Life Cycle Assessment and Life Cycle Cost of Sewage Sludge Recycling System Considering Technology Selection and Implementation Scale

by

Jiawen Zhang

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Supervising Professor: Toru Matsumoto, Ph.D.

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Table of Contents

List of Table

List of Figure

ABSTRACT

With the acceleration of economic development and urbanization in China, wastewater generation has sharply increased. The wastewater system begins to change from “emphasizing the sewage, despising the sludge” to “emphasizing both the sewage and sludge”. With the stricter emission standard and waste management policy in China, the landfill of sludge disposal is expected to be restricted in the future. Previous studies mostly focused on the environmental and economic performances of the sewage sludge recycling system, in which sewage sludge is solid waste by a biological treatment of wastewater. The State Council of China published the first relevant industrial policy in 2015. Subsequently, relevant policies including technological guideline, emission standards and subsidy are promulgated from 2015 to 2020.

However, policy impacts and comparative analysis integrating environmental, economic, and social performances of the applications to the potential resource recovery were ignored. Therefore, this dissertation considered the following: (1) environmental and economic performances of sewage sludge recycling scenarios, such as sludge-to-electricity, fertilizer, building material, and biogas, were evaluated by life cycle assessment and life cycle cost approaches by replacing the traditional and similar commodities on the market with a system expansion. (2) The whole life cost of these scenarios was applied to include externality, which represents the monetization of emissions to integrate environmental and economic impacts. Major pollutants that contributed to the external costs of these scenarios were identified. The net present value of each sewage sludge recycling scenario was compared, which provided the basis for technology improvements and policymaking. (3) Through a life cycle cost of policy scenario analysis, we found that waste disposal subsidy was more vital for sewage sludge recycling system than the corporate income tax and environmental protection tax. Based on this, an evaluation system of sewage sludge recycling system integrating environment, economy, and policy aspects were proposed.

In previous studies, researchers have investigated a variety of approaches to the environmental and economic analysis of sludge treatment and recycling systems but lack universal law of different capacities for environmental impact and economic of sewage sludge recycling system. The aim of the study is the analysis the scale effect of sewage sludge recycling with different technological selections on the environment and economy. Moreover, to achieve carbon neutrality, the cost-benefit impact of introducing a carbon trading mechanism into sewage sludge recycling was analyzed. The avoided carbon emission by by-production of sewage sludge recycling system becomes a part of the income of sewage sludge recycling system via selling carbon emission quota. The results show that the break-even scale of incineration, aerobic composting, used in material (brick), and anaerobic digestion are 54,899, 6707, 48,775, and 4425 t/y, respectively. The break-even scale of each system decreased with the introduction carbon trading system to the sewage sludge recycling system. These findings could provide some fundamental and technical information for the decision-making of sewage sludge recycling systems.

Finally, the environmental and economic evaluation system of sewage sludge recycling system, which focuses on energy and resource recovery, is constructed considering the technology selection, the implementation of scale, and introduced carbon trading mechanism. The policy analysis proves the market potential under the support of relevant policies. The evaluation system provided references for governments and industries and promote the construction of "zero-waste" city with the wastewater system.

Key word: *Sludge Recycling, Life cycle assessment, Life cycle cost, Policy analysis, Technology Selection, Implementation of scale, Carbon Trading Mechanism*

1 Introduction

1.1 Background

1.1.1 Introduction and status of sewage sludge treatment system

(1) Sewage sludge

Sewage sludge is the product resulting from the sedimentation of the suspended solids during the wastewater treatment at the wastewater treatment plants (WWTPs). The conventional activated sludge process produces primary sludge from the sedimentation tank and produces excess activated sludge from the aerated activated sludge tank, which is called sludge. The sludge consists of aggregates made of functional microorganisms and secreted extracellular polymeric substances suspended in wastewater, and the biological aggregates in the activated sludge tank are designated as flocs. The floc has a very complex internal structure, and the water strongly binds to the solid phase, making it difficult to mechanically release from the solid surface. Thus, the sludge has a non-Newtonian fluid behavior in the form of a suspension and has a form of cake in the dehydrated viscoelastic properties. Primary sludge is generated by post mechanical treatment after the primary stage whereas waste activated sludge (WAS) is generated via biological treatment at secondary stage in WWTPs. Usually WAS is used for resource recovery or energy generation. As known as Fig. 1.1, the mixed sludge of WWTPs is consisting primary sludge and WAS.

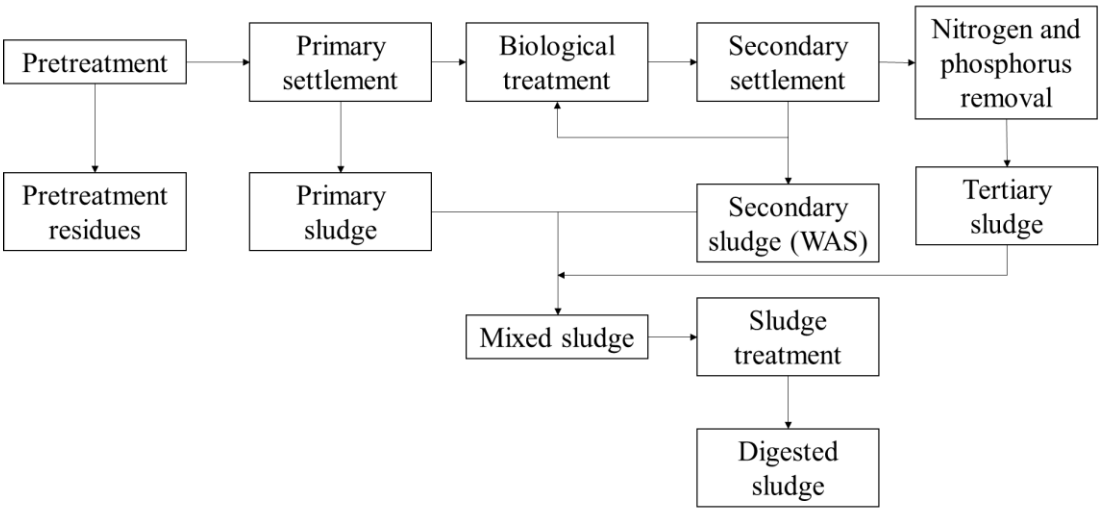

Fig. 1.1 The sewage sludge generated in WWTPs (Source: Raheem et.al, 2018)

A typical WAS composition includes 59-88% dry weight per volatile solid (w/v) biodegradable organic matters, composed of 50-55% C, 25-30% O, 10-15% N, 6-10% H with little amount of P and S.

Sludge generation in China steadily rose annually from 11 million tons in 2005 to 21 million tons in 2010 that sludge has 80% water content. Subsequently, the dry sludge production in China is summarized in Fig. 1.2 and has an average annual growth of 9% from 2011 to 2017. In China, generally speaking, more sludge is generated in the eastern part than the western part.

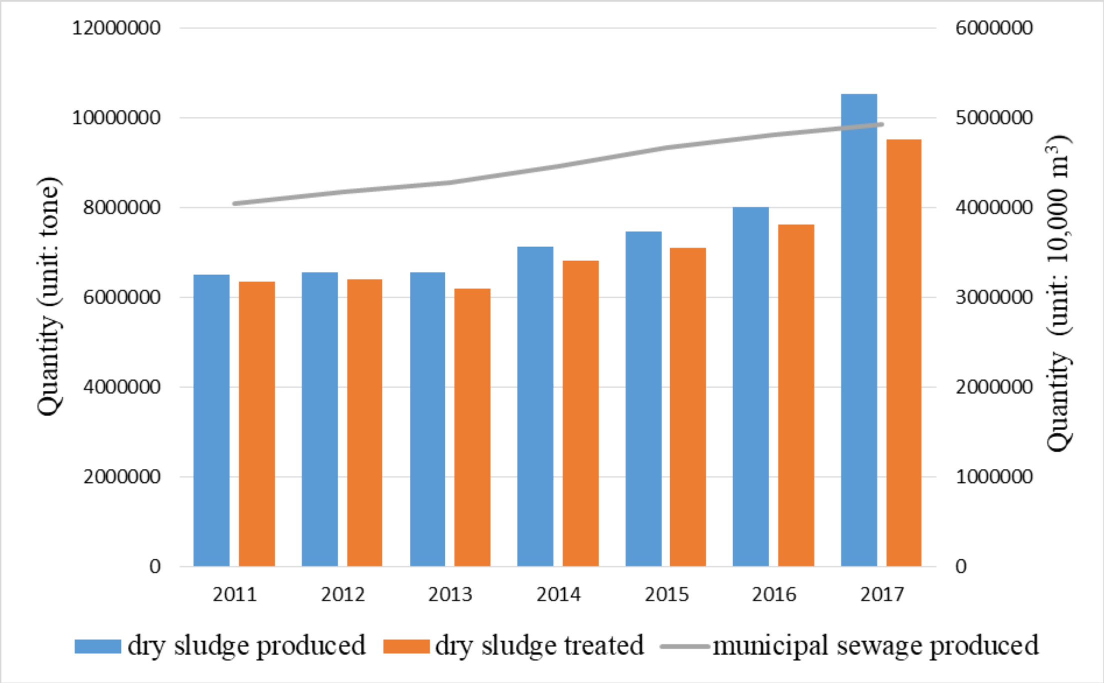

Fig. 1.2 Dry sludge produced, dry sludge treated and municipal sewage produced in China

Population density, industrialization and urbanization in the eastern part are higher than that in the western part (Yang et al., 2014), which lead to larger sewage production. Besides, the sewage treatment rate of the eastern part is higher than that of the western part. In China, there is no obvious difference of wastewater treatment process in different areas, so total sludge production usually increases with sewage amount and sewage treatment rate. Similarly, total sludge production in the southern part is higher than that in the northern part. As to administrative areas, the largest sludge production area is East China, while the lowest is Northwest China. For provinces in China, Guangdong is the largest total sludge production province, and Qinghai is the smallest.

As to Per Capita sludge production, it is higher in the eastern part than the western part due to the more developed economy. It has a nearly positive correlation with Per Capita GDP. For different administrative areas, the highest area is East China, and the lowest is Northwest China. In particular, Northwest China and Central China has the same Per Capita GDP, while their Per Capita sludge production shows a great difference. This is due to variations in industrial structure. In Northwest China, urbanization is low, and its economy development heavily depends on high energy and resource consumption. As to provinces in China, Shanghai is the largest Per Capita sludge production province, while Gansu is the lowest (Fig. 1.3).

These sludges necessitate massive space and cause greenhouse gas (GHG) emission for landfill

(Wei et al., 2020). Leachate pollution of landfill contaminates groundwater such as heavy metal, endocrine disrupting compounds, and pharmaceutical and personal care products (Elmi et al., 2020; Hospido et al., 2010). In addition, soil ecosystems can also be affected by heavy metal accumulation. Inappropriate treatment causes secondary pollution that severely threatens hu-man health, including heavy metal, organic pollutants, pathogens, and dioxin (Liang et al., 2021; Chu and He, 2021).

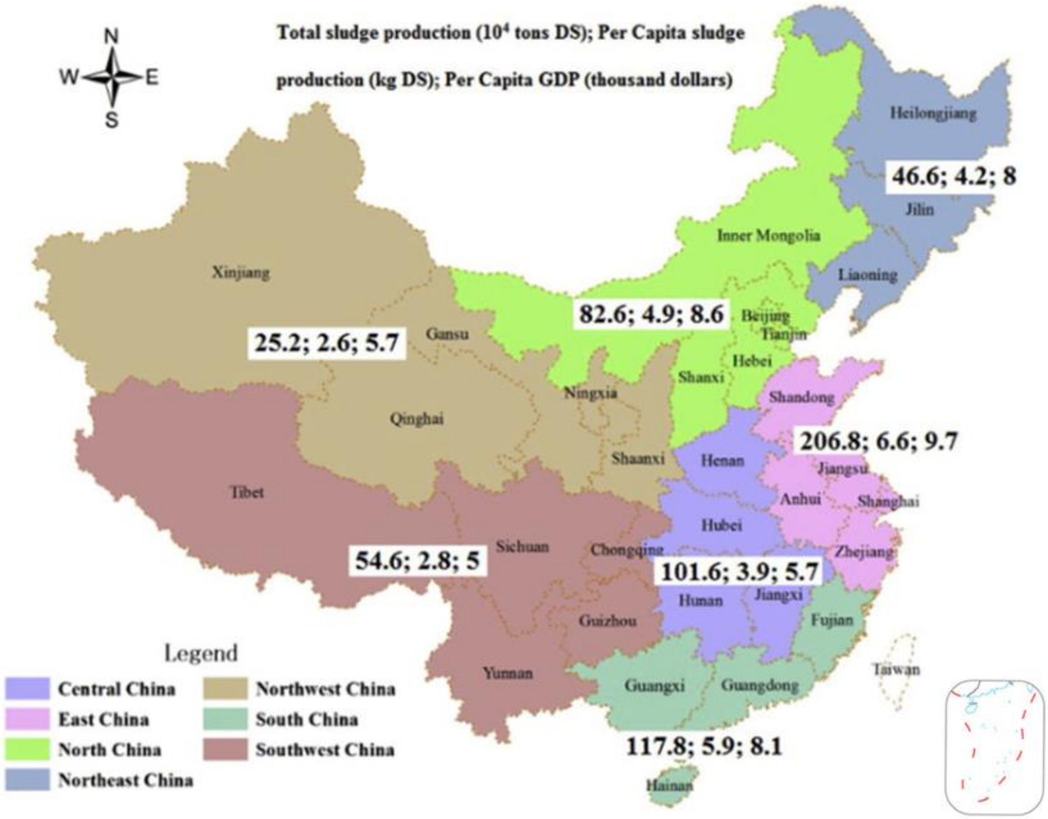

Fig. 1.3 Total sludge production, Per Capita sludge production and Per Capita GDP of different areas in China in 2013 (Source: Yang et.al, 2015)

(2) Sewage sludge treatment

WAS is the most vital by-product generated from WWTPs, whilst it causes human health problems as a potential source of secondary environmental pollution. Therefore, its proper disposal and treatment carry the utmost significance. Diverse sludge treatment technologies are available for safe disposal, resource recovery, and power generation. Commonly used technologies for energy/resource recovery, as presented in Fig. 1.4, include anaerobic digestion (AD), incineration, pyrolysis, and gasification. According to sludge treatment and recycling recorded in guideline promulgated by the Ministry of Ecology and Environment of the People's Republic of China, the priority technologies are AD, incineration, aerobic composting and industrial material substitution (brick).

About 1.977×10^6 dry solids (DS) per ton and 2.406×10^6 DS per ton of sewage sludge were reportedly produced in Japan in 2000 and 2017, respectively corresponding to an increase of 21% DS/t of sewage sludge over just 17 years. And it is estimated that the amount of the produced sewage sludge in Europe in 2020 will amount to 13 million tons dry matter of sludge (45–56 g dry matter of sludge per capita per day) (Mininni et al., 2015).

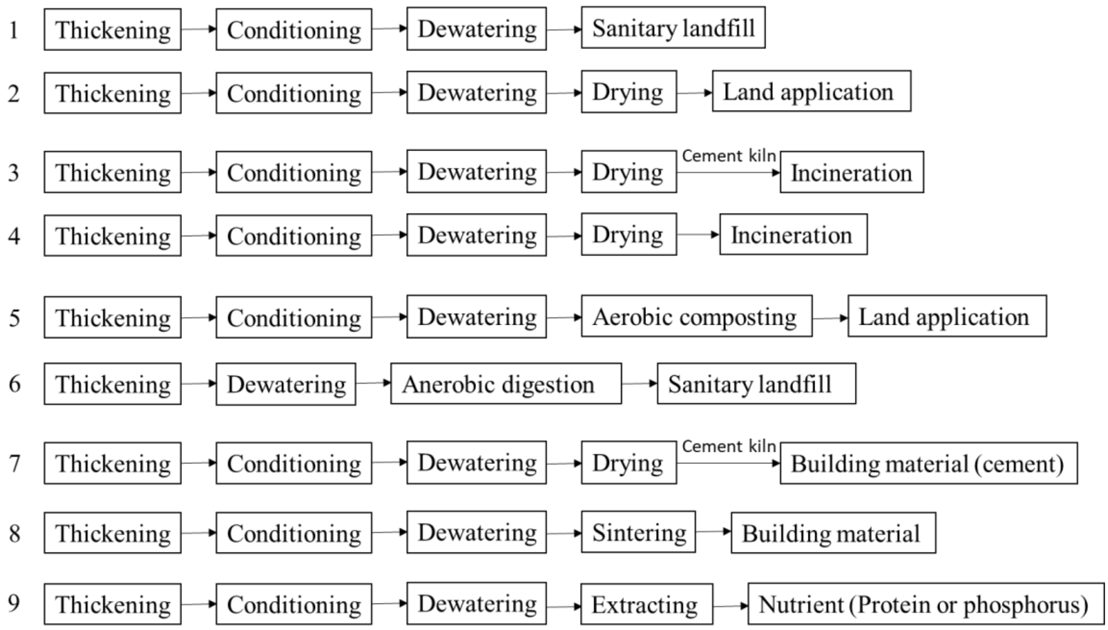

Fig. 1.4 Current sewage sludge treatment technologies in China

1.1.2 Introduction and status of sewage sludge recycling system (SRS)

There are many sludges derived resource recovery options including recovery of biogas, fuel gas, electricity generation, production of construction material, nutrient recovery, biofuel recovery (syngas, bio-diesel, bio-oil), recovery of hydrolytic enzymes, polyhydroxyalkanoates (PHA) (for bio-plastic manufacturing), bio-fertilizers, bio-sorbents etc. using abovementioned treatment methods. Fig. 1.5 also shows the routes of resources recovery from waste sludge.

(1) Situation of SRS in other countries

Considering the current legislation and sustainable development, the most common options for wastewater sludge management are reuse in agriculture and thermal treatments aimed at energy recovery and material substitution, such as incineration, combustion, wet oxidation, pyrolysis, gasification or co-combustion with other materials. In the US, the land application, landfill and incineration are the major ways of handling sewage sludge (Gude, 2015). The sewage sludge production in Lithuania was 82,000 t/y with 60% to storage and landfills, 14% to agriculture

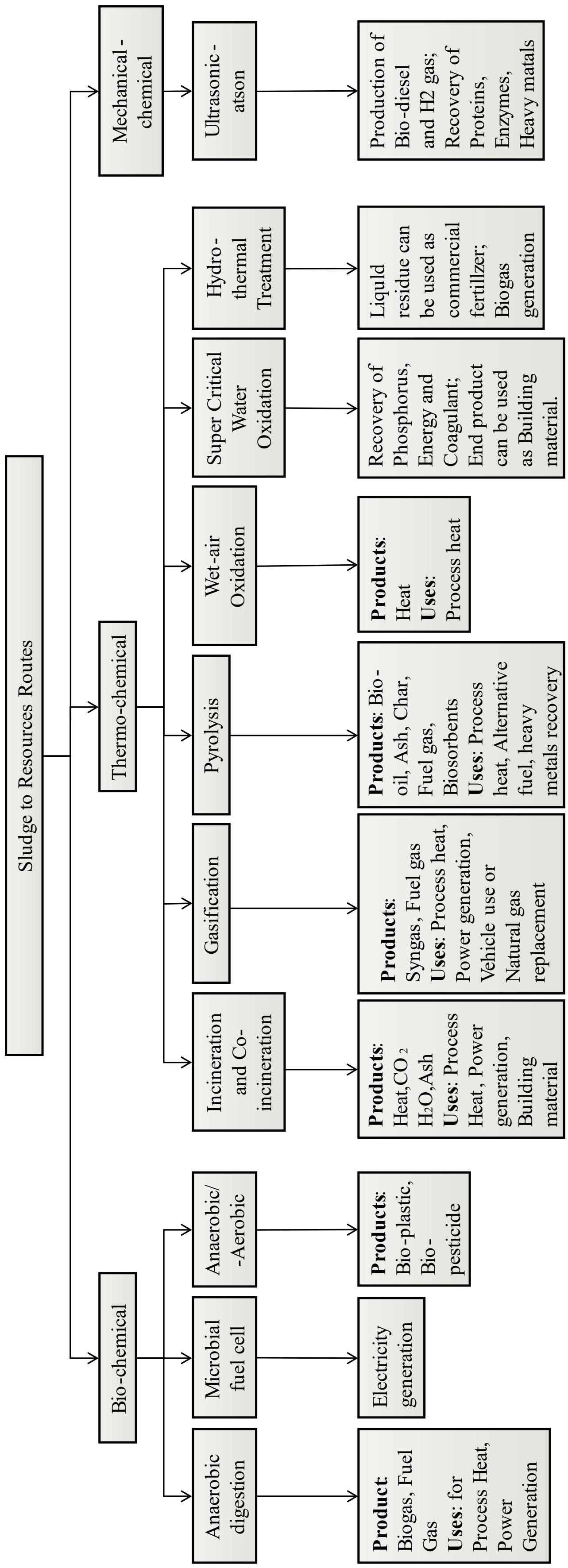

Fig. 1.5 Routes of resource recovery from waste sludge (Source: Tyagi et.al, 2013)

26% to be composted (Praspaliauskas and Pedisius, 2017). The recycling production of sludge includes building material substitution, fertilizer in agriculture and urban greening, electricity, and biogas, etc.

As shown in Table 1.1, the main methods of sewage sludge management in the EU remain agricultural use and incineration. A significant abandonment of sludge landfilling is noticed in most European countries, whereas only three countries report a slight increase in landfill use (Italy, Denmark, and Estonia). Despite the fact that approximately about 40% of the total sludge produced in the EU is used for agriculture purposes, the individual EU countries are very different in terms of the amount of sewage sludge that is directed to the soil. Some EU Members have adopted stricter limit values for contaminants than those contained in the Sewage Sludge Directive (SSD). Others added some new contaminants. The several Member States are taking into consideration the environmental risk after applying sludge to agricultural land and have even banned its use, while others use it wisely and are still improving sludge management. However, in other countries for instance in Finland and Belgium, less than 5% is used for agricultural purposes. In Greece, Netherlands, Romania, Slovenia and Slovakia sludge are not used in agriculture. In Poland, a gradual decrease in landfilling of sewage sludge and an increase in their thermal conversion has been observed. In non-EU countries, such as Norway and Switzerland, larger amounts of sludge are applied in agriculture. According to Directive (1999), Norway had an official target to recycle 60% of bio-solids to farmland, which was reached already; in 2008 as 80% of bio-solids were recycled to farmland or green areas. Regarding EU-15 countries, agricultural recycling (direct or after composting) and incineration seem to be the two main practices that will be further adopted.

Table 1.1 Trends on sludge disposal methods applied in EU-15 countries between 2000 and 2009 (Source: EC, 2009)

Country	Agricultural use (%)	Compost (%)	Incineration (%)	Landfill (%)
Austria	4	−5	−12	−5
Belgium	3	0	−1	−19
Denmark	−1	0	−22	4
Finland	−9	17	0	−6
France	−3	20	2	−16
Germany	−4	−8	28	−6
Greece	−2	−1	46	−56
Ireland	28	0	0	−44
Italy	−4	−10	−2	12
Luxembourg	−15	22	11	−18
Netherlands	0	2	16	−14
Portugal	71	0	0	−77
Spain	11	0	−4	−2
Sweden	8	2	0	−12
UK	13	1	0	−3

Fig. 1.6 presents the summary statistic for sludge recycling in Japan (MILT, 2018). The process of sewage sludge treatment is mainly carried out using five processes corresponding to different treatment technologies: 2.6%, 6.0%, 6.8%, 10.8%, and 73.9% of sewage sludge disposal systems are treated by drying, dewatering, melting, composting and incineration, respectively. Sewage sludge used for agriculture and greening is still a common recycling method. The major recycling method of sewage sludge in Japan is making building materials and making fossil fuels after 1997. The contents of sewage sludge recycling in Japan are roughly classified into green farmland utilization, construction material utilization, energy utilization, and valuable resource recovery. And processing processes, such as drying, composting, incineration, and melting, are applied to each form of sewage sludge raw material and are recycled as fertilizer, building material, fuel, and the like.

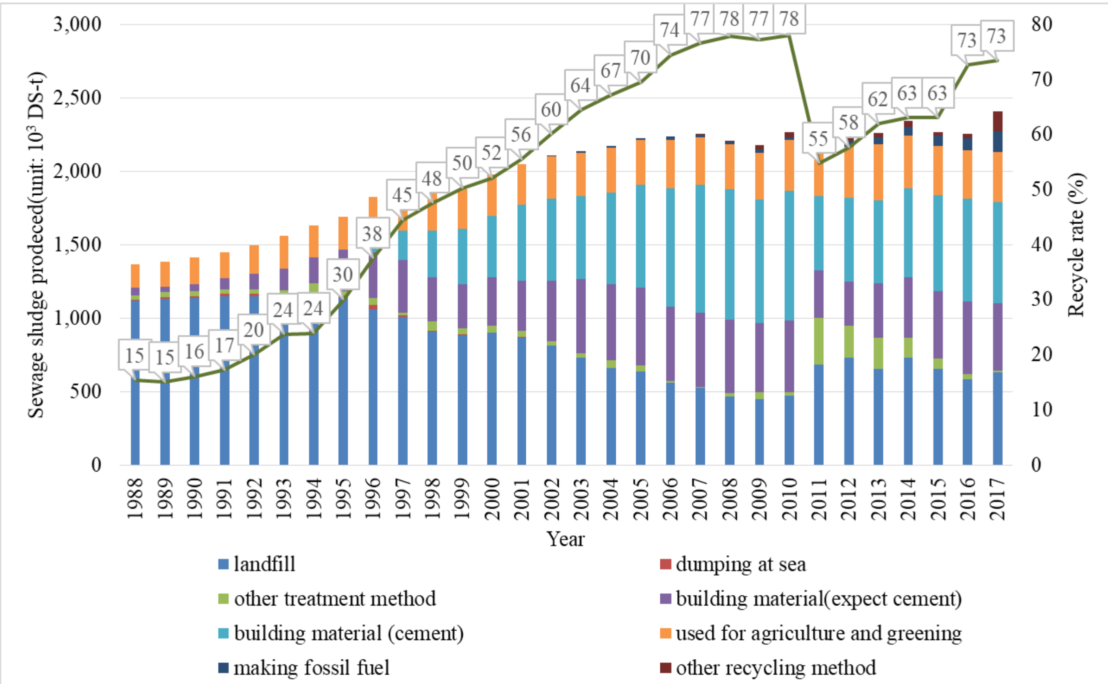

Fig. 1.6 The situation of sewage sludge produced and recycle proportion in Japan
(Source: MILT, 2018)

(2) Situation of SRS in China

The State Council of China published "Action Plan for Prevent and Control of Water Pollution" in 2015. It was the first time to focus on and promote sewage disposal in the policy. From 2015 to 2020, it is clearly stipulated in the policy that attention should be paid to the disposal and recycling of sludge in the wastewater treatment system, as shown in Table 1.2. The financial subsidy for sewage sludge recycling was proposed in 2019 to accelerate the utilization of sewage sludge. Hence, the target rate of sewage sludge harmless treatment was achieved to 90% for cities at the prefecture level and above in 2020 and further improved. At the same time, the scale of landfill would be reduced in prefecture-level cities of Eastern China and large and medium cities of Western China and Central China.

