SUSTAINABILITY EVALUATION FRAMEWORK FOR EFFECTIVE DECISION-MAKING IN URBAN WASTEWATER AND SLUDGE TREATMENT SYSTEMS

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Sustainability Evaluation Framework for Effective Decision-making in Urban Wastewater and Sludge Treatment Systems

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CERTIFICATE OF ORIGINALITY

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ABSTRACT

Sustainable water and wastewater systems are of increasing importance globally when facing the challenges of rapid urbanization, population growth and economic development. In the mission of achieving more sustainable urban water and wastewater management, application of water systems with the best-available environmental, economic and social performance is necessary. An evaluation framework that considers the characteristics of urban cities, designs of water systems and paradigm shift in wastewater treatment is demanded for. This research study developed a sustainability evaluation framework using life-cycle assessment (LCA) based techniques to assist effective decision-making in urban wastewater and sludge treatment systems.

An eco-efficiency analysis (EEA) framework was developed through integrating LCA and life-cycle costing (LCC) techniques to evaluate sewage sludge treatment options in urban cities. The framework was demonstrated in a case study of six sewage sludge management scenarios in Hong Kong. Consideration of land resource, which could be trivial in rural areas, was revealed to be crucial in urban cities. Furthermore, detailed assessment based on actual data of transportation distances was significant to avoid up to 187,000 tonnes inaccuracies in estimated GHG emissions. Sludge treatment scenario adopting anaerobic digestion (AD), dewatering, incineration and reuse in cement production was the most favorable option in the case study. By the inclusive evaluation of sludge treatment scenarios instead of individual treatment technologies, the EEA provides comprehensive and informative results and is widely applicable for sustainable urban sludge management.

An innovative EEA framework was developed for evaluating non-potable water supply systems. Four scenarios including freshwater flushing, seawater flushing, greywater recycling using aerobic membrane bioreactor (MBR) and anaerobic fluidized-bed MBR (AFMBR), were analyzed in a case study in Hong Kong. The EEA framework included detailed engineering

designs of the systems for building comprehensive and reliable inventories. Results revealed the AFMBR greywater reuse scenario to be the most eco-efficient option as the system is capable of energy recovery, recycling of water resource and reduction of sewage treatment loadings. This study demonstrated the EEA framework as an effective tool to guide water management towards sustainability and provided a basis for further research on the application of greywater recycling systems on a larger scale.

A life-cycle data envelopment analysis (LC-DEA) framework was developed for evaluating sludge-to-energy (STE) systems. The framework highlighted the strong linkage between sludge treatment and energy systems and included all essential performance metrics, namely volatile solids reduction, energy recovery, energy use, chemical consumption, sludge residues generation and direct environmental emissions, in benchmarking the efficiency of STE systems. Results showed that 44% and 69% of the sixteen STE systems were efficient in terms of overall and pure technical efficiency, respectively. The LC-DEA also informed the appropriate strategies for improving efficiency, such as increasing energy recovery, reducing energy use and scaling up/down the systems, for the less efficient systems. The framework is widely applicable for guiding decision-making on enhancing STE systems worldwide.

In summary, this research study contributed to the development of LCA-based sustainability evaluation framework that informs decision-making in sewage sludge treatment, non-potable water supply systems and STE systems in urban cities.

PUBLICATIONS ARISING FROM THE STUDY

Peer-reviewed journal articles

[1]. Alvarado, V., Hsu, S. C., **Lam, C. M.** and Lee, P. H. (in press). "Beyond Energy Balance: Environmental Trade-offs of Organics Capture and Low Carbon-to-Nitrogen Ratio Sewage Treatment Systems". Environmental Science & Technology.

[2]. **Lam, C. M.**, Hsu, S. C.*, Alvarado, V., and Li, W. M. (in press). "Integrated Life-cycle Data Envelopment Analysis for Techno-environmental Performance Evaluation on Sludge-to-energy Systems". Applied Energy.

[3]. Alvarado, V. I., Hsu, S. C., Wu, Z., **Lam, C. M.**, Leng, L., Zhuang, H., and Lee P. H. (2019). A Standardized Stoichiometric Life-Cycle Inventory for Enhanced Specificity in Environmental Assessment of Sewage Treatment. Environmental Science & Technology, 53(9), 5111 – 5123

[4]. **Lam, C. M.**, Ling, L., Chen, P. C., Lee, P. H., and Hsu, S. C. (2017). Eco-efficiency analysis of non-potable water systems in domestic buildings. Applied Energy, 202, 293-307

[5]. **Lam, C. M.**, Lee, P. H., and Hsu, S. C. (2016). Eco-efficiency Analysis of Sludge Treatment Scenarios in Urban Cities: the Case of Hong Kong. Journal of Cleaner Production, 112, 3028 – 3039

Manuscript in preparation

[6]. **Lam, C. M.**, Hsu, S. C.*, and Cai, H. "Comparing recreational water quality health indicators: an agent-based modelling approach for public health decision-making".

Conference presentations and proceedings

[7]. **Lam, C. M.**, Hsu, S. C., Alvarado, V., and Li, W. M. (2018). "Toward Sustainable Sewage Sludge Management in Hong Kong: An Eco-efficiency Approach Using LCA and DEA" International Water Association (IWA) World Water Congress & Exhibition, 16 – 21 September, Tokyo, Japan.

[8]. Alvarado, V., **Lam, C. M.**, Lai, W. K., Cheung, K. K., Hsu, S. C., and Lee, P. H. (2017). "Roadmap towards Operational Energy Neutral Sewage Treatment Works." World Sustainable Built Environment Conference, June 5~7, Hong Kong.

[9]. **Lam, C. M.** and Hsu, S. C. (2015). "Life-Cycle Sustainability Assessment of Anaerobic Greywater Treatment for Energy Production and Water Reuse." 2nd International Conference on Sustainable Urbanization, Hong Kong.

Other papers of the author are not included in the thesis.

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A2O	Anaerobic/anoxic/oxic
AD	Anaerobic digestion
AeD	Aerobic digestion
AFBR	Anaerobic fluidized bed bioreactor
AFMBR	Anaerobic fluidized bed membrane bioreactor
AP	Acidification potential
BOD	Biological oxygen demand
CHP	Combined heat and power
COD	Chemical oxygen demand
CRS	Constant return-to-scale
DEA	Data envelopment analysis
DFE	Dual fuel engine
DMU	Decision-making unit
DRS	Decreasing returns to scale
Dry	Drying
DS	Dry solids
DT	Dry tonne
DVS	Daily volatile solids handled
Dw	Dewatering
EE	Eco-efficiency
EEA	Eco-efficiency analysis
EIO	Economic input-output
EIO-LCA	Economic input-output life-cycle assessment
EP	Eutrophication potential
FBC	Fluidized-bed combustion
FU	Functional unit
GAC	Granular activated carbon
GHG	Greenhouse gas
GWP	Global warming potential
HATS	Harbour area treatment scheme
HTP	Human toxicity potential
I	Incineration
IOA	Input-output analysis
IRS	Increasing returns to scale
LA	Land application
LCA	Life-cycle assessment
LCC	Life-cycle cost
LC-DEA	Life-cycle data envelopment analysis
LCI	Life-cycle inventory
LCIA	Life-cycle impact assessment
LCSA	Life-cycle sustainability assessment
Lf	Landfill
LNG	Liquefied natural gas
MBR	Membrane bioreactor
MFC	Microbial fuel-cell
MSW	Municipal solid wastes
NENT	North East New Territories

NIRS	Non-increasing returns to scale
NPV	Net present value
O&M	Operation and maintenace
OTE	Overall technical efficiency
PM	Particulate matters
PS	Primary sludge
PTE	Pure technical efficiency
PV	Present value
RTS	Return-to-scale
SAS	Surplus activated sludge
SCI	Stonecutters Island
SE	Scale efficiency
SENT	South East New Territories
SS	Sludge solids concentration
SSG	Sludge specific gravity
ST	Shatin
STE	Sludge-to-energy
STF	Sludge Treatment Facility (Hong Kong)
STW	Sewage treatment work
SV	Daily sludge volume
SWH	Shek Wu Hui
T	Thickening
TDSS	Daily dry solids handled
TP	Tai Po
UWWTD	Urban Waste Water Treatment Directive
VOC	Volatile organic compounds
VRS	Variable return-to-scale
VS	Volatile solid
WENT	West New Territories
WWTP	Wastewater treatment plant
YL	Yuen Long

CHAPTER 1

INTRODUCTION

1.1 Research background

Water sustainability has gained increasing attention from government authorities, decision makers, practitioners and researchers globally. Rapid population growth, economic development and urbanization have caused substantial increase in water consumption and wastewater treatment demand. As critical urban infrastructure systems, water supply and wastewater management systems provide services that support the fundamental needs of communities. At the same time, urban water systems have been identified to be a crucial factor influencing sustainable urban development in terms of the environmental, economic and social consequences.

Globally, an approximate amount of 4.6×10^{12} cubic meters of water is demanded annually (Boretti & Rosa, 2019). Domestic water use accounts for 10% of the global water use and is expected to increase threefold in Africa and Asia by 2050, mainly contributed by provision of water supply services in urban areas (Boretti & Rosa, 2019). While the water demand is increasing, global wastewater and sludge production is notably high. The global production of municipal wastewater was estimated to be 315 – 330 km^3 per year, with the United States, China, Japan and India identified to be the most significant contributors (Mateo-Sagasta et al., 2015). Sewage sludge, which is a residual semi-solid material produced from wastewater treatment, also shows increasing production rates due to the higher demand and advancement of wastewater treatment technologies. Sewage sludge production rates are particularly high in high- and middle-income countries with high wastewater treatment coverage. The EU-27 countries, the United States, and China were the largest producer of sewage sludge, generating 8.9, 6.5 and 3.0 million dry metric tonnes of sewage sludge per year, respectively (Mateo-Sagasta et al., 2015). Municipal wastewater and sludge contain substantial amounts of organic matters, nutrients and energy, which are potentially valuable resources with environmental, economic and social benefits that could be recovered (Mateo-Sagasta et al., 2015). The advancements in technologies and growing demand for sustainability facilitate the paradigm shift in wastewater management towards treatment strategies with low impacts and energy recovery.

The concept of water-energy nexus has been developed from the recognition of the interdependency between water supply, wastewater treatment, energy consumption and energy recovery. In recent decades, municipal wastewater has been viewed as a renewable energy source, as the energy embedded in the biodegradable fraction could be extracted and recovered through anaerobic treatment to yield methane (CH_4) gas, which is a useful source of energy (McCarty et al., 2011). Typical municipal wastewater contains 500 mg/L organics, measured in chemical oxygen demand (COD). Assuming that 3.86 kWh of energy could be recovered from 1 kg of COD via oxidation, energy content available in domestic wastewater was estimated to be 1.93 kWh/m^3 (McCarty et al., 2011). Traditional wastewater treatment processes include preliminary, primary, secondary (biological) and tertiary treatment, in which energy recovery is unavailable. The aeration systems for supporting the microbial community in biological treatment reactors, such as activated sludge systems, are the most energy-intensive operation in conventional municipal wastewater treatment plants (WWTPs) (Pakenas, 1995; Pescod, 1992; Rosso et al., 2008). Switching from aerobic to anaerobic secondary treatment processes offers an alternative to enable energy recovery and reduce energy consumption of WWTPs simultaneously (McCarty et al., 2011). Anaerobic wastewater treatment technologies have been rapidly and maturely developed for full-scale applications. Anaerobic baffled reactor and anaerobic fluidized bed membrane bioreactor (AFMBR) are examples of anaerobic reactors that could recover energy from wastewater (Daverey et al., 2019). Combinations of various technologies, such as integrated applications of AFBR and partial nitritation AFMBR, have been used to enhanced energy recovery (Alvarado et al., 2020).

Parallel to the higher need for wastewater treatment and more stringent standards for effluent quality, global sludge production has been increasing dramatically (Øegaard, 2004). Municipal sewage sludge production rate was estimated to range from 50 to 90 g dry matters per person per day (Murray et al., 2008; Rulkens, 2008). In typical wastewater treatment processes, sewage sludge is comprised of primary sludge and surplus activated sludge (SAS) from primary and secondary clarifiers, respectively. Sewage sludge contains up to 95% of water content, nontoxic organic carbon, nutrients, toxicants and pathogens (Rulkens, 2008). Before the implementation of stricter environmental regulations by institutions, such as European Commission, landfilling and ocean disposal were possible sludge handling options (Tyagi & Lo, 2013). With increasing attention on the potential environmental and human health risks associated with sludge disposal, more advanced treatment processes have been adopted for sludge stabilization, toxicants and pathogens removal and energy recovery. Sludge treatment processes that enable energy and/or resource recovery include anaerobic digestion (AD), composting, incineration, gasification and pyrolysis (Tyagi & Lo, 2013). Sludge drying and dewatering are commonly adopted for reducing water content of sludge such that the transportation and disposal costs could be reduced.

Advanced municipal wastewater and sludge treatment technologies have been maturely developed and are available for full-scale application in WWTPs. Besides technical feasibility, the sustainability performance of different wastewater and sludge management strategies is critical to the overall sustainability of urban development. Selection of water supply and wastewater treatment systems with the most favorable environmental, economic and social performance for sustainable urban development is of great interest to decision makers. Evaluation frameworks for quantitatively evaluating the sustainability of urban wastewater treatment systems and providing comprehensive results to inform decision-making are demanded for.

Life-cycle assessment (LCA), a widely recognized approach for quantifying the environmental impacts of products and processes, is a common tool applied to wastewater and sludge treatment sector for revealing the environmental performance of different treatment technologies and management options. Besides the environmental aspect, economic performance of management options is also an essential factor that influences decisions. Economic analysis has been coupled with LCA in eco-efficiency analysis (EEA) (Lorenzo-Toja et al., 2016), data-envelopment analysis (DEA) (Dong et al., 2017; Gómez et al., 2018), environmental-economic analysis (Lim & Park, 2007) and other research studies (Murray et al., 2008) on evaluating wastewater treatment systems. Despite extensive research has been conducted to assist decision-making on selecting the most sustainable wastewater and sludge treatment options, comprehensive evaluation framework that includes the characteristics of urban areas has not been developed.

The contribution of this thesis is twofold: First, a sustainability evaluation framework was developed for effective decision-making in urban wastewater and sludge treatment systems. Second, the demonstration of sustainability evaluation framework in case studies provided comprehensive findings and policy implications that contributed to real-world decision-making.

1.2 Research questions

This thesis aims to inform decision-making on selecting the most sustainable management strategies for water supply, wastewater treatment and sewage sludge treatment in urban areas. This was achieved by the development of sustainability evaluation framework using LCA-based techniques, which was demonstrated in case studies of treatment scenarios in Hong Kong, as well as benchmarking of systems in urban cities worldwide. This is followed by proposing an innovative evaluation approach for assessing the social impacts of fecally contaminated water bodies with consideration of human behaviors. The research questions and scope of each case study are listed below.

(1) Eco-efficiency of urban sewage sludge treatment approaches (Chapter 3)

The environmental and economic performance of six sludge treatment scenarios in Hong Kong was investigated. The scenarios included sludge treatment processes and end-of-life handling approaches, including AD, dewatering, incineration with energy recovery, landfilling and reuse as substitute for cement clinker. Environmental impacts associated with material production, energy use and recovery, as well as transportation were included in the LCA, while capital, operation and maintenance (O&M) and transportation costs were included in economic evaluation.

This study addresses the following research questions:

(i) How, and to what extent, do the essential characteristics of urban cities influence the environmental and economic impacts of sewage sludge handling options?

(ii) What is the most environmentally and economically favorable sewage sludge treatment approach for dense urban cities?

(2) Eco-efficiency of non-potable water systems in domestic buildings (Chapter 4)

This study investigated the environmental and economic performance of four water supply systems for toilet flushing in domestic buildings using an EEA framework as demonstrated in a mock building in Hong Kong. The four systems evaluated include freshwater supply, seawater supply, greywater recycling using membrane bioreactor (MBR), and greywater recycling using AFMBR. The innovativeness of this study includes the provision of single and inclusive EEA results based on environmental and economic evaluations, inclusion of detailed engineering designs of piping and treatment units, and consideration of price variations of substitute good.

This study addresses the following research questions:

(i) In a coastal urban city where seawater supply for non-potable use is available, which non-potable water supply system (freshwater, seawater or recycled greywater) is the most environmentally and economically favorable option for domestic buildings?

(ii) What are the sources of environmental impacts in different non-potable water supply systems?

(iii) How do the lifetime and price of imported water influence the economic performance of different non-potable water supply systems?

(iv) How does the emphasis on environmental and economic performance affect the favorability of different non-potable water supply systems?

(3) Techno-environmental performance of sludge-to-energy systems (Chapter 5)

This study benchmarked the techno-environmental performance of sludge-to-energy (STE) systems in urban cities worldwide. Through the emphasis on the simultaneous roles of sewage sludge handling and energy systems of STE systems, this study included influencing factors of technical and environmental efficiency for the evaluation of the overall techno-environmental performance. Besides revealing the techno-environmental performance, the study also aims to identify the areas and targets for improvements required for the relatively inefficient systems.

This study addresses the following research questions:

(i) How well is the overall techno-environmental performance of STE systems, with inclusion of all essential factors that affect efficiency, in urban cities worldwide?

(ii) What are the improvement targets for the relatively inefficient STE systems to become efficient?

1.3 Research methodology

The methodology of this thesis is depicted in Figure 1-1. LCA-based evaluation techniques were used in all studies included in this thesis. LCA was the core technique for assessing the environmental performance of water and wastewater systems. To address the research questions and achieve the goal of each study, evaluation framework was developed through the integration of LCA with other methods, such as life-cycle costing (LCC) and DEA, for obtaining more comprehensive results to effectively assist decision-making.

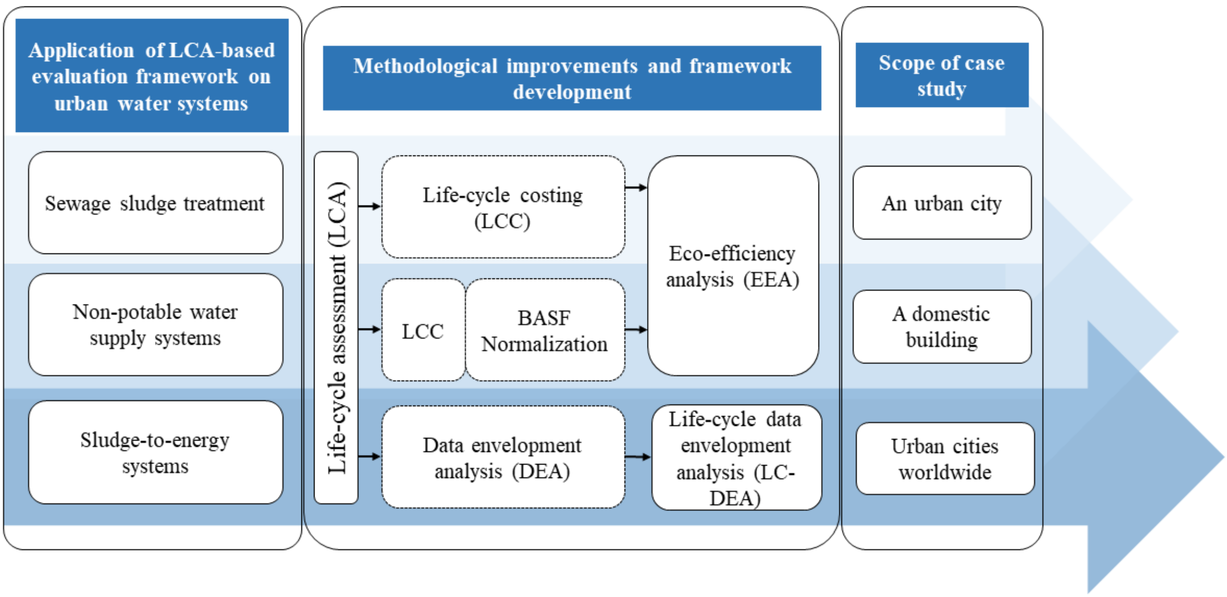

Figure 1-1 Methodology of thesis

To evaluate the eco-efficiency of urban sludge management options, an EEA framework was developed through the integration of LCA and LCC methods (Chapter 3). The EEA framework covers both the environmental and economic aspects of sewage sludge treatment processes, thus the performance of management options in these two aspects could be revealed. This study contributed to improvements in evaluation framework for urban sludge management by including essential factors that influence eco-efficiency of urban sludge treatment. Land resources are scarce and valuable in many urban cities, thus could have notable contributions to the eco-efficiency of different sludge treatment options. Another important factor to be included in evaluation is the actual transportation distances. Previous studies assumed a constant distance for sludge transportation between treatment facilities (Murray et al., 2008; Xu et al., 2014), which could lead to large deviations in estimated emissions from the actual values. To enhance comprehensiveness, this study developed an eco-efficiency evaluation framework for urban sludge management through the inclusion of real-world data of high land costs and transportation distances.

To investigate the eco-efficiency of different water supply systems for toilet flushing, an EEA framework was developed by integrating LCA and LCC using the BASF and normalization methods (Kicherer et al., 2007; Saling et al., 2002) (Chapter 4). The life-cycle environmental and economic inventories were fully backed by the designs of water supply, recycling and sewage systems. The EEA framework in this study provides clear and inclusive results in form of eco-efficiency portfolios which could be easily understood and used by decision makers. Relative emphasis on the environmental and economic aspects could be adjusted in the EEA by decision makers and other tool users based on the goals of policy formulation.

Considering the simultaneous role of contemporary sludge handling processes as energy systems, a life-cycle data envelopment analysis (LC-DEA) framework was developed by integration of LCA and DEA for benchmarking the efficiency of STE systems in urban cities worldwide (Chapter 5). The LC-DEA framework developed in this study is able to consider the technical efficiency, in terms of electricity/heat generation per unit of energy consumption and organic matters removal, as well as the environmental performance of the systems. The approach fully reflected the shift of sludge management philosophy from waste problems to waste-to-energy processes. The advantage of DEA enables the LC-DEA framework to evaluate performance metrics of STE systems measured in different units. LC-DEA results give a big picture of the relative techno-environmental performance of STE systems worldwide. The findings also provide recommendations on targets for improvements on relatively inefficient systems based on the best practice available in real-world.

1.4 Dissertation outline

After presenting in this chapter the background, research questions and scope of study, the subsequent chapters are organized as follows: Chapter 2 outlines a detailed review of literatures on LCA and other sustainability evaluation tools for water and wastewater treatment systems. Chapter 3 describes an EEA framework for evaluating urban sewage sludge treatment approaches, demonstrated in a case of sludge management in Hong Kong. Chapter 4 describes an EEA framework for evaluating non-potable water supply systems in urban cities, demonstrated in a mock building in Hong Kong. Chapter 5 describes an LC-DEA framework for evaluating the techno-environmental performance of urban STE systems, demonstrated in the application for evaluating sixteen STE systems worldwide. Chapter 6 summarizes the conclusions, contributions, policy implications, and proposed future research studies.

CHAPTER 2

LITERATURE REVIEW

2.1 Sustainability evaluation framework for urban sewage sludge treatment

To promote sustainability, wastes should be managed in an economically affordable, environmentally efficient and socially acceptable manner. LCA is a suitable tool to facilitate the development of sustainable waste management systems (Thomas & McDougall, 2005). The earliest concept of LCA emerged from energy analysis studies in the late 1960s and early 1970s. From 2002 to 2005, the Society of Environmental Toxicology and Chemistry (SETAC) published reports of their work on harmonizing the diverse frameworks and improving the LCA methodology. With the desire to codify the LCA methodology, standards for the LCA principle and requirements were specified in the International Organization for Standardization (ISO) 14000 series (ISO 14040, 2006; ISO 14044, 2006). ISO 14040 and 14044 provide a general framework without specifications for applications of LCA (Corominas et al., 2013). LCA studies have been conventionally conducted on products, but it is now gaining popularity as a tool for investigating the sustainability of different systems (Guinée et al., 2011), such as waste management and water management, by striking a balance between economic growth and environmental conservation (Chang et al., 2014). The authorities in Gipuzkoa, Spain, chose LCA as an environmental tool for decision-making, and the findings of the LCA case study on waste management planning in Gipuzkoa demonstrated a success (Muñoz et al., 2004). A research study conducted by Romero-Hernandez (2005) revealed the benefits that policy makers can gain from implementing LCA on wastewater treatment processes and suggested the application of environmental tools to optimize treatment technologies using an evaluation of economic and environmental performance.

To provide comprehensive information and guidance for decision-making, LCA has rapidly developed as a sludge management tool for evaluating the lifetime performance of sludge treatment processes. Previous studies have been conducted at divergent scopes and scales under the flexible framework of LCA (Yoshida et al., 2013). Early in 2000, a life-cycle approach for evaluating the sustainability of sludge reuse options was suggested (Bridle & Skrypski-Mantele, 2000). A few of the LCA studies have placed addition focus on specific treatment processes, such as the land application of anaerobically digested sludge (Hospido et al., 2010) and sludge treatment wetlands (Uggetti et al., 2011). Other studies compared the performance of various treatment technologies (Bridle & Skrypski-Mantele, 2000; Lundin et al., 2004). Sludge management scenarios that consisted of several treatment processes were evaluated in numerous studies. Murray et al. (2008) and Foley et al. (2010) analyzed the life-cycle inventories of the scenarios. Foley et al. (2010) carried out a study to reveal the life-cycle inventories of wastewater treatment scenarios without assessing the trade-offs between enhanced nutrient removal and environmental impacts. Conventional LCA that only focused on environmental consequences was conducted to analyze the resource consumption and environmental emissions associated with sludge handling processes (Houillon & Jolliet, 2005; Suh & Rousseaux, 2002).

Over the past years, LCA has been applied in a number of studies on wastewater treatment, but a mature framework designed specifically for compact urban cities has not yet been developed. For example, the LCIA study conducted by Suh & Rousseaux (2002) and Houillon & Jolliet (2005), excluded the land occupation impact, which has a crucial impact in compact cities. Hong et al. (2009) and Xu et al. (2014) included the impact of land use and excluded the operating costs and capital costs of infrastructures in their studies. Murray et al.

(2008) and Xu et al., (2014) assumed that the transportation distances between the treatment facilities were 25 km and 40 km, respectively. The assumptions could lead to deviations of the estimated atmospheric emissions associated with transportation from the actual values. Hospido et al. (2010) conducted an environmental assessment on the agricultural application of reused sludge, which has a restricted significance for urban sludge management because of the limited agricultural activities in urban areas.

To provide a more practical and comprehensive urban sludge management solution, the economic cost of the treatment scenarios was included in EEA using the life-cycle cost (LCC) approach. To conduct an EEA, the environmental impacts are evaluated by the LCA methodology (Saling et al., 2002) and combined with economic analysis using LCC approach (Kicherer et al., 2006). LCC methodology was adopted in addition to the traditional LCA in previous studies on sludge management. Hong et al. (2009) evaluated the global warming potential and costs of six sewage sludge treatment scenarios in Japan. Lundin et al. (2004) assessed four sewage sludge recycling and disposal options in terms of the environmental and economic aspects of sustainability. A life-cycle environmental and economic inventory was established by Murray et al. (2008) for sewage sludge treatment and end-use scenarios to facilitate informed decision-making towards sustainability. Uggetti et al. (2011) compared the technical, environmental and economic performance of sludge treatment wetlands with other sludge treatment alternatives. Xu et al. (2014) conducted LCA and LCC for assessing environmental and economic performance, respectively, of thirteen sludge treatment scenarios in China.

Based on the literature review, it was observed that most LCA studies on sludge management did not include land occupation impact, which is essential for urban cities. Emissions from transportation were commonly estimated based on single assumptions of transportation distances (Murray et al., 2008; Xu et al., 2014). The inclusion of land occupation and enhanced estimations of transportation emissions is demanded for the environmental and economic evaluation framework on urban sludge management.

2.2 Sustainability evaluation framework for greywater recycling systems for non-potable purposes

In order to assist decision-making on water management, numerous studies have been conducted to evaluate the performance of greywater reuse systems. Most studies have focused on decentralized greywater recycling systems due to the low proportion of greywater flow relative to total water consumption (average greywater:consumption ratio = 0.6) (Ghaitidak & Yadav, 2013) and the significant resource and cost burdens for the collection and conveyance process (Al-Jayyousi, 2003; Ghaitidak & Yadav, 2013; Hendrickson et al., 2015). Previous research studies comparing the environmental performance of centralized and decentralized greywater reuse systems concluded that decentralized systems are more advantageous than a centralized system in terms of energy consumption and carbon emissions (Hendrickson et al., 2015; Matos, Pereira, Amorim, Bentes, & Briga-Sá, 2014). Some studies evaluated the economic aspect of the greywater reuse systems. Friedler & Hadari (2006) investigated the economic feasibility of the rotating biological contactor and membrane bioreactor (MBR) greywater treatment systems and found that the systems would be economically feasible when the building sizes reach seven and forty stories, respectively. In the decision-making model developed by Henriques & Louis (2011), economic and financial factors were considered for prioritizing the greywater reuse and drinking water supply systems. Few studies have considered a wider scope that include the impacts of greywater recycling on municipal sewer systems through a modelling approach. The applications of on-site greywater reuse systems decrease the velocity and quantity of the flow in sewer systems but present only trivial impacts on the sizes of sewer pipes (Penn, Schütze, & Friedler, 2013). A multi-objective optimization model has been established to search for the optimal distribution of different types of greywater reuse in connection to existing sewer systems (Penn, Friedler, & Ostfeld, 2013). The study results revealed that higher flow velocity enables the maximization of water savings while reducing treatment system costs and drinking water demand.

To comprehensively evaluate the eco-efficiency of greywater recycling systems, an integrated environmental and economic evaluation tool, with the inclusion of detailed engineering design of the freshwater, greywater and wastewater systems, is needed.

2.3 Sustainability evaluation framework for sludge treatment and energy systems

The characteristics of DEA enable an objective determination of weightings between the evaluated inputs and outputs, thus giving DEA supremacy in accommodating the different factors involved in performance evaluation, such as energy use, environmental emissions and waste generation, without the need of subjective weights (Charnes et al., 1978; Kuosmanen & Kortelainen, 2005). DEA is one of the commonly adopted tools for analysing WWTPs (Torregrossa et al., 2018), water-energy-food nexus (Dai et al., 2018) and energy systems (Martín-Gamboa et al., 2017). The incorporation of DEA with LCA has been proposed in literatures for estimating the relative efficiency of the decision-making units (DMUs), which are the units of assessment that represent homogeneous entities with the same function, based on the benchmarks defined through DEA (Iribarren et al., 2010, 2014; Lorenzo-Toja et al., 2015). Combined LCA with DEA has been conducted to benchmark the environmental efficiency (Lorenzo-Toja et al., 2015) and detect daily eco-efficiency of WWTPs (Torregrossa et al., 2018). Such integrated LCA and DEA approach has also been used to evaluate the environmental performance of electricity fuel mixes in European countries (Ewertowska et al., 2016), sustainability of electricity generation technologies in the UK (Galán-Martín et al., 2016) sustainability of different biodiesel production (Ren et al., 2013) and biohydrogen production alternatives (Martín-Gamboa et al., 2016), as well as to benchmark the efficiency of wind farm for electricity generation (Iribarren et al., 2014; Martín-Gamboa & Iribarren, 2016). Despite the numerous research studies conducted using combined LCA and DEA to evaluate the performance of WWTPs and energy systems, a lack of inclusive benchmarking approach that fully covers the influencing factors on the efficiency of STE systems, such as treatment level, energy balance, environmental emissions, material consumption and waste generation, is observed.

