewater Treatment Facilities published by the Ministry of Environmental Protection (MEPPRC, 2015), the statistical results of 4,437 operating WWTPs are shown in Table. 1. The wastewater treatment processes in descending order of quantity were Anaerobic/Anoxic/Oxic (AAO), Oxidation Ditch (OD), Sequencing Batch Reactor (SBR), and Anoxic/Oxic (AO); in descending order of wastewater treatment capacity, they include AAO, OD, AO, and SBR. Therefore, three typical wastewater treatment processes, namely AAO, OD, and SBR were selected. AO was eliminated as it is similar to AAO. The sum of the three analyzed processes accounted for 71.8% of the total treated water, and the average daily treated water accounted for 74.3% of the total treated water.

Table 4.1: Statistics on different sewage treatment processes of operating WWTPs

	Number of WWTPs	Ratio *	Capacity ($\times 10^6 m^3/d$)	Ratio **
AAO	1167	26.31%	50.50	37.35%
OD	1161	26.17%	32.65	24.15%
SBR	857	19.32%	17.32	12.81%
AO	673	15.17%	18.74	13.86%
Others	579	13.05%	16.00	11.84%
Total	4437	100.00%	135.22	100.00%

Note:
* The ratio of WWTPs using different sewage treatment processes to the total WWTPs
** The treatment capacity of WWTPs using different treatment processes accounts for the proportion of total treatment capacity of total WWTPs

Table 4.2 Alternative scenario introduction

		ST technology options		
		landfill	composting	incineration
WWT technology options	AAO	S_1	S_2	S_3
	OD	S_4	S_5	S_6
	SBR	S_7	S_8	S_9

4.2.2 Selection of sludge treatment process

According to the Guideline on Best Available Technologies of Pollution Prevention and Control for Treatment and Disposal of Sludge from Municipal Wastewater Treatment Plant (MEPPRC, 2010), three typical sludge treatment processes (landfills, composting, and incineration) were analyzed.

4.3 Data sources for estimating

This study analyzed the theoretical estimation of GHG emissions from wastewater-sludge treatment scenarios in China. The estimated input data were obtained from national/industrial standards, different technical guides, and environment assessment reports. Some parameters used in this study were shown in Table. 4.3.

A 40,000 m^3/day wastewater treatment capacity was used to analyze different scenarios, as influent flow rates of small-scale WWTPs (≤ 40,000 m^3/day) (MOHURD, 2006) account for 81.1% of the total WWTPs (MEPPRC 2015), which is the mainstream treatment capacity of WWTPs built in China.

The effluent, which reflects the water quality of treated wastewater, should be under strict control before being discharged into natural waters, such as rivers and lakes. The highest discharge standard of WWTPs in China, that is 1A-level standard (Table. 4.3), was used for GHG emissions calculation.

Table 4.3 The assumed model parameter used in this study

Parameter	Symbol	Unit	Value uesd		
			AAO	OD	SBR
Flow rate	Q	m^3 / d	40,000		
Effluent [b]	BOD_{eff}	kgBOD / m^3	0.01		
	COD_{eff}	kgCOD / m^3	0.05		
	SS_{eff}	kgSS / m^3	0.01		
	N_{eff}	kgN / m^3	0.015		
	P_{eff}	kgP / m^3	0.0005		
MLSS	-	kg / m^3	2.0 ~ 4.5 [a1]	2.0 ~ 4.5 [a2]	2.5 ~ 4.5 [a3]
MLVSS	-	kg / m^3	3.4 [c]	2.9 [c]	3.4 [c]
Ratio of BOD_5 and BOD_u	f_1	-	0.68 (Metcalf et al. (21991))		
Ratio of MLVSS and MLSS	f_2	-	0.5 ~ 0.75 [a1]	0.5 ~ 0.65 [a2]	0.75 (Fan et al. (2015))
Ratio of MLSS and SS	f_3	-	0.7 (MEPPRC (2010))		
Cell-yield coefficient	Y	kg VSS / kg BOD	0.68 (WET (1998))		
Endogenous decay coefficient	k_d	1 / d	0.05 (WET (1998))		
Hydraulic retention time	HRT	day	0.46 ~ 0.75 [a1]	0.33 ~ 0.75 [a2]	0.83 ~ 1.25 [a3]
BOD removal rate	η_{BOD}	%	92.0 [d] (85 ~ 95) [a1]	97.1 [e] (85 ~ 95) [a2]	92.3 [f] (85 ~ 95) [a3]
Nitrogen removal rate	η_N	%	86.0 [d] (55 ~ 80) [a1]	78.6 [e] (55 ~ 80) [a2]	57.1 [f] (55 ~ 80) [a3]
Phosphorus removal rate	η_P	%	65.0 [d] (60 ~ 80) [a1]	93.75 [e] (50 ~ 75) [a2]	83.3 [f] (50 ~ 75) [a3]
SS removal rate	η_{SS}	%	87.0 [d]	97.5 [e]	93.3 [f]
Sludge production [g]	$X_{tre.}$	kgDS / m^3	0.5 [d]	1.32 [e]	0.2 [f]
Amount of nitrogen in the biomass	$X_{N,biomass}$	kgN / kgVSS	0.122 (Hiatt et al. (2008))		

[a1], [a2], and [a3] Technical Specifications for AAO (HJ 576-2010), OD (HJ 578-2010), and SBR (HJ 577-2010)

[b] Discharge Standard of Pollutants for WWTPs (GB18918-2002)

[c] calculated when the upper limit of the standard range was selected

[d], [e], and [f] Environmental Impact Assessment Report of different WWTP pubulished online

[g] 60% water content

4.4 Estimation method of GHG emissions from alternative scenarios

The detailed methodological description of the calculation is given in Chapter 3.

The wastewater-sludge treatment system includes a wastewater treatment process and a sludge treatment process; furthermore, GHGs can be classified into direct emissions and indirect emissions based on different emission sources. Direct emissions of GHGs include CO_2 converted by organic matter in the biotreatment process, CH_4 emitted during the anaerobic process and sludge treatment, and N_2O emitted during biological nitrogen removal. Indirect emissions of GHGs mainly include electricity consumption of mechanical equipment (such as lifting unit, aeration unit, and sludge treatment unit) and chemicals consumption (such as PAC and PAM) during treatment process.

The estimation method of direct emissions of CO_2 and N_2O was based on mass balance and active sludge/anaerobic digester model (AS/AD). Indirect GHG emissions from wastewater treatment and GHG emissions from sludge treatment were estimated by GHG emission factor method, and the emission factors used are shown in Table 4.4.

Table 4.4 The GHG emissions factor calculated in scenario analysis

Item	Emission factor	Value	Unit	Reference
Direct emission				
Sewage treatment process				
N_2O	EF_{N2O}	0.253[a]	Kg N_2O/kg $N_{denitrified}$	Foley et al. (2010)
Sludge treatment process				
Landfills	$EF_{land.,ex.N2O}$	0.042	Kg CO_2-eq./kg DS	Liu et al. (2013)
	$EF_{land.,N2O}$	0.951	Kg CO_2-eq./kg DS	De et al. (2008)
Composting	$EF_{comp.,ex.N2O}$	0.493	Kg CO_2-eq./kg DS	Liu et al. (2013)
	$EF_{comp.,N2O}$	0.656	Kg CO_2-eq./kg DS	Foley et al. (2008)
Incineration	$EF_{comb.}$	0.444	Kg CO_2-eq./kg DS	Peng et al. (2013)
Indirect emission				
Chemicals consumption				
PAM	EF_{PAM}	1.5	kg CO_2-eq./kg PAM	Carr, M. (2007)
PAC	EF_{PAC}	0.023	Kg CO_2-eq./kg PAC	Sharaai et al. (2012)
Energy consumption				
Electricity	$EF_{elec.}$	0.681	Kg CO_2-eq./kWh	NDRC (2014)
Diesel fuel	EF_{diesel}	3.261[b]	Kg CO_2-eq./kg	IPCC (2007) and NBSC (2016)

[a] In order to get the maximum GHG emissions, the upper limit is selected.

[b] calculated when diesel fuel density is 0.84kg / L

4.5 Results and discussions

4.5.1 GHG emissions of different wastewater-sludge treatment scenarios

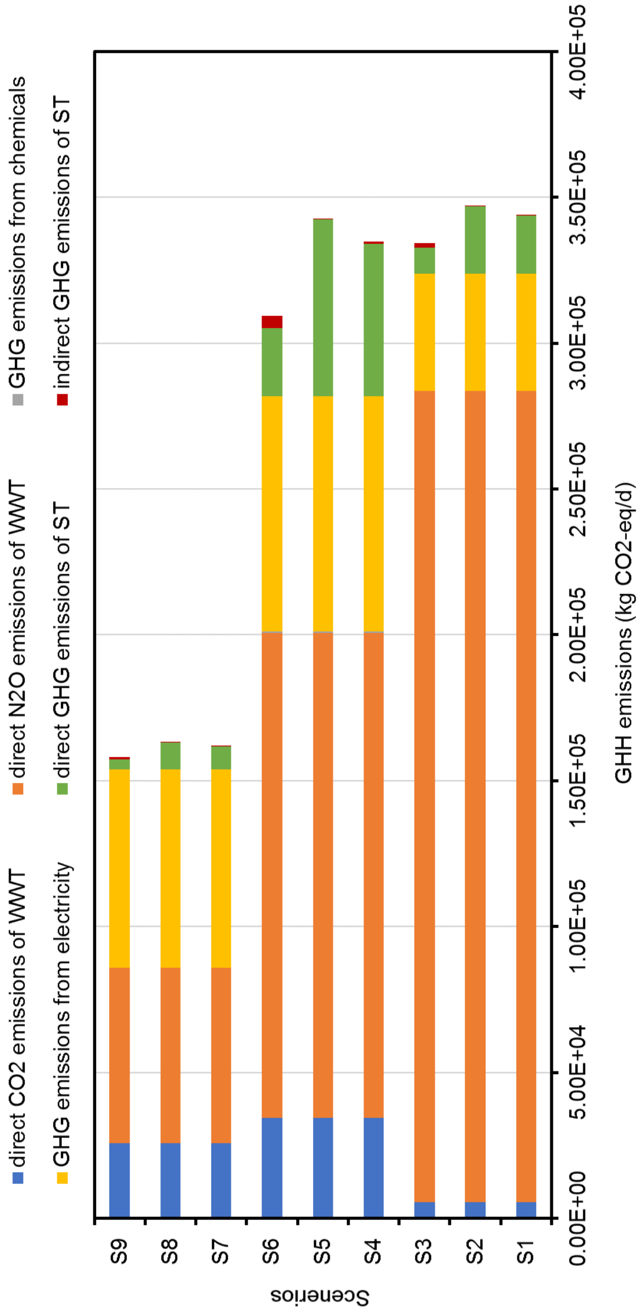

Figure 4.2 GHG emissions sources from the nine alternative technological scenarios of wastewater-sludge treatment (kg CO_2-eq./d)

Table 4.5 GHG emissions from the nine alternative technological scenarios of wastewater-sludge treatment (kg CO_2-eq./d)

			total										
				sewage treatment							sludge treatment		
					direct GHG emissions			indirect GHG emissions				direct GHG emissions	indirect GHG emissions
						CO2	N2O		chemicals	electricity			
AAO	landfill	S1	3.44E+05	3.24E+05	2.84E+05	5.64E+03	2.78E+05	4.03E+04	1.65E+01	4.03E+04	2.01E+04	1.99E+04	2.74E+02
AAO	comp.	S2	3.47E+05	3.24E+05	2.84E+05	5.64E+03	2.78E+05	4.03E+04	1.65E+01	4.03E+04	2.30E+04	2.30E+04	1.43E+01
AAO	comb.	S3	3.34E+05	3.24E+05	2.84E+05	5.64E+03	2.78E+05	4.03E+04	1.65E+01	4.03E+04	1.05E+04	8.88E+03	1.62E+03
OD	landfill	S4	3.35E+05	2.82E+05	2.01E+05	3.45E+04	1.66E+05	8.10E+04	4.10E+02	8.06E+04	5.32E+04	5.24E+04	7.23E+02
OD	comp.	S5	3.42E+05	2.82E+05	2.01E+05	3.45E+04	1.66E+05	8.10E+04	4.10E+02	8.06E+04	6.07E+04	6.07E+04	3.79E+01
OD	comb.	S6	3.09E+05	2.82E+05	2.01E+05	3.45E+04	1.66E+05	8.10E+04	4.10E+02	8.06E+04	2.77E+04	2.34E+04	4.29E+03
SBR	landfill	S7	1.62E+05	1.54E+05	8.59E+04	2.57E+04	6.02E+04	6.79E+04	3.86E+01	6.78E+04	8.05E+03	7.94E+03	1.10E+02
SBR	comp.	S8	1.63E+05	1.54E+05	8.59E+04	2.57E+04	6.02E+04	6.79E+04	3.86E+01	6.78E+04	9.20E+03	9.19E+03	5.74E+00
SBR	comb.	S9	1.58E+05	1.54E+05	8.59E+04	2.57E+04	6.02E+04	6.79E+04	3.86E+01	6.78E+04	4.20E+03	3.55E+03	6.49E+02