In China, the proportion of sludge treatment and disposal in each province are used as illustrated in Fig. 1.7. As to sludge treatment and recycling, a great variety of technologies is used in China which are related to the size of WWTPs and final disposal conditions. The accurate official data on WAS disposal is scarce, rather conflicting sources with dissimilar estimations. According to Yang et al. (2014), over 84% of WAS is disposed of by improper dumping. Concerning the proper disposal, the sanitary landfill has been the most widely used method, which accounts for 13%, followed by land application (2%), incineration (0.4%), and building material production (0.2%). Data reported by Zhang et al., 2016 showed that the most commonly employed disposal method is landfilling (63%), followed by agricultural applications (14%), and incineration (2%). Moreover, the disposal method for approximately 21% of WAS is unknown. The significant variations of data are attributed to WWTPs in China because they do not send proper reports on the treatment and final disposal of their WAS.

Table 1.2 The relevant industrial policies of sludge disposal in China

Time	Sector	Name of policy	Content
2015	The State Council	Action Plan for Prevent and Control of Water Pollution	1. Promote the sewage sludge treatment and disposal considering stabilization, harmlessness and resource utilization. Prohibit disposal of substandard sludge into agriculture land. 2. Improve the charging policy. The urban sewage treatment fee should be higher than the cost of wastewater treatment and sludge treatment and disposal.
2016	National Development and Reform Commission (NDRC), Ministry of Housing and Urban-Rural Development (MHURD)	The 13th Five-Year Plan for National Urban Sewage Treatment and Recycling Facilities Construction Plan	1. "Emphasize both sludge and sewage". Promote the stabile and harmless disposal of sludge and encourage the sludge recycling. 2. Accelerate the construction or renovation of sludge harmless treatment and disposal facilities.
2016	The State Council	The 13th Five-Year Plan for the Protection of Ecological Environment	The target rate of sludge harmless treatment and disposal in prefecture-level and above cities reaches 90%, and the Beijing-Tianjin-Hebei region reaches 95%.
2016	MHURD, Ministry of Ecology and Environment (MEE)	National Ecological Protection and Construction Planning (2015-2020)	1. By 2020, the target rate of urban sludge harmless treatment and disposal at the prefecture level and above will reach 90%. 2. Strengthen the construction of sludge treatment and disposal facilities and the ability to supervise the operation of facilities.
2017	Ministry of Industry and Information Technology (MIIT)	Guiding Opinions of the Ministry of Industry and Information Technology on Accelerating the Development of the Environmental Protection Equipment Manufacturing Industry	Demonstration of applications in key areas such as urban sewage treatment plants, industrial wastewater treatment facilities, and sludge treatment and disposal

2019	MEE, MHURD, NDRC	Three-year action plan for improving quality and efficiency of urban sewage treatment (2019-2021)	1. Promote the construction of sludge treatment and disposal facilities. 2. The local government should compensate normal costs and reasonably profits from the sludge treatment and disposal facilities to maintain the operation in principle.
2019	Seven government sectors including NDRC and MIIT, etc.	Translating Green Industry Guidance Catalogue (2019 Edition)	Introduce policies and measures in investment, price, finance, taxation, etc. to strengthen green industries such as energy conservation and environmental protection, clean production, and clean energy based on the priorities of respective fields and regional development.
2020	NDRC, MHURD	Implementation plan for the strengths and weaknesses of urban domestic sewage treatment facilities	1. Accelerate the harmlessness and reutilization of sludge disposal to further improve the rate of harmless disposal and recycling for sludge by 2023. 2. Selection of the suitable route of sludge disposal via the quality and composition of sludge and local socio-economic development level. 3. Reduce the scale of sludge landfill in economic developed area.

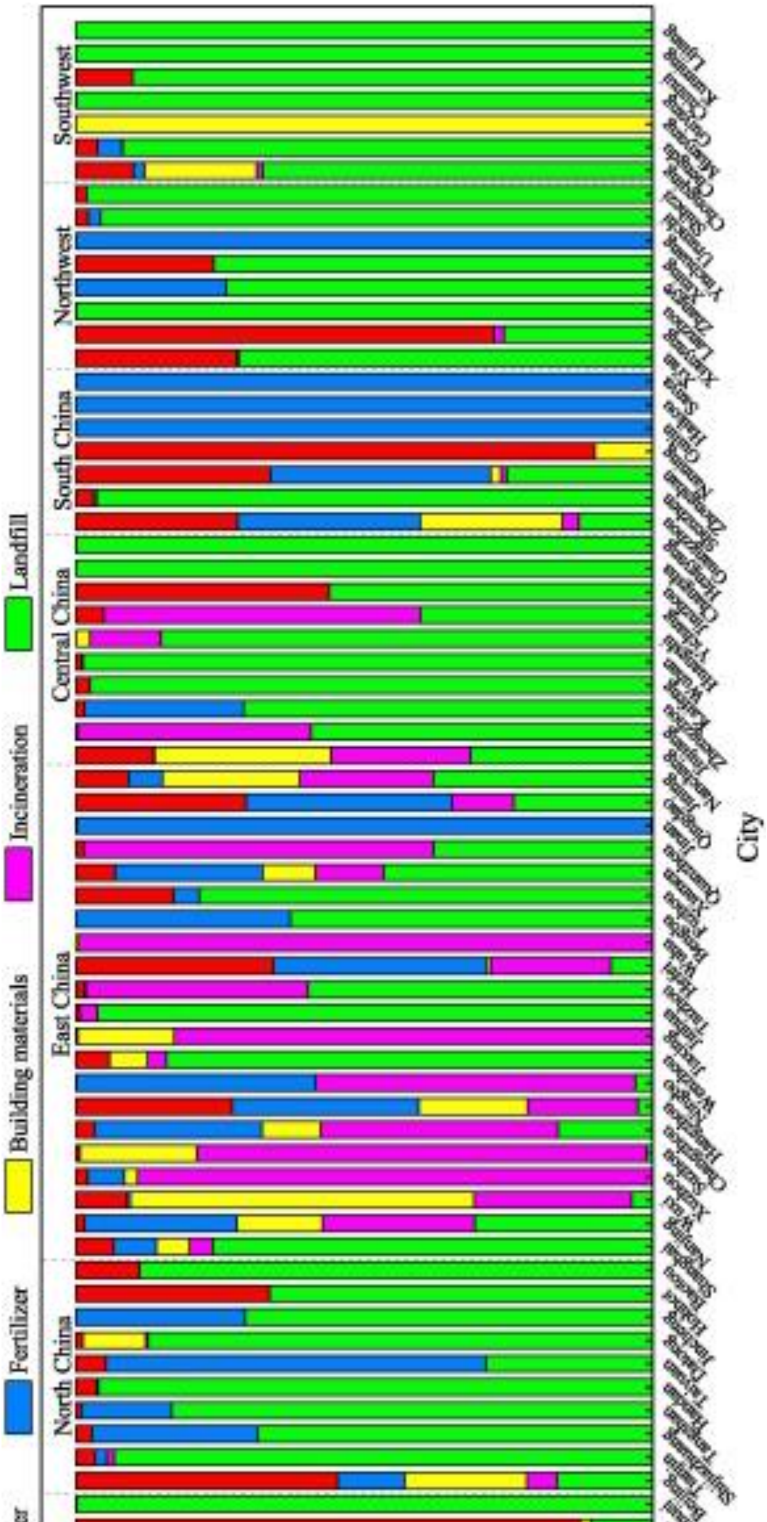

Fig. 1.7 The ratio of sludge recycling methods applied in different cities of China (Source: Zhang et al., 2016)

1.1.3 Status of environmental and economic impact of the SRS

Sewage sludge could be as a sustainable resource being in the form of nutrients and energy recovery caused by a tremendous amount of renewable organic. For example, phosphorus recovery and sludge fertilizer or soil amendment considered renewable products of sludge. The method of incineration, anaerobic digestion (AD), pyrolysis and hydrothermal carbonization (HTC) were widely used to the conversion of sludge to energy (Ding et al., 2021).

The investment fund limited the development of sewage sludge management (Yang et al., 2015). The economic impact was mainly contributed by the high initial cost and operation cost, including energy consumption in previous studies. The sludge disposal as a part of the wastewater treatment system, was public infrastructure industry in China. It belongs to social welfare areas, which most of fund from local government.

In a later comparative study, Heimersson et al. (2017) identified and explored several scenarios to handle the multi-functionality in the LCA of a sludge handling system. The authors focus on environmental assessment of strategic decisions on sludge treatment and end-use using LCA. According to the results of study, acidification potential (AP) and eutrophication potential (EP) were not strongly influenced by modelling approach. Global warming potential (GWP) and photochemical oxidant formation potential (POFP) were heavily affected by substitution applied. In contrast, Linderholm et al. (2012) investigated environmental impact of sewage sludge as a phosphorus alternative for agriculture. The study focused on secondary function like nutrient provision to soil.

In another comparative study, Johansson et al. (2008) pointed the lowest environmental impact of the studied systems was the supercritical water oxidation with phosphorus recovery, but the authors pointed sewage sludge treatment methods with land application will contribute the transfer of heavy metals to soil. Point of view of phosphorus recovery from sewage sludge, Lederer et al. (2010) evaluated six options of sewage sludge treatment used IMPACT2002+ and CML from air, water and soil three aspects. In addition, the study compared NOx emission of each option and impacts of heavy metal emissions for each category on the atmosphere, hydrosphere and soil.

On the other hand, Valderrama et al. (2013) and Hong and Li (2011) evaluated sewage sludge as an alternative raw material or fuel for industrial production based on mid-point assessment and end-point assessment. In another line of related research Cao and Pawłowski (2012) studied the energy and greenhouse gas emission footprint of two emerging sludge-to-energy system used LCA. The results showed two systems not only can be credited considerable net energy but also reducing greenhouse gas emissions.

In similar, Alyaseri and Zhou (2017) compared the fluidized bed incineration with AD. They concluded that the fronter was recommended with the attention of human health. If the focus is on resource depletion or sustainability without specific emphasis, AD would be suggested.

1.2 Significance and objective of the study

1.2.1 Significance

There is a consensus that the large quantity of sewage sludge rapidly increases and most of them do not effectively and properly treat in China. On the other hand, sewage sludge is a great source of resource and energy recovery with rich organic matter. Therefore, sewage sludge could be a biomass resource in the urban under the policy of a "zero-waste" city and increased demand for energy and resource in the urban. The verification of the environmental and economic performance of the sewage sludge recycling system is a priority. It can compare the optimal system in accordance with the environmental and economic performance. The standard of sewage sludge recycling and the average greenhouse gas emission of sewage sludge treatment at the national level have also been released by the Chinese government. Still, there is a lack of comparative analysis of the environmental and economic performance of the sewage sludge recycling system, which focuses on substitution replacing the conventional product. Currently, comparative management of sewage sludge recycling systems is lacking. The influence of related policy on the economic performance of the sewage sludge recycling system should be considered. According to the maximizing the economic efficiency and minimization of initial cost, the policymakers and investors construct the sewage sludge recycling projects. The effect of the implementation of scale on environmental and economic performance is ignored.

Finally, substitutions offset the environmental impact of conventional products. The carbon emission trading of emission allowance is the factor in promoting the development of the sewage sludge recycling system based on carbon neutrality. The management of the entire system under the condition of carbon neutrality lacks careful consideration.

1.2.2 Objective

The work of this study as follow:

(1) To estimate the environmental and economic impact of different sludge recycling systems considering the energy and resource recovery and analysis.

(2) Comparison of life cycle cost of different system under policy analysis, the influence of policy changed by tax and subsidy on the choice of system is studied.

(3) An integrated evaluation of environment and economy was constructed which environmental impact monetized as external cost. The main contribution of environmental impact to externality costs that mark potential social damages caused by pollutant discharge for sewage sludge recycling system were also determined.

(4) To clarify the relationship between the implementation of scale and unit environmental emission and economic cost. The minimization scale was determined to achieve minimize total cost, combined the initial cost, operation cost and the revenue of by-product of different sewage sludge recycling systems. In addition, after introducing the carbon trading mechanism, break-even scale of different systems was quantified.

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2 A REVIEW OF PREVIOUS STUDIES AND METHODOLOGIES

2.1 Estimation methods of environmental and economic impact of sewage sludge recycling system

2.1.1 Estimation methods of environmental impact of sewage sludge recycling system

The method of environmental impact assessment included Emergy analysis, Life cycle assessment (LCA), IPCC, Material Flow analysis (MFA) or Energy Flow analysis (EFA), etc. Among them, Emergy analysis, which emphasized energy flow, proved a series of factors to illustrate the efficiency, sustainability, environmental cost, and the benefit of the production system (Amaral et al., 2016). The IPCC were a widely standard methodology to assess the risk of climate change caused by human activity, potential impacts of climate change and possible options for prevention (Heimersson et al., 2016; IPCC, 1996, 2000, and 2006). MFA or EFA focused on the material or energy flow of the target system (Naohiro et al., 2015; Liu et al., 2021).

Life cycle assessment was a technique to assess environmental impacts associated with all the stages of a product's life from raw material extraction through materials processing, manufacture, distribution, use, repair and maintenance, and disposal or recycling (Fig. 2.1). Since this century, researchers had applied the LCA method to the selection of sludge disposal technologies and have carried out a series of analysis and research on the environmental impact of sludge disposal.

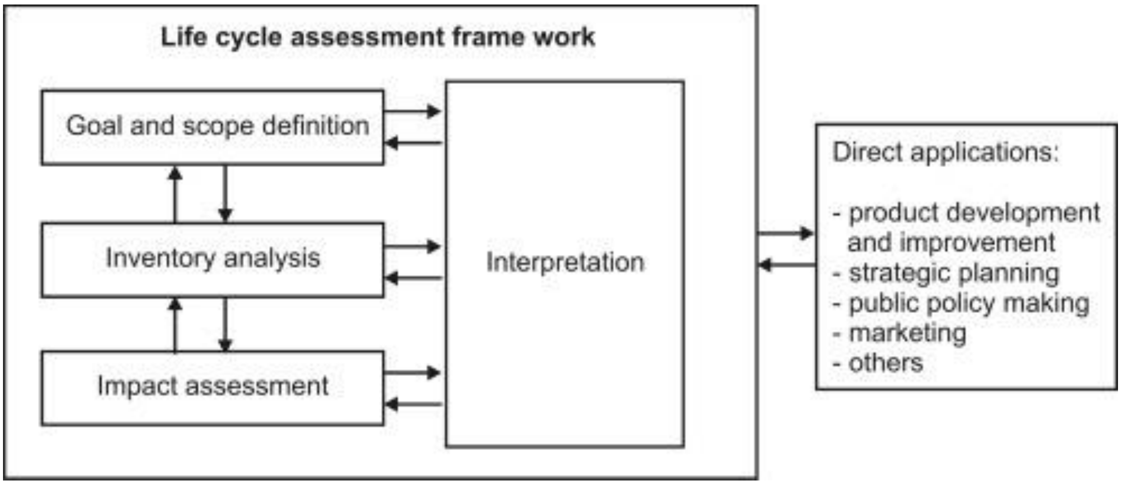

Fig. 2.1 The framework of LCA (Source: ISO 14040)

Goal and scope definition determines the scope and workload of the entire study. The purpose of the evaluation has determined by the research purpose and the reasons for the research, moreover, the scope and elementary of the study have determined based on research purpose, including definition of evaluated product system, function of product system, function unit of the study, system boundary, the type of impact selected, impact assessment method, assumptions of product system and relevant data requirements and constraints. Goal and scope definition is the first step of life cycle assessment, and it affects accuracy of the entire evaluation work process and research conclusions. It is a significant step in the life cycle assessment process.

According to determining the function units in the LCA, Life cycle inventory analysis refers to the collection and calculation of data during the entire life cycle of the product or service. Thereby, the input and output of the system is quantified. The elementary flow of each unit in the system boundary can be composed of the following aspects: input of energy and raw material, products, by-products, and direct emissions to water, atmosphere and soil.

The LCIA phase of an LCA study provides a system-wide perspective of environmental and resource issues for product or service systems. The characterization stage of LCIA evaluated the relative significance of each flow to the overall impact by converting these to an impact category's characterized unit. The weight factor of LCIA include Eco-indicator 99, CML2001, LIME, and ReCipe method, etc.

Leiden University proposed the CML2001 method according to the ISO standard, subsequently proposed the ReCiPe method based on CML. A fundamental difference between the Eco-indicator 99 and ReCiPe method is that the ReCiPe method has a problem-oriented approach to impact assessment as opposed to the Eco-indicator 99 method which has a damage-oriented approach. This means that while ReCiPe methods model the Impacts at a midpoint somewhere in the environmental mechanism between emissions and damages, Eco-indicator 99 aims at modeling damage to the projection areas: human health, natural manmade environment, and natural resources. ReCiPe can be seen as a fusion of the two methodologies, taking the midpoint indicators from CML and the endpoint indicators from Eco-indicator. The framework of ReCiPe method is shown in Fig. 2.2.

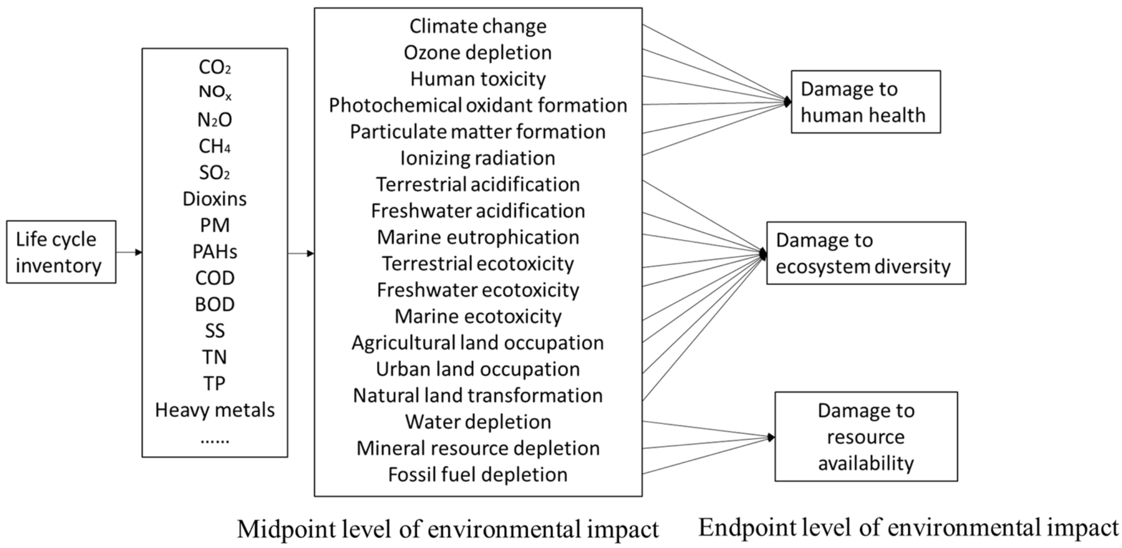

Fig. 2.2 The environmental mechanism framework of ReCiPe method

2.1.2 Estimation methods of economic impact of sewage sludge recycling system

The widely employed to evaluate the economic performance of sewage sludge treatment and disposal methodologies were cost-benefit analysis (CBA), technology cost analysis (TCA) and life cycle cost (LCC). Among them, cost benefit analysis widely was used to weigh up the economic performance of the business or policy decisions, commercial transactions, and project investments via comparing the total expected cost of each object with it total expected benefits. It can be an effective method to analyze the difference alternatives considering the social cost and benefit (Ness et al., 2007). Technology cost analysis focused on the relationship between technology values and cost values that influence the cost level and cost structure. Life cycle cost referred to the total cost produced during the entire life cycle of a product, process, and activity included research and development, mining, production, transportation, sales, use, and recycling until final disposition.

The Society of Environmental Toxicology and Chemistry (SETAC) classified LCC into Conventional, Environmental and Societal LCC. The specific industrial international standards and guidelines were developed based on the Conventional LCC method. Environmental LCC, as economic "cousin" of LCA in Rebitzer et al. (2003) did not have commonly framework and methodology. The critical relationship between the LCA and Environmental LCC was the result of Environmental LCC calculated based on monetized the environmental impacts. Meanwhile, Societal LCC was at the beginning of development. The selected economic factor for summing up the cost like Net Present Value (NPV), Net Future Value (NFV), Annual equivalent (AEW), return of investment (ROI), and the method considering the discounting rate.

Due to the partially interrelated methodologies existed, LCC and CBA was less easy to delineate. The nuances of key aspects have to be taken into account between LCC and CBA. The different of LCC and CBA in cost target is system boundary. The cost target of LCC is social cost within the system boundary comparing with CBA. The system boundary in this dissertation included the transportation, sewage sludge treatment and recycling process. For CBA, the benefit to the surroundings was not considered. For example, if the study of sewage sludge recycling focused on the negative impact that are caused by the emission of the sewage sludge recycling system, the system is taken as a product. However, when considering whether the system should be constructed or not, the construction of the system can be seen as a project and CBA would be appropriate(Hoogmartens et al., 2014).

2.2 Life cycle assessment and life cycle cost of sewage sludge treatment and recycling system

2.2.1 Life cycle assessment of sewage sludge treatment and recycling system

The summary of previous studies about system boundary was present in Fig. 2.3. In a study based in France, Suh and Rousseaux (2002) examined process of sewage sludge treatment including one main process, one stabilization process and transports of sludge. The study used SETAC/CML to assess the environmental impact. The authors pointed out that further study should paid attention to the non-point-source pollutants such as emission during transportation.

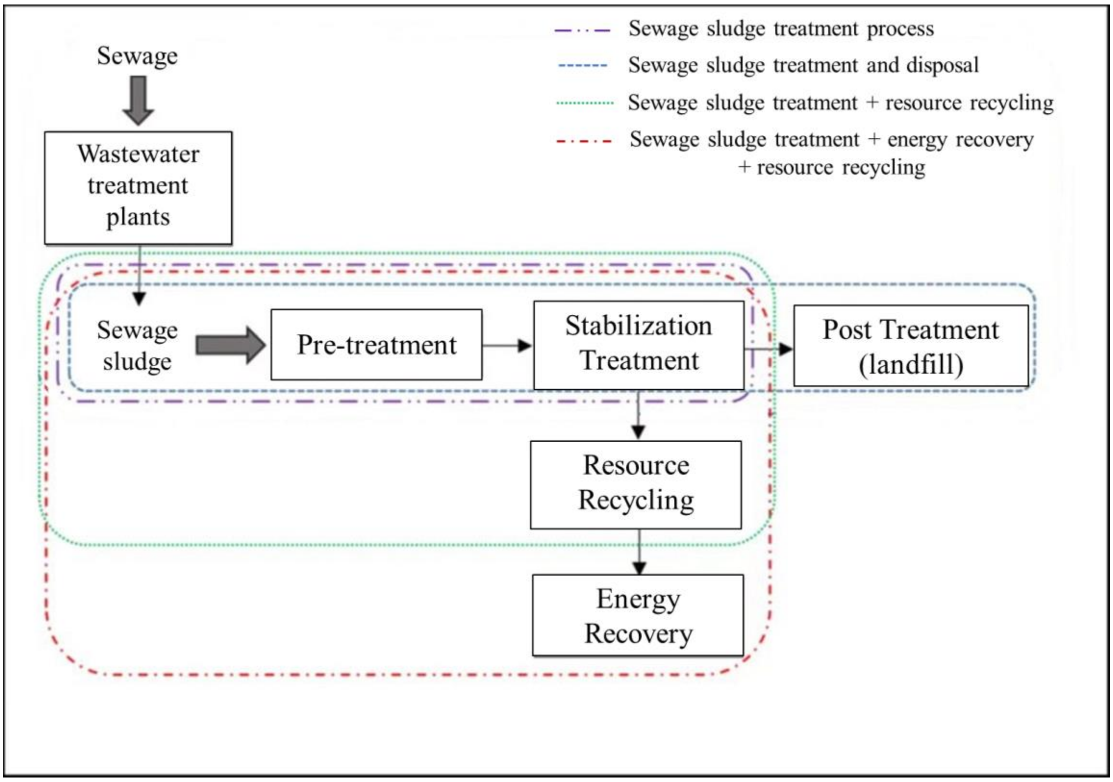

Fig. 2.3 The system boundary of LCA for STRS

Hiroko et al. (2018) evaluated five commonly applied sewage sludge treatment practices using life cycle assessment. The authors analyzed environmental impacts and the fate of carbon, nitrogen and phosphorus. The result showed the highest normalized impact potentials for all the scenarios and the relationship with fate of nitrogen and phosphorus. Further, Hospido et al. (2010) evaluated reusing of anaerobically digested sludge in agriculture from an environmental point and specifically quantified potential impacts of emerging micro-pollutants such as pharmaceuticals.