CHAPTER 3

ECO-EFFICIENCY OF URBAN SEWAGE SLUDGE TREATMENT APPROACHES

3.1 Introduction

Continuous global population growth and advancements in wastewater treatment systems have caused a significant increase in sewage sludge production worldwide. Municipal wastewater sludge contains pathogens, toxicants and heavy metals, thus poses potential hazards to human and the natural environment. Early in 1991, the recycling of sludge was encouraged by the European Union (EU) and sludge disposal to surface water was banned in 1998 (EEC, 1991). According to Fytili & Zabaniotou (2008), the sewage sludge production in the EU has been growing by 50% per year since 2005 due to the implementation of the Urban Waste Water Treatment Directive (UWWTD); and the sludge generation rates of EU members such as Italy and France in 2020 were predicted to be 1,500 Mt, 1,600 Mt of dry solids (DS) per annum respectively (Milieu Ltd. et al., 2010). In the US, sludge is generated at a rate of 6.2 dry Mt annually and continuous increase of the generation rate was expected (Kargbo, 2010). The proportion of sludge used for agricultural application is approximately 50% in both the EU and the US (Milieu Ltd. et al., 2010; USEPA, 2014). In China, the current annual sludge production of over 20 Mt was expected to increase to more than 30 Mt due to urbanization and the escalating load of wastewater treatment plants (MOHURD & NDARC, 2011). Processes adopted in China for sludge treatment include drying, thickening, dewatering, AD, incineration and composting; and the potential final destinations are agricultural application and landfill (Xu et al., 2014). Direct disposal of untreated sewage sludge has been reported in China, posing a high risk of soil, atmospheric and water pollution (Yang et al., 2012). With the recognition of the disastrous environmental and health risks, stringent sludge handling and disposal management is necessary. To provide sound evidence for strategic sludge management decisions in urban cities, an evaluation for urban sludge handling is needed for the evaluation of both the environmental and economic aspects with the inclusion of the characteristics of urban cities.

3.2 Method and data

3.2.1 Goal and scope definition

The primary goal of this study is to develop an EEA framework that is suitable for sludge management in urban cities. Characteristics of urbanized areas such as limited land areas and high land costs were considered in the EEA framework for urban sludge management in this study, using Hong Kong as an example. The impacts of transportation were estimated based on actual transportation information. Another goal of this research study is to assist decision makers in choosing the most environmentally and economically favorable sludge treatment approach for adoption in Hong Kong. Based on the actual conditions in Hong Kong, this study evaluated the environmental and economic consequences of six sludge treatment scenarios, with the aim of informing decision-making on sludge management in the city.

The six scenarios, which were defined based on actual practices and conditions, involved different combinations of treatment processes used in Hong Kong (Figure 3-1). As dewatering is a necessary process to treat sewage sludge, it was included in all scenarios and the method adopted is mechanical dewatering. In scenarios S1, S3 and S5, raw sludge is treated by AD prior to dewatering (Appendix Fig. A1-3 and A1-5) according to the real practice in the four STWs studied; treatment options without AD (Appendix Fig. A1-5) were set in scenarios

S2, S4 and S6 for comparison as most of the STWs in Hong Kong do not apply AD for sludge treatment. The sludge handling practices which have been exercising in Hong Kong are represented by scenarios S1 and S2, while the treatment processes that will be in use after the full commissioning of STF are represented by scenarios S3 and S4 (Appendix Fig. A1-7 and A1-9). Since sludge ash utilization in cement production has been investigated in previous research studies (Houillon & Jolliet, 2005; C. H. Lam et al., 2010; Murray et al., 2008), such alternative was included in scenarios S5 and S6 to explore its economic and environmental feasibility. In such scenarios, the sludge ash utilization was considered as material substitution for the clinker raw materials (Lam et al., 2010), in which no extra facility and operation requirement was added. Because AD and dewatering processes are carried out in the same STWs, no transportation is required between the two stages. For transportation from individual STW to landfill or STF, transportation distances between STWs and the nearest landfill site or STF were used in calculation. The time horizon of this study was defined as 30 years of operation of the facilities.

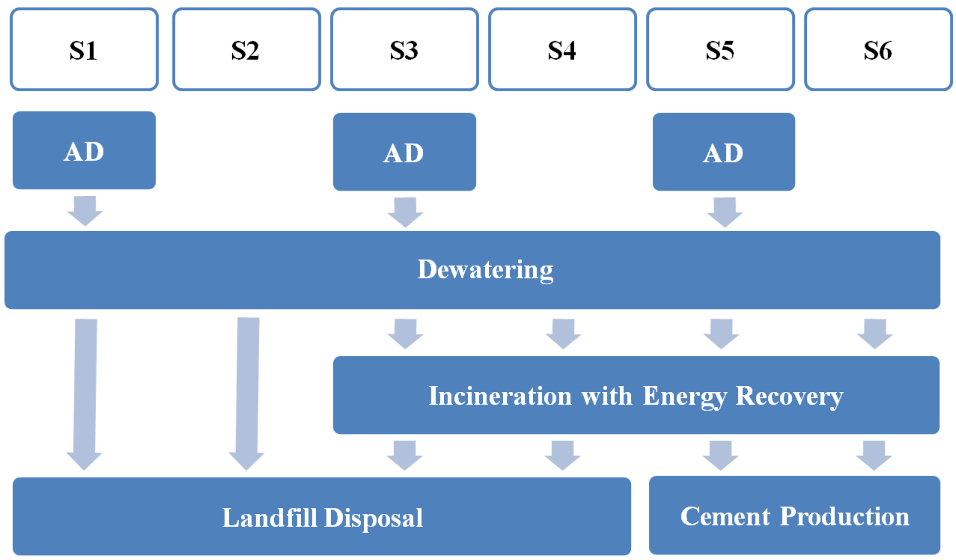

Figure 3-1 Six sludge treatment scenarios defined as the scope for LCA

A functional unit (FU) is the essential basis that enables the comparison and analysis of alternative goods and services (Rebitzer et al., 2004). Time-based FUs, which define the operational period of the facility, were used in the studies of Murray et al. (2008) and Foley et al. (2010). A volume-based approach was adopted in the study conducted by Hospido et al. (2010). A volume unit has been most frequently used in wastewater LCA, yet it is not necessarily representative because it cannot reflect the sewage characteristics (Corominas et al., 2013). Yoshida et al. (2013) showed that mass-based FUs have been applied most commonly to sewage sludge management. In this study, the FU was defined as one tonneof dry solids in raw sewage sludge, which has also been used in previous research (Hong et al., 2009; Houillon & Jolliet, 2005; Lundin et al., 2004; Rebitzer et al., 2004; Xu et al., 2014). As the mass-based FU does not completely reflect the conditions of sludge treatment, such as the influent quality and treatment efficiency, details of the processes were obtained and specified in the later parts of this study.

3.2.2 Eco-efficiency analysis framework

An EEA framework which includes LCA and LCC analyses was developed in this study. The boundary of the EEA framework covers all the processes that contribute significantly to the products or activities (Rebitzer et al., 2004). Actual operational information, emission factors from literatures and an economic input-output life-cycle assessment (EIO-LCA) tool (CMU, 2006) was used in LCA to evaluate the emissions and environmental impacts of the scenarios. Construction, O&M and transportation costs were estimated in economic analysis to evaluate the economic performance of the scenarios.

3.2.3 Life-cycle assessment

Emissions from material production, electricity balance and vehicles have been commonly considered in LCA (Lundin et al., 2004; Murray et al., 2008; Xu et al., 2014). Energy consumption and atmospheric emissions were included in the LCA. The emissions associated with electricity consumption, energy recovery from anaerobic digestion, incineration and landfilling, chemical production and fuel consumption were estimated in this research study (Appendix Part 2A). The emissions released from the construction phase of the infrastructures were excluded because such emissions have negligible contributions to the overall environmental impact (Hong et al., 2009).

Life-cycle Impact Assessment (LCIA) is the phase in which the life-cycle inventory results are processed and interpreted as environmental impacts. The aim of LCIA is to develop relative comparisons of the environmental or human health effects between the different scenarios concerned, instead of investigating the absolute damage to the environment and human health (SAIC & US EPA, 2006). Comprehensive multi-criteria LCIA, rather than mono-criterion evaluation (such as carbon footprint), has been more commonly adopted in current LCAs because the shifts of pollution can still be recognized using the former approach (Loubet et al., 2014). In this study, five life-cycle impact categories were defined: land occupation, climate change, human toxicity, acidification and eutrophication. Land occupation is a subcategory of land use impacts that considers the temporary unavailability of land as a loss of resources. Climate change is defined as the impact of anthropogenic emissions on the absorption of heat radiation by the atmosphere, which is commonly referred to as the "greenhouse effect". Adverse impacts on ecological health, human health and properties may result from climate change. The effects on human health caused by the presence of toxic materials in the surroundings were included in the human toxicity category. Acidification was primarily attributed to acidifying atmospheric emissions, including sulfur dioxide (SO_2), nitrogen oxides (NO_x) and ammonia (NH_3), which is converted to sulfuric acid and nitric acid after chemical reactions with moisture in the air or rainwater. Aquatic organism mortality, vegetation growth reduction and damage of materials are potential consequences of acidification. Eutrophication is the impact caused by excessive macronutrients. Depressed oxygen levels due to high biological oxygen demand (BOD) is a potential consequence of algal bloom, mortality of organisms and bacteria growth in aquatic habitats. Undesirable alterations to the composition of the ecological community and increased biomass production are the possible consequences of nutrient enrichment. Relevant stressors, which are the environmental releases or conditions that may contribute to the impacts, were identified and linked to the impact categories (Table 3-1).

Table 3-1 Life-cycle impact categories and relevant stressors

Impact Categories	Stressors
Land Occupation	Land area requirement
Climate Change	CH_4, CO_2, N_2O
Human Toxicity	NO_x, NH_3
Acidification	SO_2, NO_x, NH_3
Eutrophication	NO_x, NH_3

Characterization, which is the step that follows the definition of the impact categories and classification, models the potential environmental impacts using science-based conversion factors. The land occupation indicator is the multiplicative product of the land area requirements and the occupation duration, and the characterization factor for all land equals 1. The indicator for climate change expresses the levels of the greenhouse effect that were caused by the identified greenhouse gases (GHGs), and CO_2 was used as the reference GHG for the global warming potentials. Human toxicity impact was investigated and expressed in kg 1,4-dichlorobenzene equivalent. Generic acidification potential factors were used to characterize the acidifying emissions to the air, and the results were expressed in kg SO_2-equivalent. Generic eutrophication potential factors were used to convert the relevant environmental releases to kg PO_4^{3-}equivalent.

The characterized impact assessment results was normalized using a set of normalization factors (Dong & Ng, 2014) presented in Table 3-2 so that the different impact categories could be included in the assessment in a comparable manner. The normalized environmental impact assessment results would be presented as a single score to reveal the overall environmental performance of the sludge treatment scenarios.

Table 3-2 Normalization factors for the environmental impact categories

	Normalization Factor (person·year/kg)
Land Occupation	1.30E-03
Climate Change	1.38E-04
Acidification	2.59E-02
Human Toxicity	8.90E-03
Eutrophication	3.38E+00

3.2.4 Life-cycle cost analysis

Capital costs, O&M costs, and transportation costs were analyzed in the hybrid LCA conducted by Murray et al. (2008), while Lundin et al. (2004) only included the former two costs. Costs of electricity consumption, energy recovery, maintenance, materials, labor and equipment were considered in the economic assessments performed by Hong et al. (2009) and Xu et al. (2014). In this study, the construction costs of sludge treatment facilities and equipment, O&M costs, and transportation costs of the six defined scenarios were investigated. The former two components were estimated based on the guidelines given in the Handbook Estimating Sludge Management Costs (USEPA, 1985). The transportation costs were calculated using information on truck capacities and travel distances provided by the Hong Kong Drainage Services Department (DSD), as well as the price of diesel. The lifetime economic costs of the six scenarios (Appendix Part 1) over a 30-year time horizon and with 6.6% discount rate (Census and Statistics Department HKSAR, 2014) were presented in present values (PVs).

3.2.5 Data: sludge management in Hong Kong

Sludge is an unavoidable by-product of water and wastewater treatment processes. According to the information provided by DSD, Hong Kong will generate nearly 30,000 m^3 of sludge per day (EPD, 2008a) when the Harbour Area Treatment Scheme (HATS) Stage 2A is fully commissioned. All sewage sludge generated is mechanically dewatered in individual sewage treatment works (STWs) (ACE, 1999), and only sludge produced in the four major secondary STWs (Sha Tin, Tai Po, Shek Wu Hui and Yuen Long) undergoes anaerobic digestion (DSD, 2014). To explore the feasibility of sludge composting, sewage sludge is composted in a pilot study at the Ngau Tam Mei Animal Waste Composting Plant (EMSD, 2009). Landfills are the only destinations of sludge waste in Hong Kong. The current practice of co-disposal with construction wastes and municipal solid wastes (MSW) in the ratio of 1:10 is predicted to be unsustainable (EPD, 2008b); therefore a sludge treatment facility (STF) has been constructed. The STF, which is located in Tsang Tsui, Tuen Mun, uses fluidized-bed incineration technology for high-temperature combustion of sludge (EPD, 2005). To demonstrate the EEA framework developed in this study, the appropriateness of various wastewater sludge treatment options adopted in Hong Kong were examined, and the performance of six treatment scenarios was evaluated using EEA.

The environmental and economic performance of the six sewage sludge treatment scenarios applied on the four major secondary sewage treatment works, namely Sha Tin, Tai Po, Shek Wu Hui and Yuen Long STWs, were evaluated. DSD is the only governmental authority responsible for the provision of sewage treatment services in Hong Kong. Wastewater is treated in the STWs operated by the DSD prior to discharge, and the sewage sludge generated is treated on-site in the corresponding STWs. The specific information on the sludge treatment in the four STWs mentioned above was obtained from the DSD. Table 3-3 shows the data for the STWs in 2013.

Table 3-3 Sewage sludge information for the four major sewage treatment works

	Shatin	**Yuen Long**	**Shek Wu Hui**	**Tai Po**
Raw Sludge				
Daily volume (m3)	1,620	312	844	571
Percent dry solids	4.0%	3.5%	3.5%	3.0%
Percent volatile solids	66%	56%	85%	61%
Anaerobic Digestion (AD)				
Percent dry solids after AD	2.9%	2.0%	2.2%	2.5%
Percent volatile solids after AD	50%	43%	76%	59%
Percent volatile solids that can be converted into CH4, CO2 and H2O during AD	43%	77%	47%	41%
Solid retention time	10 days	N.A.	24 days	18 days
Volume of CH4 production (Volume of Biogas production) (m3)	5,600,000	616,820	1,200,000	2,000,000
Dewatering				
Method of dewatering adopted	By Centrifuges	Filter Press	Membrane Filter Press	Membrane Filter Press
Percent dry solids after dewatering	31%	33%	31%	30%
Type of chemicals added for conditioning	Polyelectrolyte	Polyelectrolyte , Ferric Chloride	Polyelectrolyte , Ferric Chloride	Polyelectrolyte
Operation				
Operation hours per day for AD	24	24	16	16
Operation hours per day for dewatering	24	8	16	16
Operation day per year for AD	365	365	365	~300
Operation day per year for dewatering	365	326	365	~300
Transportation				
Final destination of sludge	SENT	NENT	NENT	SENT
Distance of transporting	38	24	8	29
Volume of truck (m^3)	13	20	12	12

3.3 Results

3.3.1 Life-cycle assessment results

3.3.1.1 Electricity balance

The electricity consumptions (Appendix Table A1-48) of the six scenarios are presented in Figure 3-2. Scenarios except S2 achieve energy positive operation by energy recovery from methane production in AD, heat energy recovery in sludge incineration and offset from clinker substitution in cement production, as well as trivial energy recovery from landfill gas. The dewatering process and maintenance of high combustion temperature by auxiliary fuel in incinerator are energy demanding. The net energy consumption of the sludge handling scenarios in ascending order is S1 < S5 < S3 < S2 < S6 < S4.

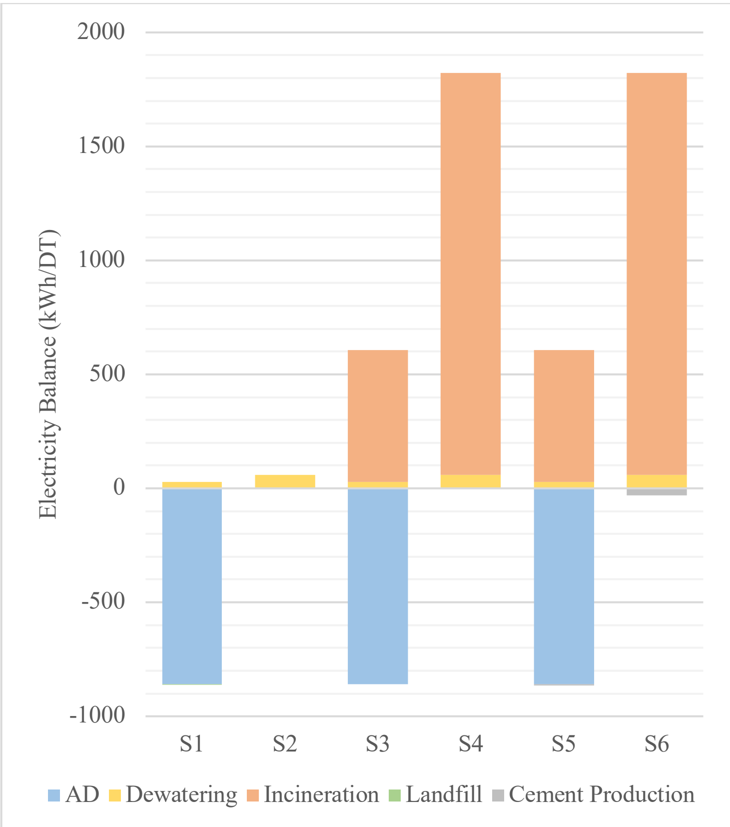

Figure 3-2 Electricity balance (kWh/DT) of sludge treatment processes

3.3.1.2 Atmospheric emissions

Air emissions from the sewage sludge treatment processes are listed in Table 3-5 (Appendix Part 2A). Emission of greenhouse gases (GHGs) including CO_2, CH_4 and N_2O is the most significant in amount among the other atmospheric emissions, followed by the less significant NO_x and SO_2 emissions. Release of NH3, particulate matters (PM) and volatile organic compounds (VOC) shows relative insignificance generally.

3.3.1.3 Life-cycle Impact Assessment (LCIA)

Five life-cycle environmental impact categories (land occupation, global warming potential, human toxicity, acidification and eutrophication) were assessed (Appendix Part 2B), and the results are presented in this section (Table 3-6). The final disposal of sewage sludge at landfill sites in all scenarios was the primary contributor to land occupation. The highest degree of land occupation by landfill disposal was shown in S2 and S4 in all of the four STWs, while S5 and S6 do not contribute to such impact as the final product of the treated sludge is used in clinker substitution in cement production. The emission offset from energy recovery and material substitution have been included in environmental impact evaluation. Scenarios with AD application (S1, S3 and S5) have lower adverse impact than those without AD (S2, S4 and S6). For the comparison of environmental impacts measured in different units, normalization of LCIA results was conducted and normalized results are shown in Table 3-7. The ascending order of the normalized environmental impact of the scenarios is S5 < S3 < S1 < S6 < S4 < S2.

Table 3-4 Atmospheric emissions (kg/DT) inventory of sludge treatment scenarios in the four STWs

	Shatin STW						Yuen Long STW					
	S1	S2	S3	S4	S5	S6	S1	S2	S3	S4	S5	S6
GHG	5.68E+02	4.80E+03	8.50E+02	5.85E+03	8.42E+02	5.84E+03	1.31E+04	4.44E+04	1.34E+04	4.55E+04	1.33E+04	4.55E+04
NOx	2.21E-01	7.87E+00	6.49E+00	1.41E+01	6.48E+00	1.41E+01	2.55E+01	8.78E+01	3.18E+01	9.41E+01	3.18E+01	9.40E+01
SO2	-3.82E-01	7.66E+00	7.61E+00	3.52E+01	7.56E+00	3.49E+01	4.37E+01	1.49E+02	5.17E+01	1.77E+02	5.16E+01	1.77E+02
NH3	9.69E-02	4.71E-01	9.69E-02	4.71E-01	9.69E-02	4.71E-01	1.23E+00	4.20E+00	1.23E+00	4.20E+00	1.23E+00	4.20E+00
PM10	4.71E-01	2.19E+00	4.71E-01	2.19E+00	4.71E-01	2.19E+00	1.35E+01	4.50E+01	1.35E+01	4.50E+01	1.35E+01	4.50E+01
PM2.5	1.94E-01	8.73E-01	1.94E-01	8.73E-01	1.94E-01	8.73E-01	5.10E+00	1.70E+01	5.10E+00	1.70E+01	5.10E+00	1.70E+01
VOC	8.18E-01	3.74E+00	8.17E-01	3.74E+00	8.17E-01	3.74E+00	6.75E+00	2.24E+01	6.75E+00	2.24E+01	6.75E+00	2.24E+01
	Shek Wu Hui STW						Tai Po STW					
	S1	S2	S3	S4	S5	S6	S1	S2	S3	S4	S5	S6
GHG	3.38E+04	7.76E+04	3.42E+04	7.88E+04	3.42E+04	7.88E+04	5.81E+02	4.04E+03	8.93E+02	5.18E+03	8.90E+02	5.17E+03
NOx	6.64E+01	1.54E+02	7.27E+01	1.61E+02	7.27E+01	1.61E+02	7.47E-03	6.67E+00	6.30E+00	1.30E+01	6.29E+00	1.30E+01
SO2	1.13E+02	2.63E+02	1.21E+02	2.90E+02	1.21E+02	2.90E+02	-8.13E-01	6.50E+00	7.18E+00	3.40E+01	7.13E+00	3.38E+01
NH3	3.28E+00	7.42E+00	3.28E+00	7.42E+00	3.28E+00	7.42E+00	1.05E-01	4.20E-01	1.05E-01	4.20E-01	1.05E-01	4.20E-01
PM10	3.48E+01	7.92E+01	3.48E+01	7.92E+01	3.48E+01	7.92E+01	5.26E-01	1.84E+00	5.25E-01	1.84E+00	5.25E-01	1.84E+00
PM2.5	1.32E+01	2.99E+01	1.32E+01	2.99E+01	1.32E+01	2.99E+01	2.10E-01	7.36E-01	2.11E-01	7.37E-01	2.11E-01	7.37E-01
VOC	1.73E+01	3.93E+01	1.73E+01	3.93E+01	1.73E+01	3.93E+01	9.47E-01	3.10E+00	9.46E-01	3.10E+00	9.45E-01	3.10E+00

Table 3-5 Life-cycle impacts on land occupation, climate change, human toxicity, acidification and eutrophication of the sludge treatment scenarios

	Land Occupation (Acre•yr/DT)				Climate Change (kg-CO2 eq/DT)			
	ST	YL	SWH	TP	ST	YL	SWH	TP
S1	5.61E+03	4.12E+03	1.14E+04	6.70E+03	5.68E+02	1.31E+04	3.38E+04	5.81E+02
S2	2.59E+04	2.41E+04	2.59E+04	2.68E+04	4.80E+03	4.44E+04	7.76E+04	4.04E+03
S3	5.61E+02	4.12E+02	1.14E+03	6.70E+02	8.50E+02	1.34E+04	3.42E+04	8.93E+02
S4	2.59E+03	2.41E+03	2.59E+03	2.68E+03	5.85E+03	4.55E+04	7.88E+04	5.18E+03
S5					8.42E+02	1.33E+04	3.42E+04	8.90E+02
S6					5.84E+03	4.55E+04	7.88E+04	5.17E+03

	Acidification (kg SO2 eq./DT)				Human Toxicity (kg 1,4-DCB eq./DT)			
	ST	YL	SWH	TP	ST	YL	SWH	TP
S1	-4.56E-02	6.39E+01	1.66E+02	-6.10E-01	2.75E-01	3.07E+01	8.01E+01	1.95E-02
S2	1.41E+01	2.19E+02	3.85E+02	1.20E+01	9.49E+00	1.06E+02	1.86E+02	8.04E+00
S3	1.23E+01	7.63E+01	1.78E+02	1.18E+01	7.80E+00	3.83E+01	8.76E+01	7.56E+00
S4	4.59E+01	2.51E+02	4.17E+02	4.39E+01	1.70E+01	1.13E+02	1.94E+02	1.56E+01
S5	1.23E+01	7.62E+01	1.78E+02	1.17E+01	7.79E+00	3.83E+01	8.76E+01	7.56E+00
S6	4.57E+01	2.50E+02	4.16E+02	4.36E+01	1.70E+01	1.13E+02	1.94E+02	1.56E+01

	Eutrophication (kg PO43- eq./DT)			
	ST	YL	SWH	TP
S1	8.99E-02	9.09E+00	2.37E+01	1.63E-02
S2	2.81E+00	3.13E+01	5.50E+01	2.39E+00
S3	2.29E+00	1.13E+01	2.59E+01	2.22E+00
S4	5.01E+00	3.35E+01	5.72E+01	4.60E+00
S5	2.28E+00	1.13E+01	2.59E+01	2.21E+00
S6	5.00E+00	3.35E+01	5.72E+01	4.59E+00

Table 3-6 Normalized life-cycle impacts of the sludge treatment scenarios

	S1	S2	S3	S4	S5	S6
Land occupation	9.03	33.32	0.90	3.33	-	-
Climate change	1.66	4.53	1.71	4.68	1.71	4.68
Acidification	1.48	4.08	1.81	4.90	1.80	4.90
Human toxicity	0.25	0.69	0.31	0.76	0.31	0.76
Eutrophication	27.80	77.35	35.24	84.82	35.23	84.78
Overall	40.23	119.96	39.97	98.49	39.06	95.11

3.3.2 Life-cycle cost analysis results

Table 3-4 presents the total costs of the sludge treatment processes in the four STWs studied (Appendix Part 1). AD adopted in S1, S3 and S5 and sewage sludge ash utilization in cement production in S5 and S6 contribute to the economic benefits of \$52 M (million) USD and \$0.1 – 0.6 M USD respectively. In the scenarios with AD application, the dewatering process costs \$9 – 11 M USD and the incineration stage costs \$19 M USD; whereas in scenarios without AD, the costs are twofold and fivefold higher respectively. The landfill disposal costs in S3 and S4 with incineration are one-tenth of the landfill costs in S1 and S2 without incineration. The landfill costs of scenarios with AD (S1 and S3) are 25% of those without AD (S2 and S4) mainly due to the 70% volume reduction achieved by the AD process. However, the landfill cost reduction is not exactly equal to the volume reduction because of cost components other than land cost, such as the costs of grading earthwork, monitoring wells and excavation equipment, included in the total landfill cost. The total economic costs of the six scenarios in ascending order are S5 < S3 < S6 < S4 < S1 < S2.

Table 3-7 Life-cycle cost (USD) inventory of the sewage sludge treatment scenarios

	AD	Dewatering	Incineration	Landfill	Cement production	Total
S1	(52,000,000)	11,000,000	-	906,000,000	-	865,000,000
S2	-	22,000,000	-	3,620,000,000	-	3,641,000,000
S3	(52,000,000)	9,000,000	19,000,000	95,000,000	-	71,000,000
S4	-	23,000,000	104,000,000	366,000,000	-	492,000,000
S5	(52,000,000)	9,000,000	19,000,000	-	(100,000)	(24,000,000)
S6	-	23,000,000	104,000,000	-	(600,000)	126,000,000

3.4 Discussions

3.4.1 Environmental impacts of urban sludge treatment approaches

3.4.1.1 Land occupation

The impact on land occupation that resulted from the defined sewage sludge treatment scenarios, operating for 30 years, is presented in Figure 3-4. Apparently higher degrees of land occupation were observed in S1 and S2, while only one-tenth of the impact level was shown in S3 and S4. The land area required for landfill disposal of the treated sludge was the dominating factor of the land occupation for S1 to S4, while only insignificant area of land was required in S5 and S6 as sludge ash was used for cement production at the final stage. The reason for the significant difference of land use impacts between scenarios with (S1 and S2) and without incineration (S3 and S4) was the 90% waste volume reduction achieved by the incineration process. Impact on land occupation was lower in scenarios applying AD than those without AD (that is S1 < S2 and S3 < S4) because volume of sludge was notably reduced by organic solids destruction in AD.

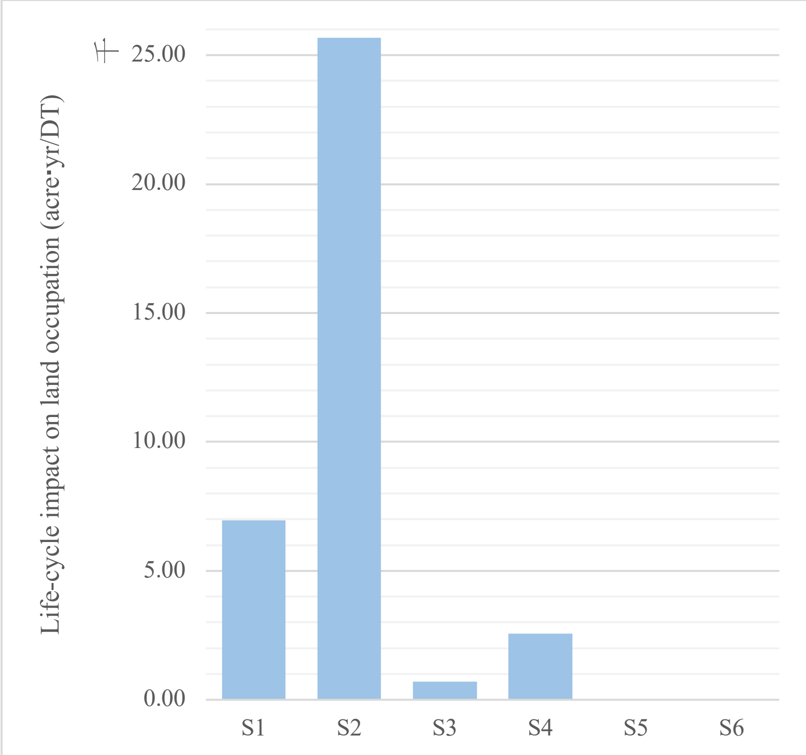

Figure 3-3 Life-cycle Impact on Land Occupation (acre•yr/DT) of the Sludge Treatment Scenarios

3.4.1.2 Climate change, acidification, human toxicity and eutrophication

The life-cycle environmental impact of sludge management scenarios is presented in Figure 3-5, and the impacts on acidification, human toxicity and eutrophication demonstrate similar trend. S1 contributes to the lowest environmental impacts, followed by S5 and then S3. As incineration was not used in S1, the atmospheric emissions from combustion of organic matters can be avoided, thus leading to the minimal environmental impact of the scenario. The application of AD treatment in these three scenarios recovers energy from waste sludge and reduces the volume thus the loading rate of the other treatment processes after AD, so a remarkable amount of environmental releases was avoided. The application on cement production in S5 further offset part of the emissions and avoid landfill disposal of the final product, therefore allowing the scenarios to perform better in the environmental aspect than S3. The explanation for the difference of environmental performance applies to the four impact categories, with NO_x emission being the major contributor to the impacts, other than land occupation.