GHG emissions from different sources for the nine wastewater-sludge treatment scenarios (S1 to S9) are shown in Table 4.5. The GHG emission ranges (with different sludge scenarios) of SBR, AAO, and OD are 58-60 kt CO_2-eq. per year, 122-127 kt CO_2-eq. per year, and 113-125 kt CO_2-eq. per year, respectively. The direct GHG emissions of SBR (33.87 kt CO_2-eq. per year) are much less than AAO (109.78 kt CO_2-eq. per year) and OD (89.86 kt CO_2-eq. per year), while the indirect GHG emissions are similar, namely 24.86 kt CO_2-eq. per year (SBR), 14.94 kt CO_2-eq. per year (AAO), and 30.18 kt CO_2-eq. per year (OD). The ratio of direct to total GHG emissions were calculated to be 88% (AAO), 75% (OD), and 58% (SBR). The contribution of GHG emissions from wastewater treatment accounted for many of the total emissions, which were 94.8% for AAO, 85.8% for OD, and 95.6% for SBR.

In the nine wastewater-sludge treatment scenarios, SBR-Incineration (S9) scenario had the least amount of GHG emissions, while the AAO-Composting (S2) scenario had the most GHG emissions. The total GHG emissions, in descending order, were S2, S1, S5, S4, S3, S6, S8, S7, and S9. The total emissions of SBR scenario were less than AAO and OD, even under different sludge treatment scenarios.

The ratio of GHG emissions from sludge treatment and total emissions were approximately 5.2% (AAO), 14.2% (OD), and 4.4% (SBR). GHG emissions from different sludge treatment scenarios (same wastewater treatment), in descending order, are composting, landfills, and incineration. The reduction rate of GHG emissions under incineration scenario, when compared to landfills were 2.8% for AAO, 7.6% for OD, and 2.4% for SBR; when compared to composting were 3.6% for AAO, 9.6% for OD, and 3.1% for SBR. Therefore, the effect of sludge treatment process selection in reducing GHG emissions is positive, without changing the wastewater treatment process.

4.5.2 GHG emissions from different sources

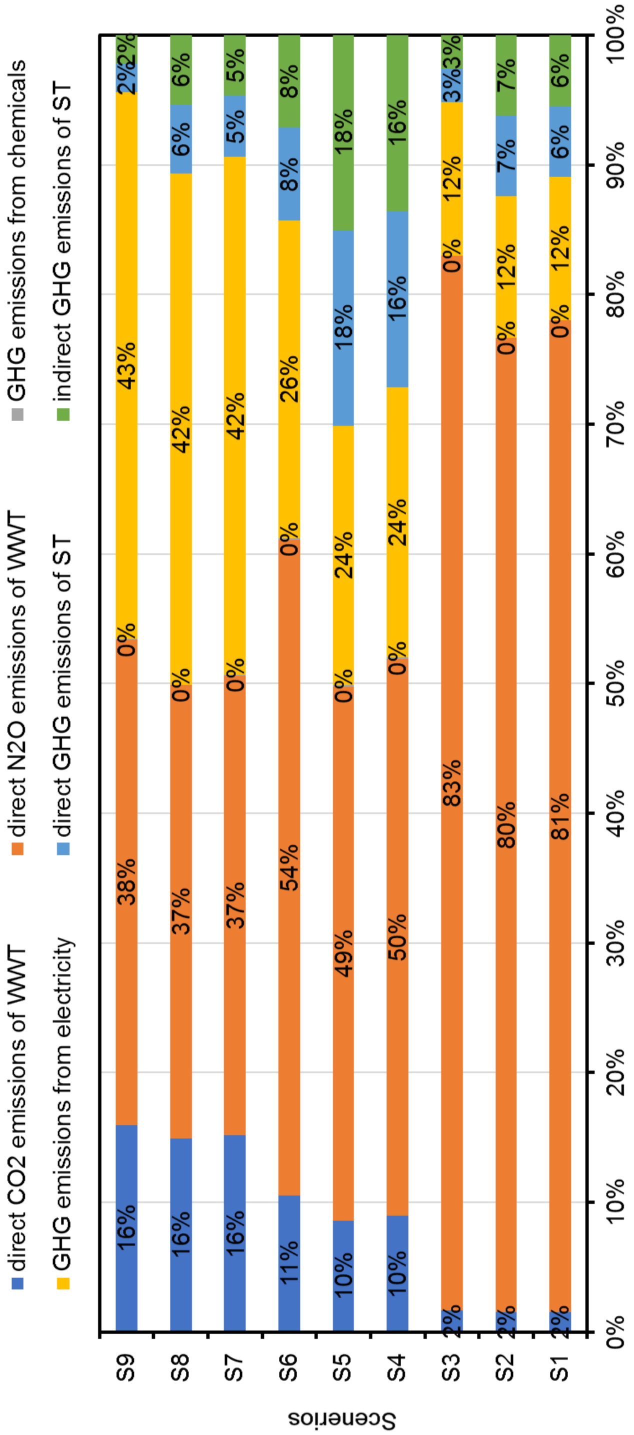

Figure 4.3 Contributions of the GHG emission sources in nine alternative scenarios

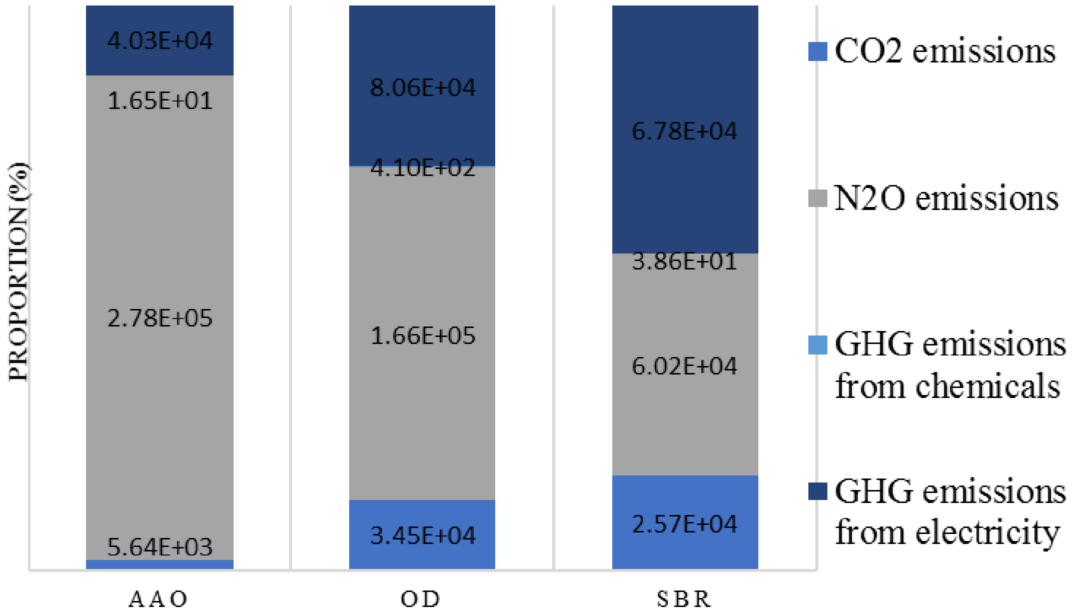

Figure 4.4 Contribution of GHG emission sources from different WWT technological options

The GHG emissions of wastewater-sludge treatment system were divided into six emission sources among nine different scenarios. The six emission sources were CO_2 from wastewater treatment, N_2O from wastewater treatment, chemicals consumption from wastewater treatment, electricity consumption from wastewater treatment, direct GHG emissions from sludge treatment, and indirect GHG emissions from sludge treatment, as shown in Figure 4.3.

The contribution of each emission source (in descending order) in the AAO scenario were 80.12-83.12% (N_2O), 11.62-12.05% (electricity), 2.66-5.77% (direct emission from wastewater treatment), 1.62-1.67% (CO_2), 0.004-0.49% (indirect emission from wastewater treatment), and less than 0.005% (chemicals).

The contribution of emission source (in descending order) in the OD scenario were 48.52-49.62% (N_2O), 23.54-24.07% (electricity), 15.66-17.72% (direct emission from wastewater treatment), 10.09-10.31% (CO_2), 0.01-0.22% (indirect emission from wastewater treatment), and 0.12% (chemicals). S6 scenario showed a different result (OD-incineration): the contribution of CO_2 (11.16%) is more than direct GHG emissions from sludge treatment (7.58%), and the contribution of indirect GHG emissions from sludge treatment increased to 1.39%.

The contribution of each emission source in the SBR scenario, in descending order, was 41.62-

42.94% (electricity), 36.95-38.11% (N_2O), 15.26-16.26% (CO_2), 2.25-5.64% (direct emission from wastewater treatment), 0.01-0.41% (indirect emission from wastewater treatment), and 0.02% (chemicals).

4.5.3 The CO_2 emissions from wastewater treatment and the N_2O from sludge treatment

The IPCC does not consider the CO_2 emissions from wastewater treatment, and NCSC does not consider the CO_2 emissions from the wastewater treatment and the N_2O emissions from sludge treatment.

As described in Figure 4.2, the contribution ranges of GHG emissions from CO_2 were 1.62-1.69% for AAO, 10.09-11.16% for OD, and 15.77-16.26% for SBR. Therefore, it is necessary to estimate CO_2 emissions from wastewater treatment when estimating GHG emissions, at least in OD and SBR systems.

The direct GHG emissions from sludge treatment account for 2.66-6.62% (AAO), 7.58-17.72% (OD), and 2.25-5.64% (SBR) of total wastewater sludge treatment systems. Moreover, it accounts for more than 95% of the GHG emissions from the sludge system.

4.5.4 Possible contribution of ignored direct CO_2 emissions for evaluating GHG emission from WWTPs

According to China's statistics, total GHG emissions from the wastewater treatment industry in 2005 was 114 million tons CO_2-eq., and the ratio of three processes (AAO, OD, and SBR) treatment capacity and the total processing capacity were 37%, 24%, and 13%, respectively. The result of this study revealed that the contribution of direct CO_2 emissions to GHG emissions in three processes were 2%, 12%, and 17%, respectively. Therefore, it can be inferred that in the scenario of calculating direct CO_2 emissions from the sewage treatment when calculating GHG emissions, total GHG emissions from the wastewater treatment industry in 2005 should be 123 million tons of CO_2 equivalent, an increase of approximately 8% compared to the previous GHG emissions.

4.6 SUMMARY

In this study, nine scenarios of different wastewater-sludge treatment processes were analyzed to estimate the GHG emissions. According to national statistics, the limiting design values of mainstream WWTPs were defined as the limit values in the scenario study. The wastewater flow rate was assumed to be 40,000 m^3/d, and the 1-A standard was assumed as an effluent limit. Results show that direct emissions of CO_2, N_2O, and indirect emissions of electricity consumption are significant contributors to the GHG emissions of wastewater-sludge systems. The total GHG emission ranged from 58-127 kt CO_2-eq. per year, with the lowest GHG emissions obtained from the SBR-Incineration scenario and the most significant GHG emissions obtained from the AAO-Composting scenario.

N_2O emissions and electricity consumption are the primary sources of the GHG emissions, and the sum of the contributions of these two sources exceeds 70% in all scenarios. CO_2 emissions have not been considered in GHG emissions estimation of IPCC, as it is of a biogenic origin. This study highlights that not considering CO_2 emissions in the results of GHG emissions estimation may cause deviations in the results.