In a study based in China, Liu et al. (2011) proposed an industrial symbiosis model, which uses industrial waste heat to dry sludge while using dry sludge and other fuel instead of fossil fuel. The authors assessed energy consumption and environmental impacts of seven scenarios of the model using LCA. The result showed that the model will reduce the global warming potential, but acidification and human toxicity will get worse.

Lastly, Xiao et al. (2018) compared different technologies from environmental and economic performance using LCA and evaluated the most optimal sewage sludge treatment in Xiamen. The authors create the particle swarm optimization (PSO) model based on the LCA result to find a sustainable sludge management system. Finally, the studies indicated the water content affects the human toxicity of environmental impact in dewatering process. The main characteristics of previous study was presented in Table, including the country, function unit (FU), method, compare scenario and result.

Table 2.1 The main characteristics of previous sludge treatment and recycling system in LCA

	Country	FU	Method	Compare scenario	Result
Hospido et al., 2010	Spain	10 L of mixed sludge (primary sludge: secondary sludge =7:3)	CML 2000	Non-digestion and four anaerobic digestion (AD) and land application scenarios with different temperature and sludge retention time (SRT)	1. The digested sludge applied on agricultural soil reduced the environmental impact of eutrophication and toxicity, comparing with undigested sludge. 2. The results of global warming for digested scenarios were higher than undigested scenario due to the energy consumption and without the energy avoided.
Hospido et al., 2005	Spain	1t thickened mixed sludge	CML 2000	Four scenarios including the land application, incineration and pyrolysis comparing with and without substitutions	The effective utilization of sewage sludge was both energy and material recovery.
Hong et al., 2009	Japan	1t dry solid	USES-LCA model	Six scenarios considering the drying, composting, incineration and melting	1. The scenario of melting was the optimal environmental scenario. 2.To cut down on environmental impact, the efficiency of thickening process, flocculants and electricity generated rate should be increased.
Suh et al., 2017	France	1 ton (dry weight) of the mixed sludge	SETAC /CML	Five scenarios combined with thickening, dewatering, stabilization (lime-stabilization, composting, AD and incineration) and main treatment (landfill and	The combination of AD and land application was most environmentally friendly scenario.

	Country	FU	Method	Compare scenario	Result
				agricultural land application)	
Li et al., 2017	China	1t dry solid	CML 2000	Five scenarios of different AD technology considering biogas recovery.	Biogas production was the most sensitive factor determining the results of LCA.
Li et al., 2018	China	1t thickened sludge	CML 2000	Three scenarios about AD and pyrolysis (PY)	The AD scenario was recommended for low organic content sludge. On the contrary, AD+ PY scenario was an optimal scenario.
Xu et al., 2018	China	One dry tone of sludge	ReCipe	Five scenarios including landfill, composting, incineration, hydrothermal-pyrolysis technology (HPT) with different water content	HPT had the lowest consumption of land sources and a relatively small environmental impact in five scenarios.
Luo et al., 2021	China	1 metric ton (dry weight) sludge	ReCipe	The sludge pyrolysis system under various conditions such as sludge moisture content and system size of the plant, etc.	The sludge with lower moisture content and higher organic content and large sludge pyrolysis system had lower global warming potential.
Xu et al., 2014	China	1t dry sludge	ReCipe & IMPACT 2002+	Thirteen scenarios combined with thickening, dewatering, drying, AD, incineration, landfill and agricultural use.	The most suitable method was the scenario combined gravity thickening, AD, dewatering and incineration considering the energy recovery.
Han et al., 2021	China	1t sludge with 80% moisture content	CML 2015	Six scenarios including landfill, incineration, used in material for brick and cement, land application and co-generation.	The scenarios of AD and used in material (brick) had the less environmental impact.

2.2.2 Life cycle cost of sewage sludge recycling treatment and recycling system

The circular economy applied to sewage sludge offers an alternative to the traditional model, considering the energy and material recovery from the sewage sludge. However, the application of sewage sludge recycling is limited by the economic risk existing and the lack of environmental and social benefits assessed. The State Council of China has pointed out that sewage sludge should be stabilized, harmless, and recycled before landfill and prohibited to land application if the sludge treatment is not up to the standard in 2015. It proposed to revise the urban sewage treatment fee, among which the urban sewage treatment fee shall not be lower than the cost of sewage treatment and sludge treatment and disposal. From 2011 to 2015 in China, the investment ratio of sludge treatment in wastewater system was only 8% (Yang et al., 2015). Due to the fund shortage, the sewage sludge recycling system should be considering the economic performance to better operating in a long term. The result of life cycle cost of sewage sludge recycling system provided a reasonable reference in economy for cities and town government to lighten the financial stress.

Li et al. (2017) evaluated five different anaerobic digestion pathways based the environmental and economic assessment using life cycle inventory assessment (LCIA) and net present value (NPV). The authors pointed the energy output related to biogas production is the most sensitive parameter.

Piao et al. (2016) used economic efficiency analysis (EEA) to assess the cost of electricity and chemical consumption, transportation, disposal cost of dewatered sludge, and the benefit of biogas.

Xu et al. (2014) analyzed the LCC of six sewage sludge treatment scenarios with and without AD using the invested data, the scenario with the least LCC was gravity thickening, AD, dewatering, and landfill. Compared with the result of Hong et al., 2009 which assessed LCC via construction and equipment cost, energy consumption, maintenance cost, flocculants and labor cost, and treatment cost, this result was lower than the result in Japan. The authors analyzed the reason could be the unit price and did not consider the carbon tax.

Taboada-Santos et al., 2019 analyzed the payback time included the cost of electricity, hygienization cost for composting of digested sludge and cost or revenue of electricity. Techno-economic analysis (TCA) was used to evaluate the economic feasibility of energy and P recovery from municipal sewage sludge about 16 scenarios in Bagheri, et al., 2022. And finding of that study prove none of the scenario was economically feasible without the subsidy from the government. In Yang et al., 2020, the authors implied cost-benefit ratio to evaluate the technologies' economic performances of four scenarios included continuous thermal hydrolysis, AD, aerobic composting, and dry-incineration.

2.3 Factors affecting the environmental and economic impact of SRS

2.3.1 Composition of sewage sludge

The composition of sewage sludge influenced the agronomic efficiency, environmental safeness, and economic feasibility of fertilizer substituted, such as organic matter and heavy metal. (Rodrigues et al., 2021) The theoretical thermal energy of incineration was equivalent to the high calorific value of the dried sludge, which was related to the organic content. (Hao et al., 2020)

More recently, Li et al. (2018) compared three pathways from life cycle environmental impacts and energy efficiency. The studies calculated the environmental impacts of three pathways under 40%, 50%, 60%, and 70% of organic content of the sludge. Syed-Hassan et al., 2017 discussed the properties of the sewage sludge relevant to the thermochemical conversion (pyrolysis, gasification and incineration), such as nontoxic organic carbon compounds, nitrogen and phosphorous containing components. The composition of sewage sludge in China was presented in Table 2.2.

Table 2.2 The composition of sewage sludge in China

Factor	Primary sludge	Waste activated sludge	Digestion sludge
pH[a,b]	5.0-8.0	6.5-7.5	6.5-7.5
Dry solid[a,b](DS, %)	2.0-8.0	0.5-0.8	6.0-12.0
Volatile solid (VS, %)	60-80	60-80	30-60
BOD_5/VS[b]	0.5-1.1	-	-
COD/VS[b]	1.2-1.6	5.0-8.0	-
Organic matter rate[a,b] (%)	60-90	70-85	30-60
Cellulose rate[a] (%)	8-15	5-9	8-15
Hemicellulose rate[a] (%)	2-4	-	-
Lignin rate[a] (%)	3-7	-	-
Fats rate[a] (%)	6-35	5-12	5-20
Protein rate[a] (%)	20-30	32-41	15-20
C/N[b]	(9.4-10):1	(4.6-5.0):1	-

a. Yan, Z., (2014); b. Lin and Liu, (2000).

2.3.2 Technological options

Xu et al. (2014) created thirteen scenarios for sewage sludge treatment considering gravity thickening and the difference between with or without anaerobic digestion. The authors used the ReCiPe and IMPACT 2002+ methods to assess environmental impacts of all scenarios from 18 midpoint categories. A later study by Piao et al. (2016) compared difference sludge management alternatives, which treatment from wastewater to sludge, used LCA provided a valuable example for wastewater treatment plants in a large city

On the other hand, Houillon and Jolliet (2005) compared six wastewater sludge treatment scenarios and focused on energy and emissions contributing to global warming over the whole treatment life cycle. Hong et al. (2009) estimated environmental and economic impacts of six commonly alternative scenarios in Japan used LCA in Fig. 2.4. The results of USES-LCA model showed dewatered sludge melting was an environmental optimal and economically affordable method. The authors indicated heavy metal of human toxicity is the main influencing factor of environmental impacts for melting process.

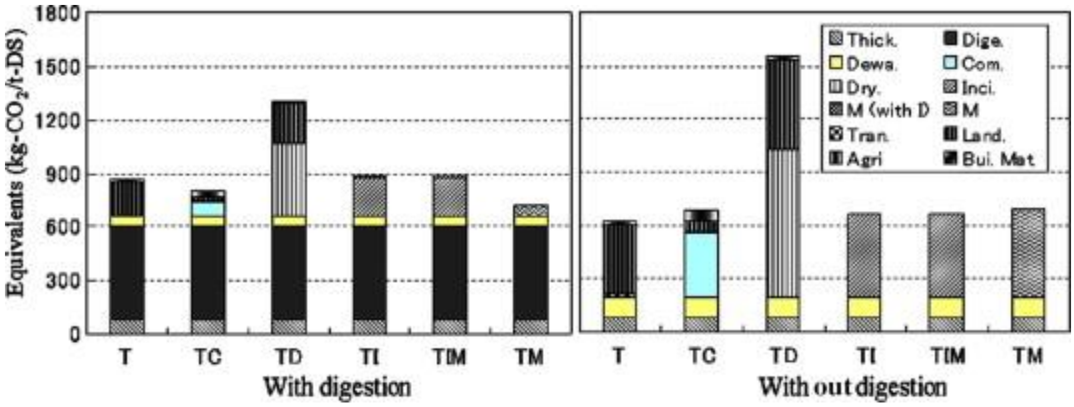

Fig. 2.4 Comparison of different sludge treatment scenarios in global warming.
(Source: Hong et al., 2009)

2.3.3 The scale of operation

According to the sensitivity analysis of the size of the facility, Houillon and Jolliet (2005) proposed the conclusions could not generalize to all sizes of facilities. Chen et al. (2021) examined the economic efficiency of sludge co-process with municipal waste and the limitation of the study, which did not take into account the different capacities of incinerators. Or else, the analysis was conducted on the large-scale of sewage sludge treatment and recycling system to avoid the effect of implementation scale. Some researchers show that energy consumption and operation costs are related to the implementation scale. Luo, et al. (2021) presented the result of environmental and economic analysis for full-scale sludge pyrolysis systems and proved the larger pyrolysis system of centralized sludge handling was more economically favorable in Fig 2.5. Kumar et al. (2020) discussed the relationship between the payback period and return of investment and plant capacity. In term of the cost of equipment and size of equipment, it was carried out by power function.

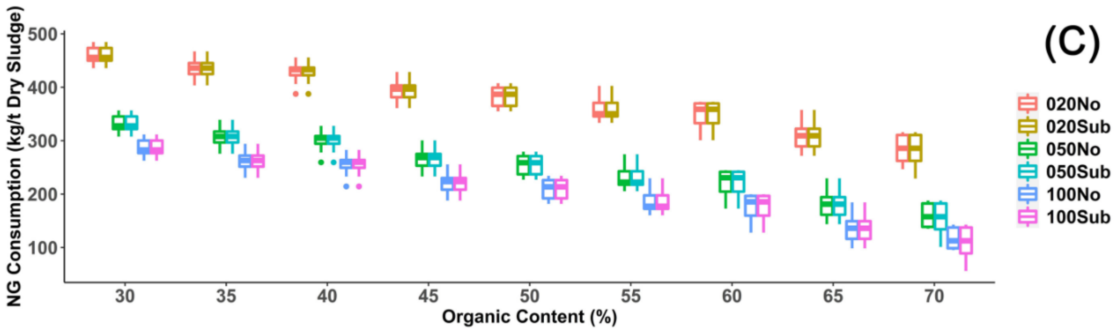

Fig. 2.5 The consumption of natural gas for pyrolysis system in different system size. (Source: Luo, et al., 2021)

2.4 Innovations of this study and the framework of this dissertation

2.4.1 Innovation

(1) Quantifying the implementation of scale affect the environmental and economy of sewage sludge recycling system.

Previous studies did consider the specific relationship between the implementation of scale and environmental and economic performance of sewage sludge recycling system. The aim of this study is to analyze the implementation of scale that affects the environmental and economic performance of sewage sludge recycling system with different technological selections (i.e., incineration, aerobic composting, used in material, and anaerobic digestion). First, we collected and analyzed the environmental and economic performance of the sewage sludge recycling system. Next, we determined the effects of implementation scale and environmental and economic performance of the sewage sludge recycling system. In addition, the second objective was to identify the impact of the scale after introducing a carbon trading mechanism into the sewage sludge recycling system. This study provided a quantitative analysis of the effect of implementation scale of different sewage sludge recycling system through defining the break-even scale.

(2) Improving an LCA and LCC evaluation model of sewage sludge recycling system considering the implementation of scale and technology selection.

LCA of sewage sludge recycling system is primarily based on the environmental emission during the operating process to ensure the environmental impact of the system. The aim of LCA was to compare the environmental impact of different cases and analyzed the potential reduction of environmental impact. However, in the face of development challenges of the amount of sludge upsurge, pollutant reduction, and energy conservation, this emphasis on the technology selection and implementation of scale with energy or resource recovery will highlight its shortcomings.

The economic performance of the sewage sludge recycling system is calculated by LCC covering the cost and revenue of the entire system. LCC aims to assess the asset of total cost in the entire life cycle, including the initial cost, maintenance cost, operation cost, and asset's residual value at the end of its life. The related policy and the revenue of alternative carbon emission allowance of sewage sludge recycling management will influence the operation cost of the system.

Hence, the following has been conducted in this dissertation:

1） The environmental performance of the sewage sludge systems focused on energy and resource recovery considering the carbon price and the trading of carbon emission allowance was analyzed. This helps to understand and compare the environmental impacts and provides suggestions for introducing the caron trading mechanism into the sewage sludge recycling system in the future.

2） The economic estimated model by LCC was comparing the total cost and payback period of different systems considering the relevant policy affected. On the basis of the carbon trading mechanism, the break-even scale was determined, which could provide suggestions to policy decision-makers and investors.

3） An integrated LCA and LCC estimated model considering the technology selection and the implementation of scale was constructed, which can normalize the environmental and economic impact assessment of each case study to supply the feasibility of comparing the different systems.

2.4.2 Framework

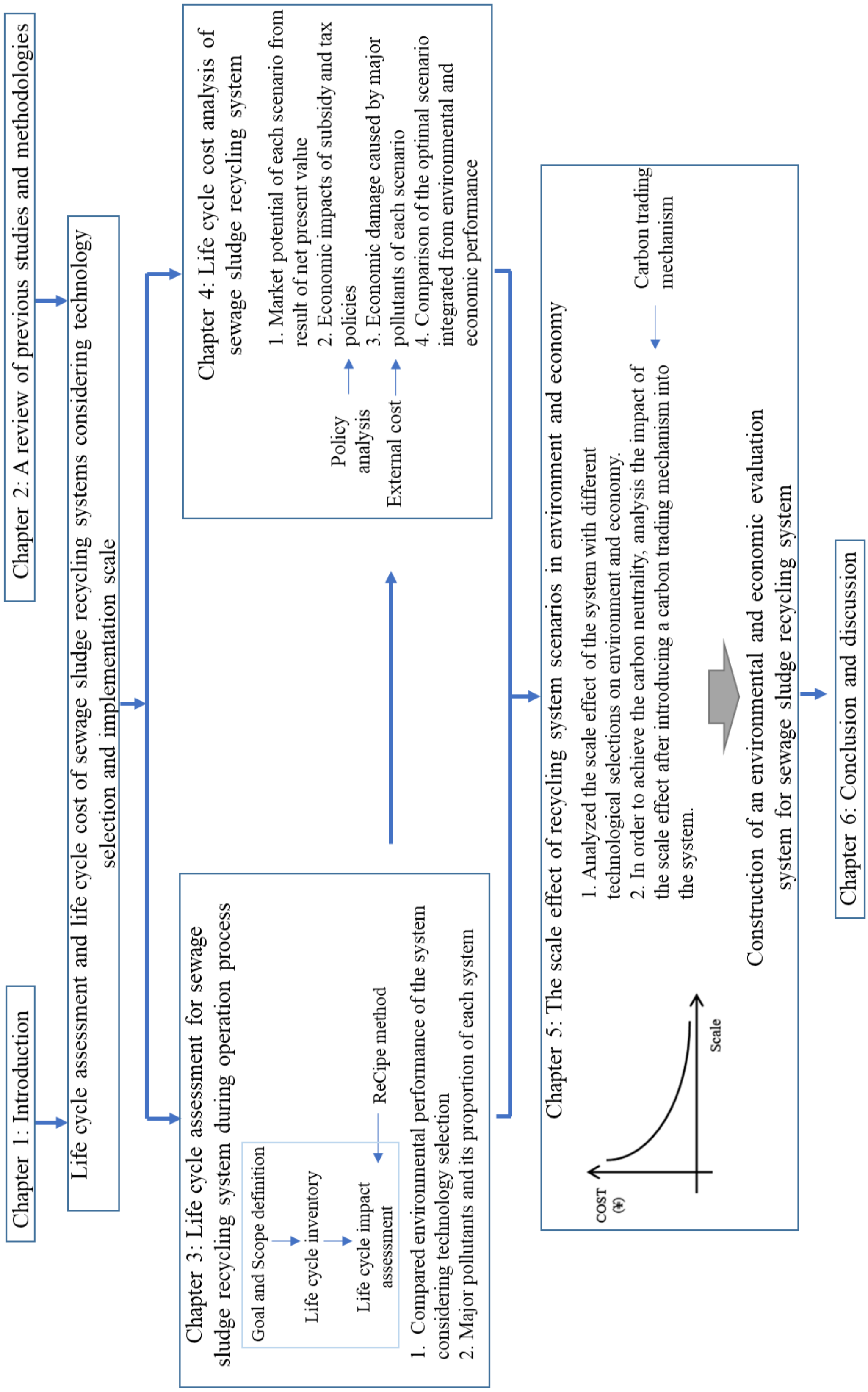

Fig. 2.6 The framework of this dissertation

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3 LIFE CYCLE ASSESSMENT FOR SEWAGE SLUDGE RECYCLING SYSTEM DURING OPERATION PROCESS

3.1 Boundary definition and function unit

The goal of this Chapter is to compare and assess four sewage sludge co-processing with agricultural wastes scenarios from an environmental performance and to compare them with difference policy scenarios. The function unit is processing 1t of sewage sludge with 80% moisture content. The system included transportation, drying, pretreatment, production processing and air pollution control treatment. Final production (i.e., electricity, fertilizer, brick, biogas, and fertilizer in greening) generated in scenarios' reutilization process were considered to replace the traditional and similar commodities on the market in full (Fig. 3.1). In order to improve the biogas production ratio, straw added to the scenario 4,5 and 6, as the material of co-digestion. Scenario 1,2 and 3 added the straw incineration to be consistent with the system boundary of each scenario. To avoid the allocation of any by-products in the process, system expansion was applied. The construction phase is not examined in this study because it had a negligible impact and GHG emissions at this stage were less than 5% (Liu, et al., 2013).

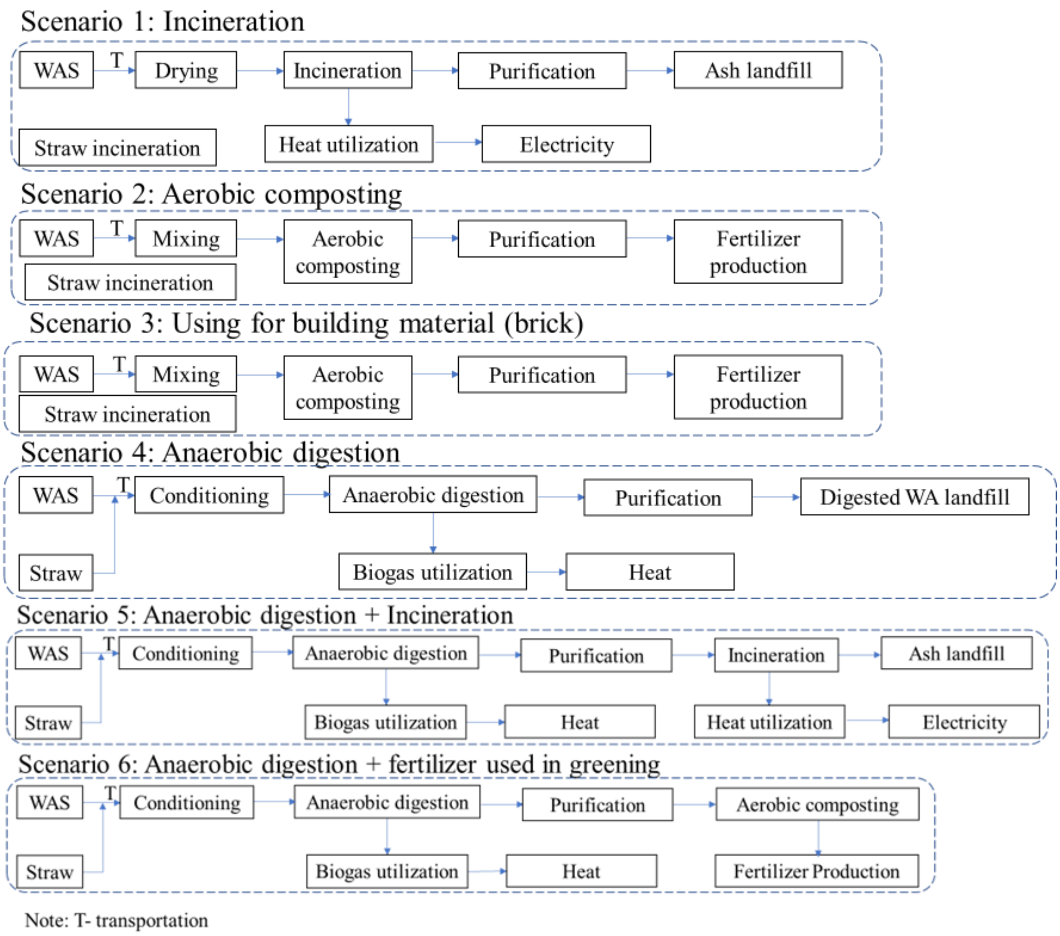

Fig. 3.1 System boundaries of six scenarios of WAS recycling system

3.2 Scenario introduction

3.2.1 Scenario 1: Incineration

Scenario 1 (Incineration) is an effective and general method to reduce quantity and toxicity of WAS and to recover electricity from waste heat. It included the straw incineration individual which the amout of straw equal to co-process with sludge in Scenario 3, 4, 5 and 6. Calorific value of MAS over 5000 and 3500 kJ/kg could mono-incineration and incineration with auxiliary fuel, respectively under the standard GB/T 24602-2009. To ensure stable and efficient combustion of MAS, it needs to be further dried up to 10-30% after mechanically dewatered (80-85% moisture content). Thence, 1 ton of sewage sludge with 80% mositure rate was burn to generate 9.6 kwh electericity. (Liang et al., 2021). Electricity and auxiliary fuel are major consumption during drying process and combustion. The main source of pollution is fuel gas.

3.2.2 Scenario 2: Aerobic composting

Scenario 2 (Aerobic composting) is included primary composting and secondary composting. The amount of straw incinration individual is identical like scenario 1. The WAS produced fertilizer is in line with relevant standard (CJ/T 309-2009 and 20076359-T-333). The substitution of fertilizer was calculated based on the content of nitrogen (N%) in fertilizer. According to the standard of sludge in agriculture, the N% of the unit sludge fertilizer was equal to 8% unit chemical fertilizer. Therefore, 1tone of sewage sludge could relace 7.86 kg of fertilizer used in agricultural. Electricity is a major combustion. The gas emission during composting and heavy metal from WAS are the major pollutants (Seleiman et al., 2020).

3.2.3 Scenario 3: Using for building material (brick)

Scenario 3 (Using for building material) is sintering technology with WAS: other raw material = 1:9 at 800-1060°C in line with GB/T 25031-2010.The produced brick is in line with relevant standard (GB/T 25031-2010 and GB/T 5101-2017) and can replace conventional brick mainly used as construction material. According to the EIA reports and previou studies, 1 ton of sewaage sludge replaced clay as raw material and could generate 351 tons of fired bricks (Limami et al., 2021). The system of scenario 3 included the straw incineration individual. The coal combustion and flue gas emission are the major pollutants.

3.2.4 Scenario 4: Anaerobic digestion

Scenario 4 (Anaerobic digestion) is widely used in processing sludge to recovery methane and reduce sludge volume. WAS with additives fed into anaerobic reactor of 20d about 35-37°C to produce biogas with 55-65% methane in line with T/CECS 496-2017 (Di Capua, F., et al., 2020). The biogas production rate of this case was 41.6m3/t based on the data of EIA. Electricity is a major combustion. Emissions from anerobic reactor is the major pollutants. The municipal sludge compost and straw, not only make the water in sludge decrease quickly but also make full use of sludge microorganisms, a large number of nutrients. This method also make straw decomposition.

3.2.5 Scenario 5: Anaerobic digestion + Incineration

Scenario 5 (Anaerobic digestion + Incineration) is anerobic digestion add incineration involved five steps: thickening, anaerobic digestion (AD) with straw, thermal drying, and incineration. Scenario 5 was a hypothetical scenario in our study without projects in operation (Liu et al., 2021). Base on the previous study, 1 tone of digested sludge with 30% moisture rate could burn to generate 855.8 kwh. Electricity is a major combustion. Emission from anaerobic reactor and incinerator after AD are the major pollutants.