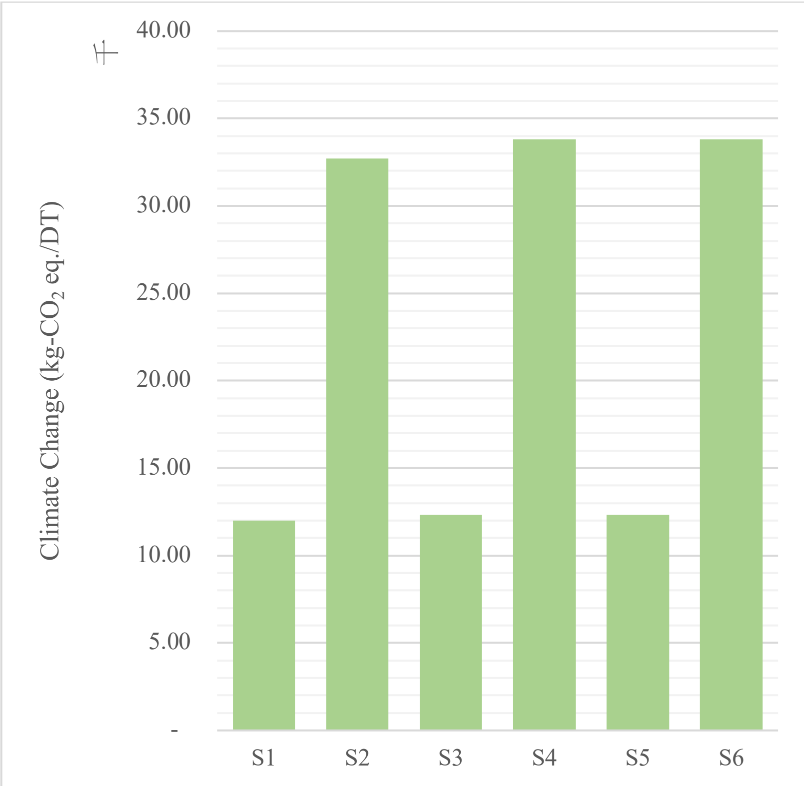

Figure 3-4 Life-cycle impact on climate change (kg-CO_2 eq./DT) of the sludge treatment scenarios

3.4.1.3 Normalized life-cycle impact

Although the scenarios performed consistently in the impact categories of climate change, acidification, human toxicity and eutrophication, the performance pattern in the land occupation impact was totally different. For comprehensive comparison between the overall environmental consequences of the scenarios, the normalized life-cycle impacts were analyzed and presented in Figure 3-6. Before the inclusion of land occupation impact, the most environmentally favorable scenario was S1 followed by S5 and then S3. However, remarkable land impact was observed in S1 and S2, while S3 and S4 presented less significant land impact and the remaining two scenarios showed insignificant land occupation impact. After taking the land impact into consideration, different overall environmental performances were resulted (S5 < S3 < S1 < S6 < S4 < S2). The most favorable overall normalized impact was presented by S5 because energy recovery was achieved in AD and incineration, and application in cement production avoided the requirement for landfill disposal. Although 90% volume reduction had already been attained by incineration in S3, the landfill requirement for the disposal of sludge ash made the scenario less favorable than S5. Energy recovery and volume were achieved by AD in S1, but the absence of incineration caused a notable requirement for landfill disposal, thus resulting high impact on land occupation. Therefore, the environmental performance of S1 become less favorable after the inclusion of land impact. For S2, S4 and S6, the absence of the AD process caused high loading rate and environmental burdens in the treatment stages, thus significantly higher adverse impacts were observed.

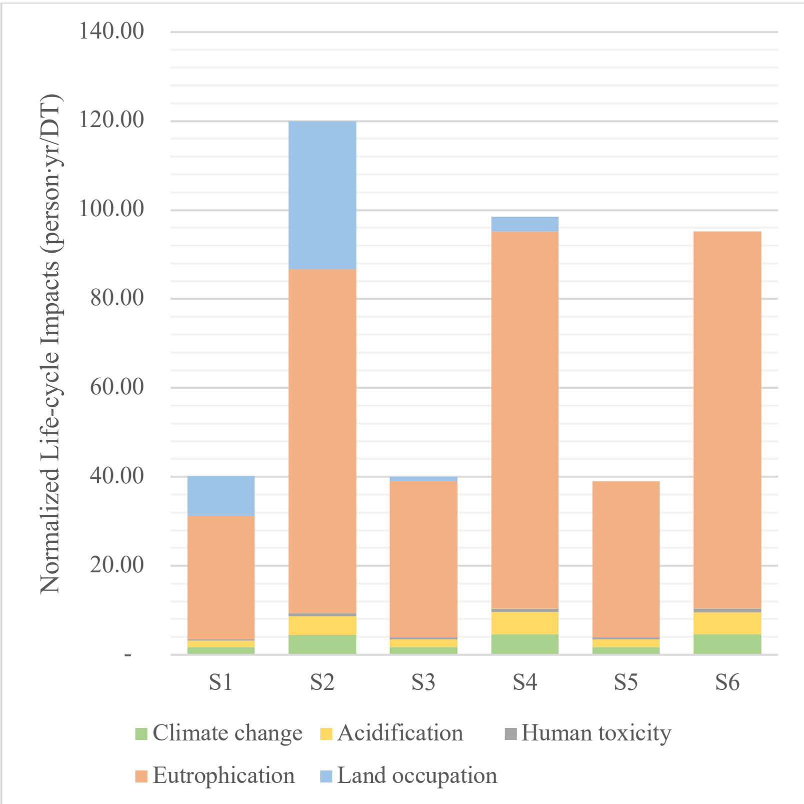

Figure 3-5 Normalized life-cycle environmental impacts of the sludge treatment scenarios

3.4.2 Economic impacts of urban sludge treatment approaches

The total life-cycle costs (Appendix Table A1-44 to A1-47) of the six sludge treatment scenarios in the Sha Tin, Tai Po, Yuen Long and Shek Wu Hui STWs are presented in Figure 3-3.

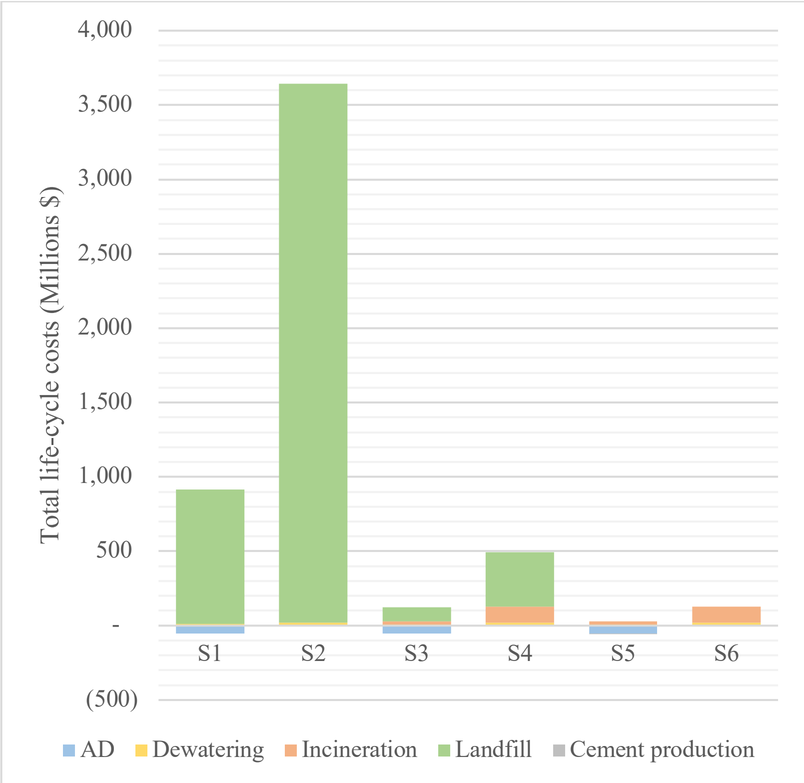

Figure 3-6 Total life-cycle costs of sludge treatment scenarios

In the AD process, biogas, which primarily contains methane (CH_4) and carbon dioxide (CO_2), is produced and collected for energy recovery. The CH_4 in the biogas can be burned as fuel for electricity and heat production. Electricity generation for the self-sustainability of the AD facilities and provision to the public electricity grid in the case of surplus electricity production was assumed in this study. Heat energy from sludge combustion in incineration is also recovered for electricity generation. Offsets to the electricity cost from energy recovery were considered in the economic analysis, thus earnings were observed to reduce part of the total costs of the relevant scenarios. Nine million cubic meters of CH_4 were produced annually, which resulted in \$4 M USD per annum as the total electricity cost offsets and earnings. Economies of scale were one of the contributors to the economic advantage of the AD system in Sha Tin STW, which produced the largest volume of raw sludge among the four STWs. The normalized AD costs (US\$/ m^3 of raw sludge) were US\$ -1.99 / m^3 for the Sha Tin STW and US\$ 0.54 / m^3 for the Yuen Long STW, which received 1,620 m^3 and 312 m^3 of raw sludge for treatment, respectively. For dewatering, the costs of the process normalized by the volume of inlet sewage sludge (US\$/ m^3 of inlet sludge) were revealed to be higher in the Yuen Long and Shek Wu Hui STWs. This was the result of the application of ferric chloride ($FeCl_3$) in the two mentioned STWs, and the chemical costs were US\$1.15 per pound. The dewatering cost, which constituted 0.59% to 17.63% of the total costs, were relatively insignificant when compared with those of the other sewage sludge treatment processes.

The incineration cost is mainly contributed by the high capital cost, including the installation cost and the costs for the building and foundation, priced at \$9.71 M USD in total. The second contributor to the incineration costs was the elimination of nitrogen oxides (NO_x), which has been identified as one of the major atmospheric pollutants in flue gases. The following chemical equation represents the complete combustion of sewage sludge ($C_5H_7O_2N$):

$$C_5H_7O_2N + \frac{27}{4}O_2 \rightarrow 5CO_2 + \frac{7}{2}H_2O + NO_2.$$

The products of the process include carbon dioxide (CO_2), water vapor (H_2O) and nitrogen dioxide (NO_2). According to the Guidance Note on the Best Practicable Means for Incinerators (Sewage Sludge Incineration) (EPD, 2010), the daily and half-hourly average concentration limits for the emission of NO_2 are 200 mg/m^3 and 400 mg/m^3 respectively. Selective catalytic reactors are commonly used for NO_2 removal, and the capital and operating costs of the equipment are US\$ 45/kW of capacity and US\$ 2,165/t of NO_x elimination, respectively (Yam & Leung, 2013). Lower incineration cost was achieved in S3 and S5 compared to S4 and S6 because AD is adopted in the prior two scenarios, enabling the destruction of organic solid contents in sewage sludge before incineration. Such treatment process reduces the volume of sludge, thus the loading rate of the incinerator. Therefore, a remarkable reduction in incineration cost (-$85 M USD) can be observed in S3 and S5.

Landfill costs were predominant in the total costs in S1 and S2, in which sludge incineration was not employed, for all of the STWs. The total costs of S2, followed by S1, was the highest among the treatment scenarios because of the overwhelming landfill costs. This is attributable to the large sludge end-product volumes and high land costs in Hong Kong. The percentage volumes of the sludge end products to the untreated raw sludge volumes are presented in Table 8. For all STWs, the volumes of the treated sludge in S1 and S2 are much larger than the incineration ash in S3 and S4. Volume reduction of sludge apparently was better achieved in S1 and S2 due to the 90% volume reduction in the incineration process (EPD, 2005). Because Hong Kong is a densely populated city, land is a scarce resource, and land costs are high. As mentioned above, the price of industrial land in Hong Kong was estimated to range from HK$500 to HK$1,200[1] per square foot. More expensive disposal costs, and therefore total costs, in S1 and S2 resulted from the sludge disposal volume and the volume-sensitivity of the landfill costs. Better volume reduction was observed in scenarios with AD application than those without (S1 < S2 and S3 < S4). Because the sludge incineration ash was utilized in clinker substitution in S5 and S6, no disposal of end-product was required. Thus, the ratios of inlet and outlet volume are not listed in Table 3-8.

[1] Exchange rate: US$1 = HK$7.75.

Table 3-8 Percentage volume of the sludge end product for disposal to the inlet sludge volume (% volume of raw sludge)

	End-product	ST	YL	SWH	TP	Average
S1	Sludge cake	2.57%	1.65%	1.95%	2.30%	2.12%
S2	Sludge cake	11.86%	9.67%	4.41%	9.19%	89.55%
S3	Ash	0.26%	0.17%	0.19%	0.23%	0.21%
S4	Ash	1.19%	0.97%	0.44%	0.92%	0.88%
S5	Ash	Used for clinker substitution in cement production				
S6	Ash	Used for clinker substitution in cement production				

The best economic performance was observed in S5 because energy recovery was achieved and material substitution in cement production was the final destination of end-product. Energy recovery was achieved by AD and incineration, thus leading to economic benefits from surplus electricity generation. AD was also adopted for VS content reduction, which consequently lowered the NO_x emission from organic substance combustion in the incineration stage. Most importantly, the use of sludge incineration ash as clinker in cement production avoid the landfill disposal requirement in the scenario. The second most economically favorable scenario was S3, which adopted similar sludge treatment processes as S5 except for the final stage. Instead of utilization in cement production, the final destination of sludge ash was landfill disposal in S3, which adds the landfill disposal cost to the scenario.

3.4.3 Outlooks on sludge management

As sludge is an unavoidable by-product from wastewater treatment, sludge management is a common issue faced by different countries worldwide. A change of disposal practice was observed in the European countries. In the EU-15 countries between 1992 and 2005, the percentage of countries adopting landfill disposal decreased from 33% to 15% significantly (Kelessidis & Stasinakis, 2012). Kelessidis & Stasinakis (2012) revealed that most of the European countries have abandoned the disposal of sludge in landfills, while a slight increase in landfill disposal of sludge was observed for some other countries, such as Italy, Denmark and Estonia. In the US, as the water and wastewater treatment is energy demanding, contributing to more than 40% of the energy usage of the country, advancement on sewage sludge management for energy conservation has been reviewed (National biosolids partnership, 2013). For example, AD has been recognized as a widely adopted approach to turn sludge into source of energy and developments in microbial fuel-cell (MFC) have been investigated in research studies to improve the energy efficiency of energy extraction from sewage sludge (National biosolids partnership, 2013; Zhang et al., 2012). However, full-scale application of energy extraction from sludge using MFC has not yet achieved technically. The above observations reveal that the common future direction is to minimize landfill disposal of sludge and to utilize sludge as a source of energy. This matches the findings of our study, including the use of incineration for thermal energy recovery and reduction of landfill disposal loads, as well as the adoption of AD to recover energy from methane production.

3.5 Sensitivity analysis

3.5.1 Sensitivity to land use in urban city

To reveal the crucial influence of land resource requirement on the total life-cycle cost in urban sludge management, sensitivity analysis was conducted based on the high land cost in Hong Kong for the scenarios (Table 3-9). The land cost was varied by 5% (Xu et al., 2014) and the variation in total life-cycle costs of the scenarios was observed for the comparison on the sensitivity on land cost between different scenarios. The sensitivity to land cost in ascending order is S3 < S4 < S1 < S2, while S5 and S6 without landfill disposal requirement were considered to have negligible land resource demand. Highest sensitivity was observed in S1 and S2 because the landfill requirement of these two scenarios were the highest, and land cost was the dominating factor of landfill cost and the total cost in these scenarios. S3 and S4 had lower sensitivity due to the reduction in volume and therefore land requirement by incineration. Scenarios without AD had higher sensitivity to land cost (that is S2 > S1, and S4 > S3) because the volume in such scenarios was larger than that in scenarios adopting AD.

Table 3-9 Sensitivity to urban land cost

	Unit	S1	S2	S3	S4	S5	S6
Land Cost Variation	%	+5%	+5%	+5%	+5%	N.A.	N.A.
Total Cost Variation	%	+4.59%	+4.91%	+3.52%	+3.65%	N.A.	N.A.

To recognize the sensitivity of land occupation to raw sludge volume, sensitivity analysis was conducted to evaluate the sensitivities of different environmental impacts to a 5% increase of raw sludge volume input Table 3-10. Among the five environmental impact categories, land occupation was the most sensitive to the incoming volume of sewage sludge. In the normalized life-cycle impact assessment, the 5% increase in input sludge volume could be reflected by 5% in the land occupation category, while only 0.01% to 0.73% of the influence could be reflected in other impact categories. Thus, the substantially high sensitivity of land occupation impact was considered to be essential to the overall environmental impact of the scenarios. In the comparison between sensitivity to raw sludge volume among the six scenarios, S5 and S6 that did not adopt landfill disposal presented the lowest sensitivity, followed by S3 and S4 which employed sludge incineration to reduce sludge volume by 90%. S1 and S2 showed the highest sensitivity to raw sludge volume because sludge was only dewatered with or without AD before disposal to landfill sites.

Table 3-10 Sensitivity of Environmental Impacts to Raw Sludge Volume

	Unit	S1	S2	S3	S4	S5	S6
Raw Sludge Vol. Variation	%	+5%	+5%	+5%	+5%	+5%	+5%
Land Occupation	%	+5.00%	+5.00%	+5.00%	+5.00%	N.A.	N.A.
Climate Change	%	+0.02%	+0.01%	+0.13%	+0.17%	+0.13%	+0.17%
Acidification	%	+0.02%	+0.01%	+0.59%	+0.73%	+0.59%	+0.73%
Human Toxicity	%	+0.02%	+0.01%	+0.02%	+0.01%	+0.02%	+0.01%
Eutrophication	%	+0.02%	+0.01%	+0.02%	+0.01%	+0.02%	+0.01%
Overall	%	+1.06%	+1.40%	+0.15%	+0.22%	+0.05%	+0.05%

The influence of land cost on the total costs of the six scenarios was further investigated. The investigation revealed that holding all the other conditions unchanged in this case study, the land cost had to be reduced to 19% of the original (US$ 910,000/acre) in order for S3 to achieve net-zero life-cycle cost. If the life-cycle cost of S1 had to achieve net-zero, the land cost has to be reduced to 1% of the original (US$ 48,000/acre). In the previous study conducted by Murray et al. (2008), final landfill disposal cost was not included in the economic analysis, and the dewatering treatment followed by landfill disposal was observed to be the most economically favorable option. The land cost was also not included in the LCA conducted by Xu et al. (2014), thus the influence of land cost in China was not revealed and compared with this study.

3.5.2 Sensitivity to transportation distance

A noteworthy amount of GHGs, mostly CO_2, was contributed by the transportation of sewage sludge from one treatment facility to another. Because of the large variations in the travelling distances between the four STWs and the facilities for further treatments (STF or landfill sites), different data inputs of the travelling distances between the STWs and treatment facilities were studied. As the CO_2 emissions contributed up to 98.27% of the total transportation emissions, the impact on climate change was focused on. Previous studies have included transportation in the air emission calculations. Travelling distance between treatment facilities was assumed to be 25 km by Murray et al. (2008) and 40 km by Xu et al. (2014) in their case studies in China. Actual road transportation distances were obtained and used for the estimates of environmental impacts in this study (Table 3-11).

Table 3-11 Actual transportation distances (km) between STWs and treatment facilities

	ST	YL	SWH	TP
S1	38	24	8	29
S2	38	24	8	29
S3	44.2	25.8	39.6	50.5
S4	44.2	25.8	39.6	50.5

S5	44.2	25.8	39.6	50.5
S6	44.2	25.8	39.6	50.5

The investigation of the influence of input transportation distance data on climate change impact was conducted by substituting the actual travelling distances with the assumed 25 km and 40 km distances. Absolute values of the deviations of GHG emissions from the actual releases are presented in Figure 3-7. The errors ranged from 0.91 to 145.28 kg-CO_2/DT of sludge for the 40 km assumption and 1.05 to 106.48 kg-CO_2/DT of sludge for the 25 km assumption. The maximum deviation in estimating CO_2 release from transportation over 30 years of operation, assuming that all of the conditions remain unchanged, reached 187,000 tonnes and 137,000 tonnes for the 40 km and 25 km assumptions respectively. Even for the most accurate scenarios (0.91 and 1.05 kg/DT deviation from the actual CO_2 emission), the accumulated errors reached 1,000 tonnes for the two assumptions. Because the inaccuracy in the emission estimation associated with transportation was substantial in the 30-year accumulation, the assumption for the uniform travelling distance between the STWs and destinations for treatment was proven to be unsuitable for the environmental impact evaluation for multiple STWs with different locations. Thus, the acquisition of real operational data and the separate calculation for the STWs in this study showed remarkable significance.

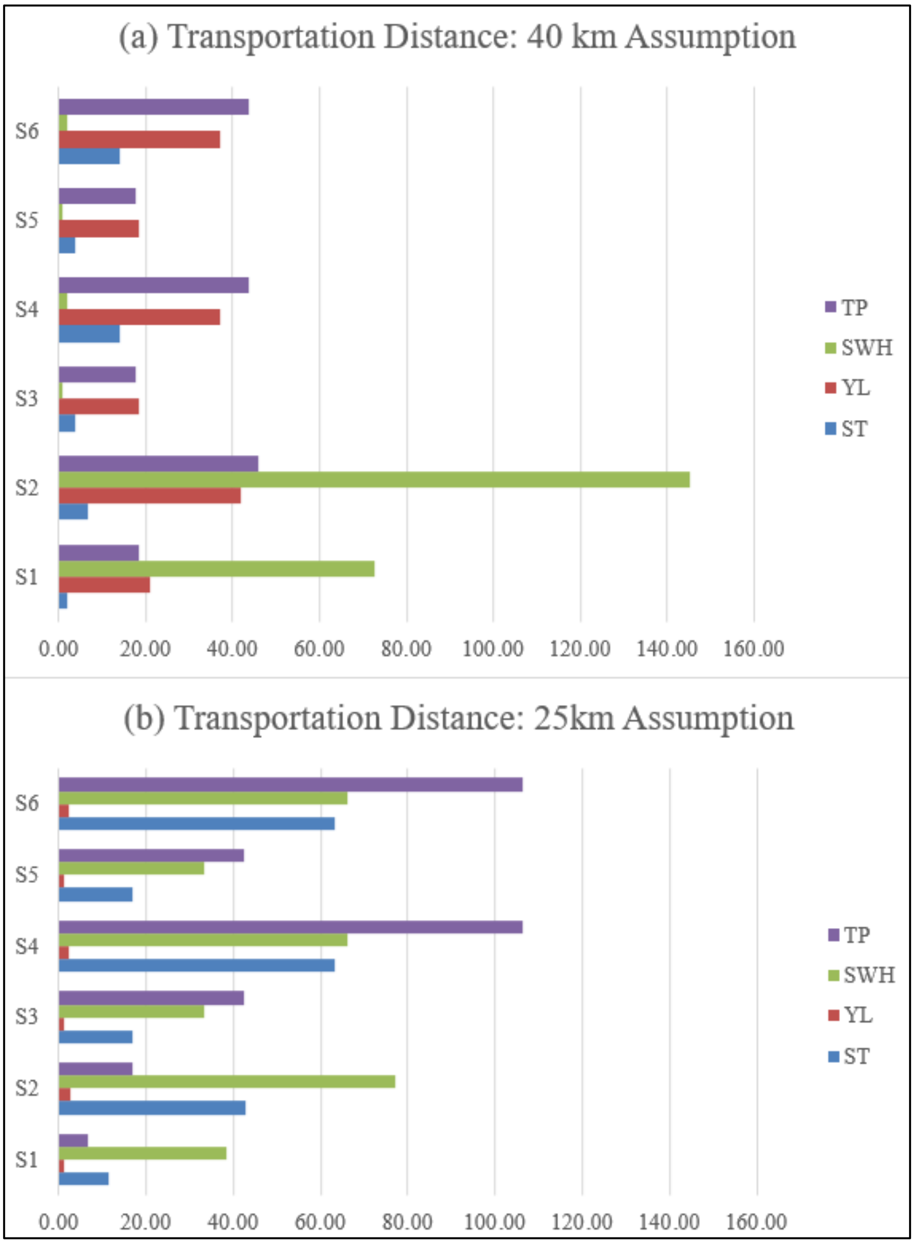

Figure 3-7 Absolute values of the inaccuracies with the assumptions of (a) 40 km and (b) 25 km transportation distances

3.6 Summary

To promote better water management and sustainable development in urban cities, both the economic and environmental impacts of the sewage sludge handling alternatives were evaluated using a case study based on the current conditions in Hong Kong. The most significant observation was the substantial influence of the land cost on sludge management in urban cities. As shown in the sensitivity analysis, the total costs of the scenarios without incineration was notably higher than those adopting incineration, and the total costs were influenced by 3.5 – 4.9% by a 5% alteration in land cost. Land occupation was also shown to be the most sensitive impact category to a change in raw sludge inlet rate. Therefore, the inclusion of land use was concluded to be an essential factor in urban sludge management. The detailed and separate performance evaluations for the four sewage treatment works were also notably significant. An important example is the consideration of the actual transportation distances between the STWs and the treatment facilities for the estimation of atmospheric emissions. The use of single assumption for the transportation distance revealed remarkable deviations in the GHG emissions from the actual emissions estimated with real transportation distances, especially when accumulated emissions over 30 years was considered. The evaluation of the sludge management scenarios, rather than individual treatment technology, is important because different performances were observed for the same treatment technology in different combinations. More comprehensive results can be obtained using scenarios because the combinations of treatment processes better represent actual conditions.

The demonstration of application of this eco-efficiency tool on urban sludge management in Hong Kong revealed S5 (AD, dewatering, incineration, cement production) to be the most favorable option for the city as the scenario showed the best economic and environmental performance among the six scenarios. The second-best option was shown to be S3 (AD, dewatering, incineration, landfill disposal). As the market size for clinker material replacement by sludge ash and the suitability of the actual sludge conditions for such application are subjected to further investigation, S3 is determined to be a favorable backup solution if any technical problem is identified for S5.

The economy, environment and society are the three pillars of sustainable development. The economic and environmental aspects were analyzed in this study. However, the social aspect has not yet been included. Because sustainability will be the focus of future urban development, the inclusion of social impact evaluation to provide a Life-cycle Sustainability Assessment (LCSA) tool is definitely the trend of the future. Computer modeling approaches can be incorporated into the LCA tool to simulate human behavior and responses to different scenarios. The approach can also be adopted for obtaining a series of optimized scenarios for different priorities as determined by the decision makers. In addition, because the environmental emissions were observed to be sensitive to the transportation distances between the STWs and treatment facilities, the optimization of facility locations and transportation networks would be a meaningful topic for future studies. Such improvements to LCA would lead the future trend in sustainable town planning and management in urban cities around the world.

CHAPTER 4

ECO-EFFICIENCY OF NON-POTABLE WATER SYSTEMS IN DOMESTIC BUILDINGS

4.1 Introduction

Anthropogenic activities are major contributors to intensifying water scarcity and uneven distribution of water resources around the globe. The World Economic Forum (2015) has listed the water crisis as the global risk of the most devastating impact. In 2015, 663 million people (approximately one-tenth of the world population) lacked access to safe drinking water, and 2.4 billion people lacked improved sanitary facilities (WHO & UNICEF, 2015). The vast majority of the population using substandard drinking water and unimproved sanitary facilities were in sub-Saharan Africa and Southern Asia, revealing the unbalanced distribution of water resources. In developing countries, population growth and rapid urbanization are major drivers for increasing water demand as the intense urban development activities are often coupled with escalating consumer demand and improving living standards (WWAP, 2014). Safe water resources are available in most of the developed countries, thus the water crisis is often not an immediate risk encountered in such regions, yet sustainable water management is still necessary for avoiding imprudent water resource exploitation and maintaining both the quantity and quality of the water supply (WWAP, 2014).

Traditional water management approaches generally aim at maintaining water supply with stable quantity and quality for water-related services and is demand-driven (Al-Jayyousi, 2003; Haasnoot et al., 2011). Conventional water and wastewater treatment infrastructures are centralized and large-scale developments, as well as large energy consumers for many municipal governments. Water treatment utilities in the United States contribute to 30-40% of total energy demand, and thus are the largest consumers among the publicly owned utilities (USEPA, 2016). In the United States, drinking water and wastewater treatment systems account for approximately 2% of total energy consumption in the country and emit more than 45 million tonnes of GHGs per year (USEPA, 2016). While in EU countries, drinking water and wastewater treatment present net energy consumption of 34 and 88 kWh/y/person respectively, accounting for 7.6% of the overall energy consumption (EEA, 2014). To facilitate sustainability, unconventional derivatives for water and energy resources should be sought and sustainable water management approaches adopted in conjunction with changes in water utilization patterns, so that economically affordable, environmentally favorable, and socially acceptable resource utilization can be achieved.

Water and energy are both invaluable resources to maintain the well-being of humans, as well as the socio-economic development of the society. While conservation and sustainable management of both resources are gaining attention from the public, policy makers, and researchers, a novel perspective of water-energy nexus has been proposed in recent years, suggesting that water and energy resources are strongly interdependent (Schnoor, 2011; WWAP, 2014). The interlinkage between the resources could be reflected in the reality that energy transformation and utilization require water usage, and energy is required for water acquisition, transmission and treatment. Wakeel et al. (2016) reviewed the studies on energy consumptions of the different stages of the water sector in different countries, and suggested that the energy efficiency and the sustainability of the water sector could be improved by the integrated management of the energy and water resources. Studies have been conducted to investigate the correlation between water- and energy-saving policies so as to promote the formulation of more integrated policies (Gu et al., 2014; Siddiqi & Anadon, 2011). The high organic content in wastewater stores a significant amount of energy, thus wastewater is now more commonly viewed as a source of energy (Frijns et al., 2013; Heidrich et al, 2011; Liu et al., 2015; Silvestre et al., 2015). Regarding such characteristic of the wastewater, a notable number of studies have been conducted to evaluate the energy consumption and environmental performance of the centralized wastewater treatment plants (Corominas et al., 2013; Longo et al., 2016; Panepinto et al., 2016; Wakeel et al., 2016).

Greywater recycling and reuse offer notable potential for reducing sewage treatment loads and water supply costs, as well as alleviating the rising demand for freshwater supply, and therefore will become a core component in sustainable water management (Al-Jayyousi, 2003; Figueres et al., 2012). In contrast to blackwater, which is sewage collected from toilets and kitchen sinks, greywater includes the wastewater generated from showers, sinks (except kitchen sinks), bath and laundry, and accounts for 50-80% of total water consumption (Al-Jayyousi, 2003; Ghaitidak & Yadav, 2013; Jamrah et al., 2007). Greywater should be viewed as a stable and valuable source of water, energy, and cost savings through recycling for non-potable uses, such as toilet flushing, irrigation and car washing. Based on the close-looped concept, which states that water consumption follows certain cycles as in natural state (Figueres et al., 2012), a conceptual framework of greywater reuse for toilet flushing and landscaping has been proposed (Al-Jayyousi, 2003). Acceptable quality of treated greywater (Santos et al., 2012), sufficient greywater source for covering water consumption, and public acceptance of greywater reuse (Jamrah et al., 2007) have been investigated and the findings confirm the great potential for promoting greywater recycling systems.

Notable efforts and contributions have been made in previous studies to evaluate the performance of greywater reuse systems, yet for the purpose of informing decision-makers and the public on the selection of sustainable water supply systems, there are some key shortcomings in the existing assessment frameworks. Firstly, only environmental or economic aspects, but not both, were evaluated in previous studies. Both aspects are important factors that directly influence the social acceptability of options, and evaluating only one of them is inadequate for supporting decision-making for sustainable water management. Secondly, detailed inventories of the evaluated systems studied in previous studies were not available. The absence of detailed environmental and economic inventories lowers the transparency of the LCA or economic analysis. The possibility of refining the developed frameworks for assessing greywater reuse systems is also hindered by the unavailability of the detailed inventory data used for evaluation.