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5. DEVELOPMENT OF AN LCA CONSIDERING SPATIAL STRATEGIES ON UWS INTEGRATING WITH RECLAIMED WASTEWATER USE

5.1 Decentralized and centralized systems for community-scale wastewater treatment

5.1.1 System boundaries and functional unit

Figure 5.1 shows the system boundaries of this study. In the DWWT (b+c), wastewater flows directly into the wastewater treatment system (b), and treated water is used on site. In the CWWT (a+b+c+a'), wastewater is collected and transported through the wastewater pipeline system (a) into the wastewater treatment system (b), and the treated water is transported by the reclaimed water pipeline system (a') to a user. The excess sludge generated from both systems is transported after dewatering to 80% water content into the sludge treatment system (c) for incineration and landfills disposal.

This study evaluated three GHGs released by wastewater treatment systems: CO_2, N_2O, and CH_4, whose global warming potential (GWP) as CO_2 equivalent (CO_2-eq) on a 100-year timescale are 1, 25, and 298, respectively (IPCC, 2007). The functional unit (FU) is defined as the population equivalent (PE) in one year, that is, 1 FU = 1 PE · a. According to CUWA (2016), the volume of treated wastewater per PE per day was 0.154 m^3. Service life was defined as 20 years (Xiaodi et al., 2019).

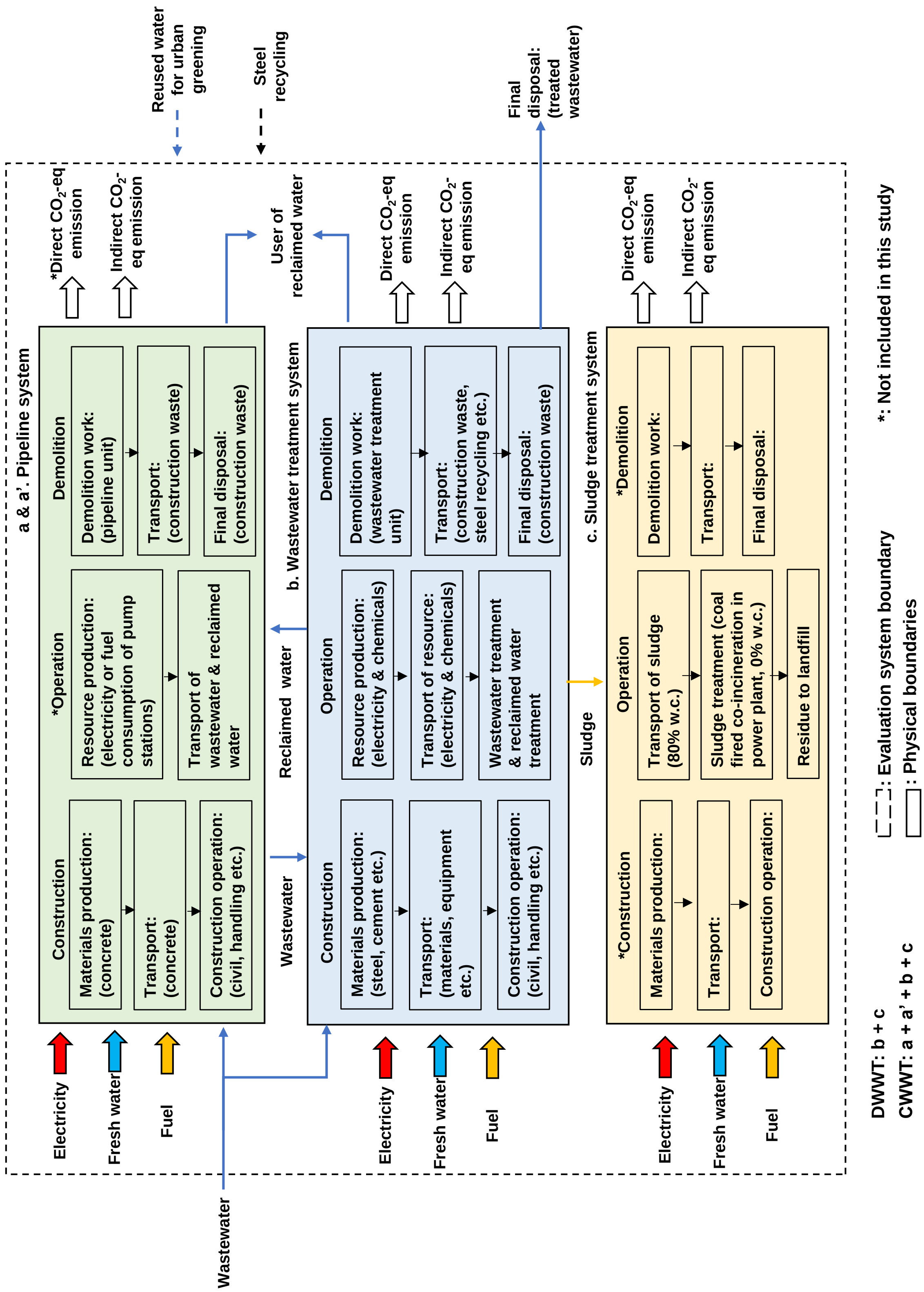


Figure 5.1 System boundaries in the case study

5.1.2 Urban pipeline systems of CWWT for wastewater collection and reclaimed water supply

The construction and demolition phases of the pipeline system were examined. As shown in Figure 5.2, the pipeline within the community boundary was not considered because of the same condition of CWWT and DWWT. The pipeline distance from Jinnan Campus to WWTP_B was estimated to be 30 km using the shortest distance along the street in Google Maps.

The calculation condition was assumed that: a) the same length of wastewater pipeline and reuse water pipeline (constructing 60 km pipeline system); b) the pipeline system (a and a') were completed in one construction operation; c) the pipeline system of 60 km was constructed using D300 mm reinforced concrete pipe (RCP) (DG/TJ 08-2222, 2016); and d) the environmental impact of the pump station was not considered because that wastewater and reuse water was transported by weight only in the pipeline.

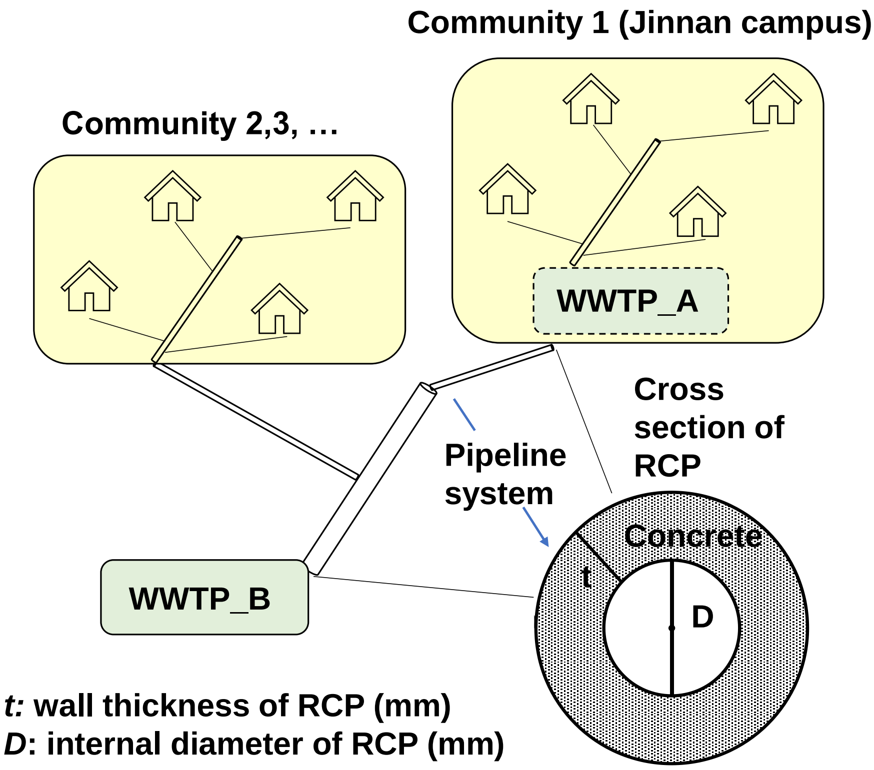


Figure 5.2 Diagram of the pipeline system

5.1.3 Defining the critical distances

The critical distance is defined as the maximum length of pipeline (L) that can be constructed by CWWT if it is lower environmental loads than DWWT.

$$\boldsymbol{E_{DWWT} = E_{WWTP\_A} + E_{sludge}} \quad \textbf{(eq. 5.1)}$$

$$\boldsymbol{E_{CWWT} = E_{WWTP\_B} + E'_{sludge} + E_{pipeline}} \quad \textbf{(eq. 5.2)}$$

$$\boldsymbol{E_{pipeline} = E_{pipe\_WW} + E_{pipe\_RW} = 2 \times (E_{MC} + E_{MT} + E_{WT})/(p \times y)} \quad \textbf{(eq. 5.3)}$$

$$\boldsymbol{E_{MC} = \sum_i A \times L \times P_i \times EF_i} \quad \textbf{(eq. 5.4)}$$

$$\boldsymbol{E_{MT} = \sum_{i,j} A \times L \times P_i \times \rho_i \times EF_{i,j} \times S_{i,j}} \quad \textbf{(eq. 5.5)}$$

$$\boldsymbol{E_{WT} = \sum_{i,j} A \times L \times P_i \times EF_w \times S_w} \quad \textbf{(eq. 5.6)}$$

$$\boldsymbol{A = \pi(D/2 + t)^2 - \pi(D/2)^2} \quad \textbf{(eq. 5.7)}$$

Subject to $E_{CWWT} < E_{DWWT}$, thus,

$$L = \frac{\left(E_{pipe\_WW} + E_{pipe\_RW}\right) \times (p \times y)}{2 \times A \times \left(\sum_i P_i \times EF_i + \sum_{i,j} P_i \times \rho_i \times EF_{i,j} \times S_{i,j} + \sum_{i,j} P_i \times EF_w \times S_w\right)}$$
$$\le \frac{\left[\left(E_{WWTP\_A} + E_{sludge}\right) - \left(E_{WWTP\_B} + E'_{sludge}\right)\right] \times (p \times y)}{2 \times A \times \left(\sum_i P_i \times EF_i + \sum_{i,j} P_i \times \rho_i \times EF_{i,j} \times S_{i,j} + \sum_{i,j} P_i \times EF_w \times S_w\right)}$$

where

E_{CWWT}, E_{DWWT}, E_{WWTP_A}, E_{WWTP_B}, E_{sludge}, E'_{sludge}, $E_{pipeline}$, E_{pipe_WW} and E_{pipe_RW}: total GHG emissions from CWWT, DWWT, WWTP_A, WWTP_B, DWWT sludge treatment system, CWWT sludge treatment system, pipeline system, wastewater pipeline, and reclaimed water pipeline, respectively, kgCO$_2$-eq/FU;

E_{MC}, E_{MT} and E_{WT}: GHG emissions from construction materials consumption, materials transport and waste transport, respectively, kgCO$_2$-eq;

p: service population, P·E;

y: service life, a;

EF_i: emission factor of material i, $kgCO_2$-eq/m^3;

$EF_{i,j}$: emission factor for material i transport by j, $kgCO_2$-eq/(t·km);

EF_w: emission factor of construction waste transport by truck, $kgCO_2$-eq/(t·km);

$S_{i,j}$: national average transport distance of material i by transport modes j, km;

S_w: national average transport distance of construction waste by truck (100 km in this study), km;

P_i: the proportion of material i (100% concrete for RCP in this study), %;

ρ_i: density of material i, t/m^3;

A: cross section area of pipes, m^2;

t: wall thickness of pipe, m;

D: internal diameter of pipe, m;

L: lenght of pipeline, m;

5.2 LCI of DWWT and CWWT

5.2.1 Pipeline systems (a and a′) for CWWT

GHG emissions during the construction operation of pipeline systems were calculated according to the empirical formulae for concrete pipes (250 mm $\leq$ D $\leq$ 1200 mm) in JSTT (2018).

Other assumptions for the calculations are described in Chapter 5.1.2 and the LCI is shown in Table 5.1.