3.2.6 Scenario 6: Anerobic digestion + fertilizer used in greening

Scenario 6 (Anaerobic digestion+fertilizer used in greening) involves AD with straw, composting and drying to used in gardens or parks. The active ingredients (%N, %P_2O_5 and %K_2O) and organic content of fertilizer used in gardens or parks is over 3% and 25%, respectively. Base on these data, 6.7kg fertilizer used in the garden could be replaced by sludge fertilizer produced by 1 tone ewage sludge. Electricity combustion and AD process contribute to the main pollutants. The fertilizer used in gardens or parks quality complies with standard GB/T 23486-2009.

3.3 Data source for estimating

Life cycle inventories for incineration, aerobic composting, the production of bricks, and anaerobic digestion were collected from enterprises via environmental impact assessment (EIA) report. Life cycle inventories of replaced main products were based on the Chinese Life Cycle Database and previous studies (Di Capua, F., et al., 2020 ; Di et al., 2005; Song et al., 2018; Luo et al., 2009). The implementation scale, efficiency of each technology, and quality of WAS may significantly affect environmental and economic performance. Therefore, each WAS from wastewater plants needs to be evaluated on a case-by-case basis (Ding et al., 2021). To perform a comparative study, we assumed that the quality of WAS complies with the standard GB 24188-2009 and that quality changes and differences in regions do not significantly influence the evaluation results in this study.

3.3.1 Collection of data during the straw treatment individual

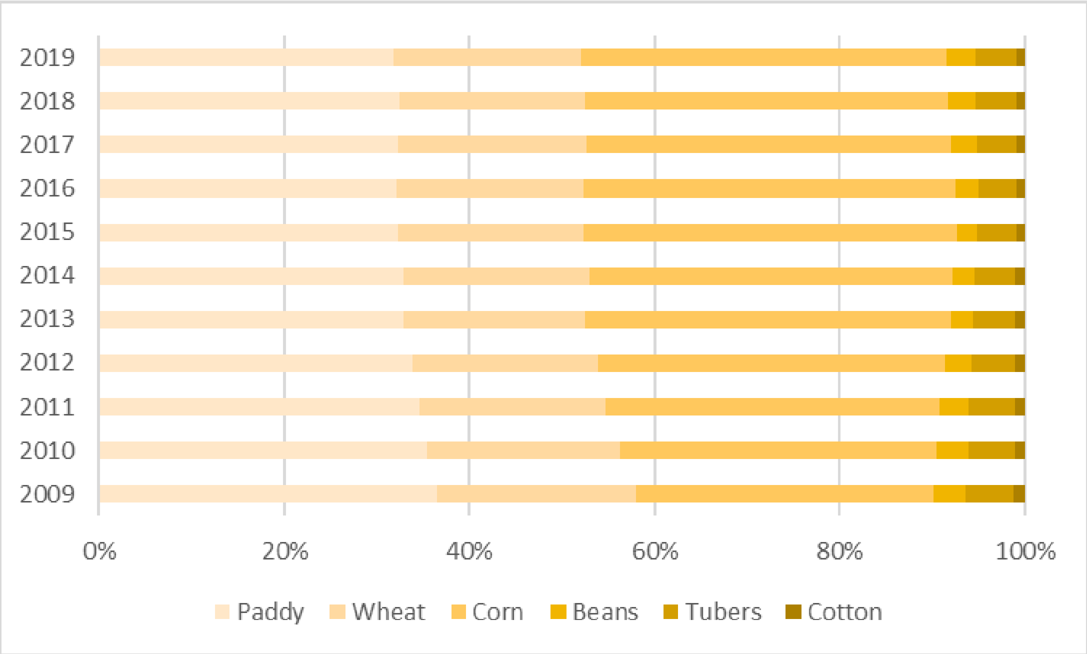

Fig. 3.2 The proportion of crops in China from 2009 to 2019

Straw is an important agricultural waste, and makes up over half of the yields for many crop types including the wheat, paddy, corn,etc (Fig. 3.2).

The amount of air emissions were produced in short time via the open burning of straw, such as carbon monoxide (CO), CO_2, CH_4, SO_2, PM10, PM2.5, NH_3, and organic carbon (OC). The Eq. 3.1 had been widely used to calculate the emission of straw open-burning (MEPCa, 2015; Guan et al., 2017).

$$M_j = P_j \times \alpha \times r \times \theta$$

$$E_{i,j} = \sum_{i,j} (M_j \times EF_{i,j})$$

$$E_{i,straw} = \sum n_j \times E_{i,j} \qquad \text{Eq. (3.1)}$$

where, M_j present the activity levels of emission source (kg), such as the activity levels of crop type j ; P_j is the crop production (kg) which from the Chinese Statistical Yearbook; α is the ratio of straw to grain ; r is the ratio of straw open-burning with the value of 0.2 from MEPCa, 2015 ; θ refer to the comustion effiency with the value of 0.9 from MEPCa, 2015 ; $EF_{i,j}$ is the emission factor of i pollutant with open-burning j crop straw (g/kg); $E_{i,j}$ is the amount of i pollutants emission under open-burning straw of crop j (g) ; $E_{i,straw}$ is the average amount of i pollutants emission under open-bunring straw (g); n_j is the ratio of crop j in the total crop. The factor of straw open-burning was present in Table 3.1. According the proportion of crops in China from 2009 to 2019 as shown in Fig. 3.2, the straw open-burning in this chapter included paddy, wheat and corn. In 2019, the ratio of paddy, wheat and corn was 32%, 20% and 40%, respectively.

Table 3.1 The ratio of straw to gain and air emission factor of straw open-burning

Crop	the ratio of straw to grain	The emission factor of pollutant (g/kg)								
		SO_2	NOx	NH_3	CO	VOCs	PM10	PM2.5	CH_4	OC
Paddy	1.323	0.53	1.42	0.53	27.7	8.45	5.78	5.67	3.9	1.36
Wheat	1.718	0.85	3.31	0.37	59.6	7.48	7.73	7.58	3.4	2.5
Corn	1.269	0.44	4.3	0.68	53	10.4	11.95	11.71	4.4	3.06

3.3.2 Collection of data during the transportation process

According to the Technical Guidelines for Compiling Air Pollutant Emission Inventory of Road Motor Vehicles promulgated by Ministry of Environment Protection of the People's Repulic of China (MEPCb, 2015), The base emission factor (BEF) of vehicles ($E_{i,BEF}$) was determined which the vehicle was in average accumulated mileage, typical urban driving conditions (30km/h) under the weather condition (temperature 15°C, relative humidity 50%), and typical load factor for diesel vehicles is 50% based on the national investigation in 2014. Different type and motor vehicle emission standard of the vehicles had different BEF. For this dissertation, we assumed the vehicles for sewage sludge transportation was 10t (heavy goods vehicles) with 75% of load factor and level 4 of motor vehicle emission standards, which other condition was same as condition of BEF.

The average speed of road freight vehicles was 49 km/h in 2020 based on the invested data of Ministry of Transport of the People Republic of China. The actual fuel consumption distribution of ordinary trucks varies greatly, the highest was 37.4 L/100km, and the lowest was only 22.2 L/100km, mainly due to the difference in the truck's own weight and cargo weight. The average actural fuel consumption was 30 L/100km from the invested data. For that reason, the environmental emission of transprotation ($E_{i,transprotation}$) was caluculated as follow and the factor presented in Table 3.2:

$$E_{i,transprotation} = E_{i,BEF} \times \omega \times s \qquad \text{Eq. (3.2)}$$

where, ω referred to diesel vehicle load factor correction factor under 75% load factor (Table 3.3) ; s is the average speed correction factor under 40-80 km/h (Table 3.4).

Table 3.2 The base emission factor of vehicles with level 4 in China

	CO	CH	NOx	PM2.5	PM10
Heavy goods vehiles (g/km)	2.2	0.129	5.54	0.138	0.153

Table 3.3 Diesel vehicle load factor correction factor(ω)

	CO	CH	NOx	PM2.5, PM10
0	0.87	1.00	0.83	0.90
50%	1.00	1.00	1.00	1.00
60%	1.07	1.00	1.09	1.05
75%	1.16	1.00	1.21	1.13
100%	1.33	1.00	1.43	1.26

Table 3.4 Average speed correction factor (s) for diesel vehicles of level 4

	<20km/h	20-30km/h	30-40km/h	40-80km/h	>80km/h
CO	1.29	1.10	0.93	0.70	0.61
HC	1.38	1.12	0.91	0.64	0.48
Nox	1.39	1.12	0.91	0.60	0.28
PM2.5, PM10	1.36	1.12	0.91	0.65	0.48

3.3.3 Collection of data during the sludge treatment and recycling process

This dissertation was based on actual data from the sewage sludge treatment project. Thus, the process of the different scenarios was based on the environmental impact assessment (EIA) of each sludge disposal plant. In order to compare different technologies, the sludge composition of each scenario used was under the standard of CJ 3025-1993 in this dissertation.

The significant variations of data are attributed to WWTPs in China because they do not send proper reports on the treatment and final disposal of their WAS. In order to carry out the inventory, data were mainly collected from Environmental Impact Assessments (EIA) for projects using different sludge treatment technologies respectively and the Chinese Life Cycle Database (CLCD). The overall inputs and outputs to be measured by the study should be element flows. A large amount of basic data is required in the sludge treatment and recycling system. Among them, are inventory of transportation process, and energy and raw material consumption, and emissions from the sewage sludge treatment and recycling process. The life cycle inventory of energy and raw material of each scenario was presented in Table S1 and S2. (In Appendix)

3.4 Result and discussions

3.4.1 Total environmental impact of different scenarios

In this study, the environmental performance of sewge sludge recyclding system was evaluated and quantified using LCA by ReCipe 2008 (Goedkoop, et al., 2009; Xiao, et al., 2018). It was the preferred methodology due to the wide range of potential environmental effects it covers, such as climate change (CC, kg CO_2 eq), terrestrial acidification (TA, kg SO_2 eq), marine eutrophication (MEP, kg N eq), freshwater eutrophication (FEP, kg P eq), human toxicity (HT, kg 1,4-DB eq), terrestrial toxicity (TT, kg 1,4-DB eq), freshwater toxicity (FT, kg 1,4-DB eq), marine toxicity (MT, kg 1,4-DB eq), photochemical oxidant formant (POFP, kg NMVOC eq), particulate matter formation (PMFP, kg PM10 eq), water depletion (WDP, m^3), fossil fuel depletion (FDP, kg oil eq), and ozone depletion (ODP, kg CFC-11 eq). The total environmental performance ($EP_{(total)}$) represented the EP of each category indicated for the sewage sludge recycling system, calculated by Eq. (3.3).

$$EP_{(total)} = \sum EP_{(j)i} = \sum \left[Q_{(j)i} \times EF_{(j)i}\right] \qquad \text{Eq. (3.3)}$$

where, $\sum EP_{(j)i}$ is the environmental performance j influenced by inventory flow i, $Q_{(j)i}$ is the amount of inventory flow i, $EF_{(j)i}$ is environmental factor of environmental impact j related to inventory flow i. Eq. (3.4) shows how to convert from inventory units to an impact category's characterized unit (abbreviated char. unit) given a characterization factor (abbreviated char. factor) and how to convert from characterized flow to a normalization flow given reference value.

$$Characterized\ flow = flow\ (inventory\ unit) \times char.factor\ \left({}^{char.unit}\!/\!{}_{inventory\ unit}\right)$$

$$Normalization\ flow = {}^{char.flow\ (char.unit)}\!\Big/\!{}_{reference\ values\ (reference\ unit)}$$

Eq. (3.4)

Normalization values of the world is chosen as reference values. Since the reference values of environmental impact in China have not yet been established, the reference values in Table 3.5 use the reference values of the world recalculated in 2014 in the ReCiPe method. Among them, it lacks the reference value of water deletion, and the data from China Water Resources Bulletin in 2016 is adopted.

Table 3.5 The reference values of environmental impact categories in the ReCiPe method

Categories	Unit	Reference value
Climate change	kg CO_2 eq/p/yr	5.53E+03
Terrestrial acidification	kg SO_2 eq/p/yr	4.21E+01
Marine Eutrophication	kg N eq/p/yr	7.34E+00
Human toxicity	kg 1,4-DB eq/p/yr	1.45E+03
Terrestrial toxicity	kg 1,4-DB eq/p/yr	8.15E+00
Freshwater toxicity	kg 1,4-DB eq/p/yr	4.55E+00
Marine toxicity	kg 1,4-DB eq/p/yr	6.76E+02
Photochemical oxidant formation	kg NMVOC eq/p/yr	5.67E+01
Particulate matter formation	kg PM10 eq/p/yr	1.41E+01
Water depletion	m^3/p/yr	9.49E+04
Fossil fuel depletion	kg oil eq/p/yr	1.29E+03
Freshwater eutrophication	kg P eq/p/yr	2.90E-01
Metal depletion	kg Fe eq/p/yr	4.45E+02
Ozone depletion	kg CFC-11 eq/p/yr	3.76E-02

In Fig. 3.3, the highest EP_{total} except WDP was observed for scenario 5 ascribed to the largest consumption of electricity and auxiliary fuel during AD and drying before incineration. However, EP_{total} of scenario 2 was the least because the energy consumption, electricity, and fossil fuel, were less than those of other scenarios. Compared with scenario 4, scenario 5 and scenario 6 had incineration and composting steps after sewage sludge digesting, thereby consuming more energy and discharging more air pollutants. The consumption of fossil fuel was the main reason for EP_{total} in scenario 3, and CH_4 contributed indirectly to electricity consumption in all scenarios.

In order to further explore the environmental performance of sewage sludge recycling system, the results of this chapter compared with the previous studies in Table 2.1 in Chapter 2. Considering the difference of system boundary and function unit between this dissertation and previous study, the specific result of each environmental category was different. Aerobic composting and used in material (brick) were optimal scenario due to their environmental friendliness in this dissertation and AD was the optimal scenario in previous studies. The difference of aerobic coposting was the differnce during the treatment process. AD caused by different function unit is more suitable for sludge treatment with low moisture content.

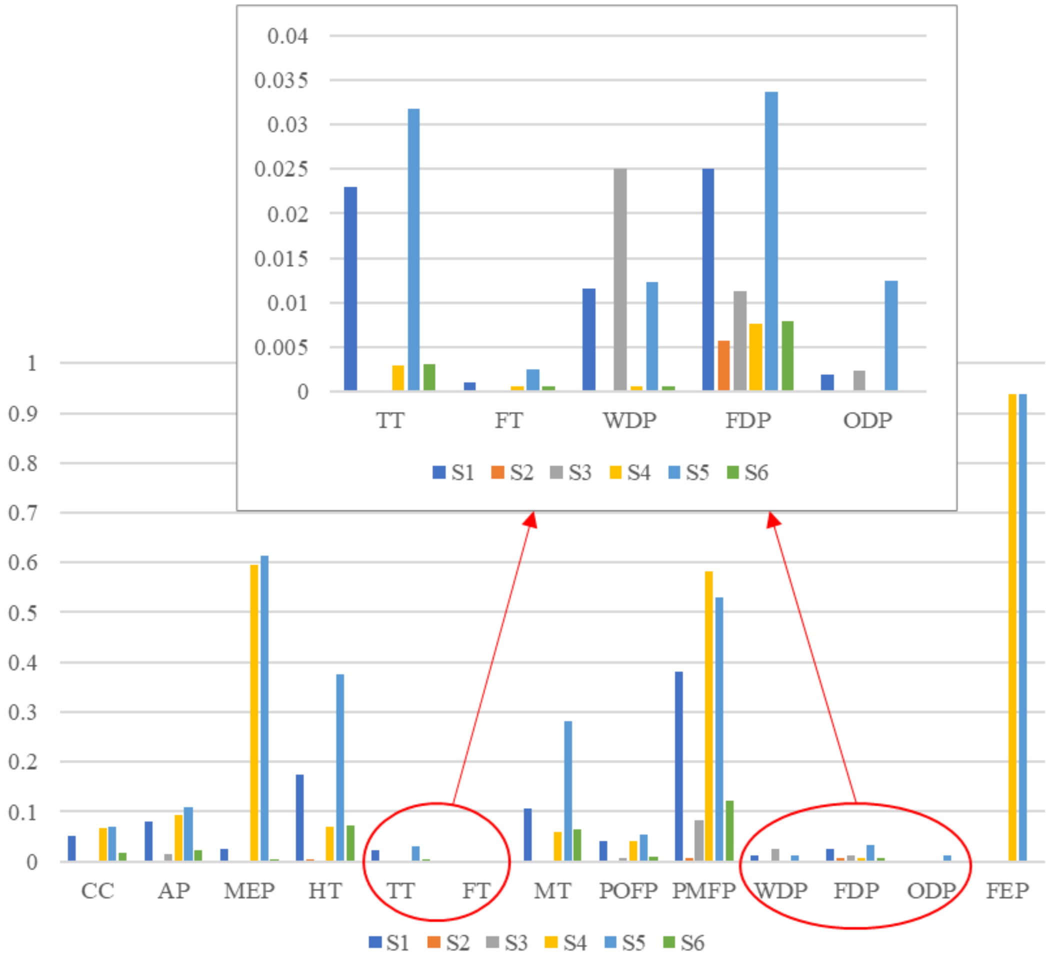

Fig. 3.3 Total Environmental impact of each scenario for sewage sludge recyling system (Note: CC: kg CO_2 eq, TA: kg SO_2 eq, MEP: kg N eq, FEP: kg P eq, HT, kg 1,4-DB eq, TT: kg 1,4-DB eq, FT: kg 1,4-DB eq, MT: kg 1,4-DB eq, POFP: kg NMVOC eq, PMFP: kg PM10 eq, WDP: m3, FDP: kg oil eq, ODP: kg CFC-11 eq)

3.4.2 Net environmental impact of different scenarios

The avoided environmental performance (EP_{avoid}) represented the EP from products replacing the same number of traditional products and calculated follow the equation present in Chapter 3.4.1. EP_{net} represented the “true environmental performance” of each scenario, calculated by Eq. (3.5) as follows:

$$EP_{net} = EP_{total} - EP_{avoid} \qquad \text{Eq. (3.5)}$$

Overall, EP_{net} of scenario 2 exhibited the lowest environmental impact in 10 environmental categories apart from TT, FT, and MT as shown in Fig. 3.4. In the environmental category of TT, FT, and MT, scenario 6 exhibited the lowest EP_{net} in six scenarios. EP_{avoid} in scenario 2 was the highest which was the reason of EP_{net} of S2 exhibited the lowest environmental impact, because the toxicity of heavy metals in producing fertilizer using chemical raw materials was more than that produced with the same nitrogen content of the sewage sludge recycling fertilizer. Scenarios 1 and 3 were the lowest in EP_{avoid}. For scenario 1, a large amount of energy consumption during the drying process led to a lower EP_{avoid}. EP_{avoid} of scenario 3, wherein sewage sludge replaced 10% clay, was lower than the others because energy consumption and pollutants were not decreased. In contrast, scenario 5 exhibited the highest EP_{net} except for CC, TT, WDP, and FDP. Scenarios 3, 4, and 5 exhibited the highest EP_{net} in WDP and FEP categories. CC, TT, and FDP categories for scenario 1 exhibited the highest EP_{net}. Therefore, scenario 2 was sustainable in environmental performance, and the sustainability of scenario 5 was questionable and needs further demonstration.

Fig. 3.4 Net environmental impact of each scenario for sewage sludge recyling system (Note: CC: kg CO_2 eq, TA: kg SO_2 eq, MEP: kg N eq, FEP: kg P eq, HT, kg 1,4-DB eq, TT: kg 1,4-DB eq, FT: kg 1,4-DB eq, MT: kg 1,4-DB eq, POFP: kg NMVOC eq, PMFP: kg PM10 eq, WDP: m^3, FDP: kg oil eq, ODP: kg CFC-11 eq)

According to the midpoint indicator which focuses on the single environmental problems, such as CC or AP, endpoint indicators present environmental impacts on three higher aggregation levels, included Human health, Ecosystems, and Resources. The net environmental impact of different scenarios in endpoint level was shown in Table 3.6.

Table 3.6 The net environmental impact of different scenarios in endpoint level

Endpoint category	Unit	S1	S2	S3	S4	S5	S6
Human health	DALY	1.84E-03	-1.04E-03	3.12E-04	3.19E-03	2.22E-03	2.88E-05
Ecosystems	species.yr	5.20E-06	-7.00E-07	4.23E-08	6.09E-06	4.58E-06	-1.25E-06
Resources	dollar	5.11E+00	-5.03E+00	2.39E+00	6.90E+00	4.50E+00	1.08E+00

3.4.3 The main pollutants of each environment categories in different scenarios

Based on the mechanism of Recipe method used in this dissertation, midpoint environmental impact divided in three higher aggregation environmental impact (i.e., Human health, Ecosystems, and Resources). The most contributed endpoint environmental impact of the scenario was human health, the pollutants of each scenario contributed were analysis.

The relative midpoint environmental impact to human health in scenario 1 were CC, HT, and PMFP. In terms of CC, CO_2 contributed impact of CC by about 97.51%. The main sources of human toxicity are heavy metals, hydrocarbons and dioxins, of which Cr (53.89%) and Hg (32.91%) are the main influences. Cr is mainly derived from energy consumption, and Hg is in the process of incineration of sludge. However, the mechanism of dioxin production is more complicated. For example, dioxin is easy to be produces when the incineration temperature is less than 800℃ in the process of burning domestic garbage. Among them, PMFP pollutants come from dust (81.39%) during energy consumption and incineration. Heavy metals, such as As, Cd, Pb, and Cr, reduced by alternative mineral fertilizer was the significant contribution to offset the human toxicity of environmental impact in scenario 2.

POFP and PMFP are major midpoint environmental impact categories of human health in scenario 3. According to analysis the release constituents, NOx (87.77%) and SO_2 (10.07%) are the principal source of POFP. Besides, Dust (87.24%) is the principal source of PMFP. HT and PMFP are most obvious midpoint environmental impact categories of human health in scenario 4, followed by CC and POFP. The main influence constituents of HT are As (87.94%). And the main reason for PMFP is PM10 (85.17%) in the process of energy production and consumption. The main influencing constituents of CC are CO_2 (94.27%) as same as scenario 1. TN (62.27%) and NH_3-N (35.98%) cause the impact of POFP during energy consumption and treatment process of scenario 4.

The major impact categories of scenario 5 that impacted human health were CC and HT. The contribution of pollutants to CC was present in Fig. 3.5. As, Hg, Pb, and V covered the 88%, 5%, 3%, and 4% of HT, respectively. In scenario 6, HT was the most positive environmental impact on human health. In the contrast, CC and PMFP were major negative impacts on human health. Dust (93%), SO_2 (9%), and NOx (6%) contributed to the impact of PMFP while NH_3 cleaned up the impact of PMFP by about 8%.

Paying attention to the significance of the comparison of greenhouse gas emissions during the sludge disposal process, which was the potential risk of sludge treatment and disposal (Fang et al., 2019). On the one hand, it could reduce other pollutants caused by energy utilization in the sludge disposal process, and on the other hand, it could promote people to develop new energy and clean energy. Therefore, climate change was an impact category that was specifically considered in research.

Fig. 3.5 presented the relative constituents of the impact category of climate change. From the characterization results of climate change, scenario 2 had a positive environmental impact. The reason for the positive environmental impact of scenario 2 was alternative mineral fertilizer to reduce the CO_2 and CH_4. For the other scenarios, the energy and resource recovery offset the impact of environemtal impact to some extent.

3.4.4 The potential for reduction in the environmental impact during different scenarios

Pollutants are mainly derived from energy consumption, and heavy metal and greenhouse emissions are high in scenario 1 and 5, because China still relies mainly on thermal power generation. Therefore, in the future production process, the use of clean energy should be gradually promoted to reduce the proportion of thermal power generation. At the same time, the incineration process should pay attention to the collection and treatment of dust in scenario 1.

For scenario 3, the sludge treatment and recycling process should improve the production technology to reduce water use and carry out clean production to reduce the environmental impact produced during the treatment process.

In view of the main pollutions of Scenario 4 and 6, heavy metals and dust are generated in energy production and pollutants of N and P in the process of anaerobic digestion. According to the alternative fertilizer of digestion sludge, NH_3 was offset by about 8% of CC. Therefore, the utilization of clean energy should be promoted, and the technology should be improved to reduce the pollutant emissions of N and P in the future.

3.4.5 Sensitivity analysis

Sensitivity analysis is significant to check the system robustness for LCA and to display the key contributors focusing on improvements for each scenario. In this study, according to varying (10%) environmental factors, that is, energy combustion and productivity of final product, the local sensitivity analysis on LCA were assessed using a one-at-a-time approach, as shown in Eq. (3.6). Moreover, sensitivity ratios of LCA were obtained (Liu et al., 2021; Edwards et al., 2018).