In this study, an EEA framework was developed and applied to inform decision-making on greywater recycling with energy recovery for non-potable use, presenting three innovative features that significantly improve the comprehensiveness of the greywater management framework: (1) integration of the environmental and economic portfolios of the greywater systems to produce eco-efficiency portfolio; (2) detailed inventory backed up by full engineering system design, and; (3) consideration of freshwater supply as a substitute good to the greywater source. This study aims to assist decision-making in the selection of a sustainable water management approach with a focus on non-potable water supply in residential buildings. An inclusive EEA framework for greywater recycling systems was developed in this study. Detailed data inventories for environmental and economic analysis were built to serve as the foundation of the EEA. Eco-efficiency scores were obtained to serve as the final results for comparison between scenarios. This study also revealed the effects of freshwater price on the selection of water management options. The EEA framework developed in this study was demonstrated in the analysis of greywater recycling for toilet flushing in a residential building in Hong Kong. The comprehensive framework was used to evaluate different greywater treatment technologies, including aerobic and anaerobic treatment, as well as freshwater and seawater supply scenarios. This framework can be applied widely for assessing water and greywater systems in other regions to assist decision-making in sustainable water management.

4.2 Descriptions of case study

4.2.1 Scope of study

An anaerobic fluidized bed membrane bioreactor (AFMBR) system is proposed for treating greywater for non-potable use and recovering energy simultaneously. The AFMBR system is selected among various decentralized treatment options because the system is energy efficient, has low rate of sludge production and is effective for treating low-strength wastewater (Bae et al., 2014; Kim et al., 2011; Shin et al., 2016). Application of the AFMBR greywater reuse system in a residential building in Hong Kong is evaluated under the EEA framework and compared with other three water supply systems in a mock case. The freshwater and seawater flushing scenarios are included for comparison because they are the existing systems in Hong Kong. Aerobic greywater treatment system is also analysed and compared with the AFMBR system to feature the difference between aerobic and anaerobic systems. The environmental impacts and the economic costs of the construction and the operation phases of the scenarios are estimated and analysed.

4.2.2 Background of case study

Due to the absence of natural lake and groundwater resources, the freshwater supply in Hong Kong mainly relies on the supply from Dongjiang River in mainland China and the rainwater from catchment (Research Office of Legislative Council Secretariat, 2015; WSD, 2014). The water consumption in Hong Kong was estimated to be 1.2 trillion cubic meters per year, with an average daily freshwater and flush water consumption of 0.13 and 0.09 cubic meter per capita (WSD, 2014, 2015). To conserve the precious freshwater resources, the city has been alleviating the rising demand for freshwater by introducing the seawater flushing system, which received international recognition for success in sustainable water management (CSB, 2004), since the 1950s. Currently about 80% of the population in Hong Kong is supplied with seawater for toilet flushing (WSD, 2016) while some inland areas in the New Territories are still using freshwater for flushing. The replacement with seawater has successfully saved more than 270 million cubic meters of potable water per year (Research Office of Legislative Council Secretariat, 2015; WSD, 2016).

The scenarios in this case study was set according to the actual conditions in a new town development project in the inland area in the New Territories in Hong Kong. A 35-storey residential building with 8 apartments per floor and 3 residents per apartment was studied (HK Housing Authority, 2016; LegCo, 2014). The daily freshwater and flush water consumption was assumed to be 130 and 83.8 liters per person respectively (LegCo, 2009; WSD, 2015). Four scenarios were defined in the case study (Figure 4-1). Greywater is defined as wastewater, except sewage from toilets and kitchen sinks, that could be recycled after appropriate treatment. The proportion of kitchen sink sewage was assumed to be 15% of the total water consumption (Butler & Memon, 2005). The background information and assumptions of the mock case are summarized in Appendix Table A2-1 to Table A2-2.

Scenario 1 – Freshwater flushing

In Scenario 1 (S1), the total water consumption of the building is solely supplied by freshwater, which is the real situation in the remote areas in the New Territories in Hong Kong. Therefore, this scenario is considered the baseline scenario. Freshwater is supplied for toilet flushing and other uses. The ratio of potable water supply from Dongjiang River to local rainwater yield is 3:1 (Research Office of Legislative Council Secretariat, 2015). All the wastewater is collected through the sewer system and then transferred to the Sewage Treatment Works (STWs) for centralized treatment.

Scenario 2 – Seawater flushing

Seawater is supplied for toilet flushing in Scenario 2 (S2) after screening and chlorine disinfection (WSD, 2016). Water consumption for other purposes is supported by freshwater. The other settings for this scenario are the same as in Scenario 1.

Scenario 3 – Aerobic greywater reuse system

Greywater, which is wastewater excluding sewage from toilet flushing and the kitchen sink, is recycled for toilet flushing in Scenario 3 (S3). Supplemented with the recycled greywater, the demand for freshwater supply, as well as the sewage treatment loading, are reduced by approximately 40%. Greywater is collected separately from blackwater for decentralized treatment at the bottom of the building. The aerobic treatment system adopted in this scenario includes the aerobic MBR as the core treatment unit and a chlorine disinfection unit. The adoption of aerobic MBR for greywater reclamation has been studied and demonstrated to be a reliable technology in previous literatures (Abdel-Shafy, Al-Sulaiman, & Mansour, 2015; Hasar & Kinaci, 2004; Jefferson, Laine, Parsons, Stephenson, & Judd, 2000; Kishino, Ishida, Iwabu, & Nakano, 1996). The treated greywater is pumped up to the residents for use. The blackwater generated from the building is collected for centralized treatment in STWs.

Scenario 4 – Anaerobic greywater reuse system

Greywater is reclaimed for toilet flushing after treatment in an AFMBR and chlorine disinfection unit in Scenario 4 (S4). The AFMBR is a reliable energy efficient treatment unit due to its low energy requirements compared to aerobic treatment systems, and its energy recovery capability through methane production (Bae et al., 2013; Gao et al., 2014; Kim et al., 2011; Ren et al., 2014). The substitution by biogas in electricity generation could reduce the consumption of conventional fossil fuels, thus is an advantageous feature of AFMBR-based systems in sustainable wastewater management (Bidart et al., 2014). The fouling of the membrane could be alleviated through enhancing filtration with granular activated carbon (GAC), which can reduce the need for backwashing and maintenance (Gao et al., 2014; Kim et al., 2011). Other advantages of AFMBR include high effluent quality and low sludge production. The other settings for this scenario are the same as Scenario 3.

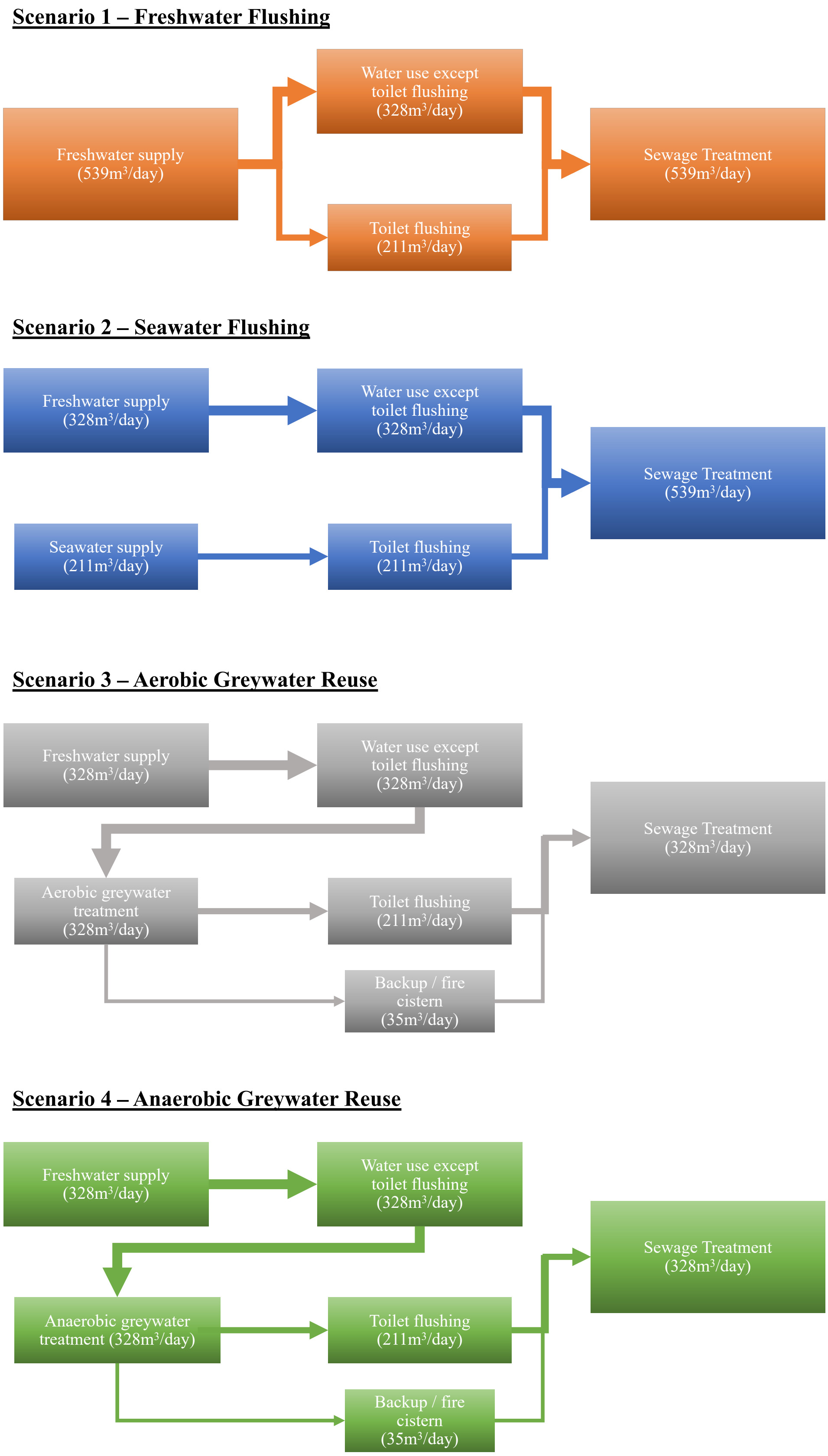

Figure 4-1 Water Management Scenarios for EEA

4.2.3 Engineering design of water systems

This study focuses on the application of the different water systems on a practical level, thus the construction of pipelines, pumping systems, water tanks, and treatment units were included in the analysis. The engineering design of the water system for the case study was based on the standards stated in China National Standard: Code for Design of Building Water Supply and Drainage (China Engineering Construction Standardization Association, 2007). The parameters used for the design of the water supply and drainage systems are documented in Appendix Table A2-3. The water supply mode used in this case study adopted the pressure-boosting system, which is a common practice for water supply in high-rise buildings in Hong Kong. The system involves pumping the water from the sump tank to the roof tank. Then the building was divided into different water-pressure zones and water is supplied through the pressure reduction valves to different groups of apartments. Four sets of vertical freshwater pipeline were designed for the scenarios, each set covering the freshwater supply for six apartments on each floor. Four additional sets of seawater and greywater pipeline were designed for the seawater flushing and greywater flushing scenarios, respectively. The Appendix Figure A2-1 shows the schematic design of one set of the pipelines. For the drainage systems, sewage from the households is collected through vertical drains and discharged into the public sewer system by gravity. The apartments on each floor were grouped into four drainage units, each unit covering six apartments. For scenarios 1 and 2, there are two sets of vertical drains: one set collects the sewage from the washing room and the other one collects sewage from the kitchen. As greywater excludes the sewage from toilets and kitchen sinks, an additional set of drains is required for the greywater reuse scenarios (scenario 3 and 4). The three sets of drains collect wastewater from: (i) the shower and washing basin in the washing room, and the washing machine; (ii) the toilet in the washing room, and; (iii) the kitchen sink. The wastewater collected by drain set (i), which is classified as greywater, is conveyed to the greywater treatment units at the bottom of the building. The treated greywater is pumped to the roof tank and distributed to the apartments for toilet flushing using a similar approach as for the freshwater supply. Details of the water systems designs are documented in the "Engineering water system designs" section in Appendix 2.

4.3 Method and data

4.3.1 Eco-efficiency analysis framework

With the rising environmental awareness of policy makers and the public, decision-support tools such as life-cycle assessment (LCA) have been widely used to assess the environmental performance of infrastructure projects or management strategies. However, the projects under assessment often involve economic investments, thus the economic performance should be considered and related to the environmental profile (Lorenzo-Toja et al., 2016; Saling et al., 2002). Early in 1992, the role of the economic market as a control to change decisions or actions has been discussed in the United Nations Conference on Environment and Development (Cope, 1993). To effectively facilitate sustainable development, the business costs and the environmental costs should be accounted for so that the market could serve as a driver to promote efficient resources consumption and waste minimization (Schmidheiny, 1992). The eco-efficiency analysis framework, which integrates the LCA and LCC, defined in the standard ISO 14045:2012 illustrates the principles for linking the environmental and economic aspects and provides guidelines for EEA at an operational level (ISO 14045, 2012).

4.3.1.1 Life-cycle assessment

Life-cycle assessment (LCA) is an approach for evaluating the environmental performance of a product or process from "cradle to grave", which includes the life-cycle phases of raw material acquisition, manufacturing, use and waste handling (ISO 14040, 2006; US EPA, 2006).

The four main phases of conducting LCA are goal and scope definition, life-cycle inventory (LCI) analysis, LCIA and interpretation (ISO 14040, 2006; ISO 14044, 2006). Besides the traditional application on manufacturing processes, LCA has become more commonly applied on energy recovery from organic wastes, such as agricultural waste (Pierie et al., 2015; Tonini & Astrup, 2012) and food waste (Ebner et al., 2014; Franchetti, 2013; Jin et al., 2015). The LCA method has also been widely adopted in water, wastewater, and sludge management (Kalbar et al., 2013; Lane et al., 2015; Opher & Friedler, 2016; Strauss & Wiedemann, 2000; Suh & Rousseaux, 2002; Yıldırım & Topkaya, 2012).

The two traditional approaches of LCA are process-based models and economic input-output (EIO) models. The process-based LCA models assess the series of processes involved in the scenarios under evaluation. The inputs and outputs of each process are identified and included in the life-cycle inventory. The advantage of such an approach is its high specificity and accuracy due to its use of primary data. Yet the process-based models also present drawbacks, including the exclusion of upstream processes outside the system boundary and the challenges encountered during collection of a huge volume of process-specific primary data (Rowley et al., 2009). Compared to the conventional process-based LCA, the EIO models enable a wider coverage of environmental impacts, especially those from upstream processes (Hendrickson et al., 1998; Rowley et al., 2009). The EIO-LCA models were developed based on an economic analysis methodology known as input-output analysis (IOA), in which the relationships between different production sectors in an economy are being studied. The limitations of EIO models include the inaccuracies induced by the assumptions of uniformity of prices and environmental emissions per unit price (Rowley et al., 2009). Wakeel et al. (2016) suggested that, besides the process-based (bottom-up) and input-output (top-down) LCAs, the hybrid approach is also a common methodology to assess the energy footprint and the environmental consequences of the water use cycles.

This study adopted a hybrid approach through the integration of the process-based and EIO-LCA models. Two LCA tools were used to estimate the environmental impacts of the scenarios, namely the software SimaPro and the Economic Input-Output LCA (EIO-LCA) tool (Carnegie Mellon University (CMU), 2006). SimaPro is an internationally recognized tool for conducting LCAs and it was used in this study for quantifying the environmental impacts of the operation phase, which is comprised of a series of processes such as water pumping, water treatment, and electricity production. The EIO-LCA tool was used to estimate the emissions from the manufacturing of components such as pipes, pumps and water tanks, during the construction phase. The total life-cycle environmental impacts were evaluated and integrated using the ReCiPe Endpoint method (Goedkoop et al., 2009).

4.3.1.2 Life-cycle costing

The life-cycle costs of the water management scenarios were estimated in the LCC analysis. Based on the engineering design of the plumbing and sewer systems, a detailed inventory of the components required for the construction phase of each scenario was built. The online cost estimation tool RSMeans and its up-to-date database were used for the LCC analysis (RSMeans, 2016). The operational costs associated with the purchase of freshwater, water treatment, electricity consumption, chemical requirements, and sludge treatment were also considered.

4.3.1.3 Eco-efficiency analysis for integrated evaluation of environmental and economic performance

According to the definition by the World Business Council for Sustainable Development (WBCSD), eco-efficiency is “achieved by the delivery of competitively priced goods and services that satisfy human needs and bring quality of life, while progressively reducing ecological impacts and resource intensity throughout the life-cycle to a level at least in line with the Earth’s estimated carrying capacity" (WBCSD, 2000). The concept of eco-efficiency focuses not only on the environmental performance of an option, but also puts emphasis on the importance of considerations from a business perspective. On a practical level, the eco-efficiency of a certain process or product is evaluated in terms of the ratio between economic value added or service provided (outputs) and the environmental burdens produced (inputs) (Kicherer et al., 2007; Saling et al., 2002; WBCSD, 2000).

The methodology adopted in this study made modifications based on the Baden Aniline and Soda Factory (BASF) and the normalization EEA approaches described by Saling et al. (2002) and Kicherer et al. (2007) respectively. This method provides a means for integrating the environmental and economic performance of the alternatives and presenting the results in eco-efficiency (EE) scores to reveal the relative eco-efficiencies.

The environmental impact category indicators EI_i (i = 1, …, I) of the scenario n (n = 1, …, N) were estimated by the LCA tools. Normalization in the LCIA is a procedure to express the category indicators in relation to a set of well-defined reference data (de Bruijn, van Duin, & Huijbregts, 2002), such as the global or regional per capita environmental impact category indicators ($\frac{GREI_i}{Pop}$). Based on the policy targets or the social preference, weightings (w_i) could also be assigned to the impact categories to reveal their different importance. Thus, the total environmental impacts were estimated as the aggregated normalized and weighed impacts:

$$EP_n = \sum_{i=1}^{I} \frac{EI_{i,n}}{GREI_i} \cdot Pop \cdot w_i$$

(Equation 4.1)

EP_n = total environmental impact of the scenario n

$EI_{i,n}$ = environmental impact in category i of the scenario n

$GREI_i$ = global/regional impact normalization reference indicator for impact category i

Pop = population of the region

w_i = weighting of impact in category i

For economic performance, the total life-cycle cost LC_n is estimated from the price (P_j) and the quantity requirements (Q_j) of the component j (j = 1, …, J).

$$LC_n = \sum_{j=1}^{J} P_{j,n} \cdot Q_{j,n}$$

(Equation 4.2)

LC_n = total life-cycle cost of the scenario *n*

$P_{j,n}$ = price of the component *j* used for the scenario *n*

$Q_{j,n}$ = quantity of the component *j* used for the scenario *n*

The environmental and economic performances were integrated and presented on an EE portfolio graph to show the overall relative eco-efficiencies of the scenarios. The two aspects were first considered to be equally important as the sustainability concept suggests, yet the tool is capable of allowing the users to change the relative importance of the two aspects based on their own preferences. According to Kicherer et al. (2007) and Saling et al. (2002), the environmental impact portfolio position (y-coordinate) of scenario n is:

$$PP_{E,n} = \frac{EP_n}{(\sum EP)/N}$$

(Equation 4.3)

$PP_{E,n}$ = environmental impact portfolio position of the scenario *n*

EP_n = total environmental impact of the scenario *n*

$\sum EP$ = summation of environmental impacts of all scenarios

N = number of scenarios

Similarly, the life-cycle cost impact portfolio position (x-coordinate) is:

$$PP_{C,n} = \frac{LC_n}{(\sum LC)/N}$$

(Equation 4.4)

$PP_{C,n}$ = life-cycle cost impact portfolio position

LC_n = total life-cycle cost of the scenario *n*

$\sum LC$ = summation of the life-cycle costs of all scenarios

N = number of scenarios

In an eco-efficiency portfolio graph, the upper-right corner indicates the highest eco-efficiency, where both the environmental burden and the economic cost are minimal. The bottom-left corner, in contrast, indicates the lowest eco-efficiency. The y and x axes display the relative environmental and economic performances of the scenarios respectively, with the value 1 representing the average performance ($\sum EP\ /N$ and $\sum LC\ /N$). The distance between the options and the diagonal of the graph represents the respective eco-efficiency of the scenarios. Such graphical presentation enables clear and easy communication to the audience.

A term R_{EC}, which is defined as the ratio of regional or global significance of environmental impacts to that of the cost impacts, is used to reflect the relative importance of the two aspects (Kicherer et al., 2007). The R_{EC} ratio is defined as:

$$R_{EC} = \frac{EP_n}{LC_n}$$

(Equation 4.5)

R_{EC} = ratio of significance of environmental impacts to cost impacts

EP_n = total environmental impact of the scenario n

LC_n = total life-cycle cost of the scenario n

The EEA tool in this study allows the adjustment of the R_{EC} ratio to reflect the emphasis of the decision-makers in their policy formulation. A ratio larger than 1 means that the environmental performance is more highly valued than the economic one, and vice versa. The portfolio positions, thus the results, can be adjusted in accordance with the R_{EC} ratio to the new positions (Kicherer et al., 2007):

$$PP'_{E,n} = \frac{(\sum PP_E)/N + [PP_{E,n} - (\sum PP_E)/N] \cdot \sqrt{R_{EC}}}{(\sum PP_E)/N}$$

(Equation 4.6)

$$PP'_{C,n} = \frac{(\sum PP_C)/N + [PP_{C,n} - (\sum PP_C)/N]/\sqrt{R_{EC}}}{(\sum PP_C)/N}$$

(Equation 4.7)

$PP'_{E.n}$ = adjusted environmental portfolio position of the scenario n

$PP'_{C.n}$ = adjusted cost portfolio position of the scenario n

PP_E = original environmental portfolio position of the scenario n

PP_C = original cost portfolio position of the scenario n

N = number of scenarios

R_{EC} = ratio of significance of environmental impacts to cost impacts

4.3.2 Data inventory

The data source for this study includes documents issued by the Hong Kong Government, experimental results, the engineering designs of the systems, and previous literature. The data was used to build the life-cycle inventory on which the eco-efficiency analyses were based.

Table 4-1 Water flow in mock building

	S1	S2	S3	S4
Freshwater supply (m^3/d)	539	328	328	328
Seawater supply (m^3/d)	-	211	-	-
Greywater supply (m^3/d)	-	-	211	211
Sewage treatment (m^3/d)	539	539	328	328
Greywater treatment (m^3/d)	-	-	247	247

Table 4-1 shows the flow rate (m^3/day) of freshwater supply, seawater supply, greywater supply, sewage treatment, and greywater treatment of the four scenarios. In S3 and S4, all the greywater from the households (247 m^3/day) was collected for decentralized treatment at the bottom of the building, yet only the quantity needed for toilet flushing was pumped up for use. The extra amount of treated greywater (36 m^3/day) was stored as backup or used for other purposes, such as irrigation and floor cleaning. Based on the engineering design of the systems for the four scenarios, the material and operational requirements of each scenario were identified and listed in the inventories of construction and operation phases (Table 4-2 and Table 4-3).

Table 4-2 Inventory of construction phase

	S1	S2	S3	S4
Freshwater supply system	539 m^3/day - Roof tank - Sump tank - Plumbing - Pump	328 m^3/day - Roof tank - Sump tank - Plumbing - Pump	328 m^3/day - Roof tank - Sump tank - Plumbing - Pump	328 m^3/day - Roof tank - Sump tank - Plumbing - Pump
Seawater supply system	N.A.	211 m^3/day - Roof tank - Sump tank - Plumbing - Pump	N.A.	N.A.
Greywater supply system	N.A.	N.A.	211 m^3/day - Roof tank - Plumbing - Pump	211 m^3/day - Roof tank - Plumbing - Pump
Drainage collection system	539 m^3/day - 2 sets of sewers - 6 Septic tanks	539 m^3/day - 2 sets of sewers - 6 Septic tanks	328 m^3/day - 1 set - 2 sets of sewers - 4 Septic tanks	328 m^3/day - 1 set - 2 sets of sewers - 4 Septic tanks
Greywater treatment unit	N.A.	N.A.	247 m^3/day - 2 equalization basins - Aerobic MBR	247 m^3/day - 2 equalization basins - AFMBR - Adsorption unit

Table 4-3 Inventory of operation phase

	S1	S2	S3	S4
Purchase of Dongjiang water (m^3/d)	404	246	246	246
Treatment of freshwater from local yield (m^3/d)	135	82	82	82
Seawater treatment (m^3/d)	N.A.	211	N.A.	N.A.
Sewage treatment (m^3/d)	539	539	328	328
Sludge treatment (kg/d)	N.A.	N.A.	19	9
Water pumping (m^3/d)	539 (freshwater)	328 (freshwater) 211 (seawater)	328 (freshwater) 211 (greywater)	328 (freshwater) 211 (greywater)
Energy consumption of treatment unit (kWh/d)	N.A.	N.A.	128	19
Energy recovered (kWh/d)	N.A.	N.A.	N.A.	177
Chlorine dosage (kg/d)	N.A.	N.A.	1.2	1.2

More details of the system requirements are shown in Appendix 2. The detailed cost information of the materials required for the construction phase is listed in Appendix Table A2-31 to Table A2-34. The parameters used for the operation costs estimation and the breakdowns of operation costs are listed in Appendix Table A2-35 to Table A2-54. The inventories of conventional air emissions and greenhouse gas emissions of the construction phase are presented in Appendix Table A2-59 to Table A2-66.

4.4 Results

4.4.1 Life-cycle assessment results

The ReCiPe Endpoint method was adopted to conduct the environmental impact assessment of the scenarios in this study. The environmental emissions estimated based on experimental results, literature review, EIO-LCA and the Ecoinvent database were analyzed using the SimaPro software and seventeen impact categories were assessed. The characterization and single score results of the construction and operation phases are presented in Appendix Table A2-67 to Table A2-70. Figure 4-2 shows the environmental portfolios of the scenarios, which reveal the relative performance of the scenarios in each impact category (Appendix Table A2-71). Within each category, the scores assigned to the scenarios range from 0 to 1. The scenario with the highest impact is assigned to score 1 and serves as the reference. The scores of other scenarios reveal their performance relative to the reference scenario. The scenario with an environmental portfolio closest to the center, which is S4 in this case study, are the most environmentally favorable. The environmental portfolios also reveal that S1 and S2 generally perform less favorably in most of the impact categories.

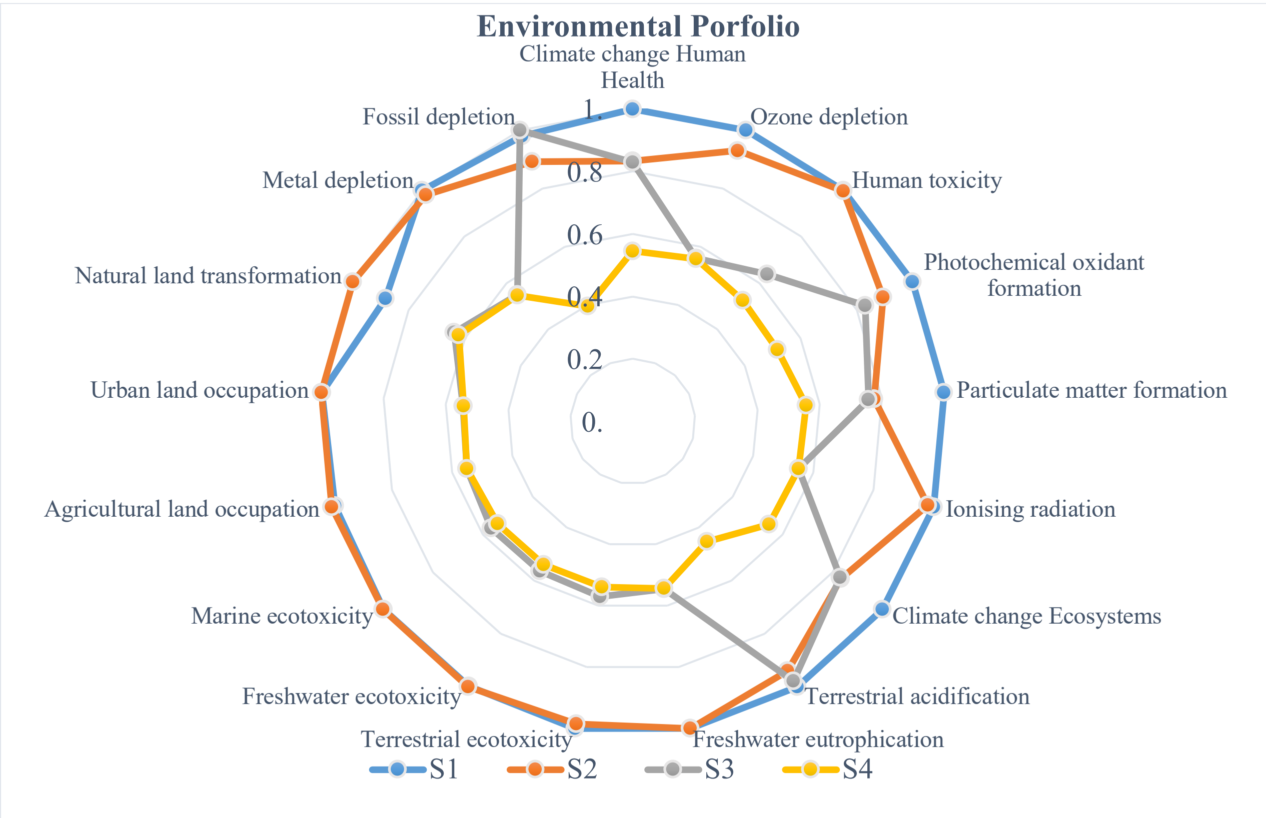

Figure 4-2 Environmental portfolio of non-potable water system scenarios

The environmental portfolio only shows the relative performance of alternatives in different impact categories; the different magnitudes of their effects on the total environmental burden are not considered. Thus, a standard LCA has been conducted to evaluate the total environmental performance of the scenarios and the results are shown in Figure 4-3 (Appendix Table A2-72). As the environmental impacts are normalized against the world reference inventories using the ReCipe method, the LCA results of the different indicators are of the same unit (points) and, therefore, a single score for each scenario could be obtained. The graph presents the summation of the normalized environmental impacts of each scenario. S4 induced the least environmental impact, followed by S2, S3, and then S1. The top three impacts caused by the water systems in this case study are on "climate change human health", "freshwater ecotoxicity" and "fossil depletion".