Table 5.1 LCI input of Pipeline Systems (a and a′) for CWWT

Life-time Item	Unit	Value	
		WWTP_A	WWTP_B
Input			
Construction			
Pipeline			
Materials consumption			
Concrete	kg/FU		0.0590
Transport of materials			
by train	tkm/FU		13.9939
by carry	tkm/FU		2.7989
Construction operation			
Pipeline length	km		30
Demolition			
Pipeline			
Transport			
Construction waste			
by truck	tkm/FU		3.7336
Demolition work			
Pipeline length	km		60

5.2.2 Wastewater treatment system (b)

The construction phase of a wastewater treatment system includes the production of six raw materials, such as steel, cement, and sand, as well as the energy consumption during transportation and construction.

GHG emissions during the operation phase include direct emissions from biological treatment and indirect emissions from resource consumption and transport energy consumption. The demolition phase consists of GHG emissions from demolition works, waste transport, and disposal. Steel can be recycled to reduce raw material production, and the recycling rate can reach 0.38 t/t and can save 60% of energy during production (Xiaodi et al., 2019).

Although IPCC (2006) defines direct CO_2 emissions from wastewater treatment as of biogenic origin, Yingyu et al. (2013) indicated that 4–15% of direct CO_2 emissions come from fossil fuels through applying isotope tracer technique. The theoretical maximum emissions of direct CO_2 from the wastewater treatment process were calculated using mass balance method (Liang et al., 2021). According to operating parameters of WWTP_A and WWTP_B, direct CO_2 emission intensities of 0.0207 and 0.1653 $kgCO_2$-eq/m^3 were obtained, respectively. The higher emission intensity of WWTP_B is a result of longer hydraulic retention time (HRT) and higher pollutant removal rates (BOD and ammonia nitrogen). Similarly, the scale effect causes the two systems to consume electricity, chemicals, and fresh water at different rates during the operational phase.

(1) Nankai WWTP as DWWT

i) introduction

Nankai WWTP (WWTP_A), located at the Jinnan Campus, was defined as a community-based DWWT, and its design inflow is 1,000 m^3/d. Treated effluent from Cloth Media Filter disinfected by sodium hypochlorite (NaClO) for reuse on campus. In addition, Anaerobic-Anoxic-Oxic (AAO) was determined as secondary treatment in order, to resist the impact loads resulting from the periodic variation of influent, to meet C-Level of discharge standards (DB 12/599-2015, 2015) and to meet quality standard for Reclaimed Water use (GB/T 18920-2002, 2002).

ii) Data Sources of Inventory input of WWTP_A

Material consumption was obtained from statistics data by field investigation, with missing data estimated based on design drawings, engineering budget reports, and construction work records. Chemical and energy consumption, influent and effluent water quality, treatment water volume, and sludge production were obtained from a one-year survey covering November 2018 to November 2019.

Figure 5.3a A view of Nankai WWTP's structures (2019-03-28)

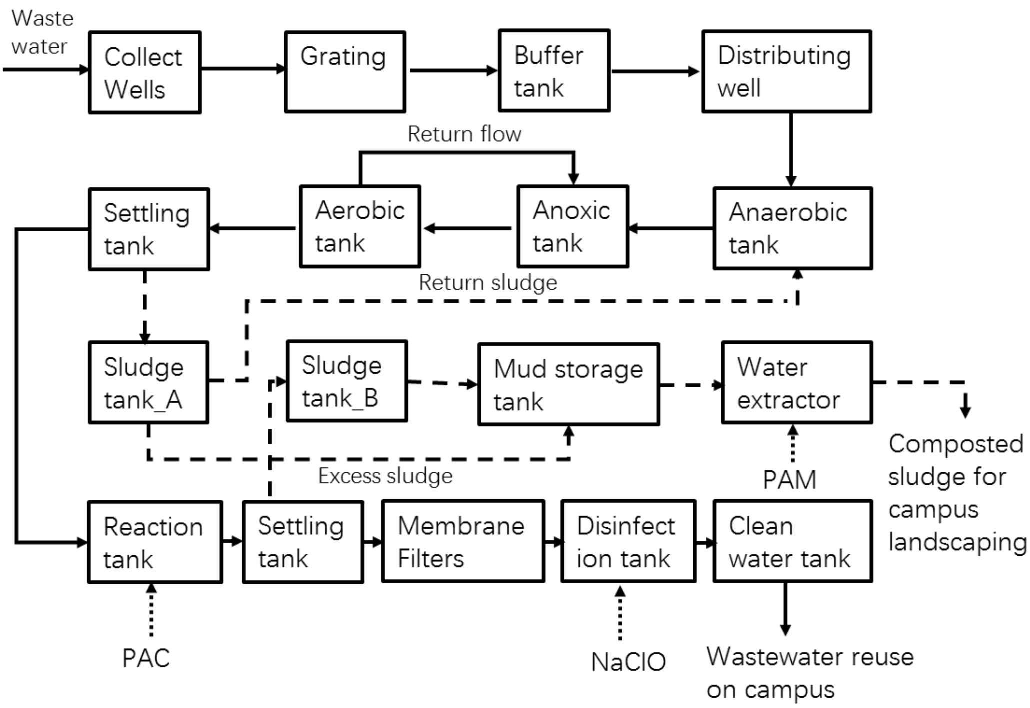


Figure 5.3b Process flow of Nankai WWTP (AAO + cloth media filter + disinfection by adding NaClO)

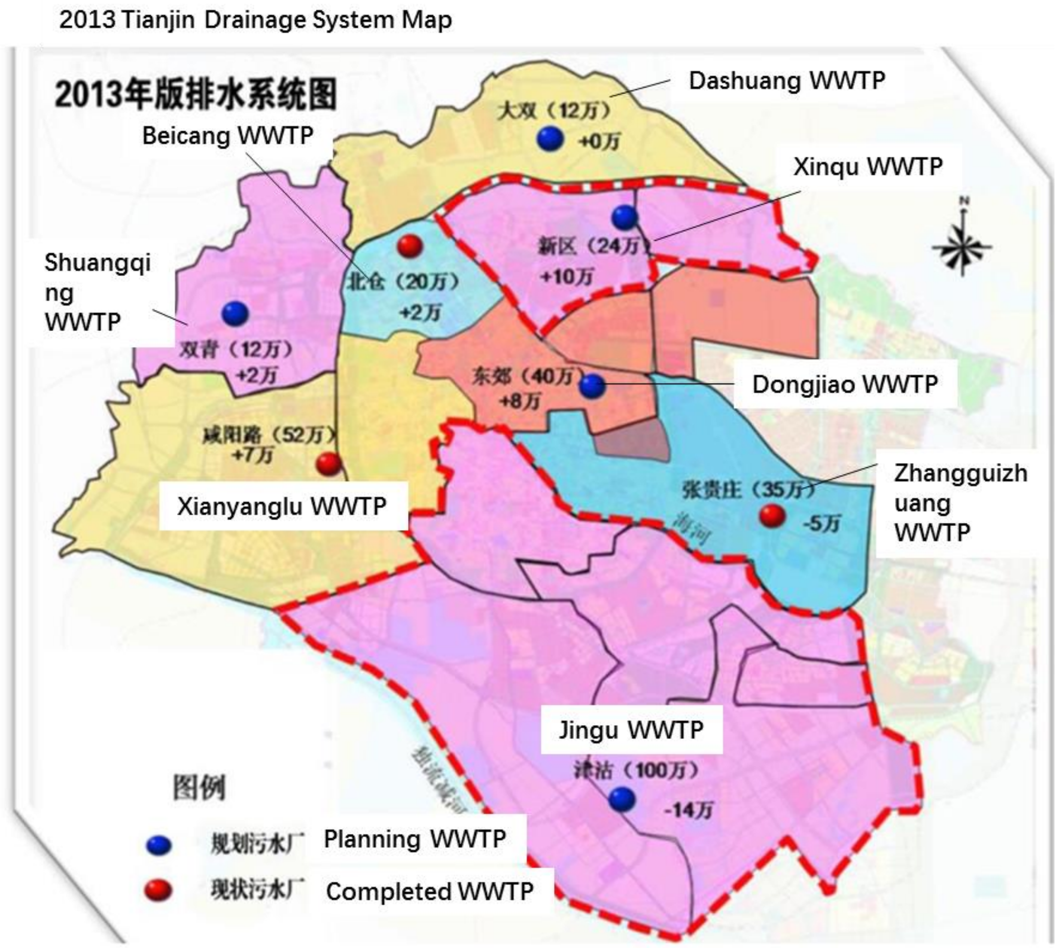
(2) Jinnan WWTP as CWWT

i) introduction

Jinnan WWTP (WWTP_B), located in Jinnan District, operating AAO, and receives domestic and industrial (7.5% of influent) wastewater from a service area of 283 km^2 (as shown in Figure 5.4a). Treated effluent from advanced treatment (Deep-bed Filter and O_3) after ultraviolet disinfection, of 30% flows into Reclaimed Water use system, and the rest is discharged into Dagu River (NCMEDRI, 2017) (as shown in Figure 5.4b). Discharge meets stricter A-Level standard than C-Level (WWTP_A) due to its larger size (>10,000 m^3/d) (DB 12/599-2015, 2015).

ii) Data Sources of Inventory input of WWTP_A

Material consumption for WWTP_B was obtained from a previous study by Hou et al. (2015), estimating the material intensity for different scale WWTPs. Chemicals and energy consumption, water quality of influent and effluent, treated wastewater quantity, and sludge production during the operation phase were obtained from the environmental impact assessment report of WWTP_B (NCMEDRI, 2017).



Source: EIP for Jingu WWTP (2015)

Figure 5.4a WWTPs' service area in urban areas of Tianjin

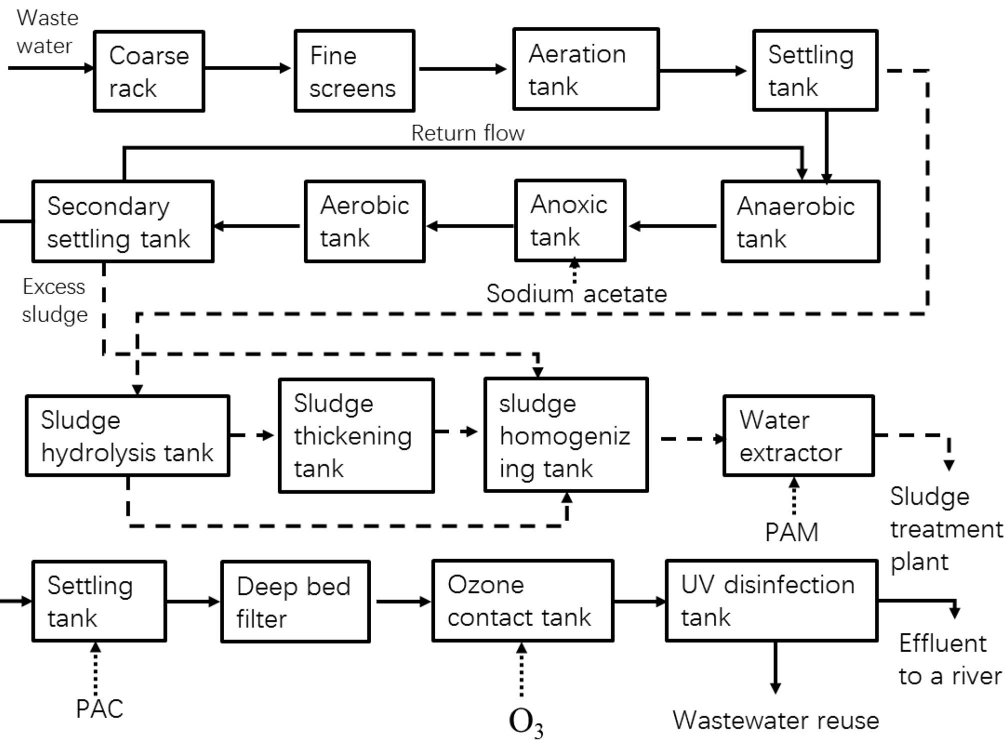


Figure 5.4b Process flow of Jingu WWTP (AAO + deep-bed filter + O_3 + ultraviolet disinfection)