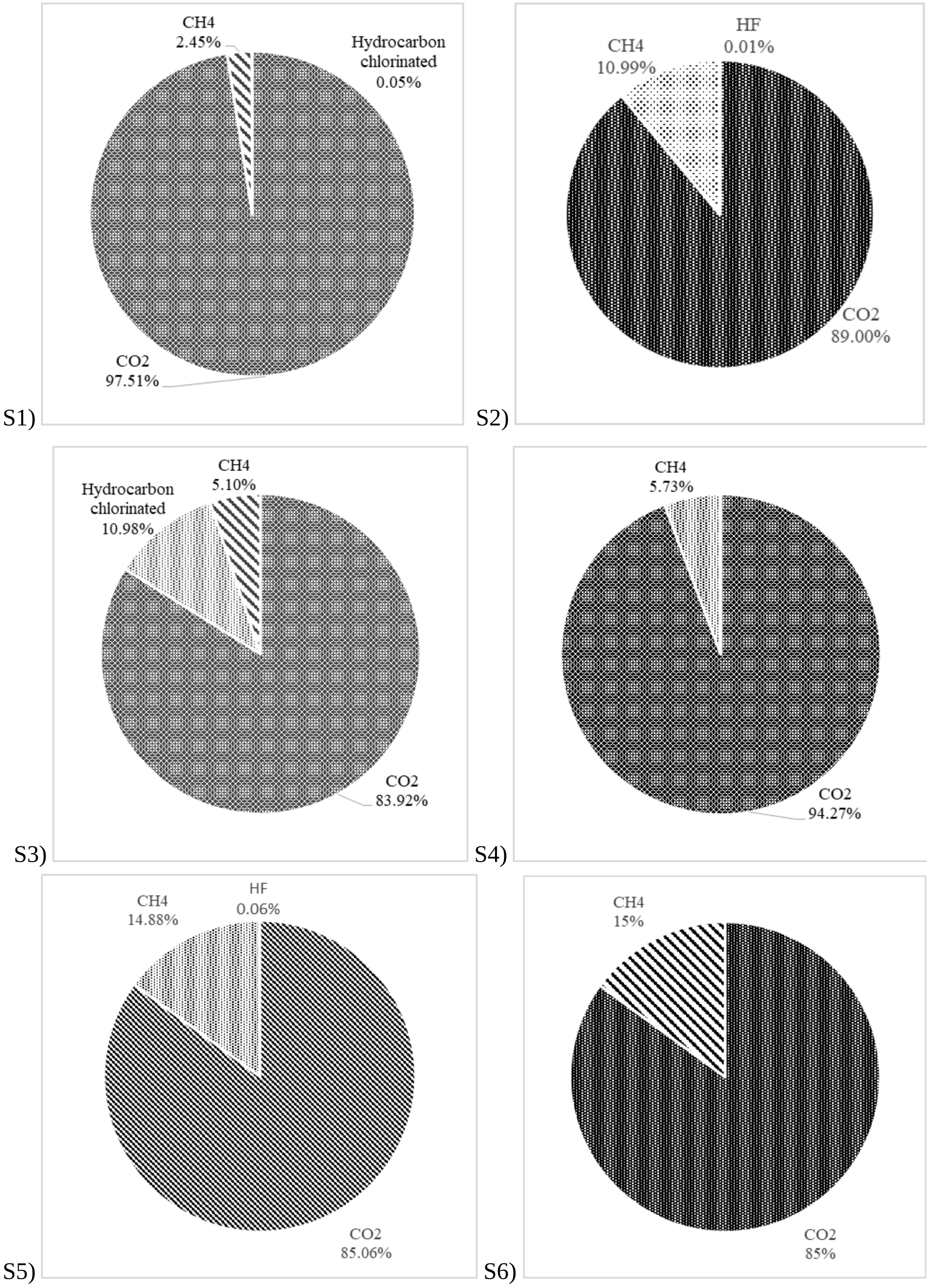

Fig. 3.5 The relative constituents about the impact category of climate change

$$Sensitivity\ analysis = \frac{\Delta result / result}{\Delta factor / factor} \qquad \text{Eq. (3.6)}$$

The power consumption was more sensitive to the impact of environmental benefits for scenario 4, particularly PMFP, whereas the environmental benefits of scenario 2 are more sensitive to production efficiency, particularly CC (Fig. 3.6). Scenario 3 was the least sensitive to power consumption and production efficiency. The environmental indicators of toxicity (HT, TT, FT, and MT) in each scheme were more susceptible to power consumption and production benefits than other environmental indicators.

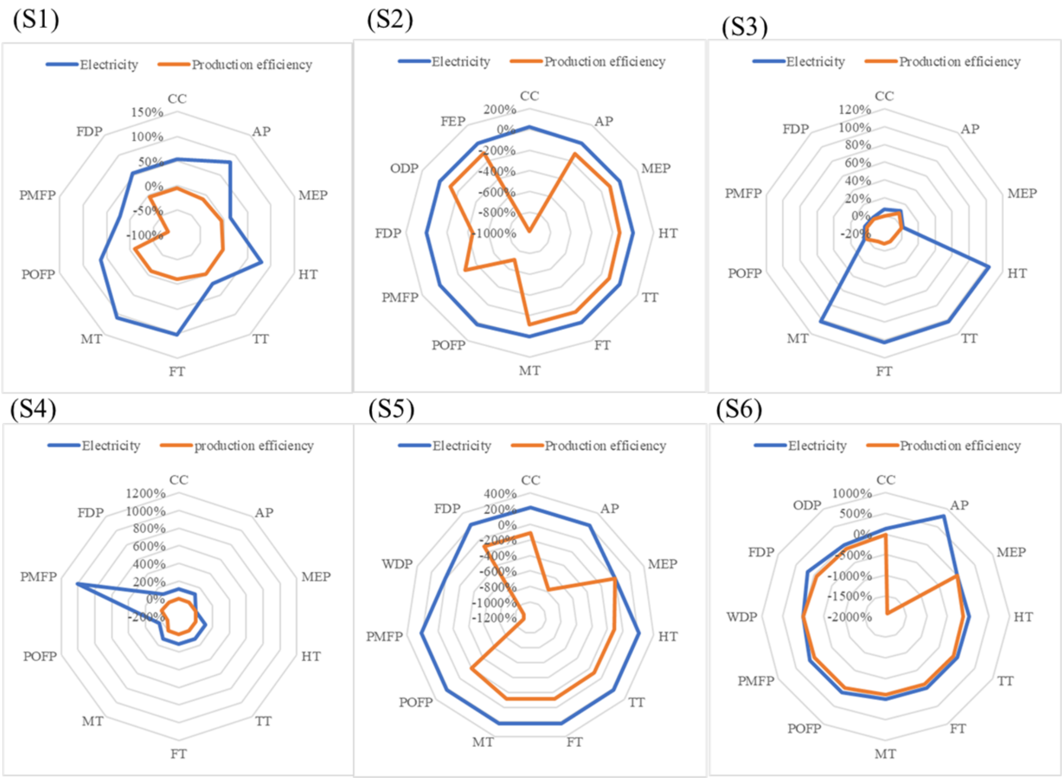

Fig. 3.6 Sensitivity analysis of each scenario in environmental performance

3.5. Summary

This Chapter analyzed which scenarios of sludge treatment and recycling system are more sustainable during the period of operation from an environmental perspective via case study. Human health is the primary impact category of overall environmental impact. Scenario 2 (Aerobic composting) is the most environmentally friendly scenario to fewer emissions and less consumption of energy. Scenario 3 (Used in material) is the optimal scenario based on less environmental impact.

According to the endpoint environmental impact results, human health is the primary endpoint environmental impact category for the scenarios. In the midpoint characterization results of human health, S1 has significantly higher environmental impacts on CC and HT than other scenarios and S3 has obvious environmental impacts on POFP and PMFP.

In Scenario 1, pollutants are mainly derived from energy consumption, and heavy metal and dust emissions are high because China still relies mainly on thermal power generation. Therefore, in the future production process, the use of clean energy should be gradually promoted to reduce the proportion of thermal power generation. At the same time, the incineration process should pay attention to the collection and treatment of dust in scenario 1.

Combined with the environmental assessment results of the four schemes, in addition to S2 is that the scenario has least environmental impact. The reason for the least environmental impact of S2 is that sewage sludge considered as waste is free of any environmental burdens when entering sludge-based fertilizer production in scenario 2. Therefore, the production of sludge-based fertilizer instead of mineral fertilizer should be promoted in the fertilizer industry.

In view of the above analysis, scenario 3 is the optimal solution among the four scenarios because of its relatively small environmental impact. Comparing the impact of human health, PMFP, POFP, and ODP are major impact categories. And the treatment process is the main source of these environmental impact. It should carry out clean production to reduce the dust and nitrogen oxides during the treatment process.

Scenario 4 should focus on CC, HT, POFP, and PMFP which cause damage to human health. MEP and FEP are also obvious impact categories of scenario 4. In view of the above analysis, the main pollution of scenario 4 comes from heavy metals, non-methane volatile organic compounds (NMVOC) and dust in energy production and pollutants of N and P in the process of anaerobic digestion. Therefore, the utilization of clean energy should be promoted, and the technology should be improved to reduce the entire environmental impact, especially pollutants emissions of N and P.

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4 LIFE CYCLE COST ANALYSIS OF SEWAGE SLUDGE RECYCLING SYSTEM

4.1 System boundary

The system boundary was same as described in the Chapter 3.1. The analysis in this chapter was used as the unit of 1t sewage sludge with 80% moisture content. The analyzed process included transportation, drying, pretreatment, production processing and air pollution control treatment. The transportation distance between each technology was assumed to be 100 km. Operation costs in this study included transportation, raw material, energy consumption, tax and labor costs. The production rate of energy and by-products recovery was the same as in Chapter 3. The sale incomes represent all incomes from final product sales. Energy and by-products recovered in the sewage sludge recycling system were sold completely, regardless of market demand.

The Life Cycle Cost (LCC) was estimated included the capital cost, fix-operating cost, variable operating cost, transport cost and revenue from the sales of recovered products (Nave et al., 2019). In this study, the LCC of sewage sludge co-processing with agricultural wastes recycling system referred to capital cost, transportation cost, operation cost included energy consumption, chemical and raw material consumption and labor capital, revenue from the by-product, tax and subsidy (Tarpani et al., 2018). WLC included LCC and externality cost to merge the comparative performance and presented “true cost” of each scenario was applied according to BSI ISO 15,686–5. The externality cost of each scenario was estimated by Ecotax 2002 method in Sweden (Martinez-Sanchez et al., 2017; Wu et al., 2005). The goal of this chapter in the dissertation was comparative the economic performance of different sewage sludge recycling systems including the external cost that are caused by the emissions.

4.2 Data Source of Life cycle cost of sewage sludge co-processing with agricultural wastes recycling system

Economic data on budget costs were from market investigations and EIA reports of each enterprise. Final product and raw material prices were obtained from the public domain reflecting their market price in 2019 (NDRC, 2019). Sewage sludge disposal subsidy lacks a standard for each city in China; hence, this study referred to the subsidy in Chongqing (205 CNY/t). The discount rate of capital cost and externality cost were 10% and 4%, respectively (Huysegoms et al, 2018).

The other costs involved enterprise income tax, potential environmental protection tax, and externality cost (Table 4.1). The environmental protection tax and externality costs were based on the Environmental Protection Law in China and China Renewable Energy Outlook (CREO) 2017, respectively (EPTL, 2018). For missing data of externality cost, the reference of monetization value of per unit total emissions was from a previous study (Edwards et al., 2018). An exchange rate of 1 AUD = 4.6 CNY was used, and all financial factors in Table 4.1 were transformed to

CNY.

Table 4.1 The accounting price for externality cost of this dissertation

Pollutant	Cost (CNY/kg)	Pollutant	Cost (CNY/kg)
To air			
SO_2	27.2	NH_3	176.7
NOx	35.8	CO_2	0.3
CO	60.1	CH_4	7.7
PM10	36.4	VOC	5.2
Cd	796.3	Dioxins	3,812,009,315
As	1638.9	Pb	3175.0
Ni	76.5	Cr	4094.6
Hg	606,222.5		
To water			
As	277.4	Pb	709.0
Cd	7449.3	Cr	467.5
Ni	8.4	Hg	19,984.8
Zn	8.4	Ammonium	21.8
Nitrite	8.5	Phosphate	21.1

(Data Source: Edwards et al., 2018; Liu et al., 2021)

4.3 Policy analysis of sewage sludge co-processing with agricultural wastes recycling system in economy

In order to promote the development of sewage sludge recycling in China, the relevant policy supports this industry such as subsidy, enterprise income tax and environmental protection tax. Enterprise income tax is a kind of income tax levied on the production and operation income and other income of enterprises. It was stipulate as 20% of income in new version in 2017. Enterprise income tax of each technology was exempted in the first three years and levied at 12.5% in the second three year for the enterprise of environment protection (EITL, 2000). Environment protection tax illustrates the guideline and set the tax on air emission, water emission and solid waste, and noise, which was published in 2018.

Potential changes in relevant government policies would influence the NPV of each scenario via the tax part of cash inflow. Therefore, we examined the NPV of each scenario in the following cases:

(1) Baseline: WAS disposal subsidy was provided, a partial exemption of enterprise income tax,

and exemption of environmental protection tax.

(2) Case 1: A 50% reduction in environmental protection tax.

(3) Case 2: No exemption of enterprise income tax

(4) Case 3: No sewage sludge disposal subsidy.

(5) Case 4: Sewage sludge disposal subsidy was provided, and enterprise income tax and environmental protection tax were exempted.

4.4 Result and discussions

4.4.1 External cost of sewage sludge recycling system

The total externality cost of each scenario was classified into four groups, such as water quality, human health, climate change, and indeterminate (Edwards et al., 2018; Martinez-Sanchez et al., 2017). The monetized value of total nitrite, total phosphate, NH_3-N, and heavy metal emitted to water contributed to the externality cost of water quality. The externality cost of climate change included the monetized value of CO_2 and CH_4. The monetized value of heavy metals and other air emissions, including NOx, dioxins, PM10, VOC, CO, SO_2, and NH_3, made up the human health contribution to externality cost. The externality cost was calculated as Eq. (4.1).

$$Externality\ cost_j = \sum e_{j,k} \cdot P_k \qquad \text{Eq. (4.1)}$$

Where, $e_{j,k}$ was the amount of emission k in scenario j in Table 3.5; P_k was the per-unit price of emission k in the scenario in Table 4.1.

The most influential category of externality cost in all scenarios was human health, as shown in Fig. 4.1. Major contaminants that are significant contributors to externality costs were identified for each scenario, including climate change, water quality, human health and unclassified. For human health, SO_2 and NOx were the major contributors, and ammonium polluted into water was the biggest contributor for water quality. Approximately 95% of the external cost of human health in scenario 4 originated from SO_2, NOx, and Hg. The main pollutants affecting human health in scenarios 1, 5, and 6 were the same as in scenario 4. In addition to SO_2 and NOx in scenarios 2 and 3, NH_3 was the main pollutant. The external costs of scenario 5 were twice that of other scenarios presented in Fig 3, and the external costs of scenarios 2 and 3 were significantly lower than the other scenarios

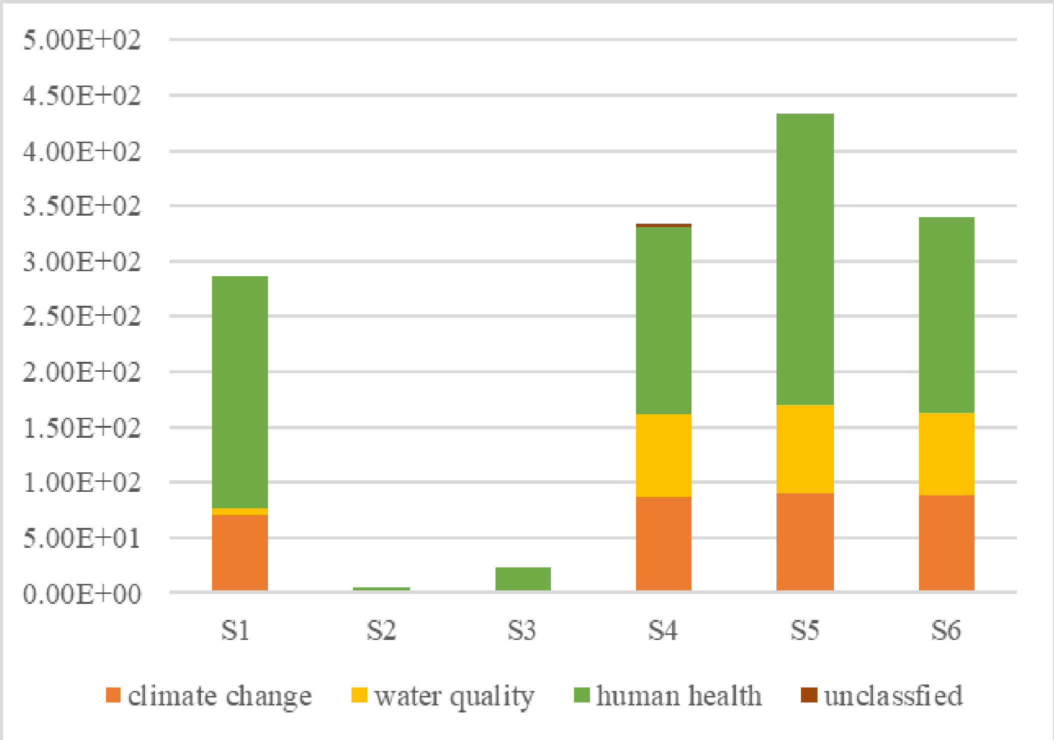


Fig. 4.1 Contribution of environmental emissions to externality costs (unit: yuan)

4.4.2 Life cycle cost of sewage sludge recycling system

In terms of the LCC of each sewage sludge recycling system, LCC was undertaken and indicated by net present value (NPV) as Eq. (4.2). The CI_i was the sum of cash inflow in the year i, such as the sewage sludge disposal subsidy and final product incomes of each system. The CO_i was the sum of cash outflow in the year i, such as capital cost, operational cost, and taxes. If the NPV of the fifth year is greater than zero, the sewage sludge recycling system presented commercial feasibility with higher NPV in view of market experiences in China (Liu, et al., 2021). Break-even year represented the first year, wherein cash inflow is greater than cash outflow (NPV>0).

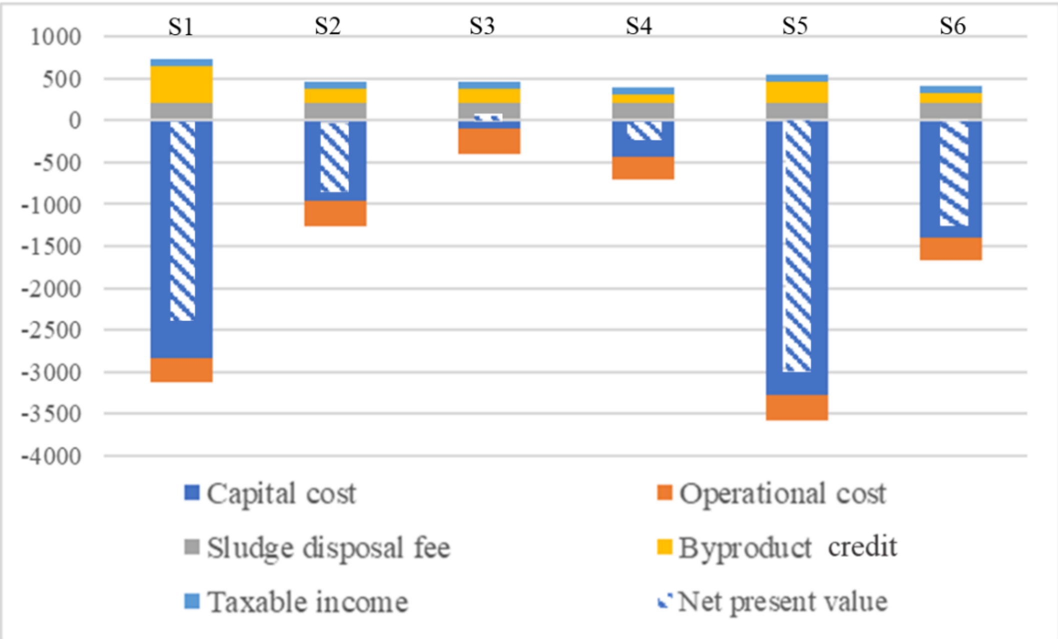
$$NPV_i = \sum_{i=0}^{n}(C_{Ii} - C_{Oi})(1 + 10\%)^{-i} \qquad \text{Eq. (4.2)}$$

As presented in Table 4.2, the break-even years of scenarios 3 and 4 presented favorable market expectations in the third and fifth years, respectively. Scenarios 1, 2, and 6 individually became NPV-positive in 10, 8, and 14 years, respectively.

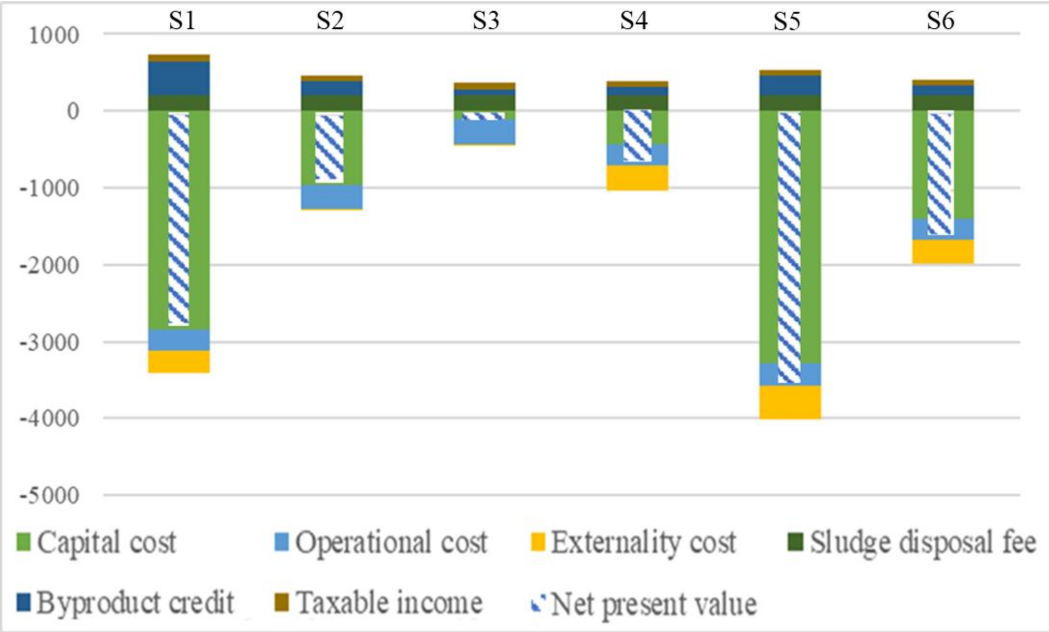
Table 4.2 Break-even year of policy impact of six scenarios for LCC (unit: year)

	S1	S2	S3	S4	S5	S6
Baseline	10	8	3	5	-	14
Case 1	12	8	4	>30	-	>30
Case 2	-	-	-	-	-	12
Case 3	16	>30	-	-	-	11
Case 4	8	7	3	4	2	4

The economic performance of scenario 3 was the best due to the lowest investment cost compared with other scenarios, with the lowest sales income. Although sales income of biogas and electricity in scenario 5 were the highest, the break-even year was over 30 years owing to the highest investment cost with a new plant. Hence, the break-even year of scenario 5 was reduced with the existing incineration plant to recycle electricity, which has not been studied so far. Expect for scenario 3, the capital investment cost covered over 40% in NPV for other scenarios.



(a)



(b)

Fig. 4.2 NPV_1 of six WASR scenarios by LCC and WLC method

(a) LCC; (b) WLC (unit: yuan)

The key operational cost, which was transportation cost, was further analyzed to provide suggestions for improvement, as shown in Fig. 4.2. Result demonstrate that transportation cost covered over 56% in operation costs. Therefore, the location of reutilization and transportation path optimization can be rationally mapped out in future sewage sludge recycling system management. For scenarios 2 and 3, the water cost was the major contributor apart from transportation cost, wherein the increasing moisture rate of sewage sludge when leaving wastewater treatment plants must be considered. Transportation and water costs should be balanced in future management because the volume was relative to the moisture rate of sewage sludge, and transportation cost increased with the volume of sewage sludge. For other scenarios, the energy costs, such as electricity, natural gas, and coal, were major contributors in addition to transportation costs, and their economic performances can gain profit from improving energy efficiency. According to the WLC results (Fig. 4.2), external cost which integrated the environmental impact via pollutants monetized did not obviously affect on the net present value in the first year of each system compared with LCC, and only scenario 3 had a sustainable development considering the break-even year.

4.4.3 Policy impact of sewage sludge co-processing with agricultural wastes recycling system in economy

All scenarios could achieve a positive break-even in 10 years, as presented in Table 4.2, with the full support of the government in developing sewage sludge recycling system via granting subsidy and tax exemption. Meanwhile, scenario 5 exhibited the highest commercial feasibility, and other scenarios except scenarios 1 and 2 had market potentials in China. The five scenarios would not survive with enterprise income tax without exemption, except scenario 2. It might not reduce 50% of environmental protection tax for scenarios 1, 2, and 3, and other scenarios will be difficult to break even in 30 years. If the government does not provide subsidies, most scenarios will not survive. By accounting, the subsidy of sewage sludge recycling system will be 268.1, 134.5, 69.8, 63.9, 660.2, and 267.2 in CNY from scenarios 1 to 6, when the NPV of the fifth year was assumed to be zero. The result will provide a feasible reference to policymakers for sewage sludge recycling management of different scenarios. Nevertheless, current government policy changes have a universal and evident impact on sewage sludge recycling system management, and scenarios 2 and 3 are more adaptable.

4.4.4 Sensitivity analysis

The sensitivity analysis of life cycle cost was calculated via varying the economic factor, such as operation cost, sales income and externality cost, which was similar as describing in Chapter 3.4.5. Table 4.3 shows the sensitivity to the main parameters related to the NPV of LCC and WLC. Product revenue and disposal fees were the main sensitive parameters of the sewage sludge recycling system.

Table 4.3 Sensitivity analysis of each scenario in economic performance

		Trans-portation	Raw material	Energy consumption	Labor cost	Externality cost	Sewage sludge disposal fee	Sales income
S1	LCC	9.0%	0.6%	3.6%	0.4%	-	10.8%	25.8%
	WLC	8.4%	0.5%	3.3%	0.3%	8.3%	17.9%	20.9%
S2	LCC	26.4%	-	3.7%	2.7%	-	30.8%	26.1%
	WLC	26.2%	-	3.6%	2.7%	0.8%	30.6%	25.9%
S3	LCC	271.9%	58.9%	69.2%	120.5%	-	726.8%	268.0%
	WLC	206.9%	42.2%	49.7%	87.8%	31.5%	722.5%	171.2%
S4	LCC	66.3%	8.6%	16.3%	5.6%	-	84.2%	41.3%
	WLC	41.3%	5.3%	10.0%	3.4%	41.4%	49.7%	24.8%
S5	LCC	105.8%	14.2%	47.4%	9.3%	-	147.0%	187.8%
	WLC	31.9%	4.0%	13.7%	2.6%	73.8%	37.7%	47.0%
S6	LCC	70.3%	9.2%	17.3%	6.0%	-	90.0%	51.6%
	WLC	57.7%	7.4%	14.1%	4.9%	20.4%	71.9%	119.1%

4.5 Summary

According to the results of LCC, it proves existing the economic benefit of sewage sludge recycling system for investors, which consider the external cost caused by environmental impact. It will attract the funds from the investors and lighten the financial stress for local government in environmental protection, which make the sewage sludge recycling more sustainable.