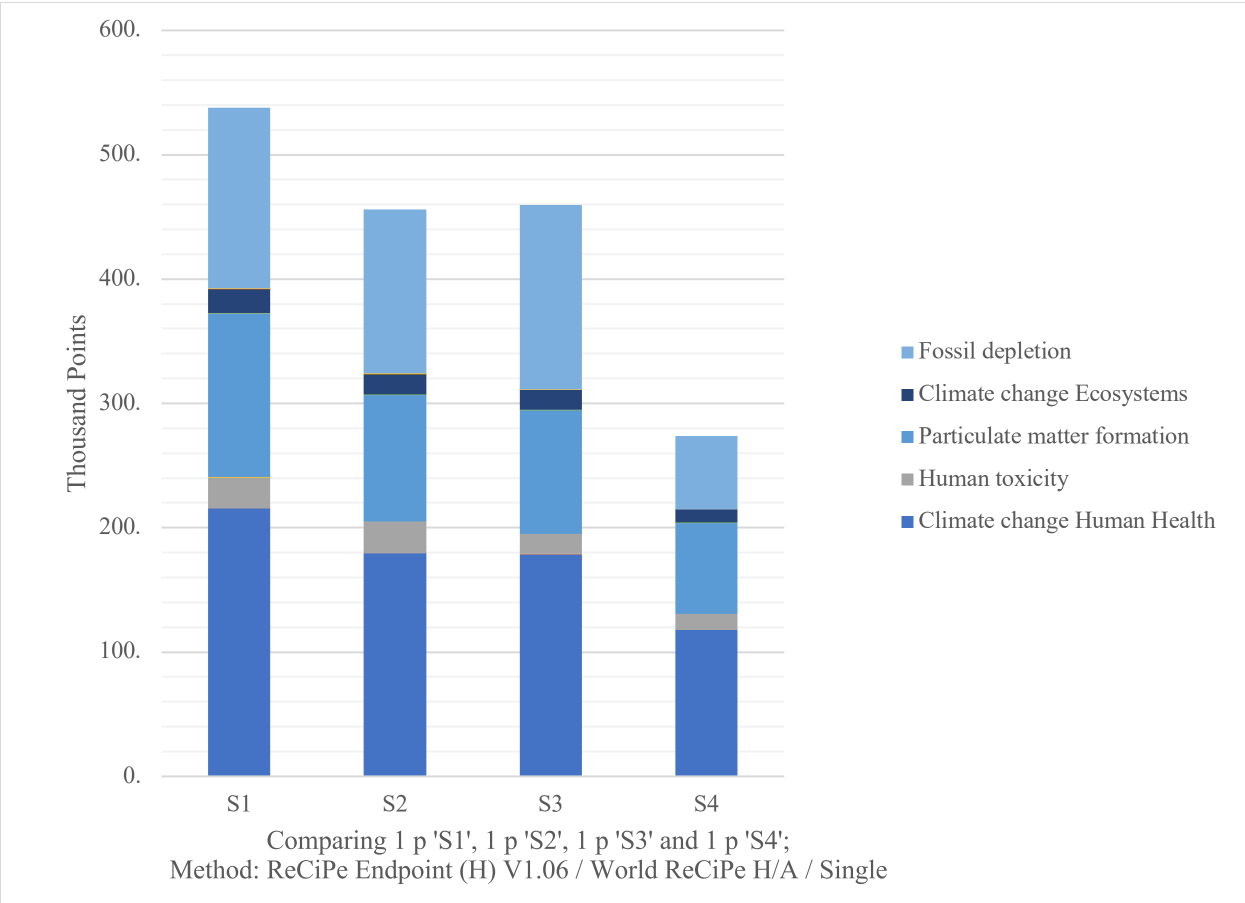

Figure 4-3 Life-cycle assessment results of non-potable water system scenarios

4.4.2 Life-cycle economic analysis results

The economic analysis results are shown in Figure 4-4. The total costs are estimated as the summation of the construction and the 20-year operation costs in net present values (NPV) (Appendix Table A2-55). The construction cost of S1 is the lowest, followed by S2 and S3, while that of S4 is the highest. The scenarios in ascending order of operation costs are S4, S2, S3 and S1. When considering both cost components, S2 and S4 has the lowest total cost and S1 presents the highest total cost.

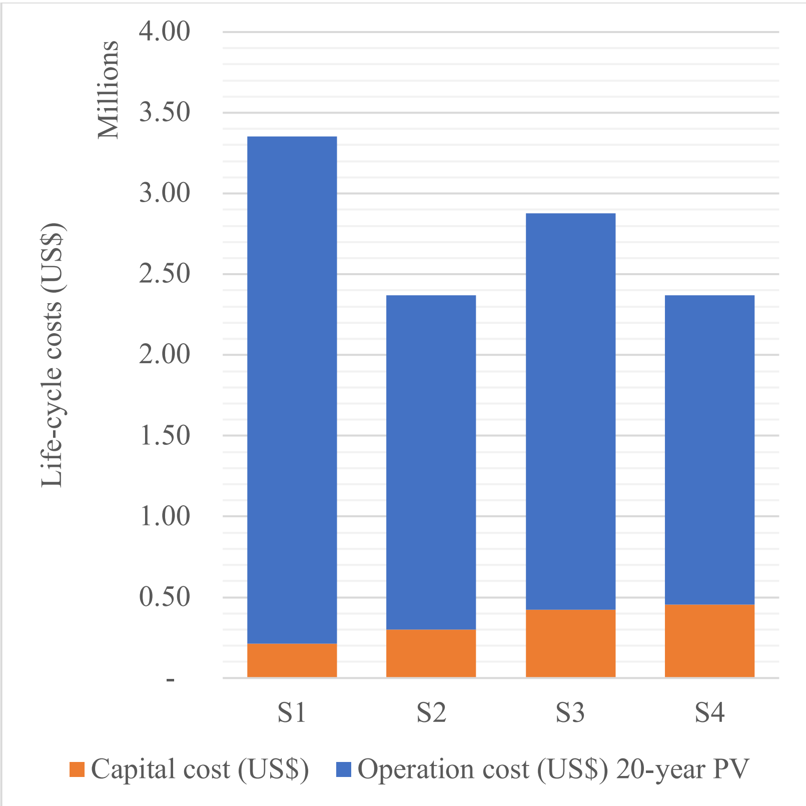

Figure 4-4 Life-cycle costs of non-potable water system scenarios

4.4.3 Eco-efficiency analysis results

Assuming that the environmental and the economic implications are of equal importance and complimentary, the LCA and the economic analysis results were linked in the

EEA (Appendix Table A2-73). The value 1 represents the average performance, and the vertical and the horizontal axes of the eco-efficiency portfolio show the relative environmental and economic performance of the scenarios respectively. With consideration of both aspects, the eco-efficiencies of the four water management scenarios in descending order are S4>S2>S3>S1 (Figure 4-5).

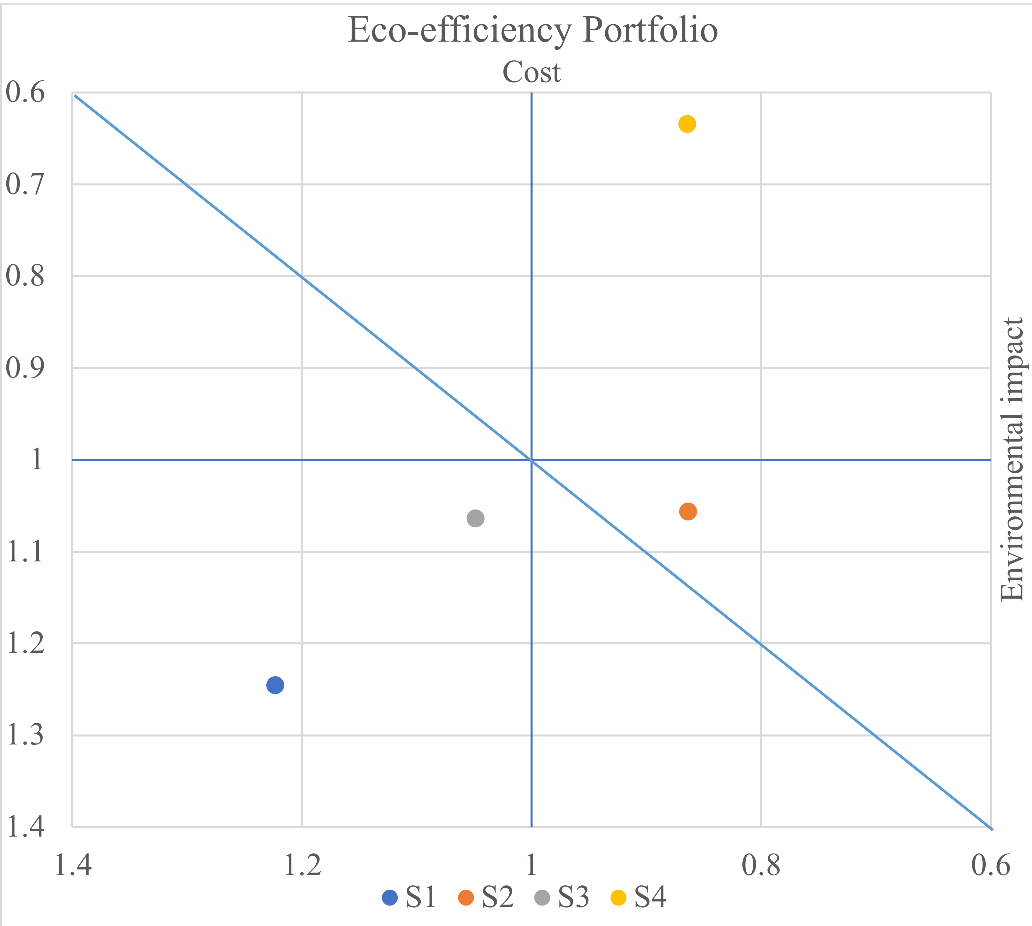

Figure 4-5 Eco-efficiency of non-potable water system scenarios

4.5 Discussions

4.5.1 Environmental impacts of non-potable water systems

The environmental impacts of the construction and the operation phases were evaluated in the environmental impact assessment. For the construction phase, the environmental emissions in descending order are S4>S3>S2>S1 (Figure 4-6 (a)).

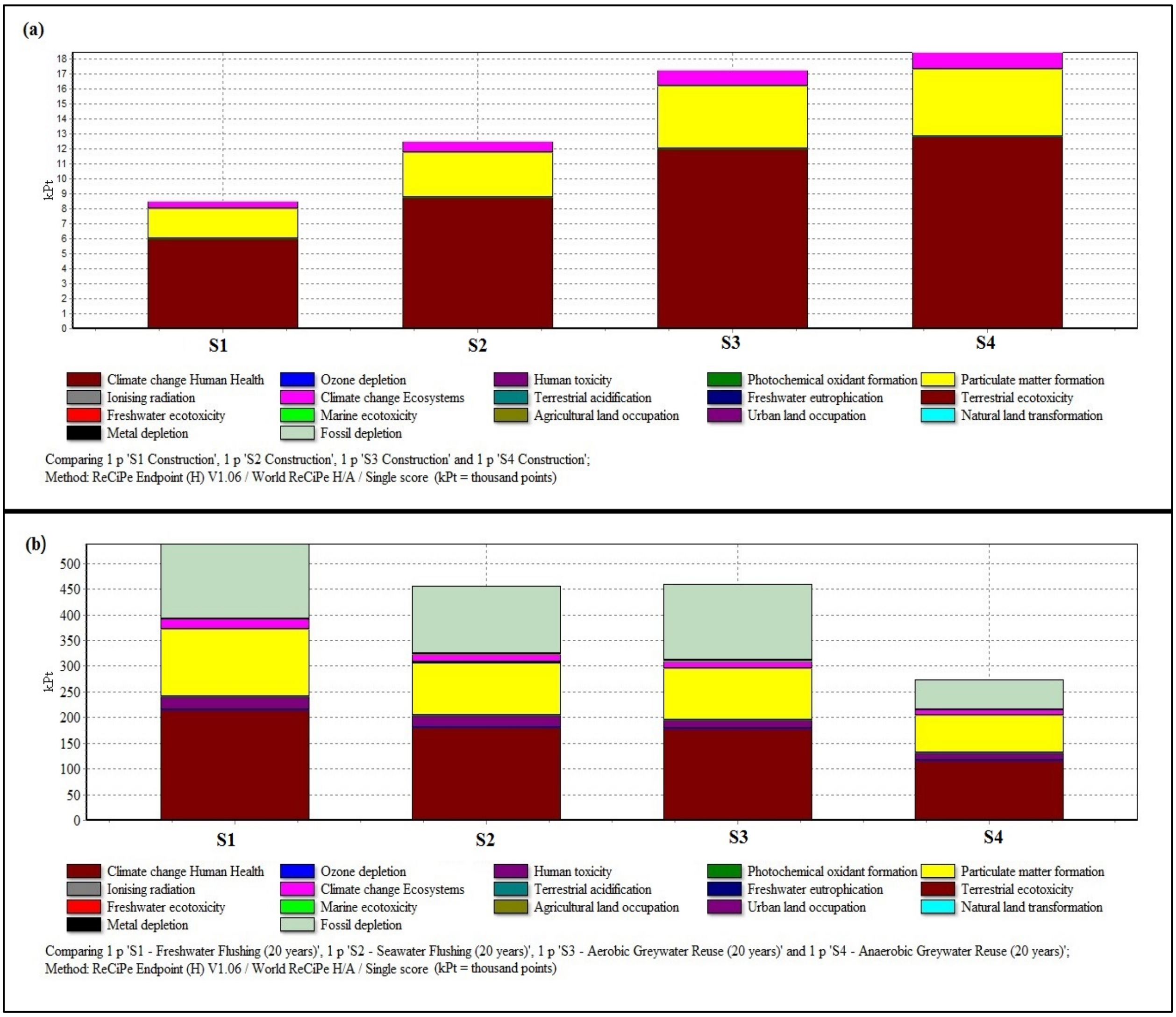

Figure 4-6 Life-cycle environmental impacts of (a) Construction and (b) Operation phases

The freshwater flushing scenario (S1) has the lowest environmental releases during the construction phase because it only requires a single plumbing system for water supply that covers toilet flushing and other water uses (Table 4-2). The water supply plumbing system for the other three scenarios are similar: one pipeline system for supporting general domestic water demand except toilet flushing, and one pipeline system for supplying either seawater or treated greywater for flushing. The higher environmental impacts of S3 and S4 relate to the material requirements of the greywater treatment units, which were mainly composed of the aerobic or anaerobic bioreactors, the adsorption unit, and the equalization basins.

In the operation phase, the S4 presented the lowest environmental impact, followed by S2 and S3, while S1 showed the highest impact (Figure 4-6 (b)). There are three primary factors contributing to the environmental impacts associated with the operation phase, namely the drinking water purification, electricity production, and sewage treatment. Because of the high freshwater demand in S1, potable water treatment is the major cause of emissions in this scenario, contributing to 43.6% of the impact. In comparison to S1, the drinking water requirement in S2 is lower but the electricity consumption is higher due to the dual pumping systems for freshwater and seawater. Due to the recycling of greywater for non-potable uses, the sewage treatment loadings for S3 and S4 are lower than those for the other two scenarios. Both the greywater recycling scenarios require electricity for the operation of the on-site treatment units, yet the electricity consumption of S3 is higher than in S4. This difference derives from the energy-intensive aeration requirement for the aerobic greywater treatment unit in S3, as well as the energy recovery via methane production achieved by the anaerobic system in S4.

After integrating the environmental impact assessment results of the construction and operation phases with the ReCiPe method, S4 was revealed to be the most environmentally friendly scenario, followed by S2 and then S3, leaving S1 to be the least favorable option to the environment (Figure 4-3).

4.5.2 Economic costs of non-potable water systems

Different from the other three scenarios that require a separate seawater or greywater system, S1 only requires a single plumbing and tank system for conveying freshwater for all potable and non-potable uses. Thus, the material requirements and capital cost (US$211,000) could be the lowest among the scenarios. The plumbing design for the greywater systems is the same as the seawater system, yet the greywater treatment units have a notable impact on the capital cost. The costs of aerobic (US$177,000) and anaerobic greywater treatment units (US$222,000) were estimated to account for 42% and 48% of the total capital cost of S3 and S4 respectively.

For S2, S3 and S4, a proportion of freshwater demand is replaced with either seawater or treated greywater, with the potential to significantly lower freshwater cost. S4 has the lowest operation cost among the scenarios due to several reasons. Firstly, the greywater recycling strategy alleviates the freshwater demand of the scenario, so the expenditure on purchasing Dongjiang water and treatment of locally yielded freshwater could be reduced. Secondly, as the anaerobic greywater treatment by AFMBR has a low sludge production rate, the treatment cost for sludge produced on-site is lower compared to S3. Lastly and most importantly, energy could be recovered by the AFMBR through methane production, thus earnings could be made from electricity generation. Although the same amount of freshwater demand could be avoided, the aerobic greywater recycling scenario S3 has a higher operation cost than S2 and S4, mainly due to the aeration energy requirement (Lo et al., 2015). The relatively high sludge production rate in greywater treatment increases the sludge treatment cost of the scenario, yet the economic impact on the operation cost is trivial. The high operation cost of S1 is mainly attributed to the freshwater cost, more specifically the purchase of Dongjiang water. This study has taken a conservative estimation: the reduction of freshwater demand in S2, S3 and S4 is shared by proportion among the Dongjiang (75%) and the local supply (25%). If the greywater reuse, together with other sustainable water management strategies, could be applied on a large scale, the freshwater demand could be remarkably reduced and reliance on Dongjiang water could be lessened. As the price of Dongjiang water (US$1.11/$m^3$) is more than double the treatment cost of local freshwater (US$0.51/$m^3$), if 100% of the freshwater demand reduction is borne by the purchase of Dongjiang water, the costs of S2, S3 and S4 could be further decreased (Research Office of Legislative Council Secretariat, 2015).

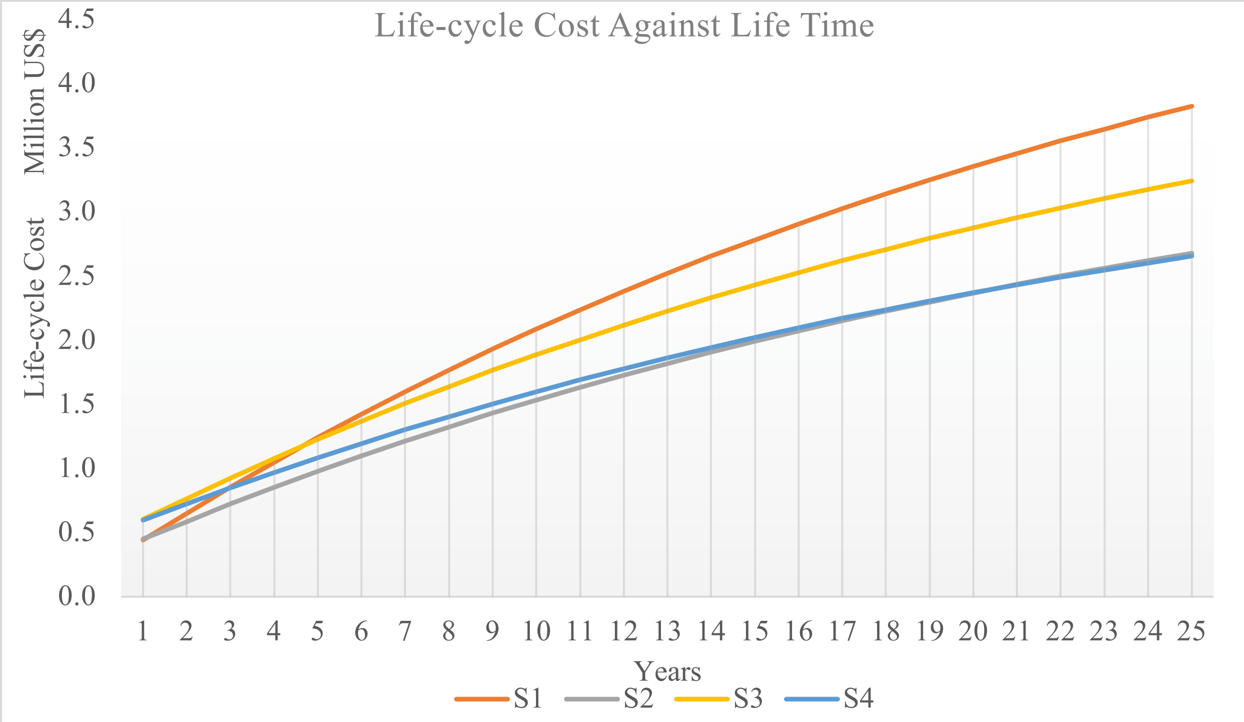

Figure 4-7 Life-cycle cost against life time

The capital cost was considered as a single investment during the first year of the time period in this study and the operation cost was an annual expenditure. The total life-cycle cost was calculated as the summation of the capital cost and the annual operation cost, and was presented in NPV. The time period of operation is a crucial factor in determining the most economically favorable option. Thus, this study also investigated the time factor in the economic analysis (Appendix Table A2-56). Figure 4-7 shows the change in total life-cycle cost according to the length of time period considered. In the first year of operation, the scenarios that adopted greywater recycling (S3 and S4) show less favorable economic performance at the total cost of US$600,000, which is US$150,000 more expensive than the other two scenarios. However, starting from the fifth year of operation, the freshwater scenario (S1) becomes the least economically favorable option. The anaerobic greywater reuse system is the most economically advantageous option when applied for 21 years or longer.

The purpose shared between the implementation of seawater flushing in Hong Kong and the proposal of greywater recycling system for toilet flushing is the alleviation of freshwater demand. Greywater and seawater could be regarded as substitute goods to freshwater, thus a rise in freshwater price could lead to an increased demand for the utilization of seawater or greywater, and vice versa. The influence of Dongjiang water price on the life-cycle costs of the scenarios was investigated (Figure 4-8 and Figure 4-9).

Figure 4-8 shows that when the Dongjiang water price is the same as the local freshwater treatment cost (US$0.51/$m^3$, equals to HK$4/m^3 [1]), the freshwater flushing option (S1) becomes economically competitive within a short period of operation (Appendix Table A2-57). The total life-cycle cost of S1 is the lowest among the water management alternatives in the first two years of operation. Seawater flushing is the most economical for operation periods ranging from 3 to 20 years. The case with the assumption of a doubled Dongjiang water price was also analyzed (Figure 4-9). The high price of imported water caused the freshwater flushing option (S1) to be the least economically favorable for operation of more than 2 years (Appendix Table A2-58). For an operation period of 21 years or above, the most economically feasible option would be anaerobic greywater reuse (S4) no matter whether the change in imported water price is halved or doubled.

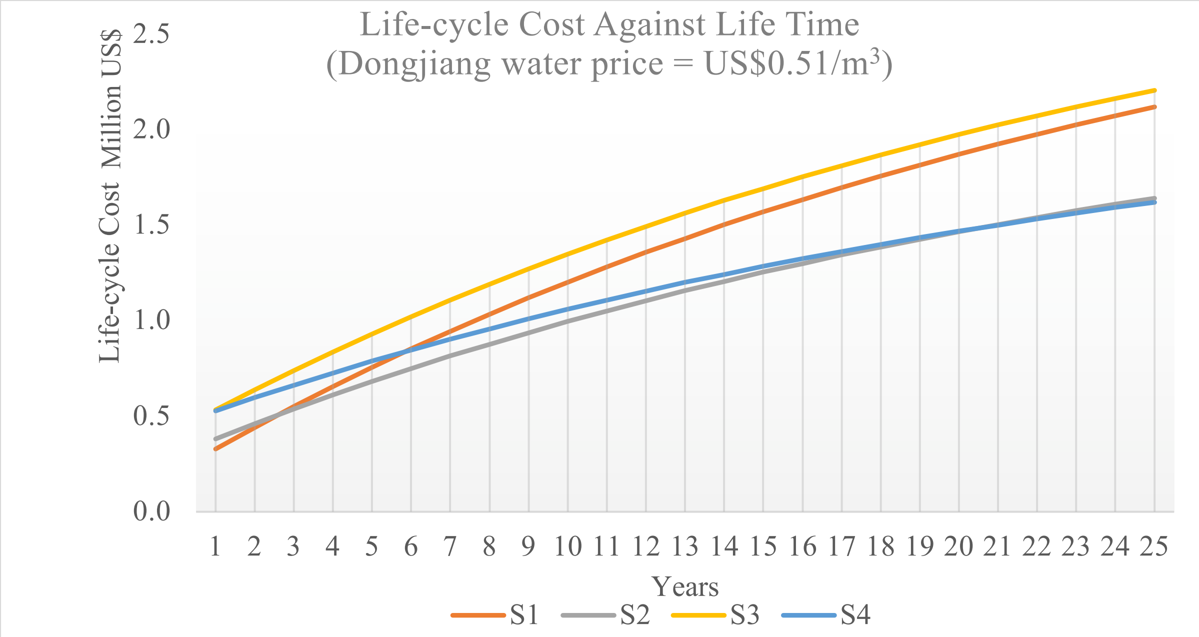

Figure 4-8 Life-cycle cost against life time (Dongjiang water price = US$0.51/$m^3$)

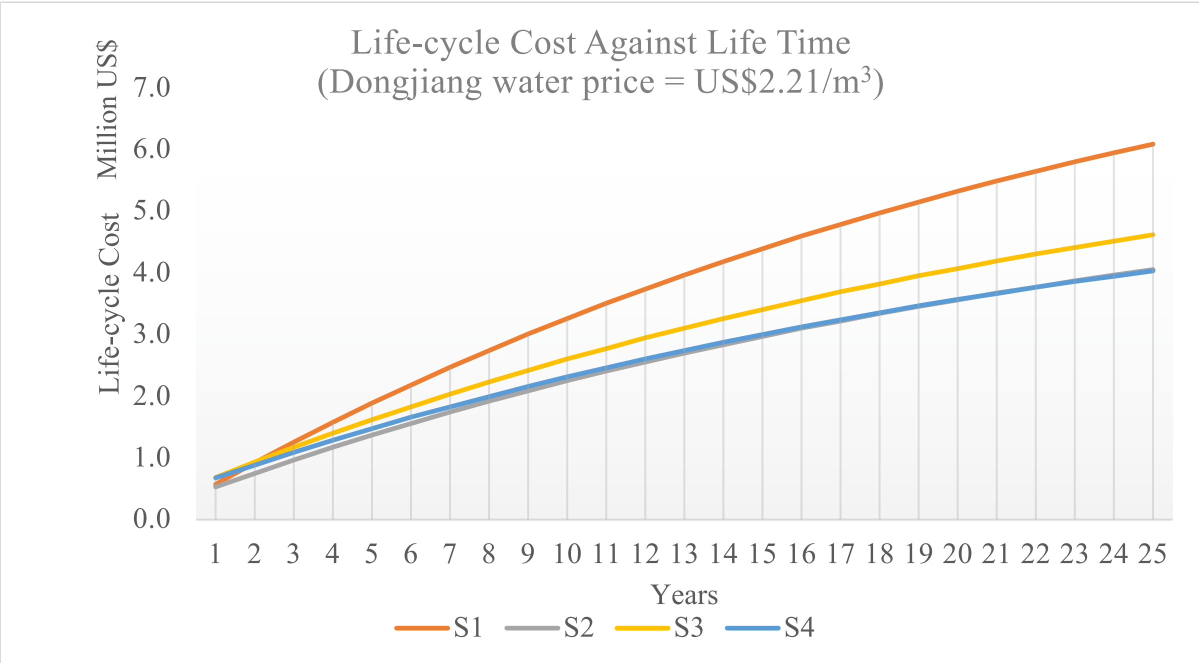

Figure 4-9 Life-cycle cost against life time (Dongjiang water price = US$2.21/$m^3$)

4.5.3 Eco-efficiency of non-potable water systems

The EEA results reveal S4 to be the most eco-efficient water management option, which coincides with the advantageous economic and environmental performances of the scenario as shown in the economic analysis and environmental impact assessment. Its ability to recover energy and water resources along with its ability to reduce freshwater demand and sewage treatment requirement are key features that favor the implementation of an anaerobic greywater reuse system. S2 is the second most eco-efficient scenario, followed by S3. Although the two scenarios performed similarly in terms of environmental impacts, S2 is more economical as it does not require the construction of the greywater treatment system and avoids some costly operation processes, such as aeration and disinfection. However, it should be noted that seawater flushing is not a common practice in the world and Hong Kong is one of the few regions that has adopted such a practice. Based on the EEA results, the seawater flushing option nonetheless remains the second-best strategy to be adopted in regions where a seawater system is technically feasible. For inland regions, an aerobic greywater reuse system could be a second favorable choice. A freshwater system is the least favorable option among the scenarios in terms of both the economic and environmental performance.

Sensitivity to R_{EC} Ratio

The sensitivity of the results to the R_{EC} ratio was tested as the relative importance of the environmental and economic performance may influence the final results of the eco-efficiency of the scenarios. Equal importance of the two aspects was first assumed in this study, and then two R_{EC} ratios, including 100 and 0.01, which represent high significance of the environmental and the economic influences respectively, were tested (Figure 4-10, Appendix Table A2-73). The denotations S1', S2', S3' and S4' represent the S1, S2, S3 and S4 with changed portfolio positions due to different R_{EC} ratios used. The eco-efficiency portfolio positions of the scenarios change accordingly with different R_{EC} ratios, yet the anaerobic greywater recycling system remained the most favorable option and the sequence of the scenarios in terms of eco-efficiency remained unchanged in both situations. As a high R_{EC} ratio represents high importance placed on environmental performance, the difference in environmental profiles (Y-coordinates) of the scenarios were amplified when the R_{EC} ratio is 100 (Figure 4-10 (a)). In contrast, the divergence between the economic profiles (X-coordinates) of the scenarios was enlarged when R_{EC} ratio is 0.01, which places much greater weight on the economic aspects than on environmental impact (Figure 4-10 (b)). The results of the sensitivity test reveal that the anaerobic greywater reuse scenario (S4) remains the most efficiency option regardless of the R_{EC} ratio shifting from 0.01 to 100.

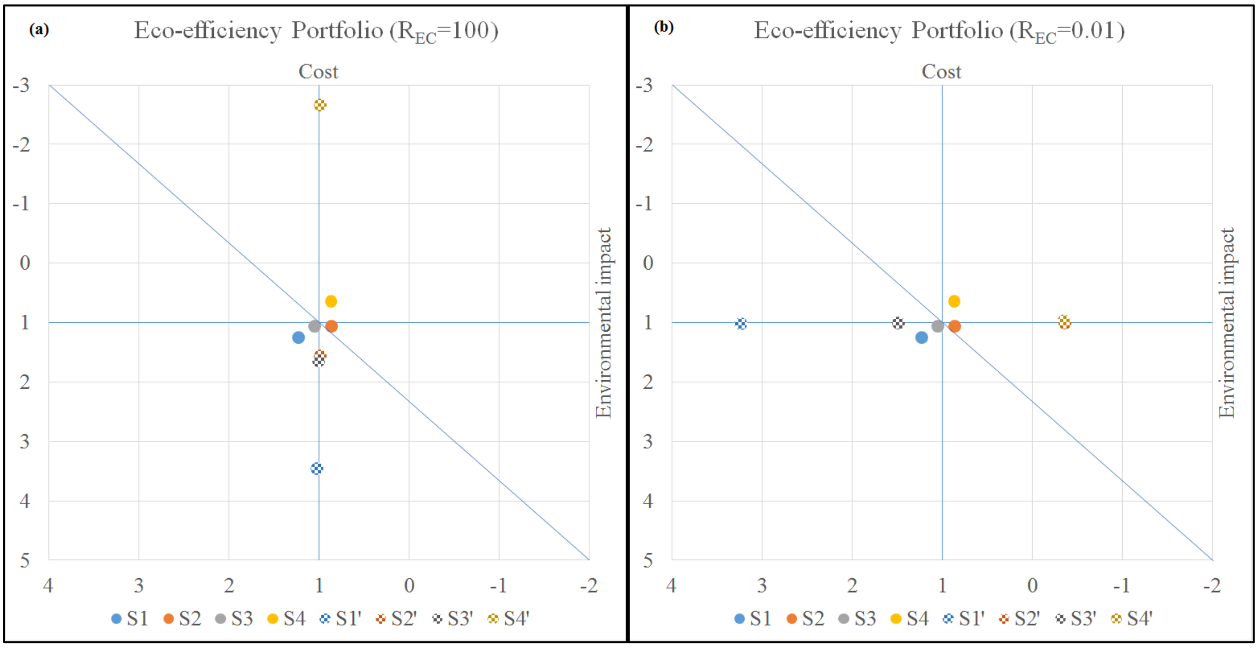

Figure 4-10 Eco-efficiency portfolio ((a) R_{EC}=100; (b) R_{EC}=0.01)

4.6 Summary

Sustainability in water management and harvesting energy from wastewater have been gaining more attention in the past decade. The anaerobic greywater reuse system employing AFMBR technology for non-potable uses have been proposed to promote sustainability, and the technical feasibility of the system has been tested. For the purposes of assisting decision-making and informing the public effectively, this study developed an EEA framework to comprehensively evaluate the eco-efficiency of decentralized water management options by linking their economic and environmental performances. The inclusive and clear results of this EEA study are presented with an EE portfolio to facilitate convenient communication to decision-makers, the public, and the other users. Unlike previous research that focused on the technical performance of the reactors, this study compared advanced greywater recycling with other options on a single-building scale with the detailed engineering designs of the systems. The EEA framework was demonstrated using a case study in Hong Kong.