(3) LCI

Table 5.2 LCI of Wastewater Treatment System (b) and resource recovery

Life-time Item	Unit	Value WWTP_A	WWTP_B
Input			
Construction			
Wastewater and sludge treatment inside of WWTPs			
Materials consumption			
Steel	kg/FU	0.7280	0.2388
Cement	kg/FU	3.5398	1.4154
Gravel	kg/FU	8.0800	13.2861
Sand	kg/FU	5.0481	7.3396
Metal pipe	kg/FU	0.0215	0.0527
Timber	kg/FU	0.0046	0.2752
Fresh water	kg/FU	3.8476	
Transport			
Construction Materials			
by train	tkm/FU	6.2488	7.2625
by truck	tkm/FU	1.4344	1.7118
Equipment			
by truck	tkm/FU	0.3803	0.0015
Construction operation			
Diesel	kg/FU	0.0086	0.0924
Electricity	kWh/FU	0.0505	
Operation			
Wastewater treatment			
COD	kg/FU	11.2350	11.2350
BOD	kg/FU	3.9323	3.9323
NH3+-N	kg/FU	1.9661	1.9661
TN	kg/FU	3.1458	3.1458
TP	kg/FU	0.1685	0.1685
SS	kg/FU	7.5836	7.5836
Energy consumption			
Electricity	kWh/FU	105.3017	22.1162
Water consumption			
Fresh water	kg/FU	151.6937	24026.3212
Chemicals consumption			
NaAc	kg/FU		4.4076
PAC	kg/FU	0.6077	4.1483
NaClO	kg/FU	0.3282	
O3 (10%, liquid)	kg/FU		0.6741
Transport			
Chemicals			
by truck	tkm/FU	0.2513	2.4172
Demolition			
Wastewater and sludge treatment inside of WWTPs			
Civil work			
Diesel oil	kg/FU	0.0077	0.0832
Electricity	kWh/FU	0.0455	
Transport			
Steel			
by truck	tkm/FU	0.0728	0.0239
Construction waste			
by truck	tkm/FU	1.6694	2.2094
Equipment			
by truck	tkm/FU	0.1200	0.0005

Continued Table 5.2

	Life-time		Item	Unit	Value WWTP_A	WWTP_B
Output						
	Operation					
		Emission to water				
			COD	kg/FU	1.872502	1.123501
			BOD	kg/FU	0.26215	0.15729
			NH3+-N	kg/FU	0.327688	0.196613
			TN	kg/FU	0.524301	0.196613
			TP	kg/FU	0.045141	0.036113
			SS	kg/FU	0.505578	0.210656
		Emission to air				
			CO2	kg/FU	1.16092	9.283029
			N2O	kg/FU	0.001201	0.001351
			CH4	kg/FU	0.007584	0.00819
	Demolition					
		Wastewater treatment unit				
			Construction waste	kg/FU	16.76679	22.11766

	Life-time		Item	Unit	Value WWTP_A	WWTP_B
Output						
	Production					
		Steel recycling		kg/FU	0.276628	0.090753
		Reclaimed water		m3/FU	56.17505	21.24314

5.2.3 Sludge treatment system (c)

As shown in Figure 5.1, only the operational phase of the sludge treatment system was considered. Beibei et al. (2013) examined the direct and indirect GHG emissions from sludge generated from WWTPs, which is dewatered to 80% water content by centrifugation, transported to power plants for coal fired co-incineration (kiln temperature is 1,000 °C) and finally landfills disposal. An emission factor of 1,318 kg CO_2-eq per ton dry sludge was obtained.

Table 5.3 LCI of Wastewater Treatment System (b)

	Life-time		Item	Unit	Value WWTP_A	WWTP_B
Input						
	Operation					
		Transport				
			Sludge (80% w.c.)			
			by truck	tkm/FU	4.4366	8.6336
		Chemicals consumption				
			PAM	kg/FU	0.0292	0.0519
Output						
		Waste				
			Sludge incineration (80% w.c.)	kg/FU	44.36555	86.3357758

5.2.4 Transportation

The national average rail transport distances for materials, equipment, and chemicals were calculated based on NBS (2014) and GB/T 50878-2013 (2013); the national average road transport distances were calculated according to NBS (2014) and MTPRC (2017). The energy consumption for both rail and road transport was obtained from GB/T 50878-2013 (2013).

The details are described in Chapter 3.2.1.

5.2.5 Data sources of GHG emission factors (EF)

As Table 5.4 shows, GHG emission from construction work of the pipeline system was obtained from a report of JSTT (2016). The energy consumption during the construction of the WWTP was calculated based on the results of Zhang et al. (2010), and the lower heating value of diesel fuel from NBS (2008).

Direct CO_2 emissions from wastewater treatment were calculated by mass balance (Liang et al., 2021), and EF of direct N_2O and CH_4 were obtained from the results of Xu et al. (2014). The EF of the sludge treatment were referenced from Beibei et al. (2013).

Table 5.4 GHG emission factors and data sources

Item		Unit	Value	Source	Item		Unit	Value	Source
Materials					Chemicals				
	Steel	kgCO2-eq/t	2600	Tao (2015)		NaAc	kgCO2-eq/kg	1.5702	City of Winnipeg (2012)
	Cement	kgCO2-eq/t	730	Tao (2015)		PAC	kgCO2-eq/kg	0.0227	Chai et al. (2015)
	Gravel	kgCO2-eq/t	2	Tao (2015)		NaClO	kgCO2-eq/kg	0.92	City of Winnipeg (2012)
	Sand	kgCO2-eq/t	2	Tao (2015)		O3 (10%, liquid)	kgCO2-eq/kg	8.01	City of Winnipeg (2012)
	Concrete	kgCO2-eq/m3	350	Tao (2015)		PAM	kgCO2-eq/kg	1.5	Chai et al. (2015)
	Fresh water	kgCO2-eq/t	0.3	Tao (2015)	GHG direct emission from wastewater treatment				
	Metal pipe (mild steel)	kgCO2-eq/t	15100	Qian et al. (2019)		Wastewater treatment			
	Timber	kgCO2-eq/t	0.0258065	Qian et al. (2019)		CO2**			
Transport						for WWTP_A	kgCO2-eq/m3	0.0206661	Zhiyi et al. (2021)
	by train*	kgCO2-eq/tkm	0.007867	GB/T50878 (2013) and NDRC (2005)		for WWTP_B	kgCO2-eq/m3	0.1652518	Zhiyi et al. (2021)
	by carry*	kgCO2-eq/tkm	0.1715851	NBS (2008) and MTPRC (2017)		N2O	g-N2O/kg-TN	0.458	Xu et al. (2014)
Energy						CH4	g-CH4/kg-COD	0.81	Xu et al. (2014)
	Diesel*	kgCO2-eq/t	3098.0991	GB/T50878 (2013) and NDRC (2005)	Waste				
	Electricity	kgCO2-eq/kWh	0.8733	NCSC (2010)		Sludge incineratior	kgCO2-eq/t	1318.47	Beibei et al. (2013)
Construction work					Production				
	WWTP*	t-diesel/m2	0.0432118	Qian et al. (2019) and NBS (2008)		Steel recycling	kgCO2-eq/t	1560	Xiaodi et al. (2019)
	Sewer syster	kgCO2-eq/m		JSTT (2016)		Reclaimed water	kgCO2-eq/t	0.3	Xiaodi et al. (2019)

*Calculations based on literature, published reports, and statistical yearbooks

**Calculations based on mass balance method using operating parameters

5.3 Case study of a community-based UWS integrated reclaimed wastewater use

5.3.1 Introduction of the case community

As an examined case study, Jinnan Campus of Nankai University, located in Haihe Education Park in Tianjin, was completed in 2015 and serving over 10,000 students. It generates 630 m^3/d wastewater from toilets flushing, washing, and cafeteria wastewater. Meanwhile, the campus needs reclaimed water for green irrigation, road washing, and replenishment of landscape water.

The Haihe Education Park is located on the south side of the middle reaches of the Haihe River between the central city of Tianjin and Binhai New Area, covering a total planning area of 37 square kilometers, with a planned schooling scale of 200,000 people, a residential population of 100,000 and 300,000 annual social training sessions. The education park is structured into a "corridor and two wings". The "one corridor" refers to the central ecological green corridor planned in conjunction with the urban ecological corridor, while the "two wings" refer to the construction areas of colleges, residences and supporting facilities on both sides of the green corridor. According to the different planning functions, the Education Park is divided into three major parts: the Higher Vocational Park, the Higher Education Park, and the Higher Research Park.

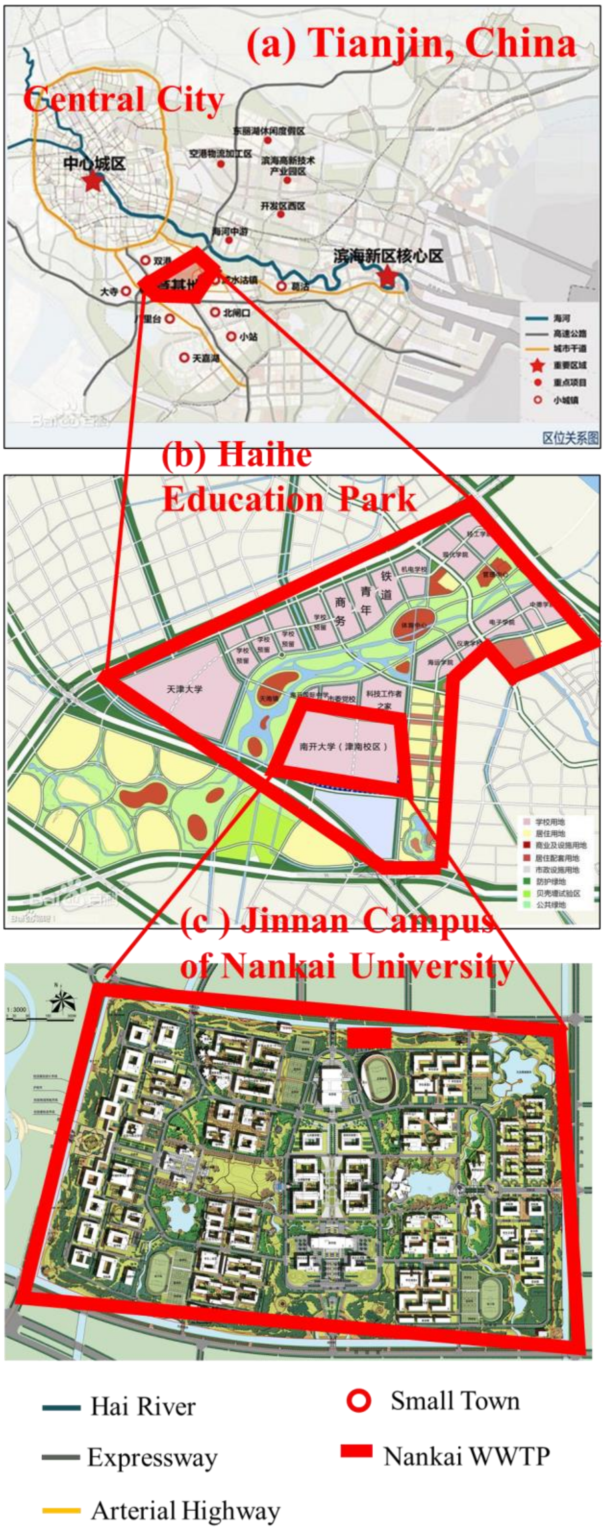


Figure 6.1 Location map of the Jinan Campus of Nankai University (c) in Haihe Education Park (b), Tianjin (a), China

5.3.2 Results

(1) Comparison GHG emissions of DWWT and CWWT

As shown in Figure 5.5, total GHG emissions are 112.8 and 93.4 kgCO_2-eq/FU for DWWT and CWWT, respectively. When considering carbon offsets from steel recycling and reclaimed water use, the values were reduced by 15% and 7%, respectively. Reclaimed water use offset 16.9 and 6.4 kgCO_2-eq/FU of GHG emissions in two WWTPs, respectively. For both systems, the operational phase contributed more than 90% of the GHG emissions.

Both construction and demolition phases contributed less than 10%. The CWWT (WWTP_B, 11.8 kgCO_2-eq/FU) increased GHG emissions from the construction phase by 123% compared to the DWWT (WWTP_A, 5.3 kgCO_2-eq/FU). The pipeline systems (a and a') contributed 73.5% of GHG emissions during construction of CWWT. The wastewater treatment system (b) during construction of CWWT emitted 3.13 kgCO_2-eq/FU, which is 40.5% lower than that of the DWWT. Material consumption of two systems contributed to 92% and 80% of the GHG emissions during the construction phase of the wastewater treatment system, respectively.