Based on the results, scenarios 3 was the preferred system for sewage sludge recycling with market potential. However, according to the national standard, considering the accumulation of heavy metals in sewage sludge in agricultural products, the fertilizers produced in scenario 2 are currently used for agricultural quantity and time constraints. We suggest that heavy metals in sewage sludge should be controlled in the future, and municipal sewage and industrial sewage should be treated separately. Meanwhile, product quality needs more control, because the heavy metals in sewage sludge are transferred to the product. As of 2019, 27% of the sludge is treated by incineration. If scenario 3 completely replaces scenario 1, the total cost will be reduced by 75%. The economic performance of scenarios 1, 5, and 6 do not support its industrialization potential. We suggest that we consider co-processing the existing incineration plant and fertilizer plant rather than constructing a new plant to reduce the initial capital investment. For scenario 4, we recommend extending the industrial chain and using digested sludge to continue producing higher value-added and more environment-friendly main products, such as those from scenarios 5 and 6 (Liu et al., 2021). In contrast, transportation cost is the main factor influencing the operating cost of all scenarios, and policy analysis has raised concerns that sewage sludge recycling companies rely heavily on government support.

There are uncertainties and limitations in this chapter. Therefore, future research should focus on evaluating the sewage sludge recycling system combined with other factors (different transportation radius, different regional characteristics, different composition of sewage sludge, and market demand for sewage sludge products) to achieve overall economic feasibility and flexibility of sewage sludge recycling system.

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5 THE EFFECT OF IMPLEMENTATION SCALE OF SEWAGE SLUDGE RECYCLING SYSTEM SCENARIOS IN ENVIRONMENT AND ECONOMY

5.1 System boundary and data source

Currently, aerobic composting, anaerobic digestion and biomass utilization, incineration and heat utilization, and using for building materials are four common scenarios of sludge treatment and recycling technologies in China. The overall scope of this study comprising dewatering, treatment and recycling process involved in each system shown in Fig. 5.1.

Owing to the situation of sludge treatment/disposal plants lacking in China (Wei et al., 2020), we assumed that the effect of the implementation of scale in the sewage sludge recycling system is similar in different countries, and the operating situation of different countries only affects the size of scale effect (Xu et al., 2014). Based on the data from Japan Sewage Works Association, we aggregated the situation of sewage sludge recycling including the scale of facilities and the consumption of energy and chemicals. We collated data of all technologies in each system, and GHG reduction through technology improvement was few than by-product offset (Wang et al., 2021). Since it has been reported in literature that three types of anaerobic digestion, including mesophilic, thermophilic, and temperature-phased anaerobic digestion, had slight differences based on the results of LCA (Lanko et al., 2020). Liu et al. (2022) reported the endpoint environmental impact and economic cost had slight difference of four types in aerobic composting. In terms of environmental impact, fluid bed incinerators were less dam-aging than multiple hearth incinerators (Alyaseri et al., 2017). Based on the data of incineration of SRTS project (165), there were 123 projects used the fluid bed incinerator, which occupied 74.5%. Accordingly, the effects of the different technologies in each scenario were ignored in this study.

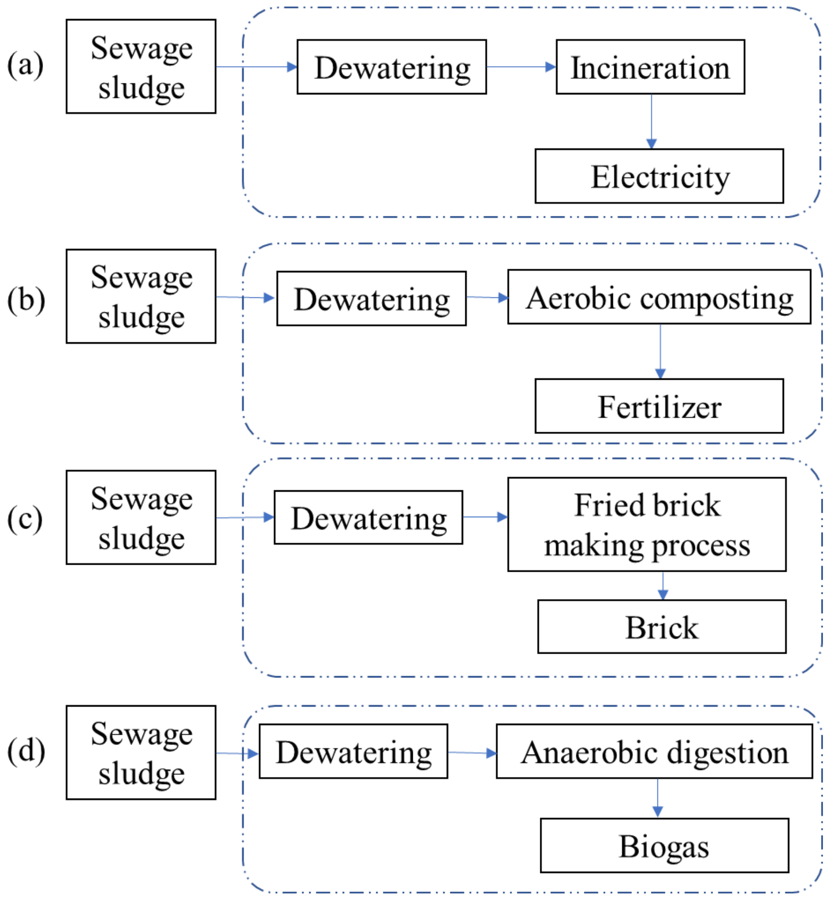


Fig. 5.1 The boundary of target sewage sludge recycling system.
(a) Incineration; (b) Aerobic composting; (c) Used in material (brick); (d) Anaerobic digestion

It is commonly used for parameter estimation in functional relationships to use the ordinary least squares method (OLS). For example, it was considered in the study of Mydland et al. (2018) that investigated economics of scale in Norwegian electricity distribution company. Referring to Fragkias et al. (2013), they examine the relationship between the city size and CO_2 emission for U.S. metropolitan areas with OLS. This study used OLS to analyze the relationship between the implementation scale and the environmental emission and economic cost of the sewage sludge recycling system (STRS) to support decision-making. The optimal system and break-even scale of each system were obtained with cost-benefit analysis under considering the effect of the implementation of scale. Following our emphasis on scale effects, we hypothesized that environmental emission and economic cost are closely related to the implementation of scale and that their relationship were measured according to a power function.

$$Y = ax^{\beta} \qquad \text{Eq. (5.1)}$$

where Y measures environmental emission and economic cost, a is a constant, x denotes the implementation of scale, β is the scaling exponent. This function acts as a baseline model to determine whether environmental emissions or economic costs were modeled with a power function relationship.

The main assumption of this Chapter included the following four points.

(1) The GHG emissions of the construction phase were not examined as they did not exceed 5% of the total impact (Liu et al., 2013).

(2) Energy and by-products recovered in the SRTS were sold completely, regardless of market demand as same as Chapter 4.

(3) The energy consumed by SRTS during the dewatering and treatment processes was derived from fossil fuels.

(4) The nitrogen content in the fertilizer generated by sewage sludge was 8%, which compared with the conventional fertilizer as same as Chapter 3 and 4.

5.2 The scale effect of recycling system scenarios in environment

5.2.1 Environmental emission of sewage sludge recycling system

In this study, we focused on greenhouse gas (GHG) emissions to present the effect of the implementation of scale and environmental emissions in each system. Owing to the lack of condition of sludge treatment and disposal plants in China, we assumed that the effect of the implementation scale in the SRTS is similar in different countries, and the operating situation of different countries only affects the size of scale effect. We aggregated the situation of sewage sludge recycling based on data from the Japan Sewage Works Association (JSWA), including the scale of facilities and consumption of energy and chemicals. Data of GHG emission factor for treatment, energy and chemical was from the Research Institute of Economy, Trade and Industry, Ministry of the Environment of Japan (MINE) and China products carbon footprint factors database. The quantity of data relevant to our subsequent validations was limited due to the lack of public data on SRTS. Considering that the chemicals used in the treatment and recycling processes are far fewer than those used in the dewatering process, only chemicals for the dewatering process were calculated. In this study, it is based on JSWA data, similar to the inventories of previous studies (Hong et al., 2009; Liu et al., 2013). The GHG emissions calculated in this study, as described in Eq. (5.2), included the chemical consumption in the dewatering process, the energy consumption of the system, and the discharge of sludge after treatment.

$$GHG = GHG_{treatment} + GHG_{energy} + GHG_{chemicals} \qquad \text{Eq. (5.2)}$$

$$= EF_{m,treatment} \times Q_{treatment} + EF_{i,energy} \times Q_{energy} + EF_{n,chemicals} \times Q_{chemicals}$$

Where, $EF_{m,treatment}$, $EF_{i,energy}$, $EF_{n,chemicals}$ were the emission factor (EF) m of the discharge of sludge after treatment, EF of the i-th kind of energy, and EF of the n-th kind of chemicals, respectively. $Q_{treatment}$, Q_{energy}, $Q_{chemicals}$ presents the amount of dry solid (DS) of sewage sludge treatment, the amount of the i-th kind of energy, and the amount of the n-th kind of chemicals. The GHG emission factor calculated was presented in Table 1.

Table 5.1 Greenhouse gas emission factor calculated in STRS

Parameter	Unit	value	Parameter	Unit	value
Energy [a]					
Heavy oil	tCO_{2eq}/kL	2.71	LPG	tCO_{2eq}/kL	3
Coal oil	tCO_{2eq}/kL	2.49	Disel	tCO_{2eq}/kL	2.58
gasoline	tCO_{2eq}/kL	2.32	Coal	tCO_{2eq}/t	2.33
Electricity	tCO_{2eq}/kwh	0.000433	Natural gas	$tCO_{2eq}/10^3Nm^3$	2.62
Chemicals [a, b]					
Ferrous chloride	tCO_{2eq}/t	0.32	Poly-ferrous sulfate	tCO_{2eq}/t	0.0308
$Ca(OH)_2$	tCO_{2eq}/t	0.45	CaO	tCO_{2eq}/t	0.75
PAM	tCO_{2eq}/t	6.5	Poly-aluminum chloride	tCO_{2eq}/t	0.41
H_2O_2	tCO_{2eq}/t	0.39			
Sludge [a]					
Incineration	tCH_4/wet-t	0.0000097	Composting	tCH_4/wet-t	0.004
	tN_2O/wet-t	0.0006042		tN_2O/wet-t	0.0003
Production [c]					
Electricity	$kgCO_{2eq}/kwh$	0.53	Nitrogen Fertilizer	tCO_{2eq}/t	10.63
Clay Brick	tCO_{2eq}/t	0.2	Biogas	$kgCO_{2eq}/t$	9.35

a. MINE (2016); b. Kainou (2010); c. CAEP (2022).

The National Development and Reform Commission issued the National Carbon Emissions Trading Market Construction Plan which the electricity sector was the first target and other sectors to be covered in the future and stated China's unified carbon emission trading market was officially established. The most widely used allocation models for carbon emission allowances are the compensation and gratis allocation models. At present, the common compensation method is the auction of carbon emissions, which is simple and efficient. GHG emissions can be avoided if by-production replaces energy or substitution (Chang et al., 2020). Avoided GHG emissions ($GHG_{avoided}$) in large implementation of scale were lower than generated emissions of production ($GHG_{production}$) by original process in every case, as shown in Eq. (5.3).

$$GHG_{avoided} = GHG_{k,production} - GHG \quad \text{Eq. (5.3)}$$

5.2.2 The scale effect of GHG emission during different scenarios

(1) The scale effect of GHG emission

Following the scaling effect in each system, the unit GHG emission is related to the implementation of scale, and it decrease rapidly and then gradually stabilize as the implementation of scale increase as presented in Fig. 5.2. When the implementation of scale is more than 209,178 t-DS, the unit GHG emission of incineration stabilized at 0.14 tCO_2/t-DS. The unit GHG emission of aerobic composting remained steady at about 0.18 tCO_2/t-DS when the implementation of scale increased over 140 kt. For the system of used in material (e.g., brick), the unit GHG emission fell to a low point around 0.01 tCO_2/t-DS over 7676 t-DS of scale. More than 40 kt of scale, the unit GHG emission reached about 0.08 tCO_2/t-DS. The energy consumption in the system was the main reason to determine unit GHG emission. of scale increase as presented in Fig. 5.2.

(2) The scale effect of GHG emission with carbon emission quota

The avoided GHG emission was calculated after introducing the carbon emission quota (CEQ) to the system. It can be seen from Fig. 5.3 that if the unit avoided GHG emission has a negative value, the unit GHG emission of system more than the GHG emission of production generated by original process (Wang et al., 2021). In contrast, if the avoided GHG emission has a positive value, the system offsets part of the carbon burden of production generated. The minimization scale of balance the GHG emission was calculated caused by the scaling effect in environmental emissions. It is apparent from Fig. 5.3 that the minimization scale of incineration, aerobic composting, used in building material, and anaerobic digestion were 31,946, 19, 33, and 82 t-DS/y, respectively. The most surprising aspect of the data is in the minimization scale of incineration is well large than other systems. The increase of renewable electricity generation and the expansion of cross-regional grid construction was the reason decrease in the GHG emission intensity of electricity generation (Peng et al., 2021).

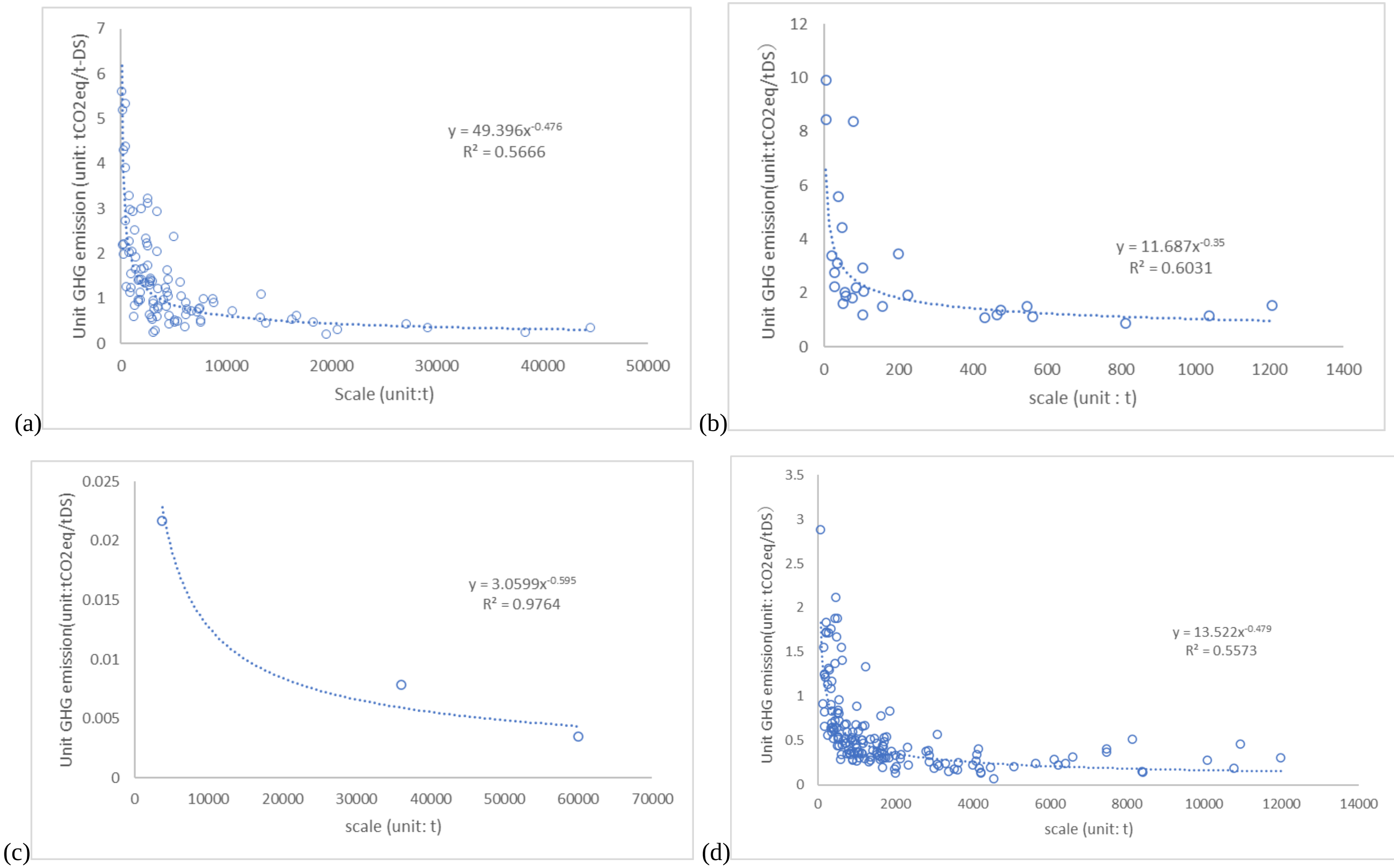


Fig. 5.2 The relationship of unit GHG emission of different system and implementation of scale.
(a) Incineration; (b) Aerobic composting; (c) Used in material (brick); (d) Anaerobic digestion

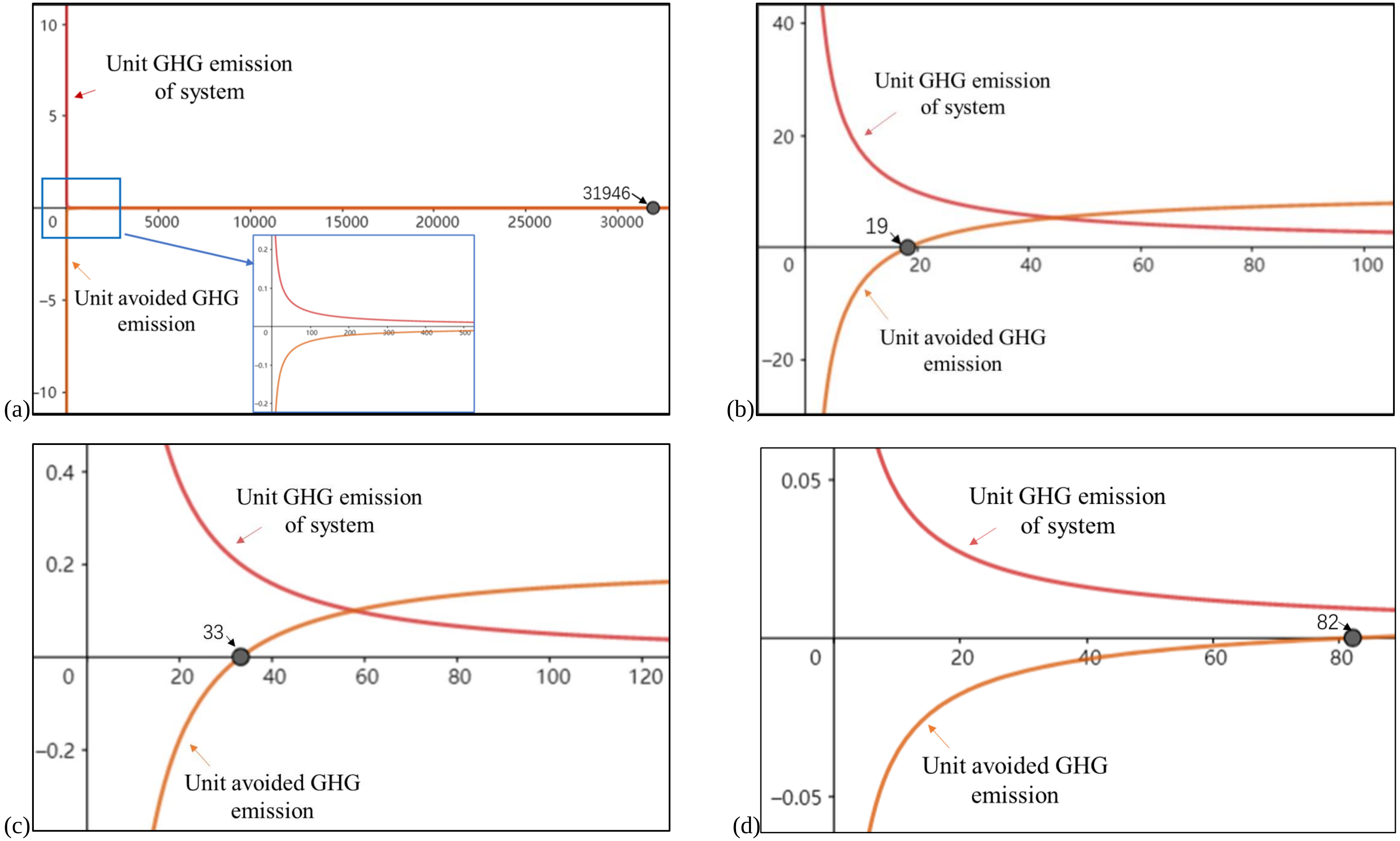


Fig. 5.3 The unit avoided GHG emission of different system.

(a) Incineration; (b) Aerobic composting; (c) Used in material (brick); (d) Anaerobic digestion

5.3 The scale of effect of recycling system scenarios in economy

5.3.1 Economic analysis of sewage sludge recycling system

The cost accounting of system divided into four parts: initial cost, operation cost and by-production profit. The total cost per unit DS of each system included initial cost of unit DS, operation cost of unit DS and by-production profit of unit DS, as presented in Eq. (5.4).

$$Cost_{total} = Cost_{initial} + Cost_{operation} + Cost_{by-production}$$

$$Cost_{operation} = Cost_{energy} + Cost_{chemicals} + Cost_{carbon}$$

$$Cost_{carbon} = GHG \times P_{carbon} \qquad \text{Eq. (5.4)}$$

In which, $Cost_{operation}$ was the operation cost per unit DS included the cost of energy consumption, chemical consumption, and carbon emission. P_{carbon} was the carbon price which is the average of market price in Emission Exchange. Economic data were obtained from market investigations and environmental impact assessment (EIA) reports of each sewage sludge recycling project in China. The finical parameters of operation cost were provided in Table 5.2.

The CST was introduced into cost accounting of sewage sludge recycling system. Carbon cost refer to the economic cost of purchasing or selling the carbon emission rights in CST. With the introduction of carbon emission quota (CEQ) into sewage sludge recycling system, the cost of difference in carbon emission via by-production of system offset should be considered into the cost accounting of system. For example, the GHG emission generated during incineration can replace the carbon credit of electricity substitution (Chen et al., 2022; Piippo et al., 2018).

$$Cost_{total}{}' = Cost_{initial} + Cost_{operation} + Cost_{by-production} + Cost_{CEQ}$$

$$Cost_{CEQ} = GHG_{avoided} \times P_{carbon} \qquad \text{Eq. (5.5)}$$

In which, $Cost_{total}{}'$ was the total cost per unit DS with CEQ. $Cost_{CEQ}$ was the cost of carbon credit via by-production substitution, $GHG_{avoided}$ was the GHG emission of by-product by conventional process.