The results of the EEA indicate that the anaerobic greywater reuse system (S4) would be the most eco-efficient option to adopt (Figure 4-5). The features of the system enable energy recovery, recycling of water resources, and alleviation of sewage treatment loading, thus giving the option both economic and environmental advantages over the other strategies. Considering freshwater and greywater as substitute goods for non-potable uses, this study also investigated the effects of changes in freshwater price on the economic favorability of the scenarios against time period. The anaerobic greywater reuse option was concluded to be the most economically advantageous scenario for operation over 21 years or longer (Figure 4-7). The study offers an improved decision-supporting tool for water management and demonstrates that the EEA framework fits the purpose of evaluating the sustainability of greywater reuse systems.

This study only evaluated water systems within a building, including the plumbing, piping, pumping, and greywater treatment components, the system for water conveyance outside the building being excluded. The economic and environmental consequences of constructing and operating the water conveyance and distribution systems also depend on the distance between facilities and the density of the community. To further refine the EEA framework developed in this study, the application of greywater recycling in a community or on a city-wide scale should be evaluated in the future. Geographic information and temporal patterns of water consumption should also be considered to further increase the comprehensiveness of the evaluation tool.

CHAPTER 5

TECHNO-ENVIRONMENTAL PERFORMANCE OF SLUDGE-TO-ENERGY SYSTEMS

5.1 Introduction

With rapid population growth and economic development, urban cities are coping with imminent waste and energy problems. Considering the high population densities and limited land resources in urban areas, the modern waste hierarchy, in addition to dealing with public hygiene problems, emphasizes energy recovery from waste as a simultaneous solution to both the waste and energy problems (Ohnishi et al., 2018; Sun et al., 2018). Sewage sludge, as an energy-rich biomass waste, is often treated as a source for renewable energy through waste-to-energy technologies, such as anaerobic digestion (AD) and incineration (Houillon & Jolliet, 2005; Milbrandt et al., 2018). The feasibility to improve energy recovery from sludge through technical means, such as altering hydraulic retention time (Ruffino et al., 2019) and implementing pretreatment (Cano et al., 2015), has been widely interested and studied. The energy-recovering treatment technologies are adopted in combinations with different techniques, such as thickening, dewatering, aerobic digestion and drying, for treating sludge in wastewater treatment plants (WWTPs) before disposal or final uses. The treated sludge can be disposed of at landfills or used for land application and cement production (Murray et al., 2008a). Numerous factors, such as the conditions of inlet sludge, types of treatment technologies, chemical dosages and configurations of processes, affect the performance efficiency of sludge treatment and sludge-to-energy (STE) systems. Although energy recovery from sludge are mainly achieved through AD and incineration, the selection and configurations of the other treatment processes influence the energy and materials inputs, as well as emissions and energy outputs of the STE systems. Therefore, it is crucial to evaluate the interrelated sludge treatment processes from a system perspective, meaning that no single process should be excluded from benchmarking the STE systems.

In the field of sewage sludge management, numerous studies using life-cycle assessment (LCA), economic analysis and other tools have been conducted to evaluate the performance of different sludge treatment approaches. The environmental impacts of conventional sludge treatment scenarios, including stabilization processes, transportation and disposal, were evaluated using LCA (Remy et al., 2013; Suh & Rousseaux, 2002). LCA has also been applied to assess sludge co-digestion and composting with food waste (Di Maria et al., 2016) and AD-based sludge treatment technologies (Sadhukhan, 2014). Economic analysis has been conducted in combination with LCA to evaluate sewage sludge management options in Japan (Hong et al., 2009), Hong Kong (Lam et al., 2016), China (Xu et al., 2014a) and Sweden (Lundin et al., 2004).

Some research studies focused on evaluating STE systems. The study conducted by Ren et al. identified the critical barriers that hindered the sustainable development of STE systems in China using grey Decision Making Trial and Evaluation Laboratory approach, followed by prioritizing STE technologies using the developed grey Multi-Criteria Decision Making method (Ren et al., 2017). Policy implications were proposed to stakeholders for developing STE technologies in China and investment should be made to technologies in descending priority of AD with gas engine, AD with fuel cell and incineration. Mills et al. used an economic and environmental LCA to compare five AD-based STE technology configurations (Mills et al., 2014). By ranking and scoring the five configurations based on their performance in net environmental impact, global warming potential, and internal rate of return with and without government incentive, Mills et al. concluded Thermal Hydrolysis Process AD with CHP followed by drying of digested sludge for solid fuel production to be the most sustainable option. Samolada & Zabaniotou (2014) conducted SWOT analysis to compare the sustainability of three thermal STE technologies, including incineration, gasification and pyrolysis, in Greece. Based on the SWOT analysis concerning four criteria, namely the effectiveness to solve wastewater problem, GHG emissions, maturity of technology and legislation, the research study revealed pyrolysis to be the most favorable option for managing sludge in Greece. A comparative LCA was conducted to investigate the environmental impacts of two STE incineration scenarios using fluidized bed combustor and cement kiln in Turkey and revealed the former scenario to be more environmentally favorable (Abuşoğlu et al., 2017). Another LCA focused on the energy and GHG footprint of STE systems applying fast pyrolysis with and without AD and concluded that the option with AD achieved better performance (Cao & Pawłowski, 2013).

Evaluating the techno-environmental performance of STE systems, which includes assessing the treatment levels achieved by treatment processes (for example, measured as volatile solid (VS) reduction), energy balance and final waste disposal, is essential for benchmarking the efficiency of STE systems. However, it has not yet been investigated in previous research studies. To fill this research gap, this study proposes a multi-criteria decision analysis using an integrated LCA and data envelopment analysis (DEA) approach to evaluate the operation efficiency of STE systems. This research study developed a life-cycle data envelopment analysis (LC-DEA) framework for benchmarking STE systems through the integration of LCA and DEA. This study contributes to illustrate the strong interrelationship between modern sludge treatment systems and energy systems, thus provides insights on decision-making for sludge management from an energy system perspective. This is the first LC-DEA study conducted on sewage sludge management that views the sludge treatment scenarios as waste treatment systems and energy systems simultaneously. This study aims to benchmark the techno-environmental efficiency of different STE systems through DEA. The environmental performance was evaluated by LCA. Areas and targets for improvement were identified for inefficient systems so that recommendations could be made to assist decision-makers or operators to enhance the efficiency of such systems.

5.2 Materials and method

5.2.1 Sludge treatment processes

Sewage sludge is the by-product of wastewater treatment processes. Containing health-threatening contents including pathogens and other organic substances, sewage sludge requires proper treatment before final disposal or reuse. Generally, primary sludge and surplus activated sludge collected from primary and secondary wastewater treatment processes will undergo a series of sludge treatment processes, such as thickening, digestion, dewatering and incineration (Figure 5-1). Electricity, fuel and chemicals are required for such processes. Energy recovery could be achieved through AD and some of the incineration technologies. The treated sludge will be dispose of at landfills or reuse for different purposes, such as land application and cement production (Murray et al., 2008a). This study focuses on the operation of sludge treatment processes, which is from the inlet of sludge to the generation of wastes, such as dewatered sludge cake and incineration ash (Figure 5-1).

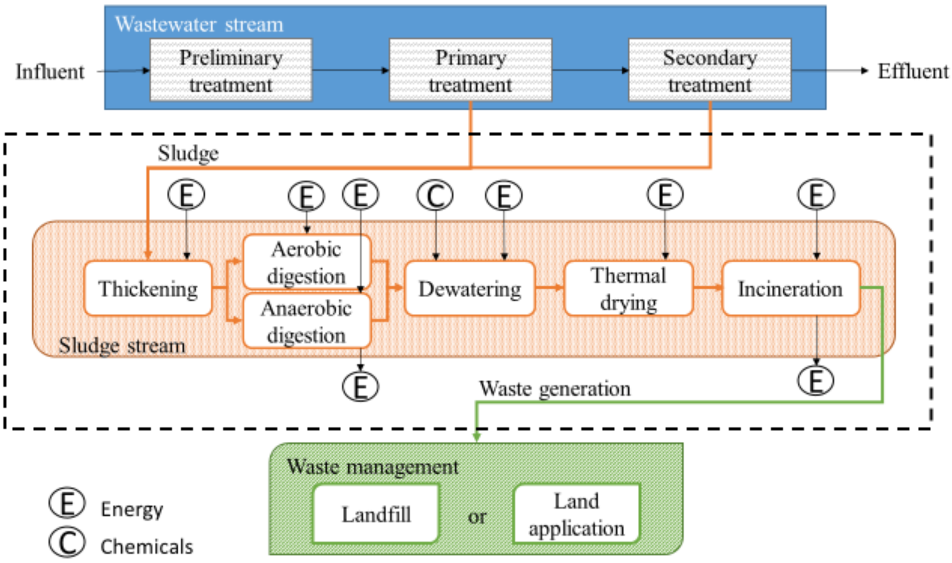

Figure 5-1 Schematic diagram of wastewater and sludge treatment processes

5.2.2 Sludge treatment processes and data in this study

Enhancing waste-to-energy systems in urban cities is of high significance due to the characteristics of high population density and lack of land resources in such areas. Waste generation and energy use of densely populated areas are immense. STE approach is a favorable solution for cities running out of landfill space, such as Hong Kong, to tackle waste and energy problems simultaneously with limited land resource. Therefore, this study focuses on the STE scenarios in Hong Kong by including nine sewage treatment works (STWs) that contribute nearly 100% of sludge generation in Hong Kong. To reveal the relative performance of Hong Kong STE systems among those in other cities and countries, this study compared the nine STE scenarios in Hong Kong with seven non-Hong Kong scenarios, which are sludge treatment systems in urban cities in different countries. All the scenarios were defined as DMUs in DEA, thus there are sixteen DMUs in this study.

5.2.2.1 Hong Kong sludge treatment and data

In Hong Kong, the wastewater and sludge treatment services provided by the STWs are managed by the Drainage Services Department (DSD) of the government. There are in total 69 STWs providing preliminary, primary, secondary, and tertiary sewage treatment in Hong Kong, with four of the major secondary STWs applying AD to treat sludge with energy recovery (DSD, 2015). Sludge thickening and dewatering are employed by the STWs to reduce the water content, and thus the volume, of sludge before disposal. A new sludge treatment facility has recently begun operating in Hong Kong to deliver waste-to-energy through sludge incineration. The facility treats 2,000 tonnes of sewage sludge per day and is capable of reducing 90% of the sludge volume (EPD, 2016).

Data of the DMUs in Hong Kong were collected from the DSD and the Hong Kong Environmental Protection Department (EPD). Nine of the local STWs that contributed nearly 100% of sludge generation in Hong Kong, namely the Shatin (DMU01), Tai Po (DMU02), Shek Wu Hui (DMU03), Yuen Long (DMU04), Stonecutters Island (SCI) (DMU05), Siu Ho Wan (DMU06), Stanley (DMU07), Sham Tseng (DMU08) and Sai Kung (DMU09) STWs, were included in this DEA. Information on raw sludge, sludge thickening, AD, dewatering and the operation of the STWs was collected from the DSD. The data on incineration and landfills, was obtained from the EPD. For the unavailable data, information from the relevant published literature and databases was used.

5.2.2.2 Overseas sludge treatment and data

To evaluate the relative performance of the Hong Kong DMUs compared to sludge treatment scenarios overseas, seven DMUs of non-local sludge treatment systems were included in this DEA. Data on sewage treatment plants located in China, Korea, Italy, Japan, and Denmark was acquired from literatures.

The study conducted by Murray et al. (2008) evaluated four biological secondary wastewater treatment plants in Chengdu, the capital of Sichuan Province, China. The four plants produced 84 dry tonnes of sludge per day. Two sludge handling scenarios in Chengdu were extracted from the study for inclusion in the DEA in this study. The two extracted scenarios involved AD (DMU10) or aerobic digestion (DMU11), followed by sludge dewatering and land application. For land application, the treated sludge was used for replacing the phosphorus and nitrogen fertilizers.

Piao et al. (2016) studied the performance of the wastewater and sludge treatment practices of the WWTPs in Korea. The data on sludge treatment from two WWTPs was extracted for DEA evaluation: WWTP-N (DMU12), which administered a conventional activated sludge process, and WWTP-S (DMU13), which employed an anaerobic/anoxic/oxic (A2O) process. In both cases, sewage sludge was treated by thickening, AD, and dewatering before landfill disposal.

The performance of wastewater and sludge treatment processes was investigated in the study conducted by Tomei, Bertanza, Canato, Heimersson, Laera and Svanström (Tomei et al., 2016). The sludge treatment data of a secondary WWTP in Italy with a treatment capacity of 70,000 person-equivalents was extracted for DEA evaluation in this study. Sewage sludge generated from wastewater treatment was treated by thickening, AD and dewatering before land application (DMU14).

The data of a sewage sludge treatment plant in Japan was extracted from the research conducted by Soda, Iwai, Sei, Shimod and Ike (Soda et al., 2010). The treatment processes included thickening, AD, dewatering, incineration, and landfill disposal (DMU15).

The performance of the Avedøre wastewater treatment plant, which is located in Copenhagen, Denmark, was evaluated by Yoshida et al. (2015). The plant was a secondary WWTP that treats 25.3 million m^3 of wastewater per year. The sewage sludge, which mainly came from the primary and secondary wastewater treatment processes, was treated by AD, dewatering, drying, and incineration, before being disposed in a landfill (DMU16).

The sixteen DMUs evaluated in this DEA study are summarized in Table 5-1. The DMUs consist of different combinations of sludge treatment technologies, including thickening (T), anaerobic digestion (AD), aerobic digestion (AeD), dewatering (Dw), drying (Dry) and incineration (I), as well as final destinations of end-products, such as landfill (Lf) and land application (LA).

Table 5-1 Summary of sludge treatment DMUs

DMUs	Treatment processes*								Remarks
	T	AD	AeD	Dw	Dry	I	Lf	LA	
01	●	●		●		●	●		Shatin
02	●	●		●		●	●		Tai Po
03	●	●		●		●	●		Shek Wu Hui
04	●	●		●		●	●		Yuen Long
05				●		●	●		Stonecutters
06				●		●	●		Siu Ho Wan
07	●			●		●	●		Stanley
08				●		●	●		Sham Tseng
09			●						Sai Kung
10		●		●				●	Chengdu S3
11			●	●				●	Chengdu S4
12	●	●		●			●		Korea N-WWTP
13	●	●		●			●		Korea S-WWTP
14	●	●		●				●	Italy
15	●	●		●		●	●		Osaka
16		●		●	●	●	●		Copenhagen

* T = thickening; AD = anaerobic digestion; AeD = aerobic digestion; Dw = dewatering; Dry = drying; I = incineration; Lf = landfill, and; LA = land application.

5.2.3 LC-DEA framework for STE systems

This study emphasizes the strong linkage between sludge treatment systems and energy systems. Figure 5-2 shows the system boundaries of the STE systems when viewed from waste treatment and energy system perspectives. The system boundary of waste treatment systems generally includes the input of waste (e.g. sludge), chemical consumption and energy use, as well as environmental impacts and waste generation. The function provided by sludge treatment systems is the reduction of organic matters. Energy recovery from sludge could be another function or the by-product of sludge treatment. The system boundary of energy systems typically includes the inputs of fuels and auxiliary energy, outputs of environmental impacts and wastes, and the generation of electricity and/or heat. Similarity between waste treatment and energy system perspectives is observed due to the overlapping inputs, outputs and function of the STE systems. While the sludge management philosophy is shifting from merely tackling hygienic problems to extracting energy from waste, STE systems are serving as waste treatment and energy systems simultaneously. Thus, the boundary of STE systems includes both perspectives to comprehensively cover the environmental and technical efficiency of the systems. Although the DMUs include different sludge treatment processes, all the DMUs serve the same functions, which are to reduce pollutants from sludge before disposal and recover energy from sludge. The DMUs also have the same types yet varying quantities of inputs, including chemical and energy use, as well as undesirable outputs, including environmental impacts and waste generation. Therefore, all these factors are included in the LC-DEA for comprehensive efficiency evaluation.

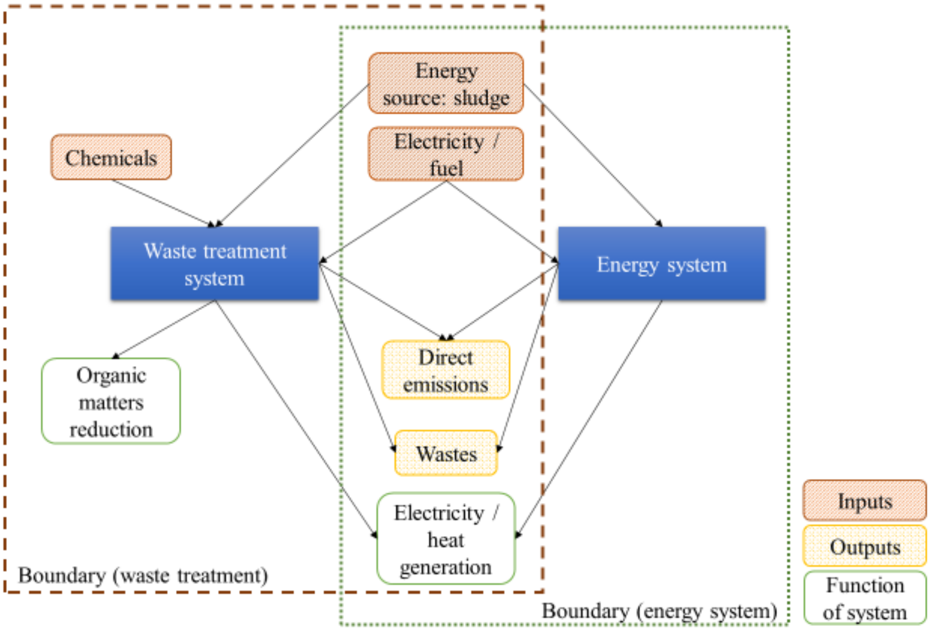

Figure 5-2 System boundaries of STE systems in perspectives of waste treatment system and energy system

The scope of this LC-DEA for STE systems includes components of technical and environmental efficiency evaluation. Common indicators for technical efficiency of energy systems include electricity generation and direct energy use (Martín-Gamboa et al., 2017), while indicators for WWTPs include contaminants removal (Castellet & Molinos-Senante, 2016; Hernández-Sancho & Sala-Garrido, 2009). Thus, this LC-DEA includes VS reduction and energy recovery as the outputs. To consider the environmental performance, this study includes chemical consumption, energy use, waste generation and environmental impacts from the treatment processes. The inclusion of such factors is consistent with previous DEA and LCA studies on WWTPs (Corominas et al., 2013; Lorenzo-Toja et al., 2015; Torregrossa et al., 2018). As the characteristics and amounts of chemicals used in sludge treatment are highly variable, this study uses environmental impacts for a unified indicator for measuring and comparing chemical consumption of the DMUs (Lorenzo-Toja et al., 2015; Torregrossa et al., 2018). Waste generation in this LC-DEA measures the volume of sludge residues to be disposed of at landfills; treated sludge reused for other purposes, such as land application, is excluded from waste generation.

The relative efficiency of each DMU with reference to the most efficient DMUs was obtained as the results of this study and the areas that required improvements were identified for the inefficient plants. This study demonstrated the application of the LC-DEA framework through benchmarking the efficiency of STE systems in Hong Kong against the overseas practices.

5.2.4 Methodology of life-cycle data envelopment analysis (LC-DEA)

5.2.4.1 Performance evaluation by DEA

DEA is a mathematical programming model for evaluation of a set of peer entities, which are referred to as DMUs, through empirically obtaining the production frontier or best-practice frontier from the observed data (Cooper et al., 2004; Zhu, 2014). In this study, the DMUs were defined as the sixteen sludge management scenarios described in Section 5.2.1. Serving as a benchmarking tool, DEA identifies the best-practice DMUs and evaluates the relative efficiency with respect to the conversion of multiple inputs into multiple outputs. Such performance evaluation and benchmarking using DEA enable operations to become more productive and efficient by revealing the strengths and weaknesses of processes, as well as identifying opportunities for improvements to meet operation targets in more efficient ways (Zhu, 2014).

5.2.4.2 DEA model

Envelopment DEA models have been developed to identify the efficient (best-practice) frontier that envelop all the observed DMUs. Assume that there are a set of n DMUs that consume m types of inputs to produce s types of outputs. DMU_j consumes x_{ij} of input i to produce y_{rj} of output r. It is assumed that $x_{ij} \geq 0$ and $y_{rj} \geq 0$, and each DMU has at least one positive input and one positive output. DEA models demonstrate two underlying properties, namely convexity and inefficiency (Zhu, 2014). The property of convexity defines the level of inputs and outputs that could be utilized and produced by DMU_j as $\sum_{j=1}^{n} \lambda_j x_{ij}$ (i = 1, 2, …, m) and $\sum_{j=1}^{n} \lambda_j y_{rj}$ (r = 1, 2, …, s), respectively. The λ_j (j = 1, 2, …, n) are non-negative scalars where $\sum_{j=1}^{n} \lambda_j = 1$. The property of inefficiency states that the same level of outputs could be produced by using more inputs, and the same level of inputs could be producing less outputs. Therefore, for DMU_j with inputs x_i and outputs y_r:

$$\begin{cases} \sum_{j=1}^{n} \lambda_j x_{ij} \leq x_i \quad i = 1, 2, \dots, m \\ \sum_{j=1}^{n} \lambda_j y_{rj} \geq y_r \quad r = 1, 2, \dots, s \\ \sum_{j=1}^{n} \lambda_j = 1 \end{cases}$$

(Equation 5.1)

The efficiency frontier to be identified by the DEA models could demonstrate different types of return-to-scale (RTS), such as constant return-to-scale (CRS) and variable return-to-scale (VRS). DMUs are identified to be operating under CRS when an increase in inputs leads to the same proportional increase in outputs, and vice versa; while operations under VRS do not demonstrate a proportionate change in outputs when inputs are altered (Castellet & Molinos-Senante, 2016). CRS and VRS measures different technical efficiencies of a DMU. CRS measures the overall technical efficiency (OTE), which is a single score representing the combined pure technical efficiency (PTE) and scale efficiency (SE) (Marti et al., 2009). The PTE purely reflects the productivity of the utilization of inputs (Kumar & Gulati, 2008), while the SE shows the level of efficiency achieved at a particular production scale (Guerrini et al., 2013). For DEA models adopting the assumption of VRS, the efficient frontier only reflects the measurement of PTE without the SE (Kumar & Gulati, 2008), while the SE can be obtained by the ratio of OTE to PTE (SE = OTE / PTE). By running the DEA model with CRS and VRS assumptions, this study investigates the OTE, PTE, and SE of the DMUs. The nature of RTS, such as increasing and decreasing returns to scale, could further be investigated. According to Avkiran (2001), the nature of returns to scale could be revealed by running the DEA model under the assumption of non-increasing returns to scale (NIRS). For the DMUs with inefficient SE, if the PTE score equals the efficiency score under NIRS, then the DMU is exhibiting decreasing returns to scale (DRS); if the PTE score is not equal to the score under NIRS, the DMU is exhibiting increasing returns to scale (IRS) (Avkiran, 2001).

DEA models could also be categorized into two types of orientations, namely input-oriented and output-oriented models. Input-oriented models target to reduce the utilization of inputs to produce the same level of outputs, while output-oriented models target to increase output levels using the same amount of inputs. Literatures indicate that the governing principle of most wastewater treatment facilities is input minimization (Guerrini et al., 2015; Sala-Garrido et al., 2012), meaning that the WWTPs aim to reduce environmental impact and resource consumption, without compromising the quality of sewage and sludge treatment services provided. Thus, input-oriented approach was used in this study.

5.2.4.2.1 Input-oriented VRS DEA model

Suppose DMU_o is the entity within the set of DMUs being evaluated by the input-oriented model. The objective of minimizing the utilization of inputs while maintaining the same level of outputs could be represented by the following equation (Banker et al., 1984; Zhu, 2014):

$$\theta^* = min\theta$$

subject to

$$\sum_{j=1}^{n} \lambda_j x_{ij} \leq \theta x_{io} \quad i = 1, 2, \ldots, m;$$

$$\sum_{j=1}^{n} \lambda_j y_{rj} \geq y_{ro} \quad r = 1, 2, \dots, s;$$

$$\sum_{j=1}^{n} \lambda_j = 1$$

$$\lambda_j \geq 0 \quad j = 1, 2, \dots, n.$$

(Equation 5.2)

θ^* represents the efficiency score of the DMU$_o$, with a value between 0 and 1. If θ^* equals 1, the input levels could not be further decreased proportionally, meaning that DMU$_o$ is efficient (on the efficient frontier).

When solving the model ((), input and output slacks may appear due to the possibility of yielding multiple optimal solutions. The input slack (s_i^-) and output slack (s_r^+) could be represented as:

$$\begin{cases} s_i^- = \theta^* x_{io} - \sum_{j=1}^{n} \lambda_j x_{ij} & i = 1, 2, \dots, m \\ s_r^+ = \sum_{j=1}^{n} \lambda_j y_{rj} - y_{ro} & r = 1, 2, \dots, s \end{cases}$$

(Equation 5.3)

The following programming model could be used to obtain any non-zero slack:

$$max \sum_{i=1}^{m} s_i^- + \sum_{r=1}^{s} s_r^+$$

subject to

$$\sum_{j=1}^{n} \lambda_j x_{ij} + s_i^- = \theta^* x_{io} \quad i = 1, 2, \dots, m;$$

$$\sum_{j=1}^{n} \lambda_j y_{rj} - s_r^+ = y_{ro} \qquad r = 1, 2, \dots, s;$$

$$\sum_{j=1}^{n} \lambda_j = 1$$

$$\lambda_j \geq 0 \qquad j = 1, 2, \dots, n.$$

(Equation 5.4)

DMUs with DEA score (θ^*) equal 1 and zero slacks are identified to be efficient; DMUs with DEA score equal 1 and non-zero slack(s) are classified as weakly efficient (Zhu, 2014). The VRS DEA model measure the PTE of the DMUs.

5.2.4.2.2 CRS and NIRS DEA models

The OTE of DMUs were evaluated using the CRS DEA model. The CRS model is the same as the VRS model described in Section 5.2.4.2.1 except the absence of the constraint $\sum_{j=1}^{n} \lambda_j = 1$. The nature of returns to scale, namely increasing and decreasing RTS, of the DMUs was revealed using the NIRS DEA model, by changing the constraint $\sum_{j=1}^{n} \lambda_j = 1$ in VRS model to $\sum_{j=1}^{n} \lambda_j \leq 1$.

After running the VRS, CRS and NIRS DEA models, the OTE, PTE and SE efficiency score, as well as the nature of RTS of the DMUs could be obtained. The targets for improvement could also be revealed through benchmarking and based on the slack values.

5.2.4.3 Performance metrics of DMUs in LC-DEA

In relation to production functions in economic theories, on which the concept of DEA is based, the performance metrics of DMUs have been commonly referred to as inputs and outputs (Zhu, 2014). For improving the efficiency of production operations, operators would aim to minimize the inputs and maximize the outputs. In DEA applications on benchmarking, the DEA inputs and outputs generally refer to metrics that operators would like to minimize and maximize, respectively. Based on the reasons described in Section 5.2.3, the inputs of this LC-DEA include i) chemical consumption (measured in environmental impacts), ii) energy use, iii) sludge residues generation, and iv) environmental impacts from treatment processes, while the outputs include i) VS reduction and ii) energy recovery.

5.2.4.3.1 LC-DEA inputs

Chemical consumption

Chemicals are commonly applied to enhance the effectiveness of thickening and dewatering of sludge. Common chemicals used in sludge conditioning include ferric chloride and polymers. According to the information provided by the DSD, ferric chloride ($FeCl_3$) and polyelectrolyte, which is a type of polymer, have been used to enhance dewatering. Information on the types of chemicals used and the corresponding dosages was collected from the DSD for the Hong Kong DMUs. Information on chemical consumption of other DMUs was estimated based on the corresponding literatures (Section 5.2.2.2) and the Handbook Estimating Sludge Management Costs (USEPA, 1985).

Chemical consumption of DMUs was measured as the life-cycle environmental impacts associated with chemical usage. The environmental impacts were estimated by LCA, which is a widely recognized methodology for evaluating the potential environmental impacts of the life-cycles of products (Iribarren et al., 2015; Xu et al., 2014a). The LCA was conducted according to the guidelines and framework specified in the ISO 14040 and 14044 standards (ISO 14040, 2006; ISO 14044, 2006). Life-cycle impact assessment was conducted with the aid of SimaPro 8.3 software, which is a commonly used LCA tool, and ReCipe Endpoint method was adopted in this study (Huijbregts et al., 2016). The total impacts were obtained through summation of the characterized impacts after normalization and represented by a unitless LCA score.

Energy use

Energy is required for the thickening, AD, aerobic digestion, dewatering, incineration and landfill disposal of sludge. For the Hong Kong DMUs, information on the electricity consumption of the sludge treatment processes is not available according to the HK DSD. Therefore, based on the operation information, such as daily sludge flow rate and amount of daily handled solids, the electricity requirements of the thickeners, digestors, and dewatering machines were estimated according to the Handbook Estimating Sludge Management Costs published by the Environmental Protection Agency in the US (USEPA, 1985). The information on electricity consumption of the sludge incinerator was collected from the HK EPD. Fuel consumption for sludge drying was estimated based on the Handbook Estimating Sludge Management Costs (USEPA, 1985). The energy requirements, including the electricity and diesel consumption of landfill operation was taken as the average values of the data obtained from published literatures (Abduli et al., 2011; Hong et al., 2010; Koroneos & Nanaki, 2012; Nabavi-Pelesaraei et al., 2017; Wei et al., 2009).

Information on energy use of other DMUs was obtained from the corresponding literatures (Section 5.2.2.2).

Sludge residues generation

Raw sewage sludge contains more than 90% of water content, which contributes significantly to its total volume. Sludge thickening and dewatering can substantially reduce the water content by more than 20%, thus reducing the sludge volume (DSD, 2009). Sludge incineration can eliminate the moisture content and combustible portion of sludge, thus achieving up to 90% volume reduction (EPD, 2018).