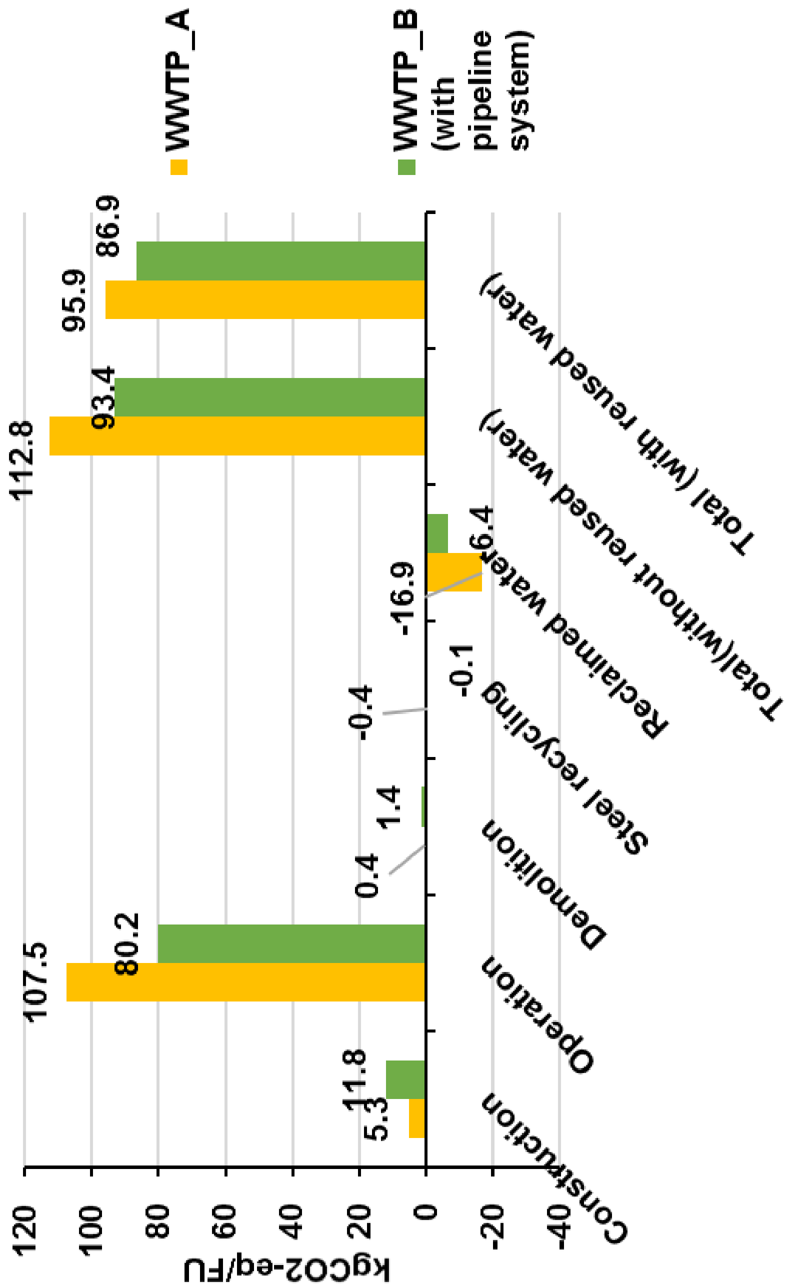


Figure 5.5 GHG emissions of different lifetime and carbon offset caused by resource recycling scenario

(2) GHG emissions from pipeline systems (a and a'), wastewater treatment systems (b), and sludge treatment systems (c)

In pipeline system (a and a'), as shown in Figure 5.6 and 5.7 (a), material consumption during the construction phase contributed 84% of the GHG emissions (7.9 kgCO_2-eq/FU), followed by waste transport during the demolition phase (7%), and material transport during the construction phase (6%).

As shown in Figure 5.6, wastewater treatment system (b) contributed the most GHG emissions, a result that did not differ between DWWT and CWWT, with 101 and 60 kgCO_2-eq/FU, respectively. As shown in Figure 5.7 (b), GHG emissions due to electricity consumption in the operation phase contributed more than 90% of GHG emissions in DWWT, followed by material consumption during the construction phase (4.8%), direct emissions during the operation phase (1.7%), and chemical consumption during the operation phase (1.3%). Electricity and chemical consumption both contributed 32.3% of GHG emissions during operation phase of CWWT, followed by direct emissions during the operational phase (16.6%), freshwater consumption during the operational phase (12.1%), and material consumption during the construction phase (4.2%).

The sludge treatment system (c) in DWWT and CWWT emitted 13 and 24 kgCO_2-eq/FU, respectively, as well as sludge treatment process causing over 90% of the GHG emissions during the operation phase, as shown in Figure 5.6 and 5.7 (c).

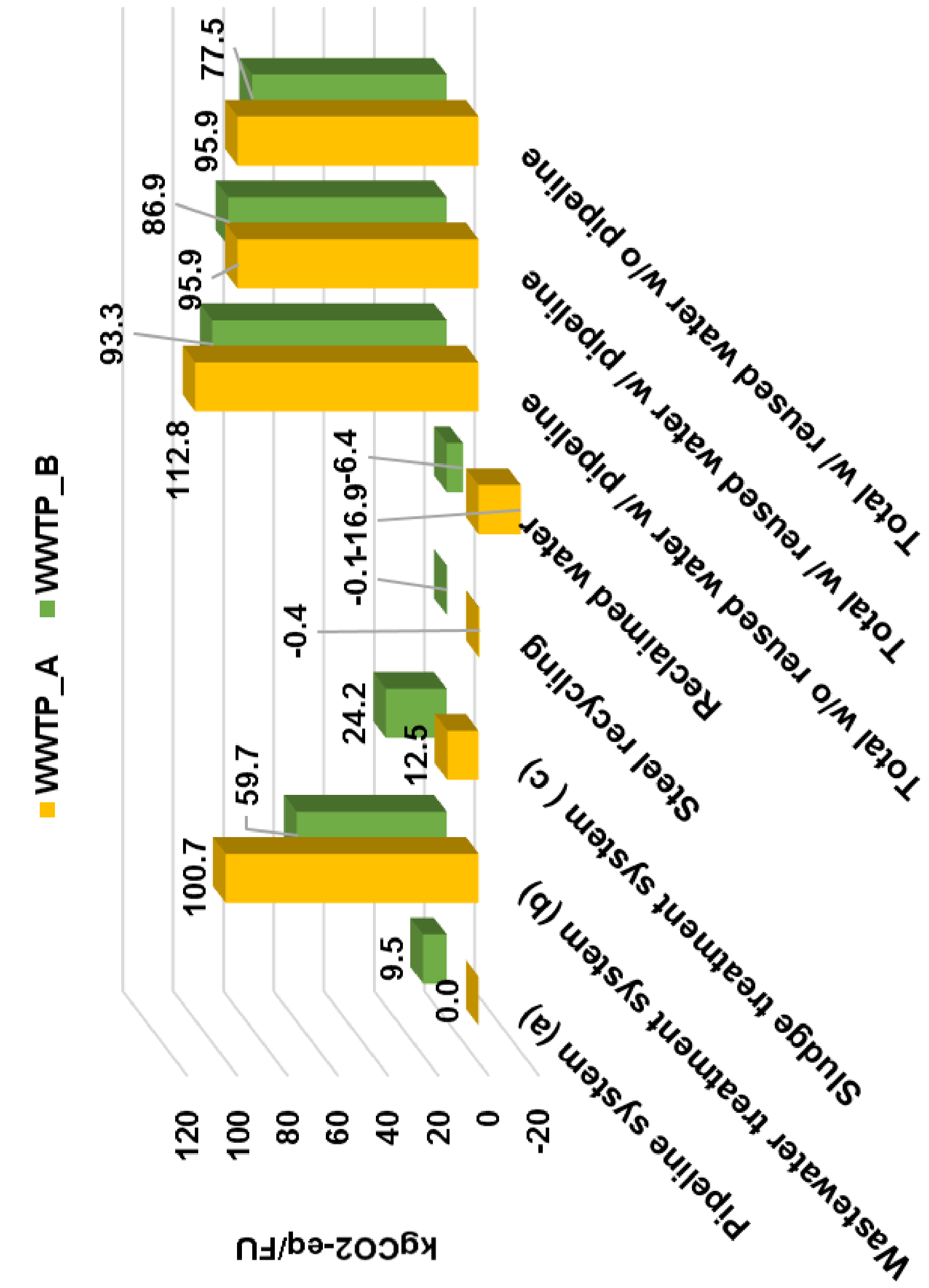


Figure 5.6 GHG emissions of wastewater treatment system, sludge treatment system, carbon offset caused by resource recycling scenario

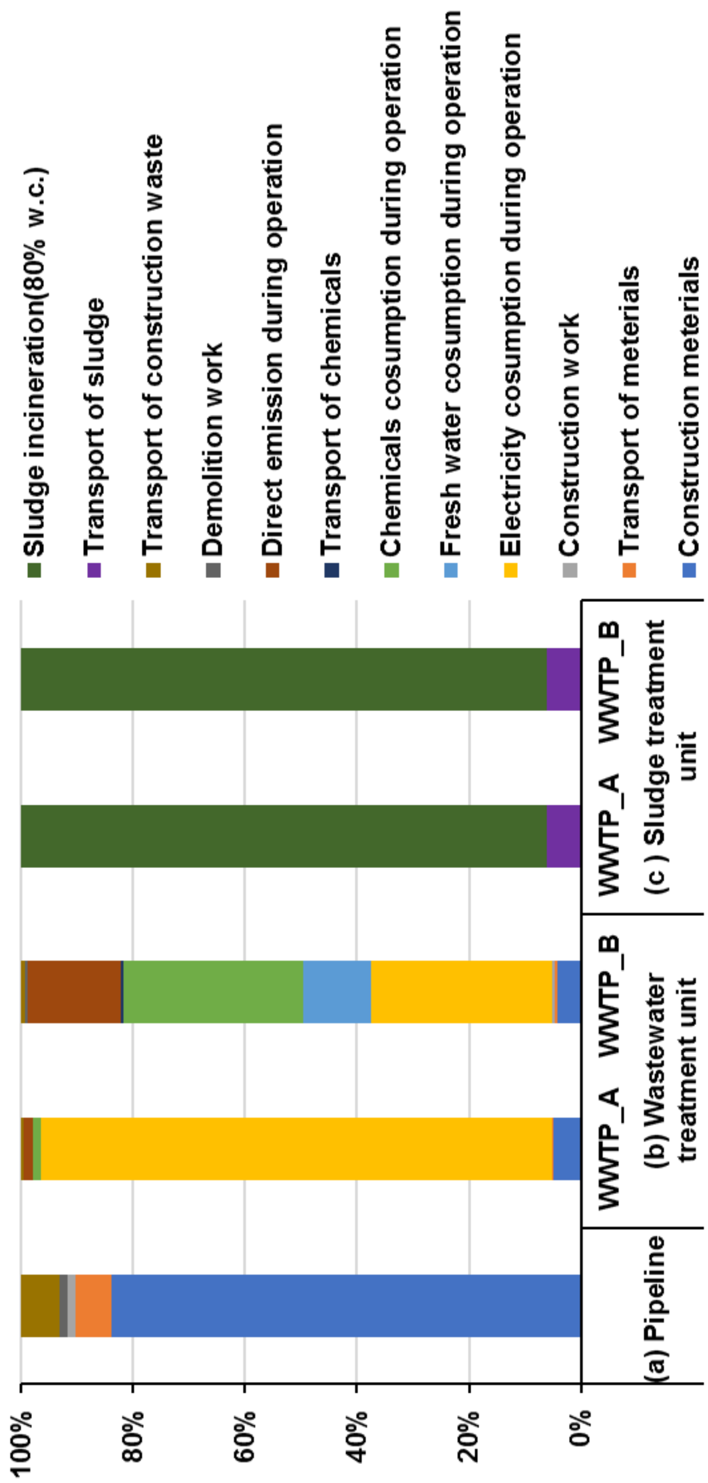


Figure 5.7 GHG emission contribution from different causes of pipeline systems (a and a'), wastewater treatment systems (b), and sludge treatment system (c)