Table 5.2 Financial parameter required to calculate STRS operation cost

Parameter	Unit	value	Parameter	Unit	value
Energy					
Heavy oil	CNY/L	4.786	LPG	CNY/m^3	12.529
Coal oil	CNY/L	2.812	Disel	CNY/L	6.925
gasoline	CNY/L	7.022	Coal	CNY/t	1696.667
Electricity	CNY/kwh	0.635	Natural gas	CNY/m^3	2.62
Chemicals					
Ferrous chloride	CNY/t	500	Poly-ferrous sulfate	CNY/t	900
$Ca(OH)_2$	CNY/t	500	CaO	CNY/t	450
PAM	CNY/t	6000	Poly-aluminum chloride	CNY/t	1200
H_2O_2	CNY/t	750	$CaCO_3$	CNY/t	400
NaOH	CNY/t	1500			
By-production					
Clay Brick	CNY/piece	0.5	Fertilizer	CNY/t	2310
Electricity	CNY/kwh	0.65			

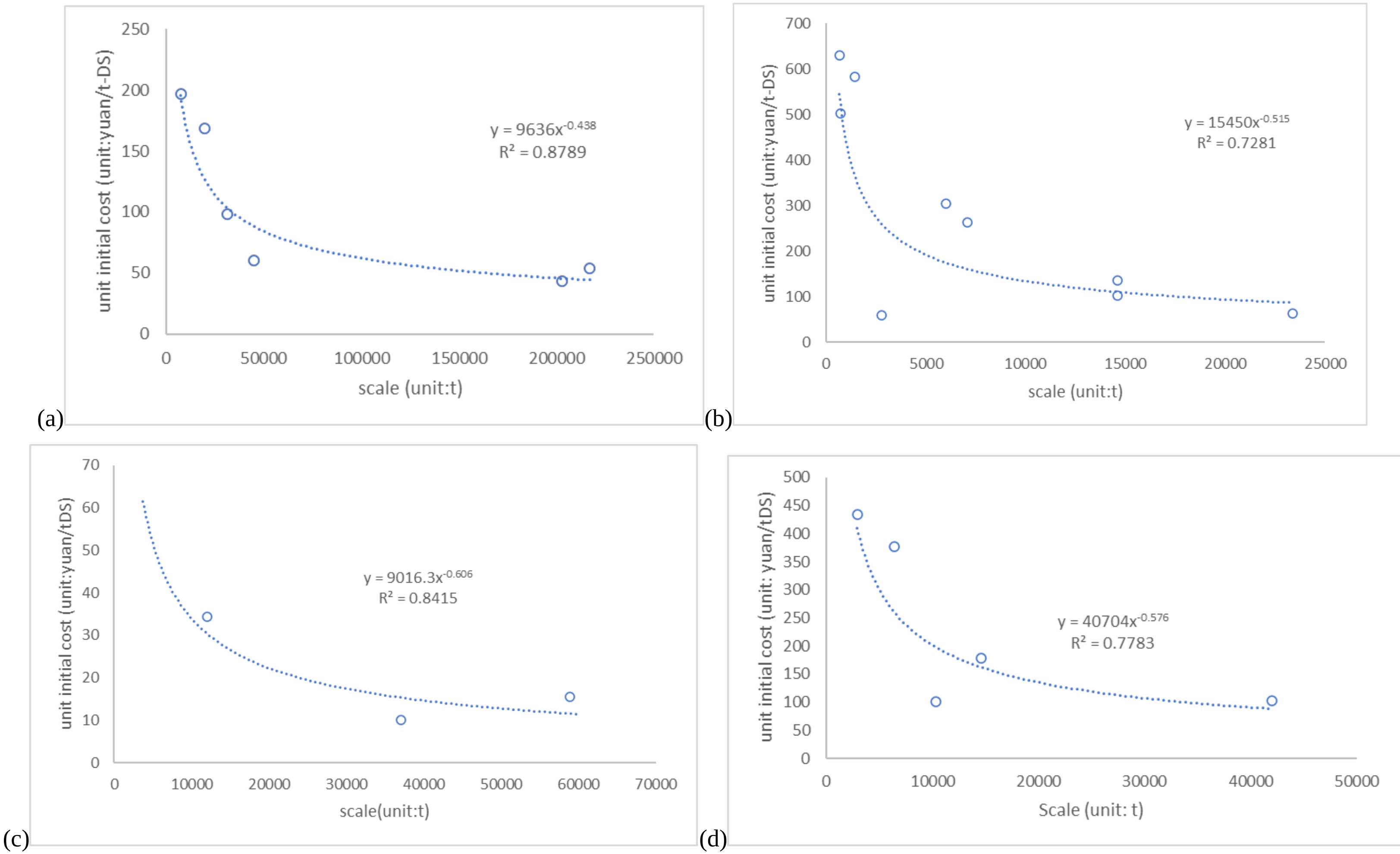


Fig. 5.4 The relationship of unit initial cost of different system and implementation of scale.
(a) Incineration; (b) Aerobic composting; (c) Used in material (brick); (d) Anaerobic digestion

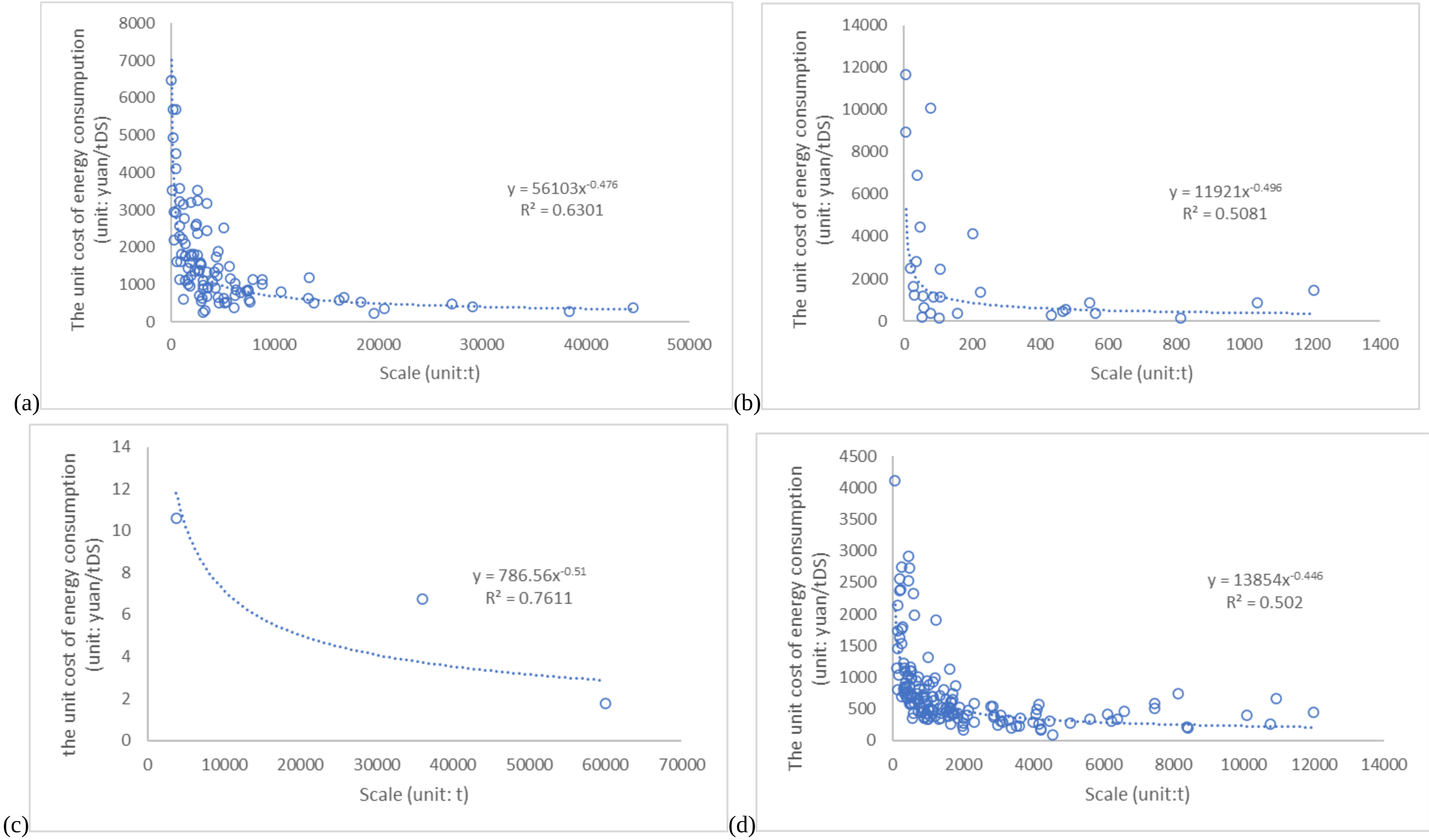


Fig. 5.5 The relationship of the unit cost of energy consumption of different system and implementation of scale.
(a) Incineration; (b) Aerobic composting; (c) Used in material (brick); (d) Anaerobic digestion

5.3.2 The scale effect of environment and economy during different scenarios

(1) The unit initial cost

Initial cost is a critical factor to direct the decision making of the investors. Investors determined whether to invest the project through the initial cost and related to how to manage the finance. In this study, we assumed the lifetime of sewage sludge recycling facility to be 30 years (Luo et al., 2021). The unit initial cost decreased with implementation of scale increase. From this Fig. 5.4., there is a significant difference between the initial cost of used in material and other three systems, which the latter had about 7 times more than the frontier. A comparison of the proportion of initial cost in the cost accounting per year revealed that the highest system is anaerobic digestion about 25%, and the lowest system is used in material about 8% under the implementation of scale about 10 kt DS.

(2) The unit cost of chemical consumption

Further statistics revealed the unit cost of chemical consumption follows the scale effect which decreases with the implementation of scale increases. The critical part of the cost of chemicals is Polyacrylamide (PAM) during the dewatering process in each system. The reduction rate of 200 kt to 1200 kt and 2 Mt to 3 Mt decrease from 76% to 27% and gradually stabilized in Fig. 5.6.

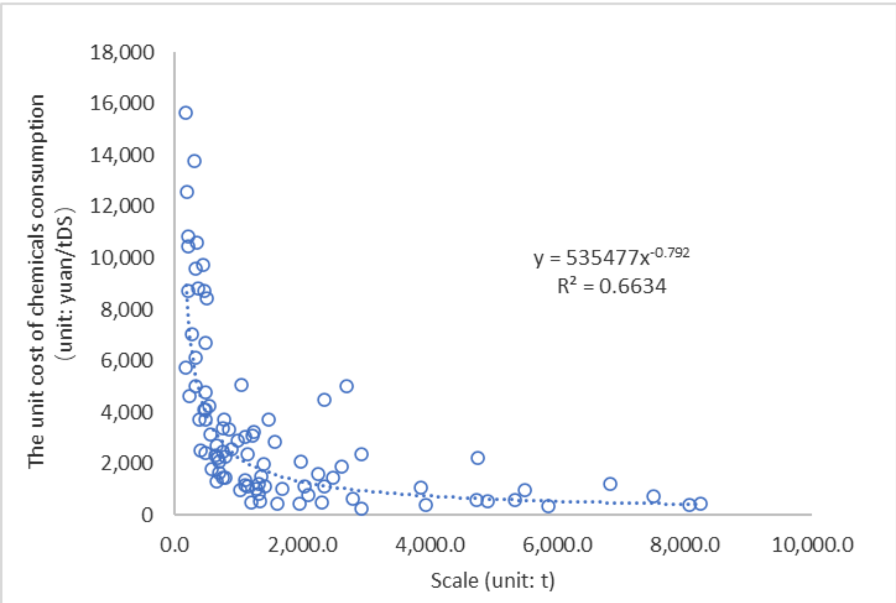


Fig. 5.6 The relationship of the unit cost of chemicals consumption and implementation of scale.

(3) The unit cost of energy consumption

In the system, the energy consumption was due to the utilization of electricity, coal, A heavy oil, liquefied petroleum gas, kerosene, diesel, gasoline, and natural gas. The price of energy was collected from published data of government and the China Petroleum & Chemical Industry

Association. The unit cost of energy consumption was obtained by diving the annual scale. The implementation of scale had a negative impact on unit cost of energy consumption, as shown in Fig. 5.5. It covered 63.7%, 24%, 1.9%, and 37.9% in unit operation cost for incineration, aerobic composting, used in material, and anaerobic digestion which treated 10 kt-DS per year, respectively.

(4) The unit cost of carbon emission

The regular pattern follows the power function as same as the GHG emission presented in Chapter 5.3.1. The cost of carbon emission referred to the direct carbon emission during the system, which did not include the tax of carbon emission. The cost accounting of the system appeared to be unaffected by the cost of carbon emission even though introducing the carbon trading mechanism.

(5) Revenue of by-production

A positive correlation was found between the implementation of scale and the unit revenue of each system. Data from this Fig. 5.7 (d) can be compared with the data in Fig. 5.7 a-c which shows that anaerobic digestion has a clear trend of increase at the beginning of implementation of scale increases. It is apparent the unit revenues of by-production of anaerobic digestion and aerobic composting are obviously more than the other two systems. The increase of productivity growth rate was obvious steady gradually with the implementation of scale increased. The phenomenon about this was the efficiency of production was low caused of the low calorific value and C/N of sewage sludge (Cydzik-Kwiatkowska et al., 2022; da Cunha et al., 2021; Deng et al., 2017).

(6) The total cost

The total cost of this study was the sum of the cost accounting and revenue of each system, and divided the annual total cost by the annual scale yielded the unit total cost. By far, the first demand for sewage sludge recycling is the minimization of the initial cost when investors or decision-makers decide to invest in the project. From Fig. 5.8, it can be seen that while the decision-maker did not consider the revenue of the system, the optimal technology is used in material (brick) with the lowest unit initial cost and unit operation cost, currently. The results of previous studies prove this point, the system of used in material was the optimal selection based on the lowest economic cost (Xu, et al. 2018).

Data from this Fig. 5.8 (a) can be compared with the data in Fig. 5.8 (b) which the proportion of initial cost and operation cost. In the system of incineration to treat 1 Gt dry solid of sludge, the operation cost covered 80%, among them 85.3% from the cost of energy consumption. 31% of the total cost of aerobic composting was the operation cost in which the cost of energy and chemicals consumption was approximately equal. The proportion of the operation cost was 17% of the total cost in the system of used in material, which the cost of chemicals mainly provided. The cost of energy and chemicals consumption inside the operation cost in anaerobic digestion were about 54% and 18%.

The revenue of anaerobic digestion has a clear trend of increase at the beginning of implementation of scale increases as shown in Fig. 5.8. It is apparent the unit revenues of by-production of anaerobic digestion and aerobic composting are obviously more than the other two systems. The optimal system based on unit revenue of by-production is anaerobic digestion which implementation of scale less than 297,255 t-DS, and the opposite is aerobic composting.

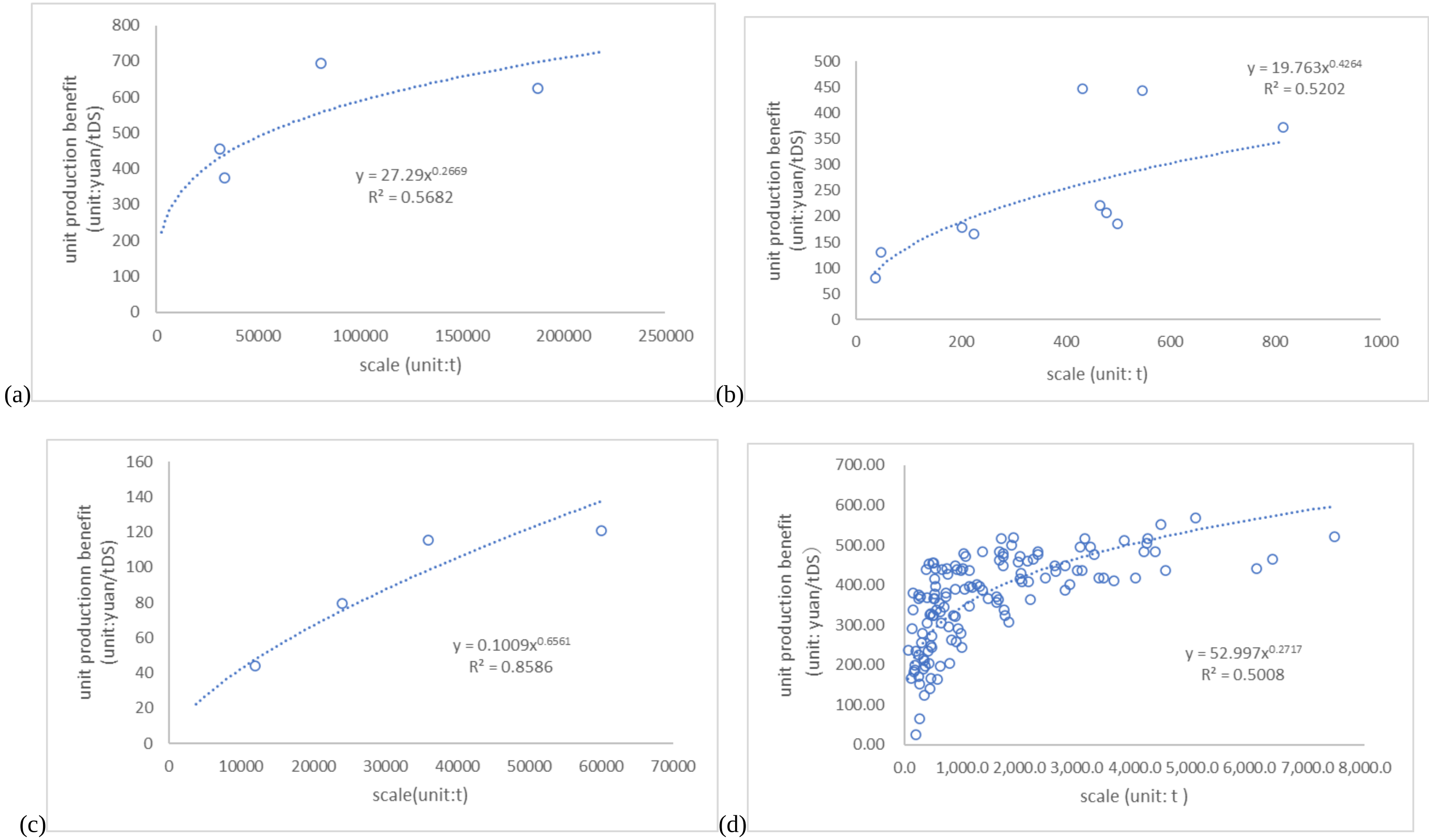
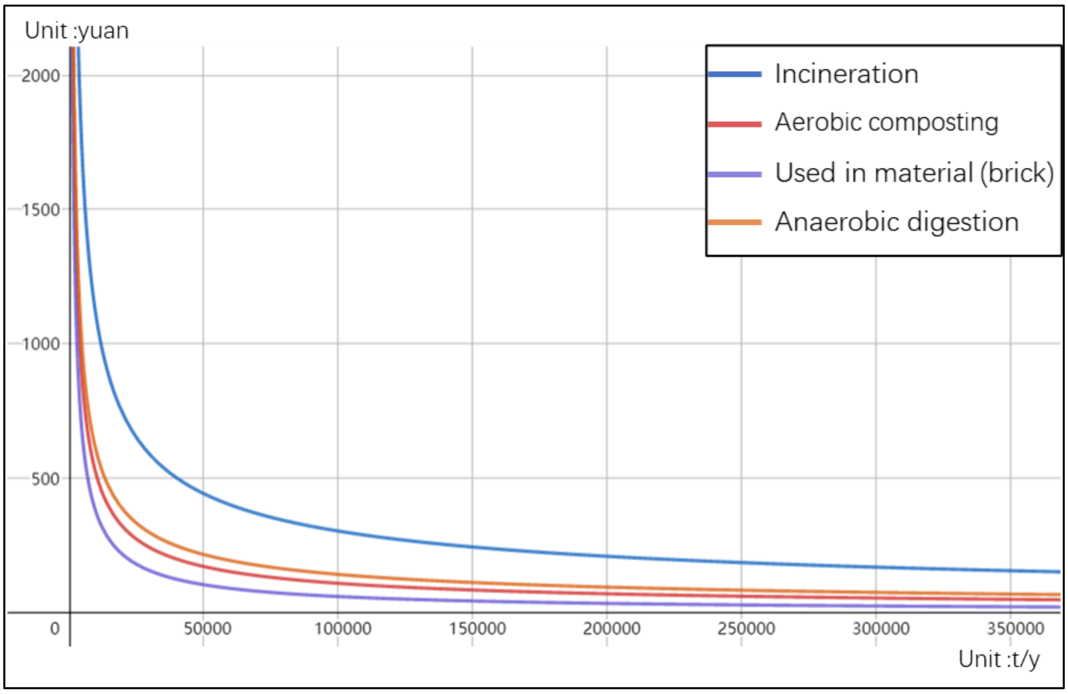
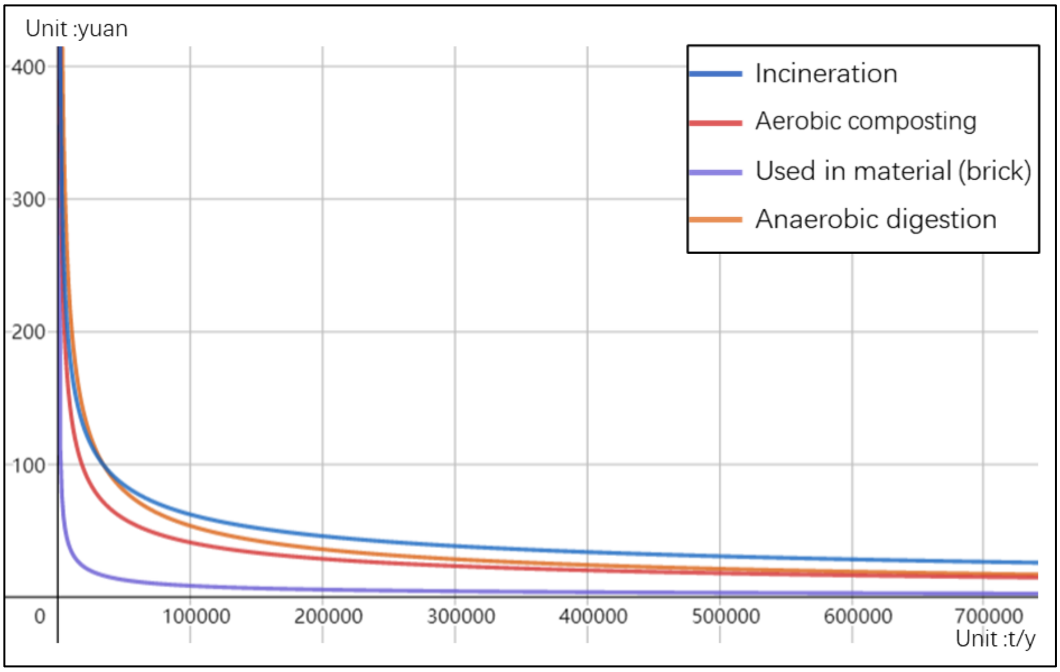


Fig. 5.7 The relationship of the unit revenue of different system and implementation of scale.
(a) Incineration; (b) Aerobic composting; (c) Used in material (brick); (d) Anaerobic digestion


Unit :yuan
2000
1500
1000
500
0
50000
100000
150000
200000
250000
300000
350000
Unit :t/y
Incineration
Aerobic composting
Used in material (brick)
Anaerobic digestion

(a)


Unit :yuan
400
300
200
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100000
200000
300000
400000
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600000
700000
Unit :t/y
Incineration
Aerobic composting
Used in material (brick)
Anaerobic digestion

(b)

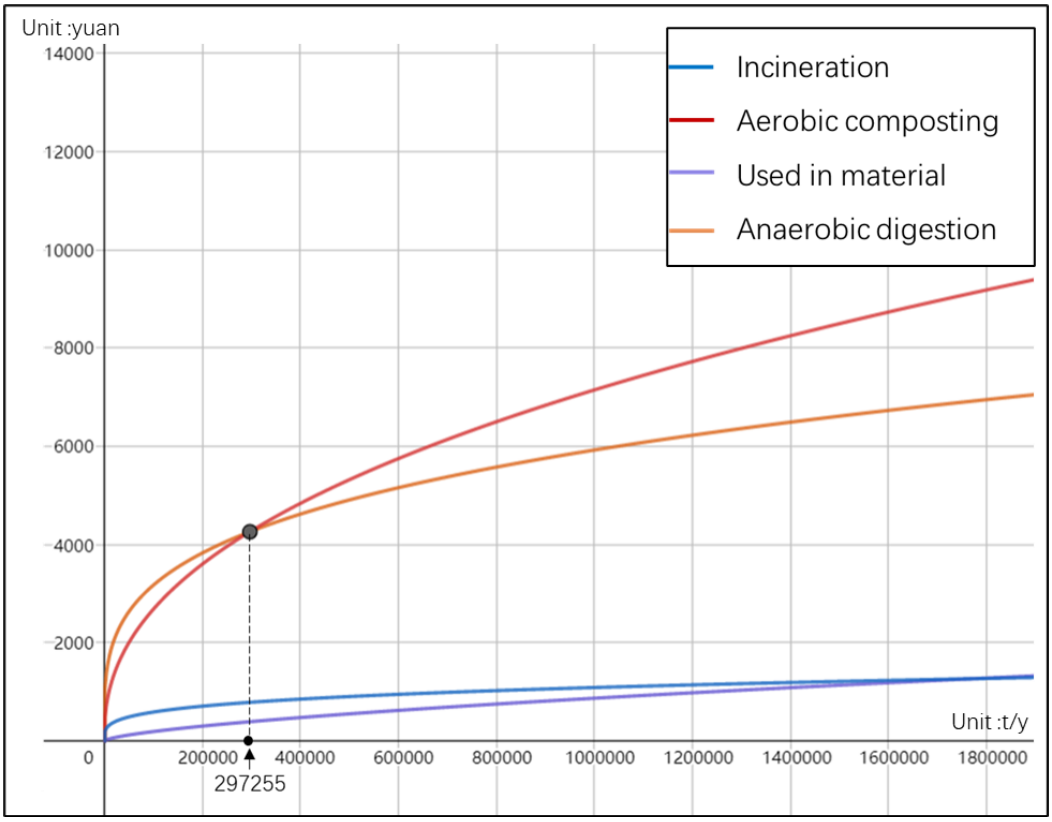
(c) 

Fig. 5.8 The unit total cost of different system.

(a) Operation cost; (b)Initial cost; (c) Revenue of by-production

5.4 The scale effect of break-even scale for different scenarios considering the carbon trading mechanism

The curve in Fig. 5.9 presented the total cost of each scenario. The break-even scale, as the zero point of the curve, referred to the revenue equal to the cost of the system. Therefore, the project had the financial value while the implantation of scale exceeded the break-even scale without government subsidies. The minimum of break-even scale was anaerobic digestion (4425 t-DS/y) and then the break-even scale of aerobic composting, used in material, and incineration were 6707, 48,775 and 54,899 t-DS/y, respectively. In summary, the results presented that the sewage sludge recycling system had negative economic performance when the implementation of scale was lower than 4425 t-DS/y. The optimal system was anaerobic digestion while the implementation of scale. between 4425 and 285,345 t-DS/y. With successive increases of the implementation of scale exceeding 285,345 t-DS/y, the system of aerobic composting was the optimal system.