The volume of sludge residues generated after dewatering for the DMUs was calculated based on the inlet sludge volume and solid concentration (USEPA, 1985). For example, in the Shatin STW, the inlet sludge consisted of primary sludge (PS) and surplus activated sludge (SAS), thus the inlet volume was calculated as the summation of PS flow (m^3/d) and SAS flow

(m^3/d) (data provided by DSD). The volume of the dewatered sludge (outlet) was calculated using the equation (USEPA, 1985):

$$\text{TDSS} = \frac{(SV)(SS)(SSG)(1000)}{100}$$

(Equation 5.5)

where

TDSS = daily dry solids handled (kg/d)

SV = daily sludge volume (m3/d)

SS = sludge solids concentration (%)

SSG = sludge specific gravity (unitless)

These calculation parameters were obtained through the following methods:

i. TDSS of the dewatering process is equal to the dry solids in the digested sludge, which was calculated based on the digested sludge volume (data from DSD), SS of digested sludge (data from DSD), and SSG of digested sludge (USEPA, 1985).

ii. SS of the dewatered sludge was provided by DSD.

iii. SSG of dewatered sludge was calculated based on the *Handbook Estimating Sludge Management Costs* (USEPA, 1985).

The volume of sludge ash after incineration was estimated to be 10% of the inlet volume (EPD, 2018). Sludge residues generation in this study only includes the treated sludge waste to be disposed of in landfills. Sludge residues, such as sludge ash, recycled for substituting other products were excluded.

Environmental impacts from treatment processes

The environmental impacts are undesirable outputs of the STE systems, thus were included as an LC-DEA input. The environmental impacts of the DMUs were estimated using LCA. The impacts of AD and incineration were estimated based on the literatures (Gould et al., 2008;

Murray et al., 2008a; Xu et al., 2014a). The impacts associated with energy use and recovery were excluded to avoid double counting as they were included as separate LC-DEA input and output. As this study investigates the efficiency of the operation stage of sludge treatment, the impacts of the construction of facilities, such as land occupation, were excluded. The environmental impacts of other sludge treatment processes were majorly originated from energy and chemical consumption. As these aspects have already been included separately in other LC-DEA inputs, they were not covered in the LCA.

5.2.4.3.2 LC-DEA outputs

VS reduction

The VS contents represent the organic solids of sewage sludge. The reduction of volatile solids before disposal is essential to avoid the odor problem. Stabilization of sludge can be achieved through AD, in which the anaerobic bacteria consume the organics in sludge for cell growth and methane production. Sludge stabilization can also be achieved through aerobic digestion without the production of methane.

The VS reduction achieved by the sludge treatment process was calculated as the difference between the VS contents of the inlet and outlet sludge.

The VS content was calculated using the equation (USEPA, 1985):

$$\mathrm{DVS} = \frac{(SV)(SS)(VS)(SSG)(1000)}{(100)(100)}$$

(Equation 5.6)

where

DVS = daily volatile solids handled (kg/d)

SV = daily sludge volume (m3/d)

SS = sludge solids concentration (%)

VS = volatile solids concentration (% of SS)

SSG = sludge specific gravity (unitless)

The SV, SS and VS of the inlet and outlet sludge were obtained from the DSD to calculate the DVS before and after treatment. SSG was calculated according to the *Handbook Estimating Sludge Management Costs* (USEPA, 1985).

The volatile solids could be ignited at the temperature of 550-600°C, thus could be eliminated in sludge incineration, which reaches temperatures above 850°C. All the VS content in sludge was assumed to be destroyed after incineration.

Volatile solids reduction is included as an LC-DEA output to reflect the function achieved by the STE systems.

Energy recovery

Energy recovery could be achieved in the AD of sludge through the production of biogas, which is a by-product of the process and contains 65% of methane (DSD, 2017). Methane (CH_4) is a clean source of energy and the gas collected from AD could be used for electricity and heat generation. The data of biogas yield from the AD process was collected from the DSD. The electricity and heat recovered from CH_4 through combined heat and power (CHP) and dual fuel engine (DFE) systems were included as a favorable output of the treatment process (Fung & Yeung, 2012). Energy recovery through sludge combustion could also be achieved in some sludge incinerators, such as the fluidized-bed incineration in Hong Kong. The electricity generation from the sludge incinerator in Hong Kong was estimated based on the information provided by the EPD. Although biogas could also be yielded from landfill gas, the organic contents in sludge was destructed in incineration prior to landfill disposal in the local scenarios, thus no energy recovery was available from landfill disposal of sludge ash.

Information on energy recovery of other DMUs was obtained from the corresponding literatures (Section 5.2.2.2).

5.3 Results and discussions

5.3.1 LC-DEA performance metrics

The performance metrics of DMUs, including VS reduction, energy recovered, chemical consumption, energy use, sludge residue generation and environmental impacts from treatment processes, were computed based on the methods described in Section 5.2.4.3 and used as LC-DEA outputs and inputs for evaluating efficiencies of the STE systems.

Table 5-2 Performance metrics as LC-DEA inputs and outputs

	Outputs		Inputs			
DMU	VS reduction (kg/yr)	Energy recovered (kWh/yr)	Chemical consumption (kPt/yr)	Energy use (kWh/yr)	Sludge residue generation (m3/yr)	Env. Impact (kPt/yr)
01	17,613,025.85	28,406,192.51	115.95	9,293,176.87	5,203.75	269.91
02	4,718,144.90	9,288,558.19	86.58	2,420,133.81	1,102.42	55.03
03	3,847,964.04	7,724,122.37	62.46	3,759,079.85	1,165.74	46.98
04	2,306,558.17	3,171,535.73	47.42	770,418.34	421.64	25.47
05	95,735,850.00	61,396,269.34	284.74	76,491,078.68	52,827.23	2,474.38
06	2,018,777.58	1,069,310.46	15.74	1,314,126.27	890.08	43.05
07	631,517.39	334,503.49	7.27	420,300.67	426.49	13.49
08	811,440.08	429,805.33	0.39	598,850.12	396.26	17.35
09	2,062,085.19	-	-	818,521.14	278.42	8.01
10	9,011,830.64	28,483,140.00	64.59	3,123,275.80	92,837.43	58.14
11	9,011,830.64	-	87.39	22,481,999.80	92,837.43	58.14
12	4,721,640.00	2,948,248.81	43.01	4,655,356.81	33,215.00	28.44
13	8,636,995.00	5,880,860.29	61.56	6,089,652.09	41,610.00	44.10
14	1,186,578.50	1,768,906.78	2.07	1,130,883.61	6,168.50	3.64
15	8,205,200.00	10,439,173.10	57.62	57,020,875.70	9,881.95	124.49
16	8,067,116.85	31,738,207.77	28.11	10,201,111.92	2,287.68	97.89

Being the DMU with the highest sludge treatment loading (annual inlet sludge flow rate of 6.4 million m^3) among the Hong Kong DMUs, DMU05 (SCI STW in Hong Kong) presented the highest environmental impact associated with chemical consumption, energy use and environmental impact from treatment processes. At the same time, DMU05 achieved the highest VS reduction and energy recovery from sludge. The high sludge flow rate of SCI STW is attributed to the large wastewater treatment capacity of 2.45 million m^3 per day, after the extension under the Harbour Area Treatment Scheme (HATS) Stage 2A in Hong Kong (DSD, n.d.). The Shatin STW (DMU01), with a wastewater treatment capacity of 340,000 m^3 per day is another large contributor to sludge production in Hong Kong, generating approximately 2.5 million m^3 of sludge annually. DMU01 showed the second highest values in all the performance metrics among the Hong Kong DMUs.

5.3.2 Performance in energy use and recovery

Four Hong Kong DMUs (DMU01, DMU02, DMU03 and DMU04) and six non-local DMUs (DMU10, DMU12, DMU13, DMU14, DMU15, and DMU16) achieved energy recovery from AD; eight Hong Kong DMUs (DMU01 to DMU08) recovered energy through sludge incineration. Electricity and fuel were consumed for sludge treatment processes in all the DMUs. The energy use and recovery per 1 m^3 of inlet sludge are shown in Figure 5-3. For the DMUs that adopted aerobic digestion for sludge treatment (DMU09 and DMU11), the energy use for aeration was intensive. While aeration is an energy-demanding process in DMU09 and DMU11, there was no waste-to-energy treatment technologies adopted in these two DMUs, making them the DMUs with the third highest and highest net energy use, respectively. The analyzed sewage sludge treatment plants in Japan (DMU15) presented the second-highest energy use per m^3 of inlet sludge. Incineration and AD were the most energy-demanding processes in DMU15. Thermal energy was consumed for sludge drying and combustion, while electrical energy was used by components of the incinerator, such as pumps and blowers; energy recovery was not available from the process (Soda et al., 2010). Although energy recovery was achieved in AD, it was outweighed by the energy use of the process. The relatively high energy use for AD was attributed to the supplementary energy used for heating the thermophilic digestion tanks (Soda et al., 2010).

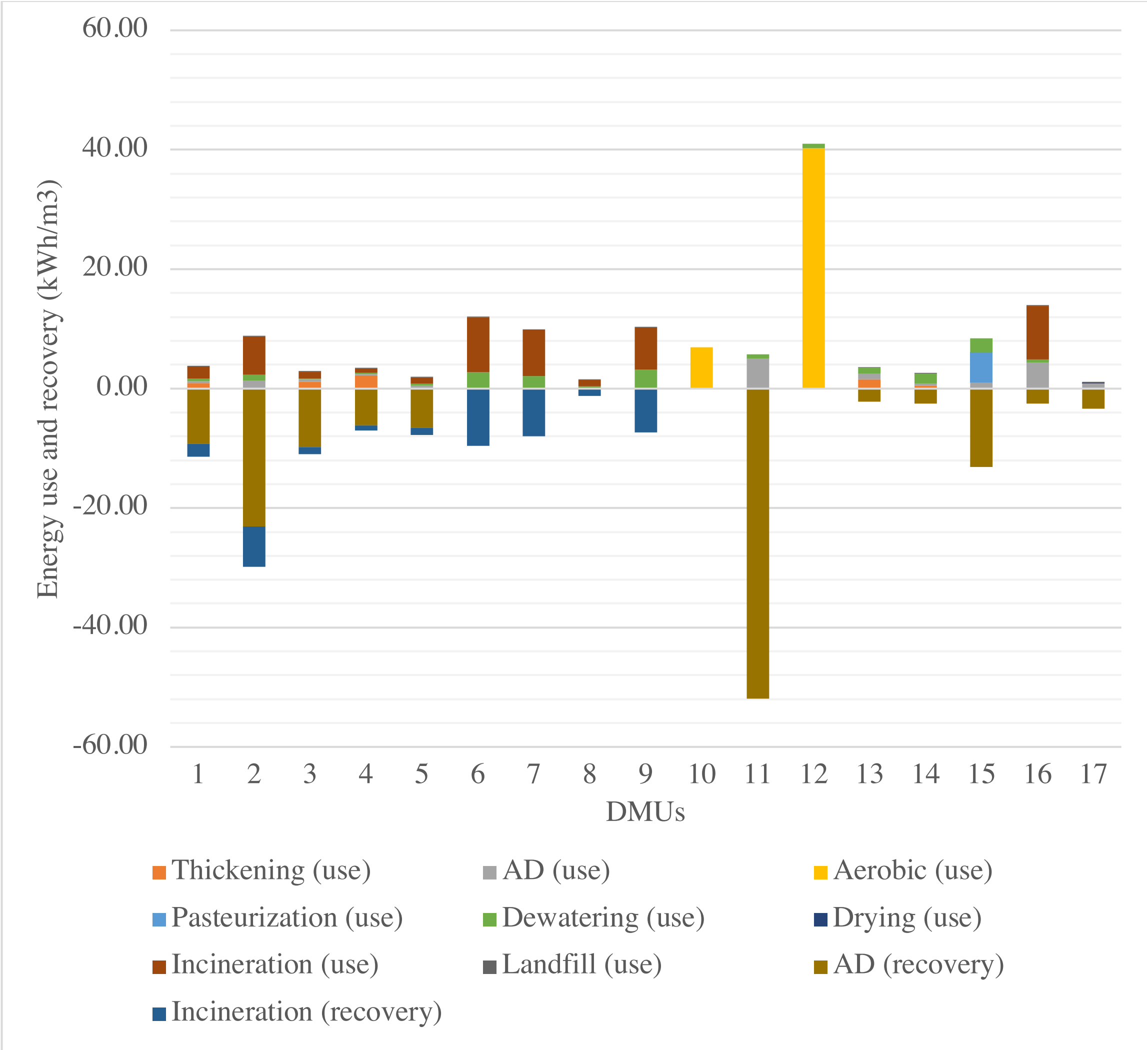

Figure 5-3 Energy use and recovery of DMUs (kWh/m^3)

Figure 5-3 shows the energy balance of the DMUs per m^3 of inlet sludge because the energy use for several processes, such as thickening and dewatering, is related to the volume of inlet sludge (USEPA, 1985). To further reveal the energy recovery efficiency, which is related to the organic contents of sludge, the energy recovery per kilogram of volatile solids in sludge is presented in Figure 5-4. For the major secondary STWs in Hong Kong, including Shatin (DMU01), Tai Po (DMU02), Shek Wu Hui (DMU03) and Yuen Long (DMU04), the energy recovered from AD ranged from 1.18 to 1.77 kWh/kg-VS, while the range for the non-local DMUs was 0.48 to 3.93 kWh/kg-VS. The energy recovery achieved by the centralized sludge incineration in DMU01 to DMU08 was 0.67 kWh/kg-VS. The information on chemical coagulants used for enhancing dewatering was obtained from the HK DSD, while that for non-local DMUs, such as DMU12, DMU13, DMU16, was obtained directly from the published literature. For other DMUs, the chemical requirements were estimated based on the mass of sludge dry solids (USEPA, 1985).

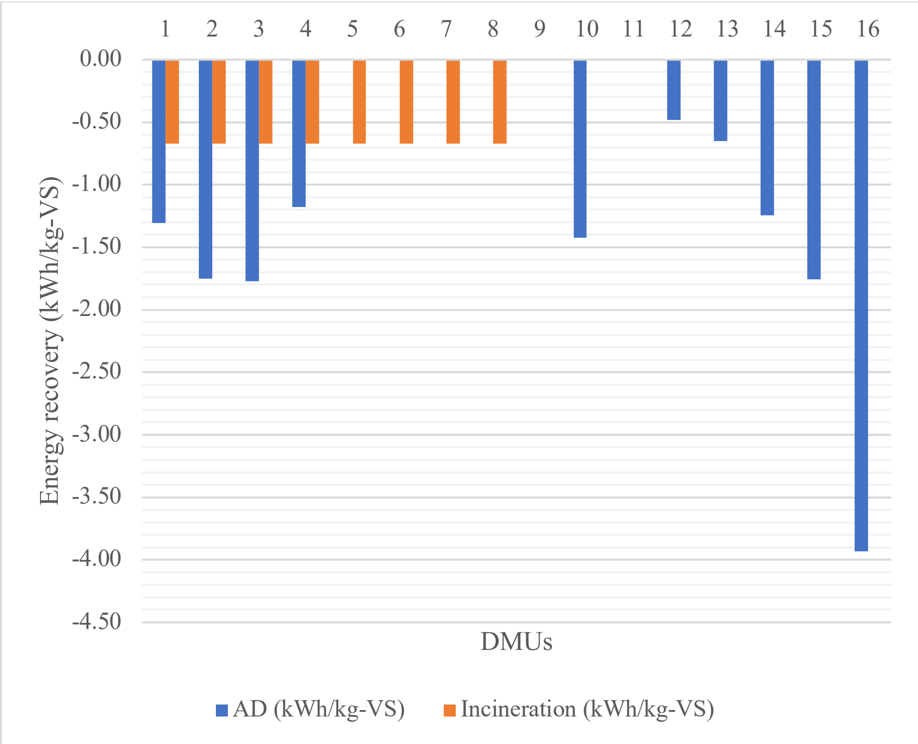

Figure 5-4 Energy recovery per kilogram of sludge VS

5.3.3 LC-DEA results

As the energy use, energy recovery, chemical consumption, waste generation and environmental performance of the DMUs depend on numerous factors—such as the inlet sludge conditions, treatment targets and treatment technologies—they could not be directly related to the flow rate or solid loading, which are the common functional units defined in conventional wastewater and sludge LCA studies. Therefore, based on the above inventory data, this study evaluated the efficiency of the DMUs in relation to the levels of treatment and energy recovery using the LC-DEA approach.

Based on the inputs and outputs in Table 2, LC-DEA results, including OTE, PTE and SE scores and natures of RTS, were obtained for the sixteen DMUs (Table 5-3). The OTE scores indicate the relative overall efficiency of the DMUs. DMUs with the maximum score 1.00 are the most efficient ones, while DMUs with scores below 1.00 are less efficient. The results show that, among the sixteen DMUs, the Shatin (DMU01), Tai Po (DMU02), Yuen Long (DMU04), and Sai Kung (DMU09) STWs, as well as the Chengdu S3 (DMU10), Italy (DMU14) and Denmark (DMU16) WWTPs achieved the highest efficiency. The DMUs with lower scores are less efficient. The most inefficient DMU is the Japan WWTP (DMU15); the most inefficient Hong Kong DMU are Stanley (DMU07), SCI (DMU05) and Siu Ho Wan (DMU06) STWs.

Table 5-3 Efficiency scores and nature of returns to scale of DMUs

DMU	OTE	PTE	SE	RTS
01	1.00	1.00	1.00	Constant
02	1.00	1.00	1.00	Constant
03	0.76	0.77	0.98	Decreasing
04	1.00	1.00	1.00	Constant
05	0.60	1.00	0.60	Decreasing
06	0.62	0.66	0.95	Increasing
07	0.57	1.00	0.57	Increasing
08	0.98	1.00	0.98	Increasing
09	1.00	1.00	1.00	Constant
10	1.00	1.00	1.00	Constant
11	0.48	0.92	0.52	Decreasing
12	0.61	0.80	0.76	Decreasing
13	0.76	1.00	0.76	Decreasing
14	1.00	1.00	1.00	Constant
15	0.42	0.74	0.56	Decreasing
16	1.00	1.00	1.00	Constant

The OTE scores represent the overall efficiency in relation to both the technical performance of the treatment processes and the scales of the DMUs. To further investigate the effect of the economies of scales, the OTE scores were decomposed into PTE and SE, which indicate the contributions of the technical aspects and the scale of treatment facilities to the overall efficiency, respectively. For example, the Shek Wu Hui STW (DMU03) showed a PTE score of 0.77 and an SE score of 0.98, meaning that both technical performance and the scale of operation had notable effects on the overall performance of the DMU. DMUs 05, 07, 08 and 13 are inefficient DMUs, each with a PTE score of 1.00, implying that the inefficiency was solely caused by the unfavorable sizes of treatment facilities. The DMUs with unfavorable operation sizes, as reflected by SE scores lower than 1.00, could be either oversized or undersized. The nature of RTS provides guidance for scaling-up or scaling-down the facilities: an increasing RTS suggests the expansion of facilities, while a decreasing RTS suggests reducing the size of facilities.

For the DMUs with inefficient PTE scores, the technical aspects of the treatment processes are required to improve. Through the benchmarking the DMUs according to different aspects of performance, the DEA results provide information on the required improvements for the DMUs to achieve efficiency. Table 5-4 shows the required improvements, including the reduction of inputs and expansion of outputs, for the DMUs to be efficient. For example, the sludge treatment process of the WWTP in Osaka, Japan, would have to reduce the chemical consumption, energy use, sludge residue generation and direct environmental impact by 42.78%, 83.04%, 25.93% and 25.93%, respectively; and increase the energy recovery by 172.24% to become efficient.

Table 5-4 Required improvements for DMUs to achieve efficiency

		Outputs		Inputs			
DMU	STWs	VS reduction (kg/yr)	Energy recovered (kWh/yr)	Chemical consumption (kPt/yr)	Energy use (kWh/yr)	Sludge residue generation (m3/yr)	Env. Impact (kPt/yr)
1	Shatin	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%
2	Tai Po	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%
3	Shek Wu Hui	0.00%	0.00%	-65.15%	-22.55%	-22.55%	-22.55%
4	Yuen Long	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%
5	Stonecutters	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%
6	Siu Ho Wan	0.00%	0.00%	-34.39%	-34.39%	-34.39%	-61.79%
7	Stanley	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%
8	Sham Tseng	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%
9	Sai Kung	0.00%	N.A.	N.A.	0.00%	0.00%	0.00%
10	Chengdu S3	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%
11	Chengdu S4	0.00%	N.A.	-26.96%	-72.32%	-56.82%	-7.93%
12	Korea N-WWTP	0.00%	0.00%	-36.77%	-31.76%	-41.17%	-20.17%
13	Korea S-WWTP	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%
14	Italy	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%
15	Osaka	0.00%	+172.24%	-42.78%	-83.04%	-25.93%	-25.93%
16	Copenhagen	0.00%	0.00%	0.00%	0.00%	0.00%	0.00%

5.3.3.1 Factors for inefficiency and recommendations for improvements

DMU15, which represents the STE system of WWTP in Japan, had the lowest OTE score and the second least favorable performance in PTE. DMU15 required improvement in energy recovery by 172.24%. The reason for the low energy recovery of DMU15 was its incapability of energy recovery from incineration in the STE system. While fuel and electricity were demanded for the energy-intensive incineration process, energy or heat recovery were not available in DMU15. The daily energy use of DMU15 was 156 MWh, while only 29 MWh could be recovered from sludge AD. Incineration was the most energy-demanding process in this STE system, contributing to 65% of the total energy use, while the energy use of AD contributed to 31% of the total consumption. The raw sludge solids contained 80% of organic matter content and AD could digest 55% of it, meaning that the digested sludge still contain a notable amount of organic matters. Upgrade of incinerators to enable energy recovery from sludge combustion were suggested for DMU15 to improve the overall energy recovery of the STE system. The energy content of organic matters in sludge could either be recovered in forms of heat and electrical energy or transferred to the environmental as an energy loss of the system. Incineration with energy recovery could achieve the former, thus improving the overall efficiency of the DMU. DMU15 required a reduction of chemical consumption for 42.78%. Chemicals were used as coagulants for enhancing the dewaterability of sludge. Previous studies have investigated the improvement of sludge dewaterability by other means, including thermal pre-treatment (Neyens & Baeyens, 2003) and lowering the pH (Chen et al., 2001; Neyens & Baeyens, 2003; Elisabeth Neyens et al., 2004). Yet the feasibility and suitability of applying such techniques in DMU15 are out of the scope of this study, thus need to be further investigated in future laboratory-based research studies. Energy use of DMU15 needed to be reduced by 83.04% to achieve efficiency. Incineration was the most energy-consuming process in this DMU, contributing to 64.80% of its total energy use. Adoption of thickening or drying to reduce the water content of sludge is favorable to the combustion process, thus is a possible means to decrease the energy requirement for sludge incineration (Mininni et al., 1997). Selection of different incineration technologies also affect the requirement of auxiliary fuels (Murakami et al., 2009). Reduction of sludge residue generation by 25.93% was recommended. As the amount of sludge residue generated was closely related to the amount of raw sludge, based on the flow rate and solid content, reduction of sludge production from the wastewater stream in WWTPs could be a possible solution for reducing the final sludge residues. Mild temperature and low organics loading tended to produce lower amount of sludge from wastewater treatment stream (Lorenzo-Toja et al., 2015). Sludge yield could also be reduced by controlling the solid retention time and dissolved oxygen level in wastewater treatment (Semblante et al., 2014). Reducing the environmental impacts originated from sludge treatment processes by 25.93% was recommended. As relatively high environmental impacts of DMU15 could be attributed to the high sludge loading for incineration. Although AD was adopted in DMU15 to eliminate a portion of VS before sludge incineration (10 tonnes of dry solids were eliminated through AD), 144 t/day of sludge from another WWTP was mixed with the dewatered sludge for incineration (Soda et al., 2010). The addition of sludge increased the dry solids from 16.5 t/day after AD to 22.3 t/day for incineration. The addition loading of sludge for incineration could be a reason for the unfavorable environmental performance. DMU15 scored 0.56 in SE with a decreasing RTS. Therefore, scaling down the STE system is recommended to improve the efficiency of DMU15.

When focusing on the pure technical efficiency of the DMUs, DMU06 (Siu Ho Wan STW in Hong Kong) was the STE system with the lowest PTE score among the sixteen DMUs. LC-DEA results revealed that, for DMU06 to achieve maximum efficiency, reduction of chemical consumption, energy use and sludge residue generation by 34.39%, and reduction of environmental impact from treatment processes by 61.79% were required. DMU06 presented the fourth highest energy use among the sixteen DMUs (Figure 5-3), with incineration contributing to 77.73% of its total consumption. The relatively high environmental impact of DMU06 could be attributed to the exclusion of AD and high environmental impacts of incineration. The STE system of DMU06 included processes of dewatering, incineration and landfill disposal. While AD was absent for eliminating part of the solids (VS), a higher solid loading would be entering incineration. The direct environmental impact induced by incineration of 1 dry tonne of sludge was more than 6 times higher than AD of same weight of sludge (Xu et al., 2014a). Although incineration eliminates all the VS in the STE systems with or without AD, higher reliance on incineration for VS elimination causes higher direct environmental impacts. Thus, application of AD for sludge treatment in DMU06 is suggested as a possible means to reduce the direct environmental impacts. DMU06 scored 0.95 for SE and presented an increasing RTS. Such results reveal that the scale of DMU06 was slightly under the optimal level, thus an increase in the size of STE system could increase the SE and, therefore, the OTE of DMU06.

5.3.4 Contributions and Limitations of LC-DEA

The waste-to-energy management approach has been widely recognized and adopted for treating sludge and other organic waste. Although environmental and economic performance of sludge management scenarios have been evaluated in numerous published studies, the strong linkage between sludge treatment and energy systems, as well as the operation efficiency benchmarking approach for STE systems, have not yet been covered in previous literatures. By the inclusion of essential factors that affect the efficiency of both sludge treatment and energy systems, and the integration of LCA and DEA, this LC-DEA study filled the research gap by providing a multiple criterion benchmarking tool that could objectively evaluate the performance of STE systems.

The LC-DEA evaluation tool developed in this study is widely applicable to guide decision-making for improving the techno-environmental efficiency of STE systems so that sustainability could be achieved. The LC-DEA framework can provide comprehensive information to decision-makers for benchmarking the sludge management systems with energy recovery in different WWTPs, which in turn can be used to prioritize appropriate remedial actions to improve the performances of relatively inefficient systems. For example, high priority for upgrade of STE system could be recommended to the authority in Japan for improving DMU15 as it was the DMU with the lowest efficiency. Targets for improvements in different aspects were identified for improving the PTE and scaling down the STE system was recommended for improving the SE of DMU15, as described in Section 5.3.3.1. This study has demonstrated LC-DEA as a suitable evaluation tool to guide decision makers, government officers and practitioners in improving the efficiency of STE systems. Such tool could also be a comprehensive approach to reveal the performance of the STE systems to the general public such that public education on waste-to-energy systems and their associated benefits could be achieved.

The limitation of this LC-DEA for STE systems is that the inadequacy in the system configurations of the treatment facilities and the technical approaches to achieve the improvement targets are excluded from the study scope. This LC-DEA study focuses on benchmarking the operation performance of the existing STE systems from a management perspective, so that the information on relative efficiency of the systems, as well as improvement targets of different aspects could be provided to decision makers. However, the alterations on system configurations of sludge treatment facilities, such as temperature and solid retention time for sludge AD, for efficiency improvement are out of the scope of this LC-DEA. Based on the recommendations for improvement obtained from the LC-DEA findings, further laboratory-based studies could be conducted to investigate the technical method for achieving the improvement targets.

5.4 Summary

The life-cycle data envelopment analysis approach developed in this study, through integration of life-cycle assessment and data envelopment analysis, for benchmarking sludge-to-energy systems is a comprehensive and widely applicable tool for guiding decisions on improving waste-to-energy systems. The development of this innovative tool contributed to fill the research gap by highlighting the strong linkage between waste and energy systems, as well as evaluating the operation efficiency with considerations of all the essential performance metrics of the sludge-to-energy systems. This life-cycle data envelopment analysis approach provides an objective basis for comparing sludge-to-energy systems with different sludge handling technologies yet having the same set of inputs and outputs.

The life-cycle data envelopment analysis approach was demonstrated in this study on benchmarking sixteen sludge-to-energy systems as a suitable tool to guide decision-making on improving the techno-environmental efficiency of the systems. The data envelopment analysis findings could firstly assist in identifying the relatively inefficient decision-making units that prioritized for improvements. The least efficient sludge-to-energy system was DMU15, which only scored 0.42 for the overall technical efficiency score, implying that the authority could consider improving the performance of the sludge-to-energy system. For decision-makers and authorities in Hong Kong, Stanley (DMU07), Stonecutters Island (DMU05) and Siu Ho Wan (DMU06) STWs should be prioritized for improvements, as these were the least efficient decision-making units (with the lowest overall technical efficiency scores) among the Hong Kong decision-making units. The findings could secondly ensure the efficient use of resources in making the appropriate improvements, as the pure technical efficiency and scale efficiency scores inform decision-makers on the sources of inefficiency (technical performance of treatment processes or size of treatment facilities). DMU15 required significant improvement on energy recovery (by 172.24%) and upgrading the sludge incinerators to models that could achieve energy recovery was recommended. Scaling up DMU07 and scaling down DMU05 could improve the efficiency of the decision-making units. With such information, the agency could further investigate whether changing the scales of the facilities is feasible in future town planning policies. For DMU06, besides increasing the operation scale, the technical performance also required enhancement.

The developed life-cycle data envelopment analysis framework will be widely applicable to different sludge-to-energy systems worldwide. While this study evaluated the performance of the sludge-to-energy systems from a management perspective, only the aspects that required improvement and the improvement targets are provided in the findings. The technical approaches to achieve the targets, which are closely related to the engineering design of the treatment facilities, were not investigated in this study. More comprehensive results could be obtained if the actual operation data of more decision-making units could be collected and included in the life-cycle data envelopment analysis.

CHAPTER 6

CONCLUSIONS

6.1 Conclusions

Sustainability evaluation framework for urban wastewater and sludge treatment systems was developed and demonstrated on sewage sludge treatment approaches, non-potable water supply systems and sludge-to-energy systems to support effective decision-making in this thesis. The LCA-based framework was developed through integration with other techniques, including life-cycle costing (LCC) and data envelopment analysis (DEA), for enhanced comprehensiveness of evaluation. The key findings and conclusions drawn from this thesis are summarized as follows.

- Land cost, land occupation and travelling distances of sludge transportation are critical factors that influence the eco-efficiency of urban sludge treatment approaches. A 5% variation of land cost could cause 3.52 – 4.91% changes in total economic costs. Land occupation is the most sensitive LCA impact category to variation of inlet sludge volume. The deviation of estimated CO_2 emissions from the actual transportation emission could reach 187,000 tonnes and 137,000 tonnes if fixed transportation distances (e.g. 40 km and

25 km) were assumed. Based on the collected data, system boundary and assumptions made, sludge treatment using anaerobic digestion (AD) followed by dewatering, incineration and reuse for cement production was the most eco-efficient option for adoption in urban cities. Due to the energy recovery in the AD and incineration, such sludge treatment approach performed the most environmentally and economically favorable and could generate US$22.05 per dry tonne of inlet sludge. Reuse of treated sludge residue for cement production avoid environmental impacts and economic costs for clinker production and, at the same time, avoid land requirement for landfill disposal. As the feasibility of reuse of treated sludge residues for clinker in cement production may depend on the market demand for clinker substitutes, sludge treatment approach using AD, dewatering and incineration followed by landfill disposal could be a backup option with the second-best eco-efficiency performance.