(3) Critical distance

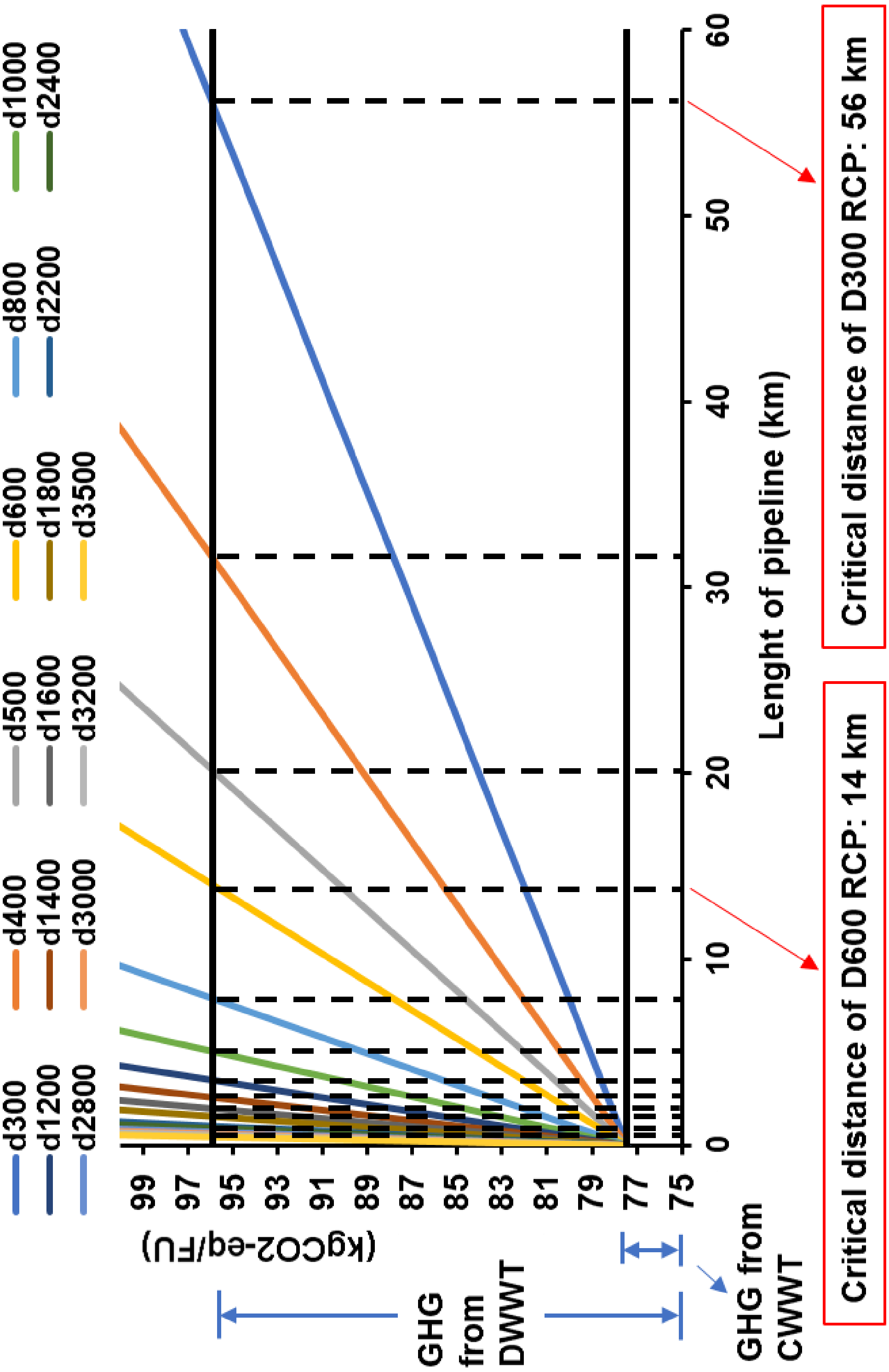


Figure 5.8 Critical distance using RCP of different internal diameter (300-3,500 mm)

As Figure 5.8 shown, the variation of the critical distance for the scenario using pipes of different diameters, from 300 mm to 3500 mm. The critical distance using RCP is 56 km for D300 mm, 14 km for D600 mm, and 0.45 km for D3500 mm. The contribution of pipeline system to total environmental loads of CWWT increases as the internal diameter becomes larger. The critical distance becomes shorter with larger internal diameters and pipeline lengths. The variation rate of emission intensity increases with larger pipe diameters.

5.3.3 Analysis of the factors affecting the critical distance

(1) Different internal diameter of pipes applied to the pipeline system

The results of pipeline system during construction and demolition were 5.8 and 0.8 kgCO_2-eq/FU, respectively (shown in Figure 5.6 and 5.7). As shown in Figure 5.8, the difference of 18 kgCO_2-eq/FU between the DWWT and the CWWT without pipeline system. The critical distances were calculated and shown in Table 5.3 and Figure 5.8. The critical distance is 56 km with a D300 mm RCP, the use of thicker (D1,600 mm) RCP leads to a reduction of the critical distance to 2 km. In Figure 5.9, the results show a positive correlation between internal diameter and emission intensity (R^2=0.9989), and a negative correlation with critical distance (R^2=0.9999).

Table5.5 GHG emission intensity and critical distances for different sizes of RCP and PCCP

Type of pipes	Internal diameter (mm)	Emission intensity (kgCO2-eq/FU/km)	Critical distance (km)
RCP	300	0.1647	56.0176
	400	0.2928	31.5099
	500	0.4576	20.1663
	600	0.6589	14.0044
	800	1.1713	7.8775
	1000	1.8302	5.0416
	1100	2.2146	4.1666
	1200	2.6355	3.5011
	1400	3.5872	2.5722
	1600	4.6854	1.9694
	1800	5.9299	1.5560
	2200	8.8583	1.0417
	2400	10.0646	0.9168
	2800	12.9618	0.7119
	3000	14.9850	0.6158
	3200	16.8398	0.5479
	3500	20.3388	0.4537
	1676	4.3125	2.1397
	2743	11.3617	0.8121
PCCP	1676 (66 inches)	9.8028	0.9413
	2743 (108 inches)	21.8074	0.4231

*Calculation conditions: concrete density is 2.6t/m3; material intensity is 100% of concrete for RCP, is 81% and 76% for D2743 mm and D1676 mm PCCP respectively (calculation also includes steel, cement and steel wire).

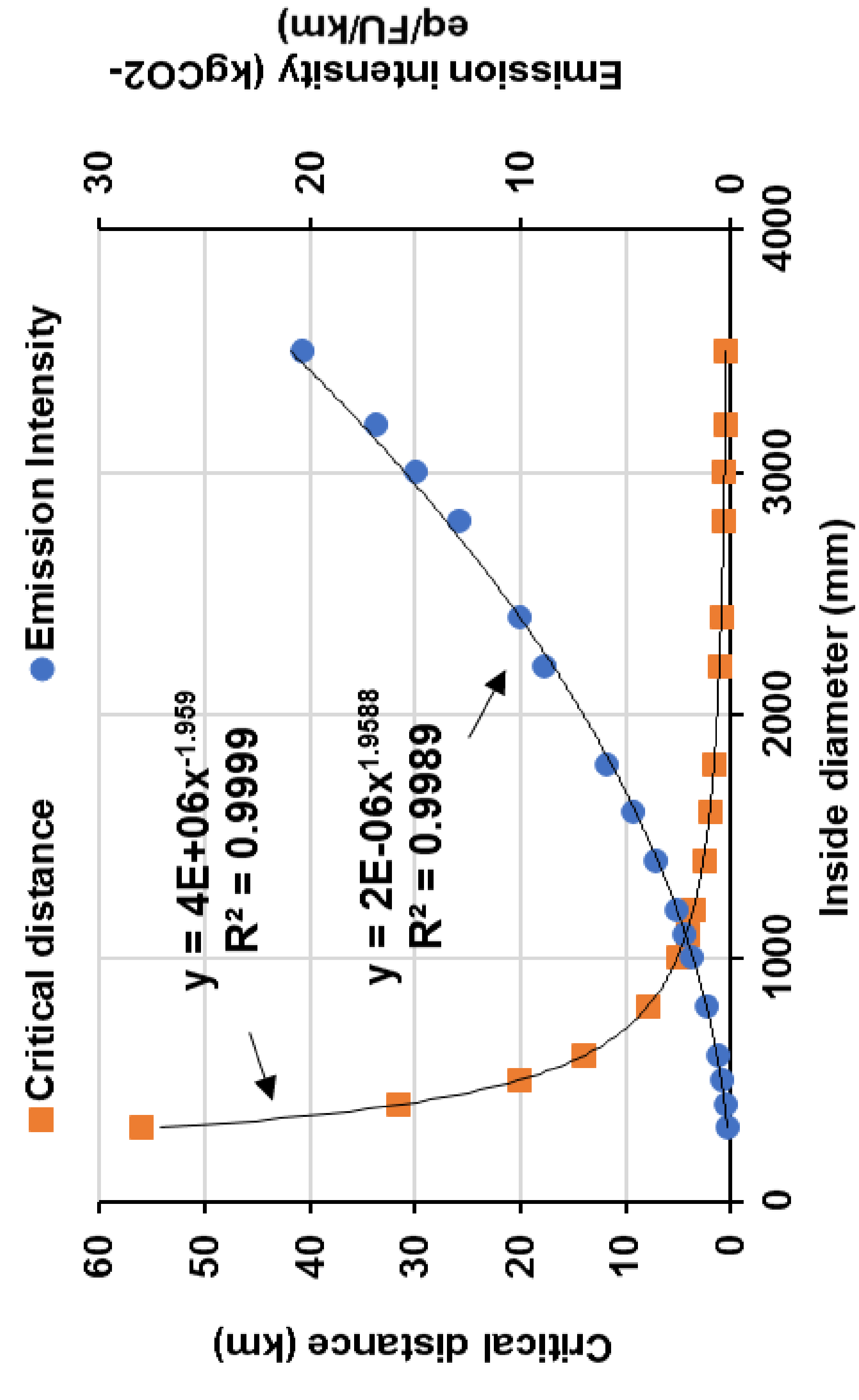


Figure 5.9 GHG emission intensity (kgCO2-eq/FU/km) and critical distance (km) vs. internal diameter (mm) of RCP, respectively.

(2) Different types of pipes applied to the pipeline system

The critical distance is 56 km with a D300 mm RCP, which will reduce 52% if the same internal diameter (D) prestressed concrete cylinder pipe (PCCP) was used instead. Internal diameter of pipeline supporting for WWTP_B was estimated 2,100–3,500 mm (DG/TJ 08-2222, 2016). Lalit et al. (2016) indicated that the PCCP manufacturing process contributed more than 90% of total GHG emissions. Using the material consumption ratio of PCCP (MTPRC, 2017), PCCP manufacturing standard (NBS, 2008), and Chinese GHG emission factors (as shown in Table 5.4), we calculated the GHG emission intensity of RCP and PCCP for different diameters, as shown in Table 5.5. The emission intensity of RCP with an internal diameter of 300 mm was 0.16 kgCO_2-eq/FU/km, while the emission intensity of PCCP with 1,676 mm and 2,743 mm was 2.1 and 1.7 times higher than that of RCP, respectively.

As shown in Figure 5.8, the contribution of pipeline system to total environmental loads of CWWT increases as the internal diameter becomes larger. The critical distance becomes shorter with larger internal diameters and pipeline lengths. The variation rate of emission intensity increases with larger pipe diameters.

(3) Geographical characteristics of the location of the case (configuration of the pumping station)

When a potential user is choosing whether to construct DWWT or whether it should be integrated into CWWT by constructing a sewer system, critical distance can be used to determine choosing DWWT if it’s located outside the critical distance, then DWWT should be chosen.

Basing on assumptions described in Section 2-3, a pipeline system (D300 mm RCP) was constructed by CWWT to serve only Jinnan Campus. In fact, CWWT need to receive more wastewater discharge units, requiring the construction of pipes with larger internal diameters, while pipes with larger internal diameters may require more complex pipe manufacturing techniques, such as PCCP.

Furthermore, the environmental impact of the pump station was not considered in the pipeline system used in this study. Imura et al. (1996) calculated the LC-CO_2 emissions of a pipeline system (pipelines and pumping stations) and indicated that the LC-CO_2 emission of pipelines was 6 times larger than that of pump stations, with pipelines contributed more than 80% during the construction phase. This means that the environmental impact of CWWT due to pipeline construction in this study was a conservative result.