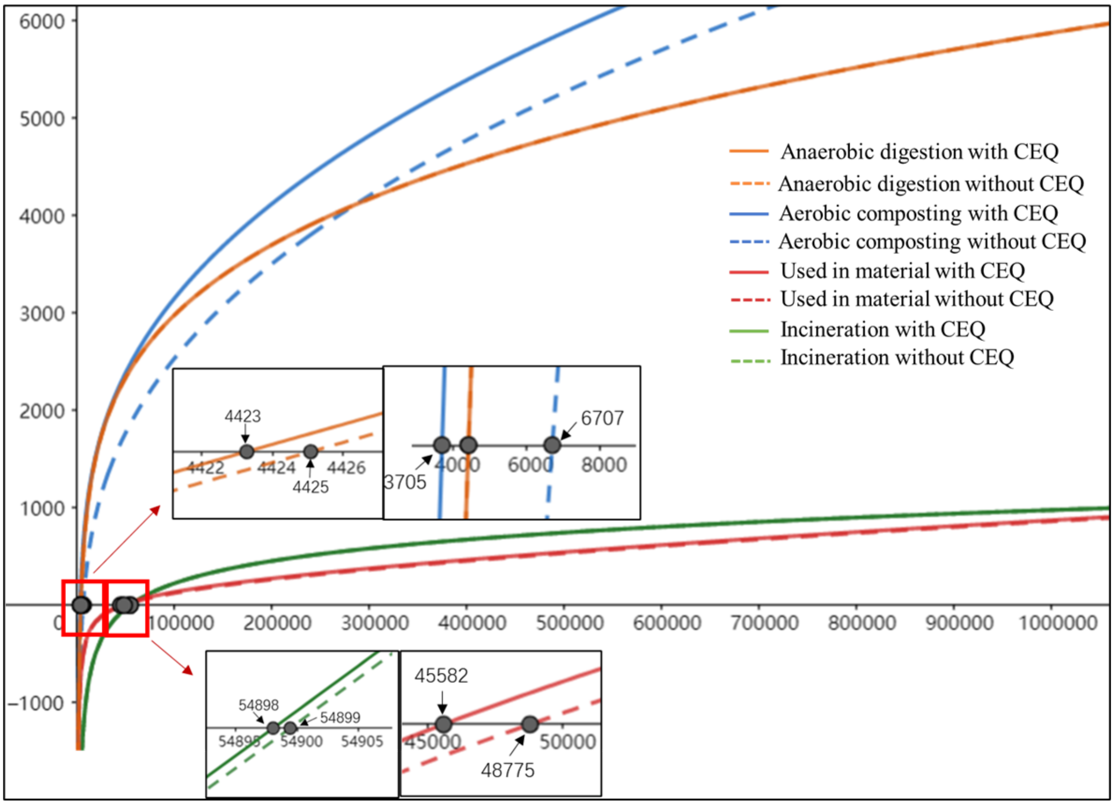


Fig. 5.9 The break-even scale of each system considering CEQ

Based on the carbon emission quota (CEQ) of production and China's carbon trading mechanism, the avoided GHG emission could obtain revenue via carbon emission trading. Subsequently, the impact of introducing the carbon trading mechanism to the break-even scale was analyzed. Comparison between the break-even scale with CEQ and without CEQ, there was a significant lessening in the break-even scale of aerobic composting. In the by-production of all systems, the nitrogen fertilizer had the highest carbon emission. Thence, the impact of the break-even scale with CEQ was obviously higher than in other systems.

5.5 Sensitivity analysis

With the introduction of the carbon trading mechanism, sewage sludge recycling facility investors can earn revenue by selling CEQ, which in turn decreases the total cost of the system. The break-even scale can be affected by the cost of carbon emissions and revenue of avoided carbon emissions. The violent fluctuation of carbon price caused by policy related to carbon emission allowance and is due to the immature market of China's emission trading scheme with an obvious policy-oriented. Therefore, we conducted a sensitivity analysis for the cost of carbon emissions and break-even scales using different carbon prices. As shown in Fig. 5.10, the rate of unit carbon emissions in the total cost had an apparent difference between incineration and aerobic composting and the other two systems. The rate of unit cost of carbon emissions in total cost increased with increasing carbon price; among them, the system with the highest ratio was aerobic composting (2.39%) and the lowest was used in building material (0.11%).

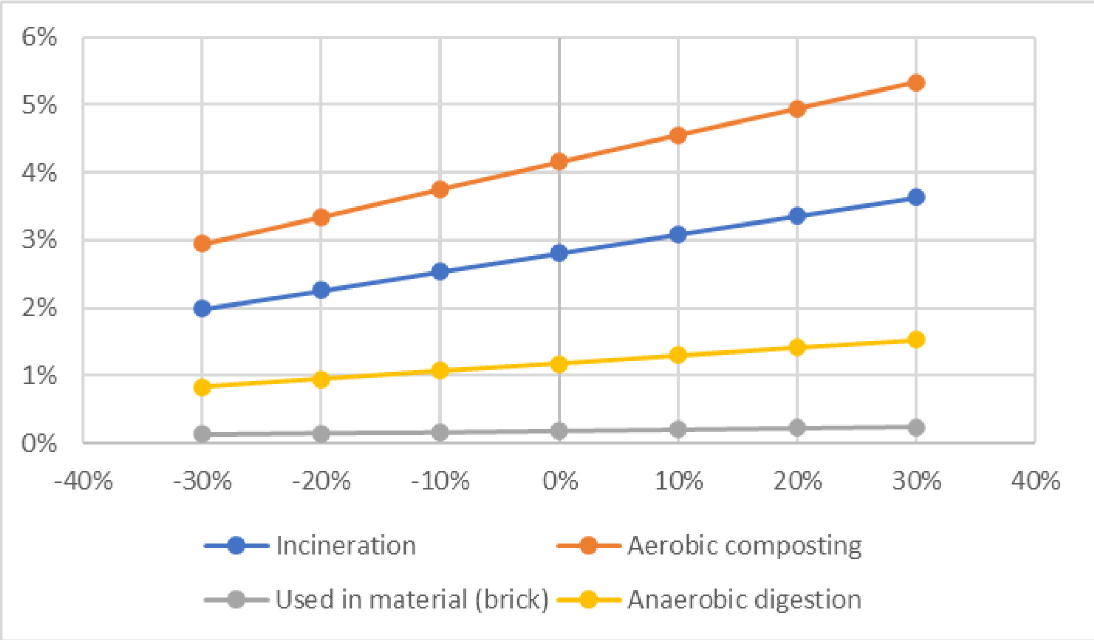


Fig. 5.10 Relationship between the rate of unit carbon cost in total cost and carbon price

With the increase in carbon price, the break-even scale without the CEQ increased slightly. Further analysis showed that the break-even scale with CEQ of aerobic com-posting and used in building material decreased owing to profit of the avoided carbon mission. Significantly, Fig. 5.11 shows that the break-even scale with CEQ of incineration and anaerobic digestion increased, which is approximate to the break-even scale without CEQ. When the break-even scale with CEQ decreased as carbon price in-creased, the unit cost of carbon emissions was larger than the profit of avoided carbon emission of product. Therefore, sewage sludge recycling technology is recommended to replace high-carbon-emission products.

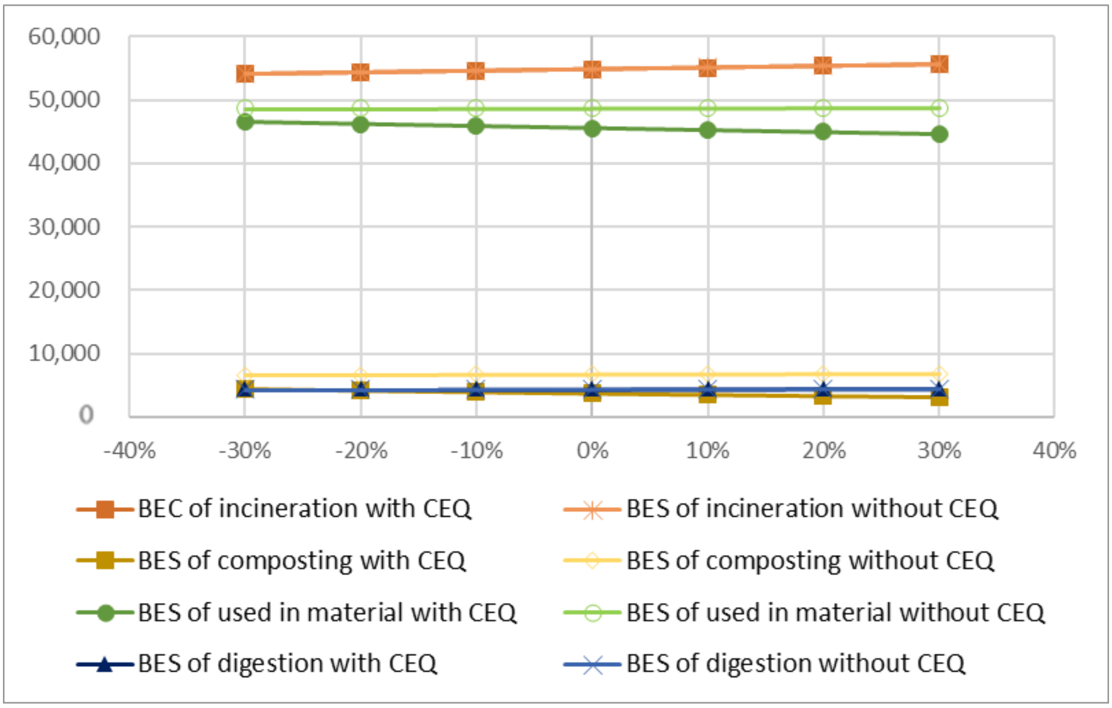


Fig. 5.11 The sensitivity analysis of the break-even scale and carbon price

5.6 Summary

This chapter provided new evidence on the implementation of scale affects the result of environmental and economic assessment of technology selection for sewage sludge recycling systems. A quantitative comparative evaluation was conducted of the GHG emission and unit total cost of each system. In the aspects of environment and economy, the deviation caused by the implementation of scale was determined for different scenarios.

The unit cost of each part decreased with the increase of the implementation scale, while the unit revenue of by-production increased with the implementation of scale. Therefore, considering the revenue of by-production, there is no economic value that the unit total cost was negative when the actual scale was smaller than the break-even scale of each system to the investors without the government subsidy. If the policy-oriented project ignored the implantation scale, it would improve the financial burden on the local government. The unit carbon emission was decreased with the implementation scale increased. When introducing the carbon trading mechanism, it is advantageous to expand the applicability of sewage sludge recycling, which minified the break-even scale under the income of avoided carbon emission allowance (Huang and Xu, 2020).

The insights gained from this chapter may be of assistance in improving the accuracy of evaluating the economic and environmental performance of the projects. From the investor's acceptance ground, the decisive factor in starting up a new project was the total initial cost of different technology selections. The optimal technology for sewage sludge recycling project was used in material (brick) which merely considered the initial cost of the project as well as ignored the impact of implementation scale. While the implementation scale was larger than 285,345 t-DS per year, the optimal technology was aerobic composting considering the revenue of by-production. And the optimal technology was anaerobic digestion when the implementation scale among 4425 to 285,345 t-DS. Hence, the technology selection and implementation scale were the critical elements of the sewage sludge recycling system strategy for decision-makers based on the scale of the city.

A limitation of this chapter is that the lack of situation of sewage sludge recycling projects in China. At the local government level, sustainable sewage sludge management should be designed considering the situation of the city. Besides, we strongly suggested that the decision-maker should pay attention to intelligence statistics and forceful administration.

5.7 Construction of an environmental and economic evaluation system for sewage sludge recycling system

The impact of scale on the environmental sustainability of wastewater treatment and sewage sludge disposal was investigated in previous studies, most of studies do not consider holistically integrated energy and resource recovery. Simultaneously, the investigations of sewage sludge recycling and sludge management option emphasized the benefits to centralization due to the economics of scale. To estimate the target scenario accurately, the environmental and economic evaluation system was constructed considering the deviation caused by the implementation of scale. In this chapter, an example of GHG was used to establish the environmental assessment system. Meanwhile, the economic assessment system was constructed included the external cost caused by environmental impact. The results of the environmental and economic evaluation model would evidence the sustainability of centralization treatment caused by the sludge production of different cities scale.

5.7.1 Construction of an environmental evaluation system for sewage sludge recycling

The environmental performance was be influenced by the implementation of scale in wastewater treatment (Cornejo, et al., 2016). Therefore, the environmental evaluation system was constructed considering the technology selection and the implementation of scale in this dissertation. Taking GHG emissions as an example, the environmental evaluation system of the sewage sludge recycling system was constructed as follows in Eq. (5.6) and Eq. (5.7):

$$GHG_{i,base} = GHG_{total} - GHG_{by-product} \qquad \text{Eq. (5.6)}$$

where, $GHG_{i,base}$ refer to the actual GHG emission of sewage sludge recycling system by technology i considering the avoided GHG of by-product, which as baseline of GHG emission in the evaluation system; GHG_{total} is the entire GHG emission of sewage sludge recycling system as describe in Chapter 5.2.1; $GHG_{by-product}$ determined as the GHG emission of by-product by conventional process.

$$GHG_i = \frac{GHG_{i,base}}{\left(\frac{S_{base}}{S}\right)^{x_{GHG}}} \qquad \text{Eq. (5.7)}$$

where, GHG_i refer to the GHG emission of target scenario; S_{base} is the implementation scale of baseline; S is the implementation scale of target scenario; x_{GHG} is the scaling exponent of GHG emission.

5.7.2 Construction of an economic evaluation system for sewage sludge recycling

The economic evaluation system of sewage sludge recycling system based on the method of Chapter 4.4.2 integrates the implementation of scale and carbon trading system. Economic evaluation system of sewage sludge recycling system based on NPV was constructed as follow. The cash inflow and outflow of target scenario was calculated in Eq. (5.8).

$$C_{I,i,base} = R_{by-product} + R_{CEQ} + S$$

$$C_{O,i,base} = C_{initial} + C_{operation} + C_{tax} + C_{CE} + C_{exteral} \qquad \text{Eq. (5.8)}$$

where, $C_{I,i,base}$, $C_{O,i,base}$ refers to the cash inflow and outflow of NPV of sewage sludge recycling system i respectively, which as baseline of cash inflow in the evaluation system, $R_{by-product}$ and R_{CEQ} are the revenue of by-product and CEQ, S is the subsidy of sewage sludge disposal, $C_{initial}$, $C_{operation}$, C_{tax}, C_{CE}, $C_{exteral}$ refer to the initial cost, operation cost, the cost of carbon emission (CE), and external cost.

In this evaluation system, external cost excepts the cost of GHG emission by introducing carbon emission trading. The effect of the implementation scale on the cost of energy and chemical consumption which are the major contributions to the operation cost represents the effect of the operation cost considering the implementation scale (Shi, et al., 2022). The cost or revenue of target scenario was calculated as Eq. (5.9).

$$C_j = \frac{C_{j,base}}{\left(\frac{S_{base}}{S}\right)^{x_j}} \qquad \text{Eq. (5.9)}$$

where, C_j, $C_{j,base}$ refer to the cost or revenue of type j in target scenario and baseline scenario, respectively, x_j refer to the scaling exponent of the cost or revenue of type j. According to the analysis in Chapter 5.3.2, initial cost, operation cost, the cost of carbon emission, and the revenue of by-product and CEQ is considered the effect of implementation scale in this evaluation.

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6 CONCLUSION AND DISCUSSION

6.1 Summary of conclusion of each chapter

Chapter 1 describes the condition of sewage sludge generated and sewage sludge treatment and recycling route in China in recent year; and introduces the status of sewage sludge treatment in each city of China. The methodology and proportion of sewage sludge recycling in other countries is introduced. Sewage sludge is not only the growing environmental problem, but also the source of energy and resource in the urban. However, the environmental and economic assessment of sewage sludge recycling system focused on the energy and resource recovery cannot be ignored. This is the significance and purpose of this dissertation.

In Chapter 2 illustrates the common methods for environmental and economic assessment of sewage sludge recycling system. The previous studies of assessing sewage sludge recycling systems via LCA and LCA are summarized in this chapter. In addition, the factors affected the environmental and economic performance of sewage sludge recycling include the composition of sewage sludge, technology selection, and the implementation of scale. According to this, the research methodology and framework of this dissertation are determined.

The environmental performance used LCA of sewage sludge recycling systems included incineration, aerobic composting, used in material (brick), AD, AD + fertilizer in greening, and AD + incineration was analyzed by case study in Chapter 3. Aerobic composting had least environmental impact of four systems via LCA. The result of LCA presented the major environmental impact category of each system and the major pollutants of specific environmental impact category. According to the endpoint environmental impact results, human health is the primary endpoint environmental impact category for the scenarios.

The economic performance of six sewage sludge recycling systems was analyzed via LCC considering the effect of policy in Chapter 4. NPV applied to calculate the LCC of scenarios. According to the external cost, the environmental and economic performance integrated by pollutants monetized. The economic feasibility of systems was analysis based on the payback year. Based on the results, used in material was the preferred system for sewage sludge recycling with market potential. AD + incineration exhibited the highest commercial feasibility, and other scenarios except incineration and aerobic composting had market potentials in China. The other five scenarios would not survive without exemption of tax, except aerobic composting. Aerobic composting and used in material were more adaptable when universal and evident impact on the other scenarios by relative policy.

In Chapter 5, the implementation of scale-affected GHG emission and economic cost of each system was quantitatively analyzed. The unit cost or revenue of each part and unit GHG emission decreased as the implementation scale increased. The break-even scale of the sewage sludge recycling system of the scenarios was determined. While the implementation scale was less than 4425 t-DS, the sewage sludge recycling system did not excite the commercial feasibility. The optimal technology for analyzed systems was anaerobic digestion whose implementation scale was between 4425 t-DS and 285,345 t-DS. The implementation scale was over 285,345 t-DS, the prior technology was aerobic composting. Meanwhile, the feasibility of sewage sludge recycling was extended after introducing the carbon trading mechanism. Based on the result of Chapter 5, Chapter 3, and Chapter 4, the environmental and economic evaluation system was established.

6.2 Summary of key findings and limitations

6.2.1 Key findings

First, this dissertation constructed the environmental and economic evaluation system of different sludge recycling systems, which focused on energy and resource recovery, considering the technology selection and implementation scale. From the economic aspect, anaerobic digestion with biogas recovery is an optimal scenario on a small implementation scale. For the large implementation scale, aerobic composting with fertilizer is the optimal scenario. This dissertation provides the evaluation model to estimate the anticipatory system based on the data of existing systems of sewage sludge recycling. The evaluation model considers the effects of the implementation scale to estimate the environmental and economic performance of the sewage sludge recycling target systems more accurately. It provided sewage sludge recycling management options considering technology selection, the implementation of scale, and carbon trading introduced to promote the sustainable development of the wastewater system.

Second, a comparison of the life cycle cost of different systems under policy analysis, the influence of policy changes by tax and subsidy on the choice of system was studied. Based on the result in Chapter 4.4.2, the enterprise income tax and relevant subsidies have a significant impact on the normal maintenance of the sludge recycling system. Currently, the sewage sludge recycling system does not exist market potential without exempting the relevant taxes. In summary, the government should issue relevant industrial policies increase the subsidy of sludge recycling to support the resource utilization of sludge. In the future, the complete guideline of relevant policies should be issued to better accomplish the circular economy of sewage sludge and promote the construction of the “zero-waste” city.

Third, an integrated evaluation of the environment and economy was constructed, and environmental impact was monetized as an external cost. The main contribution of environmental impact to externality costs that mark potential social damages caused by pollutant discharge for sewage sludge recycling system was also determined. The emissions related to human health were the major contribution in the systems. It was not only important to evaluate the system's economic performance in terms of its internal costs, but also in terms of its external costs, which relate to its environmental impacts. Based on the results of economic performance, the management gap of sludge disposal would be filled in the wastewater system management. The emissions of high monetized value should be paid attention to reduce in the sewage sludge recycling management to achieve sustainable development.

Fourth, this dissertation highlighted the quantitative relationship between the implementation of scale and unit environmental emission and economic cost. The minimization scales of different systems, that achieves minimizing the total cost, combined with the initial cost, operation cost, and the revenue of by-products of different systems, was determined. The optimal scenario is anaerobic digestion between 4423 to 285,345 t-DS/y. Aerobic composting is suitable scenario when the implementation scale is larger than 285,345 t-DS/y. The effect of CEQ on the break-even scale after introducing the carbon trading mechanism is compared. As in Chapter 5.4, it can be inferred the sludge recycling system with energy recovery or by-product subsisted would drop the break-even scale to improve the suitableness of sludge recycling system in medium and small cities. The influence degree of the minimization scale depended on the GHG emission of alternative production by the conventional processes.

6.2.2 Limitations

1) A limitation of this dissertation is that the lack of situation of sewage sludge recycling projects in China. At the local government level, sustainable sewage sludge management should be designed considering the situation of the city. Besides, we strongly suggested that the decision-maker should pay attention to intelligence statistics and forceful administration.

2) Future research should focus on evaluating the sewage sludge recycling system combined with other factors (different transportation radius, different regional characteristics, different composition of sewage sludge, and market demand for sewage sludge products) to achieve overall environmental friendliness, economic feasibility, and flexibility of sewage sludge recycling system.

Appendix

Table S1. Inventory of main energy and materials consumption of each scenario

Inventory flow	Unit	Scenario 1: Incineration	Scenario 2: Aerobic composting	Scenario 3: Used in industry (brick)	Scenario 4: Anaerobic digestion	Scenario 5: Anaerobic digestion + incineration	Scenario 6: Anaerobic digestion + green using
Input							
Electricity	kWh	108.18	3.2857	1.087	68.3288	373.129	71.851
Gas	Kg	20.6105	9.06	1.06E-04	0.0066	0.0295	0.007
Coal	Kg	62.7444	0.0029	65.8496	39.6314	176.784	41.6737
Crude	Kg	1.5578		0.0157	0.984	4.3890	1.0347
Straw	Kg	100	100	100	100	100	100
Water	Kg	33.7	0.0625	72.4638	1.8986	35.5990	1.9611
Shale	Kg			1913.043			
Output							
NH_3	Kg	0.0096	0.0070	0.005188	0.0117	0.0214	0.0175
H_2S	Kg	3.64E-04	0.0326	5.12E-04	0.02	0.016	0.055
HCl	Kg	0.01		0.007		0.0673	
HF	Kg	0.002		0.007		0.0087	
SO_2	Kg	2.3464	0.0479	0.4252	2.9225	3.2283	
CH_4	Kg	0.2813		0.0028	0.766	0.7925	0.7752
NMVOC	Kg	0.0527		5.31E-04		0.1484	
CO	Kg	0.3726		0.0017	0.4570	0.51	

Inventory flow	Unit	Scenario 1: Incineration	Scenario 2: Aerobic composting	Scenario 3: Used in industry (brick)	Scenario 4: Anaerobic digestion	Scenario 5: Anaerobic digestion + incineration	Scenario 6: Anaerobic digestion + green using
CO_2	Kg	265.7526		1.1663	314.5	326.136	
NOx	Kg	2.0039		0.297	1.9228	2.6036	
Dust	Kg	4.4312		1.0222	1.3803	6.2183	1.4643
Hg	Kg	1.00E-04	2.88E-07	9.57E-08	2.59E-05	1.19E-04	2.62E-05
Cd	Kg	9.23E-05	4.17E-08	1.38E-08	3.74E-06	9.58E-05	3.78E-06
Pb	Kg	3.87E-04	5.78E-06	1.92E-06	5.19E-04	7.35E-04	5.25E-04
As	Kg	2.16E-04	6.57E-06	2.18E-06	5.9E-04	6.1E-04	5.97E-04
Cr	Kg	1.83E-05	5.55E-07	1.84E-07	4.98E-05	5.15E-05	5.04E-05
Ni	Kg	2.70E-05	8.21E-07	2.73E-07	7.37E-05	7.62E-05	7.46E-05
V	Kg	3.12E-04	9.46E-06	3.14E-06	8.49E-04	8.78E-04	8.59E-04
Zn	Kg	2.60E-04	7.89E-06	2.62E-06	7.08E-04	7.32E-04	7.16E-04
Dioxin	Kg	1.92E-09				8.7E-09	
CODcr	Kg	1.397	0.0079	0.031	2.3	3.6762	
NH_3-N	Kg	0.1394	6.79E-04	0.003	2.0219	2.1628	2.0226
TN	Kg				2.717	2.717	2.717
TP	Kg				0.2717	0.2717	0.2717

Table S2. Inventory of by-product via conventional method

	Electricity		Heat		Fertilizer		Fired Brick	
	Unit	Value	Unit	Value	Unit	Value	Unit	Value
Input								
Electricity					kwh/t	141.9275	kwh/t	12.891
Coal	kg/kwh	0.58	kg/MJ	6.70E-04	kg/t	39.9	kg/t	0.0514
Oil	kg/kwh	0.0144			L/t	13.2792		
Gas	m^3/kwh	9.69E-05			m3/t	543.305		
K_2O					kg/t	83.3		
P_2O_5					kg/t	313		
Iron							kg/t	0.0234
Clay							kg/t	1.039
Shale							kg/t	0.463
Output								
CO_2	kg/kwh	1.07	kg/MJ	1.16E-03	kg/t	94.1	kg/t	0.0813
SO_2	kg/kwh	9.93E-03	kg/MJ	5.33E-06	kg/t	0.107	kg/t	1.09E-03
NOx	kg/kwh	6.46E-03	kg/MJ	2.11E-06	kg/t	0.7669	kg/t	6.37E-04
NO_2					kg/t	0.521		
NH_3					kg/t	56.2364		
CxHy					kg/t	0.0162		
CO	kg/kwh	1.55E-03			kg/t	0.3569	kg/t	9.45E-04
CH_4	kg/kwh	2.60E-03	kg/MJ	5.76E-07	kg/t	0.0853		
HCl					kg/t	7.72E-04		
NMVOC	kg/kwh	4.87E-04						
Dust	kg/kwh	0.0202	kg/MJ	1.83E-04	kg/t	0.4638	kg/t	4.26E-04
Na					kg/t	0.1340		
P					kg/t	0.1340		
Cl					kg/t	10.4878		
Ca					kg/t	0.1740		
Mg					kg/t	1.0072		
S					kg/t	1.0405		
As	kg/kwh	2.00E-06			kg/t	3.45E-03		
Cd	kg/kwh	1.27E-08			kg/t	3.45E-03		
Cr	kg/kwh	1.69E-07			kg/t	0.0172		

	Electricity		Heat		Fertilizer		Fired Brick	
	Unit	Value	Unit	Value	Unit	Value	Unit	Value
Hg	kg/kwh	8.78E-08			kg/t	3.28E-03		
Ni	kg/kwh	2.50E-07			kg/t	0.0133		
Pb	kg/kwh	1.76E-06			kg/t	0.0149		
Cu					kg/t	0.0172		
V	kg/kwh	2.88E-06			kg/t	4.09E-04		
Se					kg/t	1.08E-08		
Zn	kg/kwh	2.40E-06			kg/t	0.0203		
COD					kg/t	0.59		
NH_3-N					kg/t	0.4740		