5.4 Summary

While municipal WWTPs remove pollutants from water bodies and improve the water environment, they also cause other negative environmental impacts due to resource use. The reuse of WWTP effluent as reclaimed water contributes to the sustainable development of society and can offset some of the environmental loads due to alternative fresh water. CWWT have an advantage of lower operational energy consumption than DWWT. However, because of the complexity of the influent water quality, more complex processes are required to meet more stringent discharge standards, which can consume more energy and chemicals. In addition, the environmental loads of the pipeline system for CWWTs during construction phase cannot be ignored.

Some elements should be considered when available: the operation phase of pipeline system, the construction and demolition phase of sludge treatment system, and pump stations in pipeline system. In addition, this study only considered the environmental loads of comparing two systems with an indicator (GWP). A more integrated evaluation system (e.g., economic evaluation) and more indicators (e.g., eutrophication potential, water consumption) should be considered in future studies

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6. CONCLUSIONS AND DISCUSSION

6.1 Summary of conclusion of each chapter

Chapter 1:

Introduces the municipal wastewater system (UWS), a complex system that bridges wastewater, sludge, and resource recovery systems via pipelines and trucking; analyses the principles, pathways, and types of gases emitted by UWS; and describes the current status, causes, and strategies for addressing global climate change and the challenges UWS face in addressing global climate change and achieving sustainable development. UWS is an important solution to the urban water environment problem, however, the release of GHGs into the environment cannot be ignored. This establishes the significance and purpose of this paper's research.

Chapter 2:

presents methods for estimating GHG emissions from UWS and compares the advantages and limitations of each method. In addition, the choice of technology and the scale of implementation affect the GHG emissions from UWS. Previous studies on these two influencing factors are reviewed. This defines the research methodology and framework of this paper.

Chapter 3:

The IPCC process is the dominant method of accounting for GHG emissions today, however, the credibility of its results is strongly influenced by the GHG emission factors chosen. Direct emissions of CO_2 from wastewater treatment processes are assumed to be entirely of biological origin and are ignored, but in fact approximately 20% of BOD is of fossil origin from detergents, cosmetics, chemicals, etc. Therefore, this study improved a GHG evaluation system with a basic framework of LCA for the UWS, covering the whole life cycle of construction, operation, and dismantling, with the system boundary starting from the wastewater collection (enters the urban sewer system) and ending the wastewater-based reclaimed water use and the finally disposed or recycled of sludge. In addition, the calculation of direct CO_2 emissions is supplemented by an calculation based on mass balances, with input data including characteristic parameters for wastewater treatment technologies (BOD and N removal rates, HRT, MLVSS, biomass yield, etc.).

Chapter 4:

Nine alternative scenarios for the selection of different sludge and wastewater treatment technologies were analyzed according to the GHG evaluation system developed in Chapter 3, which includes three wastewater treatment technologies (AAO, SBR, and OD) and three sludge treatment technologies (incineration, composting, and direct landfills). For comparison purposes, we have assumed a treatment of 40,000 tons per day and an effluent that meets the most stringent 1-A discharge standards in China.

The results shown that the SBR-Incineration scenario has an advantage in terms of low GHG emissions, while AAO-Composting is the scenario that results in maximum emissions. The direct N_2O emissions and emission caused by electricity consumption are the main GHG emissions sources, and the sum of the contributions of two sources exceeds 70% in all scenarios. In addition, the results highlighted that not considering direct fossil CO_2 emissions may cause deviations in the estimation of GHG emission.

Chapter 5:

Based on the improved GHG accounting method in Chapter 3, which was used as a constraint, we developed an optimization model which was used to quantify the environmental loads of community-scale wastewater treatment systems integrating reclaimed water use under different management strategies, and which can provide upper limits on distances for optimizing the location of decentralized and centralized hybrid applications.

The results show that: 1) CWWT consumes only 20% of the electricity of DWWT in its operation phase, but consumes 14 times more chemicals and 158 times more fresh water than DWWT; 2) pipeline system supporting CWWT contributes 65% of total GHG emissions during the construction phase; and 3) the critical distance is 56 km when applying 300 mm internal diameter reinforced concrete pipes (RCP), and is shortened in scenarios where thicker RCPs are used and replaced with prestressed concrete cylinder pipes.

6.2 Summary of key findings and limitations

6.2.1 Key findings

1) This study develops a model for estimating GHG emissions from municipal wastewater systems based on the LCA procedure, which is a "bottom-up" model that integrates the characteristic parameters (eg., HRT and BOD removal rate) of different wastewater-sludge treatment technologies. The mass balance approach is used in the LCI analysis to calculate direct fossil CO_2 emissions from UWS for different technology options, which is a complement

and improvement to previous studies.

2) This study highlights that not considering CO_2 emissions in the results of GHG emissions estimation may cause deviations in the results. This study examines GHG emissions for nine scenarios consisting of a combination of mainstream technological routes for WWT and ST. The results show that direct fossil CO_2 and direct N_2O emissions and indirect emissions from electricity consumption are important contributors to GHG emissions. The contribution of GHG emissions from direct fossil CO_2 were 2% for AAO, 12% for OD, and 17% for SBR. In addition, GHG emission intensities are 0.14, 0.86, and 0.64 kg CO_2-eq/m^3, respectively. As in Chapter 4.5.4, it can be inferred that the total GHG emissions from the WWT industry should have increased by about 8% in 2005 compared to the previous inventory data when calculating direct fossil CO_2 emissions (theoretical maximum emissions).

3) This study evaluates the environmental loads of a large community-based UWS integrated with reclaimed water reuse, comparing operating in DWWT or CWWT. The LCIs cover energy consumption, materials consumption, and transport during the construction, operation, and demolition phases. This study provided a quantitative analysis of optimizing location of hybrid applications of decentralized and centralized in cities through defining the critical distance. This study is intended to provide researchers, managers, and decision makers with information on the environmental loads of centralized and decentralized wastewater management strategies.

In the case study of this study, the critical distance is 56 km with a D300 mm RCP, the use of thicker (D1,600 mm) RCP leads to a reduction of the critical distance to 2 km. When a potential user is choosing whether to construct DWWT or whether it should be integrated into CWWT by constructing a sewer system, critical distance can be used to determine choosing DWWT if it's located outside the critical distance, then DWWT should be chosen.

4) The case and scenario analysis data in the study are all from China, but the study area of the article is not limited to China. The improved GHG emission model in Chapter 3 of the study and the LCA built based on this emission model in Chapter 5 considering spatial strategies are generic. Data acquisition is the focus and difficulty of GHG accounting work because the data of GHG emissions from WWTPs are not mandatory to be disclosed. In this study, a generic model based on conventional mandatory disclosure data is developed, which adopts a "bottom-up" accounting approach, based on operational data at plant scale, and includes parameters characterizing different technologies. Second, in the scenario analysis in Chapter 4, the technical characteristics are taken from Chinese national standards and industry surveys to reduce errors and make the results more credible; in Chapter 5, the case and LCI data are taken from field surveys and Chinese industry surveys. However, it does not mean that the study

results only represent the Chinese situation; this paper provides a generic approach that other studies can obtain by simply replacing the data using the characteristics of the target cases or target regions when used.

5) The critical distance is defined as the maximum pipeline length (L) that CWWT can construct with lower environmental loads than DWWT. It is a spatial decision support model for optimally locating treatment plants for community-based safe wastewater reuse. The "L" calculation included WW and RW, calculated by material consumption, material transport, and waste transport. The scope of the study in Chapter 5 is community-scale and examines the community's wastewater treatment and reclaimed water use needs. The purpose of the study is to develop an LCA evaluation model under spatial strategies (decentralized and centralized systems), quantify the environmental impacts of the two options, and compare the environmental costs of the two options. It provides municipal wastewater management options that consider environmental impacts and relevant environmental information for policy development, urban planning, and future research.

6.2.2 Limitations

1) The critical distance in this study is borrowed from an economic concept to provide an environmental criterion for the safe reuse of community-scale wastewater. In addition, this study only considered the environmental loads of comparing two systems with an indicator (GWP). A more integrated evaluation system (e.g., economic evaluation) and more indicators (e.g., eutrophication potential, water consumption) should be considered in future studies.

2) In the study of city scale, different communities are connected by the pipelines, generally called the secondary network. The collected wastewater flows into a thicker pipeline (the trunk network) connected to the WWTPs. However, this study only examined the community scale case, and future studies should discuss the improvement and application of the model at the city scale. The case of pipe networks in city-scale studies is more complex, and there is the challenge of optimizing the optimal implementation scale and optimizing distances.

3) The stable control and improvement of removal rates is a difficult engineering challenge. Many factors affect pollutant removal rates, such as temperature, pH, and influent pollutant load. The factors contributing to the differences in removal rates between technologies are mainly the differences in microbial community composition structure and activity due to different operating conditions. Although the engineering specifications for each technology give a range of design values for the removal rate of each pollutant, they do not serve as parameters to characterize the differences between technologies. The data for pollutant removal

rates in this study were obtained from EIA reports for WWTPs using different technologies. While they represent each technology in actual operation, there is also uncertainty in the data. Future studies should use statistical analysis data from more wastewater treatment plant cases and analyze the effect of parameter uncertainty on the results, alternatively combining the model from this study with a dynamic kinetic model for simulation analysis.

APPENDIX

Table S-1 Equipment list of Nankai WWTP

Structures	Name of equipment	Power kw	Number of equipment	Design power kw	Actual power kw
Primary treatment					
Catchment wells	Hand-operated and electrically operated openers	0.55	2	1.1	1.1
Grille room	Rotary grate decontaminator	0.55	1	0.55	0.55
Conditioning tank	Submersible mixers	0.75	4	3	3
	Submersible sewage pumps	1.5	3	4.5	3
Secondary treatment					
Anaerobic tank	Low-speed submersible mixers	1.5	2	3	3
Anoxic tank	Low-speed submersible mixers	3.7	2	7.4	7.4
	Jet aerators	3.7	6	22.2	22.2
Aerobic tank	Mixing fluid return pumps	2.2	4	8.8	4.4
Vertical Flow Sedimentation Tanks	Centre Drive Scrapers	0.55	2	1.1	1.1
	Return sludge pumps	1.5	2	3	3
Sludge tank	Residual sludge pumps	0.75	2	1.5	1.5
Chemical dosing systems	PAC Dosing pumps	0.75	2	1.5	1.5
Advanced treatment					
Fast Reaction Cell	Fast Response Mixers	0.37	2	0.74	0.74
Slow Reaction Tanks	Slow reaction mixers	0.22	2	0.44	0.44
Inclined tube sedimentation tanks	Centre drive scrapers	0.55	2	1.1	1.1
Sludge tanks	Residual sludge pumps	0.75	4	3	1.5
	Filter cloth filter tanks	1.1	1	1.1	1.1
Filter cloth filter tanks	Mobile submersible sewage pumps	2.57	1	2.57	2.57
Clearance basins	Reuse pumps	15	3	45	30
Deodorization systems	Ion deodorisation equipment	7.5	1	7.5	2.5
Sludge pre-treatment		0.75	1	0.75	0.75
Sludge storage tank	Underwater mixers	0.75	1	0.75	0.75
	Sludge cutters				
	Sludge feeding	1.5	2	3	3
	Screw pumps	0.65	1	0.65	0.22
Sludge treatment room	Stacked screw sludge dewatering machines	0.75	2	1.5	1.5
	PAM Dosing pumps	2.2	1	2.2	2.2

Table S-2 Equipment list of Jinnan WWTP

	Models & Specifications	Unit
Submersible mixers	B=630mm N=3kW	60
Vertical turbine mixers	HCM2500	68
High Chain Gridders	PAN4-4424	4
Rotary grate decontaminators	ZGF1300x5500	2
Submersible centrifugal pumps	XFP	12
Sand suckers	PXS-11-10500	4
Sand and water separators	Q=80 m^3/h	4
Rotary drum fine-grid	B=2400mm	8
Sedimentation tank scrapers	L×B＝49.0m×8.0m	8
Sludge return and discharge pumps	VX186	18
Roots blowers	Q=5000m^3/h, H=7m, N=130kW	6
Single-stage centrifugal blowers	SG52A-CVC	7
Belt thickener and dewaterer	PDL2500+PPS11 2500	6
UV disinfection units	N=500KW	6
Hydraulic transfer pumps	KOS1070P	8