- Anaerobic greywater recycling for toilet flushing is more eco-efficient than freshwater, seawater and greywater recycling using aerobic MBR. The environmental and economic advantages of anaerobic greywater reuse system are attributed to its capabilities of recovery of energy and water, and low sludge production rate. Despite the relatively high environmental impacts in construction phase, anaerobic greywater reuse system is still considered to be environmentally friendly due to the environmental benefits from energy recovery from methane yield during the operation phase. The construction cost of anaerobic greywater reuse system was the highest (US$462,500) and the greywater treatment unit contributed to 48% of the total construction cost. Anaerobic greywater reuse system has the advantage of avoiding cost of imported water and energy recovery in the operation phase. The time period of operation and the price of imported were the crucial factors influencing the life-cycle costs of systems. Anaerobic greywater reuse system is a favorable option for non-potable water supply for mid- to long-term application (>20 years), while seawater supply is a better option for in short term. The favorability of anaerobic greywater system remained unchanged for mid- to long-term application regardless of the variations of imported water price from 50% to 200%. The ranking of eco-efficiency of different systems is robust to different levels of emphasis on the environmental and economic aspects.

- Among the sixteen STE systems evaluated, 44% and 69% of STE systems achieved best practice in terms of OTE and PTE, respectively. The STE system with the least favorable performance among the studied systems was a sludge treatment consisting of thickening, AD, dewatering, incineration and landfilling in Osaka, Japan, scoring 0.42 for OTE. An enhancement of energy recovery by 172.24% was required and adoption of incineration energy recovery were recommended. In Hong Kong, sludge treatment systems in Stanley, SCI and Siu Ho Wan STWs were identified to be the least efficient, thus had the highest priority for the authorities to implement improvement measures. Scaling up and scaling down the sludge treatment units in Stanley and Stonecutters Island STWs, respectively, could enhance the efficiency; both scale up and improvement in technical performance were required enhancing efficiency of sludge treatment system in Siu Ho Wan STW.

6.2 Contributions of research

The major contribution of this research is the development of novel sustainability evaluation framework for effective decision-making on urban wastewater and sludge treatment systems. LCA-based evaluation framework was developed and applied on sludge treatment, non-potable water supply systems and STE systems to provide quantitative findings, knowledge and policy implications to enhance decision-making towards a sustainable manner. The key contributions are highlighted as follows:

- An EEA framework for evaluating urban sludge treatment approaches was developed by integrating LCA and LCC to assist decision-making towards sustainable sewage sludge management. The innovativeness of the EEA is the inclusion of essential characteristics of urban areas, namely high land costs and limited land resources. Actual transportation distance was also identified to be an influencing factor to the environmental performance, thus was included in the EEA framework.

- An EEA framework for non-potable water supply systems in domestic buildings in urban cities was developed to inform decisions on selecting the most eco-efficient option. The novel approach evaluates the environmental and economic performance based on detailed information of the engineering designs of the systems. The influences of system lifetime, price of imported water (which was viewed as a substitute good to recycled greywater) and the relative emphasis on environmental and economic performance were investigated to enhance the comprehensiveness. Considering the practicability, the EEA provides eco-efficiency portfolios, which are clear and comprehensive results that assist decision-making and promote public education.

- Benchmarking of STE systems was achieved using the LC-DEA. This is a first-of-its-kind evaluation tool for assessing techno-environmental performance od STE systems with emphasized on the interrelationship between sludge and energy systems. The key contribution of the LC-DEA study is the development of a multiple criterion evaluation tool for an impartial benchmarking of STE systems. The LC-DEA assists decision makers to identify and prioritize improvement works of the relatively inefficient systems.

- Another contribution of the research is revealing the significance of energy recovery in the wastewater treatment sector in moving towards more sustainable urban development. The benefits of AD and incineration with energy recovery from sewage sludge, as well as anaerobic greywater recycling using AFMBR, were revealed in the EEA studies. LC-DEA on STE systems also emphasized the significance of viewing energy recovery as a important output of sludge treatment systems. The findings provide strong evidence to decision-makers and practitioners in supporting the adoption of waste-to-energy strategies in the wastewater treatment sector for promoting sustainable development and a greener future. The transparency of the evaluation framework and the intelligible results promote thorough understanding on the relationship between urban wastewater systems and sustainability.

6.3 Recommendations for future research

The organic content of municipal wastewater offers a high potential for energy recovery. Numerous wastewater-to-energy technologies have been developed for energy extraction from municipal wastewater as a valuable resource. Energy-harvesting from wastewater could be achieved using advanced technologies, such as microbial fuel cell (MFC), anaerobic fluidized-bed bioreactor (AFBR) and anaerobic fluidized-bed membrane bioreactor (AFMBR).

Technologies for recovering energy from sewage sludge, the solid by-product from wastewater treatment, include anaerobic digestion, incineration, pyrolysis and gasification. Different energy recovery technologies vary in terms of energy consumption, energy yield, organics removal and sludge residue generation, and various technologies could be adopted in combinations. The recovered energy could be used on-site in forms of heat and/or electricity, or connected to the main electricity grid. Technology selection for treatment systems and utilization of recovered energy play an important role in life cycle emissions and energy performance of wastewater-to-energy systems. Although numerous studies have been conducted to evaluate the environmental performance of wastewater treatment systems, an evaluation tool that considers the engineering design and technical aspects of wastewater-to-energy systems is lacking.

This study aims to reveal the life cycle emissions and energy recovery of different wastewater-to-energy systems through the integration of engineering design and mass flow of wastewater treatment systems with life cycle assessment (LCA). The contributions of this study include (1) capturing the impacts of upstream treatment technologies on subsequent processes, and (2) assisting the selection of the most favorable wastewater-to-energy system based on the characteristics of wastewater. The engineering design and mass flow of different combinations of technologies enable realistic configurations of treatment systems that take the characteristics of wastewater into consideration. Mass flow analysis also enables a pragmatic evaluation of downstream treatment processes with consideration of the impacts of upstream technology selection. LCA modules will be established based on the characteristics of wastewater and the key treatment parameters for each wastewater-to-energy process. System-level LCA on wastewater-to-energy treatment systems will be established by combining the modules. The mass flow information of upstream process will be fed into the LCA modules of the downstream processes for emission and energy evaluation. The modular LCA coupled with engineering design and mass flow analysis provides a comprehensive assessment framework for evaluation and informs decision-making on the selection of wastewater-to-energy systems. The modular LCA framework is presented in Figure 6-1.

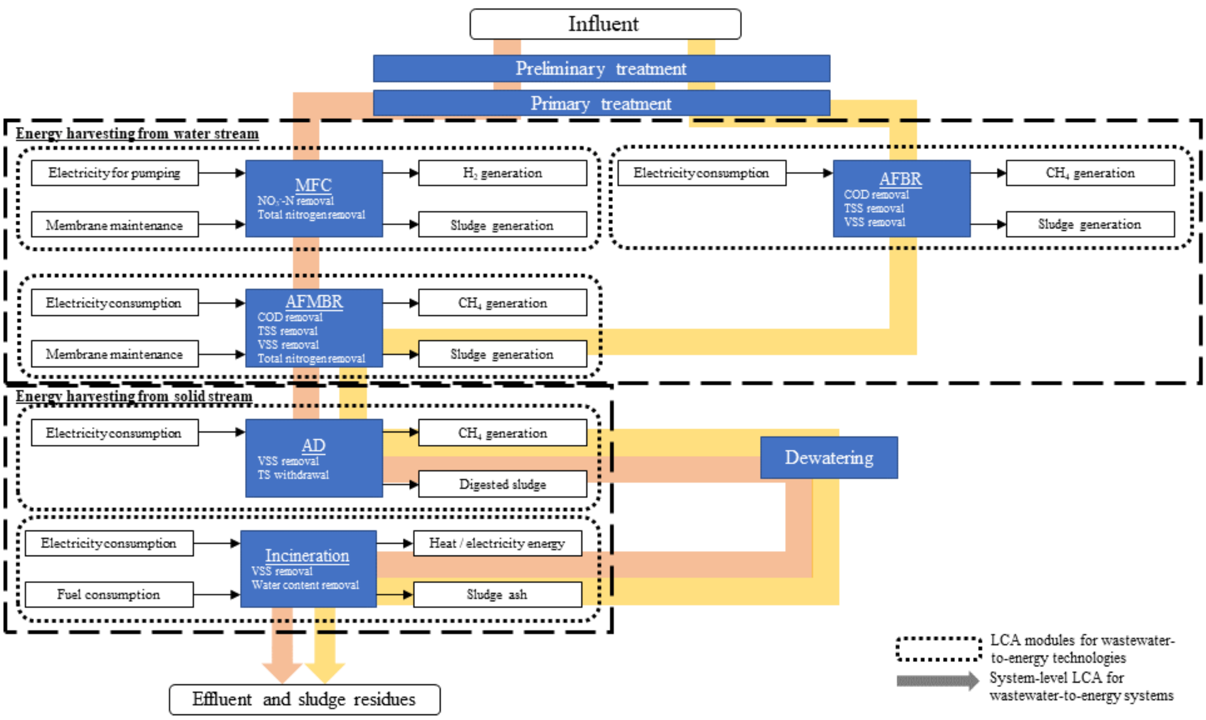

Figure 6-1 Framework of modular LCA for wastewater-to-energy systems

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APPENDICES

Appendix 1

Part 1: Economic cost analysis

A3.1. Scenario 1

Figure A1- 1 Process Diagram of Scenario 1

A1.1.1.Anaerobic Digestion (Scenario 1)

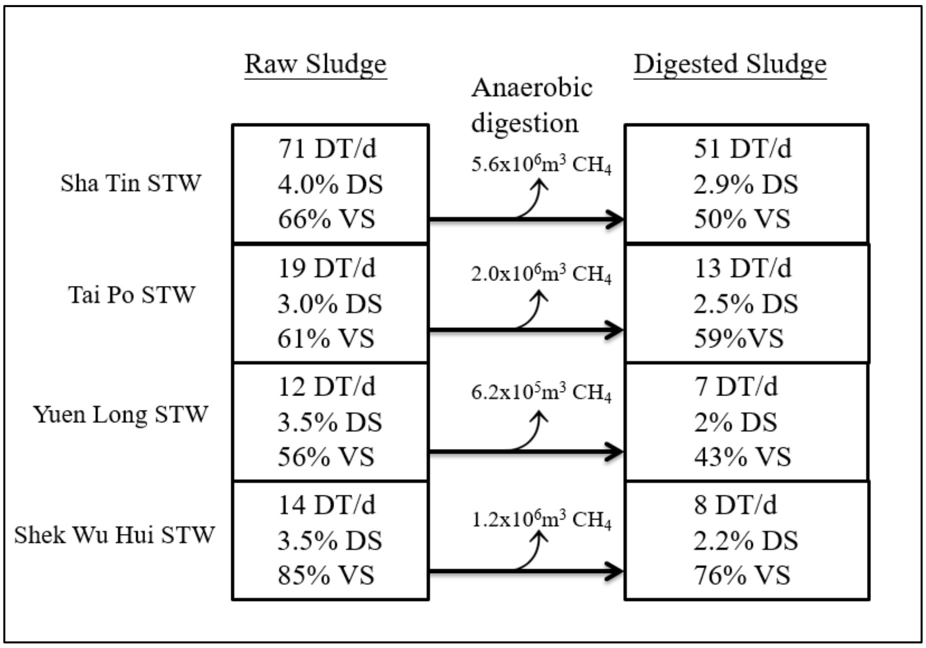

Figure A1- 2 Process Flow Diagram of Anaerobic Digestion (Scenario 1)

Table A1- 1 Methodology of Anaerobic Digestion Cost Estimation (Scenario 1)

Capital Cost[2]	The excavation and construction of reinforced concrete tanks, costs for gas circulation equipment, heat exchangers, pumps, internal piping, ancillary equipment and a two-storey control building are included in the capital cost estimation.
Operation and Maintenance (O&M) Cost[2]	O&M cost consists of wages for operation and maintenance labor, cost of electricity and maintenance material cost. Electricity generation from methane recovery is calculated as the offset of O&M cost.
Transportation Cost	After anaerobic digestion, sludge is transferred to the dewatering process. As both the anaerobic digestion and dewatering process take place on-site in the STWs, no transportation of sludge is required between these two processes.

Table A1- 2 Data and Assumptions for Anaerobic Digestion Cost Estimation (Scenario 2)

	ST	TP	YL	SWH
Daily volume (m^3)*	1,620	571	312	844
Percent DS (%)*	4.0	3.0	3.5	3.5
Percent DS after AD (%)*	2.9	2.5	2.0	2.2
Percent VS (%)*	66	61	56	85
Percent VS after AD (%)*	50	59	43	76
Operation hours per day*	24	16	24	16
Operation days per year*	365	300	365	365
Wage for labor (HK$/hr)[3]	30			
Annual methane production (m^3)*	5,600,000	2,000,000	616,820	1,200,000

*2013 data from Drainage Services Department (DSD), HKSAR

[2] USEPA (1985). *Handbook Estimating Sludge Management Costs*. Ohio: USEPA
[3] Statutory Minimum Wage (2013). Retrieved October 1, 2014 from HKASR, Labour Department Web site: http://www.labour.gov.hk/eng/news/mwo.htm

A1.1.2.Dewatering (Scenario 1)

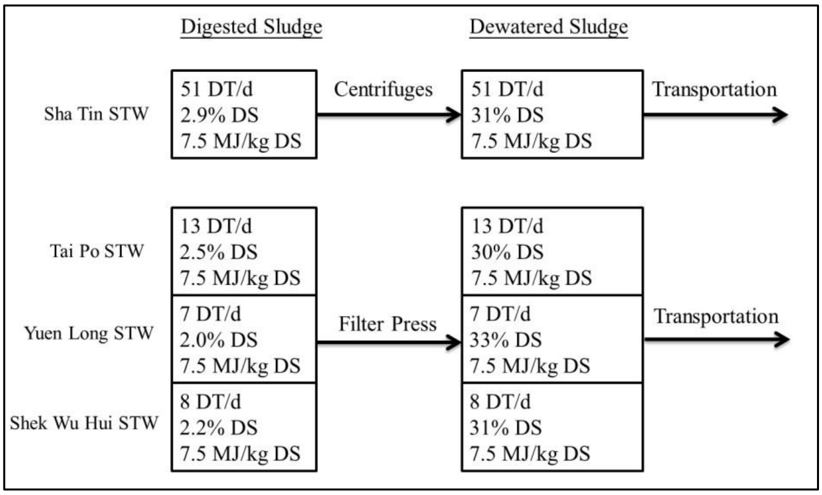

Figure A1- 3 Process Flow Diagram of Dewatering (Scenario 1)

Table A1- 3 Methodology of Dewatering Cost Estimation (Scenario 1)

Capital Cost[1]	Land cost is negligible in the capital cost estimation as the centrifuge and filter press dewatering processes are not land-intensive. Costs of the construction of buildings for accommodating the equipment were included. Capital cost for the chemical addition system was also estimated.
Operation and Maintenance (O&M) Cost[1]	Annual costs of operation and maintenance labor, electricity, parts and materials and chemical addition were included. Life-time costs of the sewage sludge treatment scenarios over a 20-year time horizon and with 6.6% inflation rate[3] are presented. The electricity cost is HK$1.108/kWh[4].
Transportation Cost	Dewatered sludge is transported to landfill site for final disposal. The transportation cost includes the cost for diesel consumption by vehicles. The estimation is based on the travelling distance, truck capacity and number of round trips.

[3] Consumer Price Indices (2014). Retrieved October 4, 2014 from Government of HKSAR, Census and Statistics Department Web site: http://www.censtatd.gov.hk/hkstat/sub/sp270.jsp?tableID=052&ID=0&productType=8

[4] FAQ (2015). Retrieved May 29, 2015 from China Light and Power, Web site: https://www.clponline.com.hk/faq/residentialcustomersfaq/Pages/ElectricityPrices.aspx?lang=en

Table A1- 4 Data and Assumptions for Dewatering Cost Estimation (Scenario 1)

	ST	TP	YL	SWH
Daily volume (m^3)*	486	171	94	253
Percent DS (%)*	2.9	2.5	2.0	2.2
Percent DS in dewatered sludge (%)*	31	30	33	31
Operation hours per day*	24	16	8	16
Operation days per year*	365	300	326	365
Wage for labor (HK$/hr)[2]	30			
Travelling distance (km)[5]	38	29	24	8
Diesel consumption (kg/km)[6]	0.168			
Volume of trucks (m^3)*	13	12	20	12
Cost of diesel (HK$/L)[7]	12.87			

*2013 data from Drainage Services Department (DSD), HKSAR

A1.1.3.Landfill Disposal (Scenario 1)

Table A1- 5 Methodology for Landfill Disposal Cost Estimation (Scenario 1)

Capital Cost[1]	Landfill disposal method involving trenching is assumed. Land cost, site improvements, installation of monitoring wells, and purchase of excavation vehicles and earth-moving vehicles are included in the capital cost estimation.
Operation and Maintenance (O&M) Cost[1]	O&M cost includes costs for labor, diesel consumption, machinery maintenance and site maintenance.

[5] Google Map (2014). Retrieved September 21, 2014 from https://www.google.com.hk/maps/@22.352734,114.1277,11z?hl=en
[6] Murray, A., Horvath, A. & Nelson, K. L. (2008). Hybrid Life-Cycle Environmental and Cost Inventory of Sewage Sludge Treatment and End-Use Scenarios: A Case Study from China. Environmental, Science and Technology, 42 3163-3169.
[7] Price Board (2014). Retrieved February 13, 2014 from, Shell Hong Kong Limited Web site: http://www.shell.com.hk/en/products-services/on-the-road/fuels/price-board.html

Table A1- 6 Data and Assumptions for Landfill Cost Estimation (Scenario 2)

	ST (SENT)	TP (SENT)	YL (NENT)	SWH (NENT)
Daily volume (m^3)	42	13	5	16
Operation hours per day[8,9]	15	15	10	10
Operation days per year[8,9]	365	365	365	365
Wage for labor (HK$/hr)[2]	30			
Cost of diesel (HK$/L)[7]	12.87			
Industrial land cost (HK$/$ft^2$)[10]	850			

[8] South East New Territories (SENT) Landfill (2013). Retrieved October 3, 2014 from HKSAR, EPD Web site: http://www.epd.gov.hk/epd/english/environmentinhk/waste/prob_solutions/msw_sent.html

[9] North East New Territories (NENT) Landfill (2013). Retrieved October 3, 2014 from HKSAR, EPD Web site: http://www.epd.gov.hk/epd/english/environmentinhk/waste/prob_solutions/msw_nent.html

[10] Ho, K. C. (2013, July 19). Investment in agricultural land in the New Territories of different grades in different price. *Epoch Times*

A3.2. Scenario 2

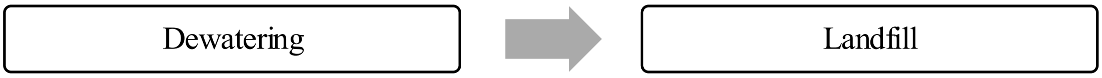

Figure A1- 4 Process Diagram of Scenario 2

A1.2.1.Dewatering (Scenario 2)

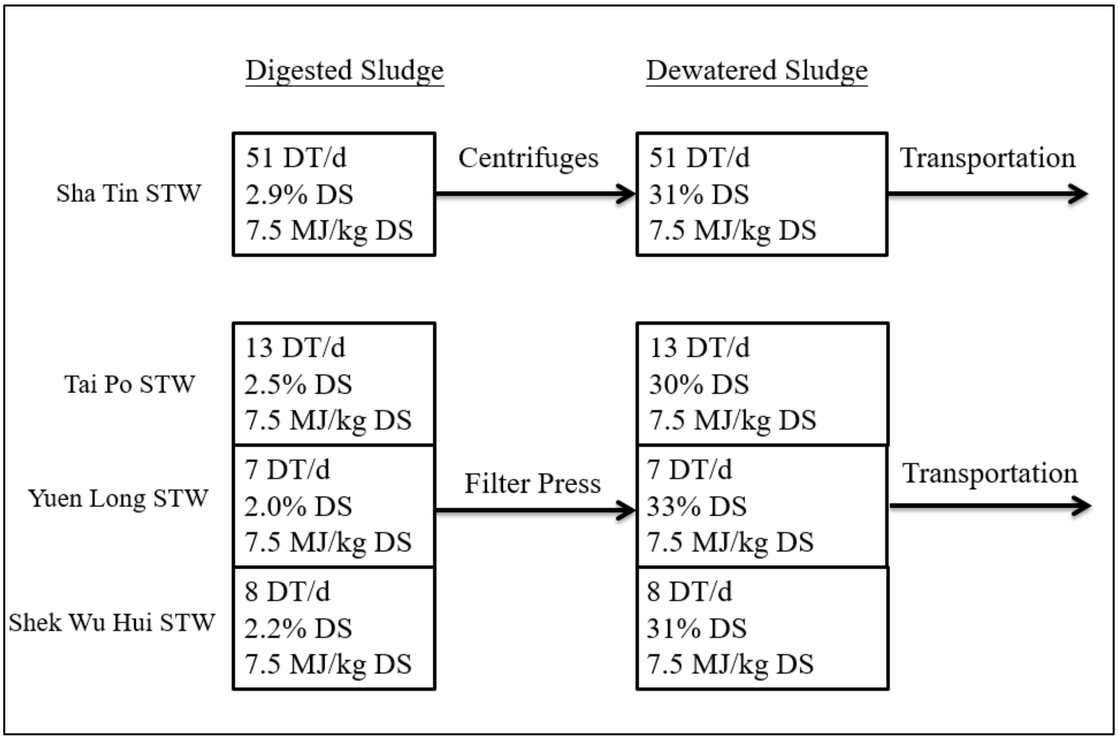

Figure A1- 5 Process Flow Diagram of Dewatering (Scenario 2)

Table A1- 7 Methodology of Dewatering Cost Estimation (Scenario 2)

Capital Cost[1]	(Table A1- 3)
Operation and Maintenance (O&M) Cost[1]	(Table A1- 3)
Transportation Cost	(Table A1- 3) The treated sludge is transported to SENT landfill for Sha Tin and Tai Po STWs, and NENT landfill for Yuen Long and Shek Wu Hui STWs for final disposal.*

Table A1- 8 Data and Assumptions for Dewatering Cost Estimation (Scenario 2)

	ST	TP	YL	SWH
Daily volume (m^3)*	1,620	571	312	844
Percent DS (%)*	4.0	3.0	3.5	3.5
Percent DS in dewatered sludge (%)*	31	30	33	31
Operation hours per day*	24	16	8	16
Operation days per year*	365	300	326	365
Wage for labor (HK$/hr)[2]	30			
Travelling distance (km)[5]	38	29	24	8
Diesel consumption (kg/km)[6]	0.168			
Volume of trucks (m^3)*	13	12	20	12
Cost of diesel (HK$/L)[7]	12.87			

*2013 data from Drainage Services Department (DSD), HKSAR

A1.2.2.Landfill Disposal (Scenario 2)

Table A1- 9 Methodology for Landfill Disposal Cost Estimation (Scenario 2)

Capital Cost[1]	(Table A1- 5)
Operation and Maintenance (O&M) Cost[1]	(Table A1- 5)

Table A1- 10 Data and Assumptions for Landfill Cost Estimation (Scenario 2)

	ST (SENT)	TP (SENT)	YL (NENT)	SWH (NENT)
Daily volume (m^3)	192	52	30	37
Operation hours per day[8,9]	15	15	10	10
Operation days per year[8,9]	365	365	365	365
Wage for labor (HK$/hr)[2]	30			
Cost of diesel (HK$/L)[7]	12.87			
Industrial land cost (HK$/$ft^2$)[10]	850			

A3.3. Scenario 3

Figure A1- 6 Process Diagram of Scenario 3

A1.3.1.Anaerobic Digestion (Scenario 3)

Process Flow Diagram of Anaerobic Digestion (Scenario 3): Refer to Figure A1- 2

Table A1- 11 Methodology of Anaerobic Digestion Cost Estimation (Scenario 3)

Capital Cost[1]	(Table A1- 1)
Operation and Maintenance (O&M) Cost[1]	(Table A1- 1)
Transportation Cost	(Table A1- 1)

Table A1- 12 Data and Assumptions for Anaerobic Digestion Cost Estimation (Scenario 3)

<table>
<tr><th></th><th>ST</th><th>TP</th><th>YL</th><th>SWH</th></tr>
<tr><td>Daily volume (m^3)*</td><td colspan="4" rowspan="9">(Table A1- 2)</td></tr>
<tr><td>Percent DS (%)*</td></tr>
<tr><td>Percent DS after AD (%)*</td></tr>
<tr><td>Percent VS (%)*</td></tr>
<tr><td>Percent VS after AD (%)*</td></tr>
<tr><td>Operation hours per day*</td></tr>
<tr><td>Operation days per year*</td></tr>
<tr><td>Wage for labor (HK$/hr)[2]</td></tr>
<tr><td>Annual methane production (m^3)*</td></tr>
</table>

*2013 data from Drainage Services Department (DSD), HKSAR

A1.3.2.Dewatering (Scenario 3)

Process Flow Diagram of Dewatering (Scenario 3): Refer to Figure A1- 3

Table A1- 13 Methodology of Dewatering Cost Estimation (Scenario 3)

Capital Cost[1]	(Table A1- 3)
Operation and Maintenance (O&M) Cost[1]	(Table A1- 3)
Transportation Cost	(Table A1- 3)

Table A1- 14 Data and Assumptions for Dewatering Cost Estimation (Scenario 3)

<table>
<tr><th></th><th>ST</th><th>TP</th><th>YL</th><th>SWH</th></tr>
<tr><td>Daily volume (m^3)*</td><td rowspan="9" colspan="4">(Table A1- 4)</td></tr>
<tr><td>Percent DS (%)*</td></tr>
<tr><td>Percent DS in dewatered sludge (%)*</td></tr>
<tr><td>Operation hours per day*</td></tr>
<tr><td>Operation days per year*</td></tr>
<tr><td>Wage for labor (HK$/hr)[2]</td></tr>
<tr><td>Diesel consumption (kg/km)[6]</td></tr>
<tr><td>Volume of trucks (m^3)*</td></tr>
<tr><td>Cost of diesel (HK$/L)[7]</td></tr>
<tr><td>Travelling distance (km)[5]</td><td>44.2</td><td>50.5</td><td>25.8</td><td>39.6</td></tr>
</table>

*2013 data from Drainage Services Department (DSD), HKSAR

A1.3.3.Incineration (Scenario 3)

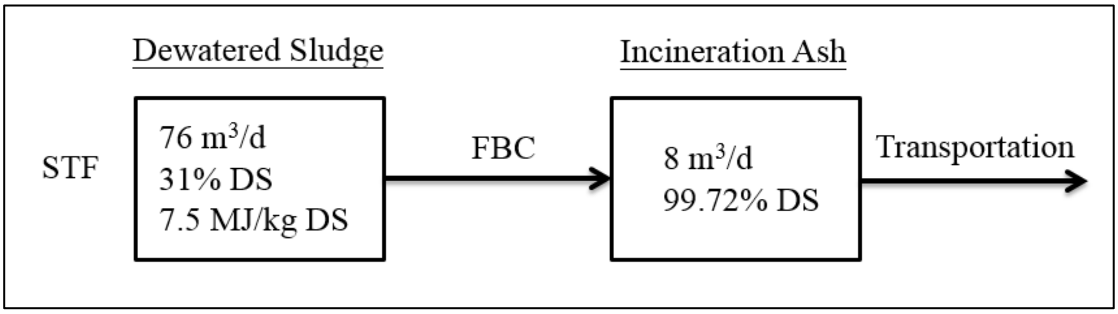

Figure A1- 7 Process Flow Diagram for Fluidized-Bed Incineration (Scenario 3)

Table A1- 15 Methodology for Incineration Cost Estimation (Scenario 3)

Capital Cost[1]	The capital cost includes the purchase and installation of the fluidized-bed incinerators, costs for ancillary equipment and construction of the building for accommodation for the incinerators.
Operation and Maintenance (O&M) Cost[1]	Sludge incineration is a centralized process in which sludge from STWs are transported to the STF for incineration. Volume reduction by 90% is achieved by this process[11]. The moisture content of the incineration ash is reduced to 0.28%[12]. The incinerators are assumed to be operating continuously due to the large fuel requirements for startup. Fuel oil is used as an auxiliary fuel for maintaining the high operating temperature. Diesel is assumed to be the fuel oil used. The treatment of nitrogen oxides (NO_x) by selective catalytic reactor to meet the air emission standard has been assumed and such abatement cost is estimated.
Transportation Cost	(Table A1- 1) The incineration ash is assumed to be transported to the WENT landfill, which is the nearest landfill to the STF, for final disposal

[11] Sludge Treatment Facility (STF) (2005). Retrieved February 11, 2014 from Government of HKSAR, Environmental Protection Department (EPD) Web site: http://www.epd.gov.hk/epd/english/environmentinhk/waste/prob_solutions/WFdev_TMSTF.html
[12] User Guidelines for Waste and Byproduct Materials in Pavement Construction (2012). Retrieved October 12, 2014 from U.S. Department of Transportation, Federal Highway Administration Web site: http://www.fhwa.dot.gov/publications/research/infrastructure/structures/97148/ss1.cfm

Table A1- 16 Data and Assumptions for Incineration Cost Estimation (Scenario 3)

	STF
Daily volume (m^3)	76
Averaged percent DS (%)	31.25
Averaged percent VS (%)	57.00
Operation hours per day[11]	24
Operation days per year[11]	360
Wage for labor (HK$/hr)[2]	30
Travelling distance (km)[5]	1.7
Diesel consumption (kg/km)[6]	0.168
Volume of trucks (m^3)	14
Cost of diesel (HK$/L)[7]	12.87
Heating value of sludge (MJ/kg DS)[6]	7,500

A1.3.4.Landfill Disposal (Scenario 3)

Table A1- 17 Methodology for Landfill Disposal Cost Estimation (Scenario 3)

Capital Cost[1]	(Table A1- 5)
Operation and Maintenance (O&M) Cost[1]	(Table A1- 5)

Table A1- 18 Data and Assumptions for Landfill Cost Estimation (Scenario 3)

	WENT Landfill
Daily volume (m^3)	8
Operation hours per day[13]	12
Operation days per year[13]	365
Wage for labor (HK$/hr)[2]	30
Cost of diesel (HK$/L)[7]	12.87
Industrial land cost (HK$/$ft^2$)[10]	850

[13] West New Territories (WENT) Landfill (2013). Retrieved October 3, 2014 from HKSAR, EPD Web site: http://www.epd.gov.hk/epd/english/environmentinhk/waste/prob_solutions/msw_went.html

A3.4. Scenario 4

Figure A1- 8 Process Diagram of Scenario 4

A1.4.1.Dewatering (Scenario 4)

Process Flow Diagram of Dewatering (Scenario 4): Refer to Figure A1- 5

Table A1- 19 Methodology of Dewatering Cost Estimation (Scenario 4)

Capital Cost[1]	(Table A1- 3)
Operation and Maintenance (O&M) Cost[1]	(Table A1- 3)
Transportation Cost	(Table A1- 3)

Table A1- 20 Data and Assumptions for Dewatering Cost Estimation (Scenario 4)

	ST	TP	YL	SWH
Daily volume (m^3)*				
Percent DS (%)*				
Percent DS in dewatered sludge (%)*				
Operation hours per day*				
Operation days per year*	(Table A1- 8)			
Wage for labor (HK$/hr)[2]				
Diesel consumption (kg/km)[6]				
Volume of trucks (m^3)*				
Cost of diesel (HK$/L)[7]				
Travelling distance (km)[5]	44.2	50.5	25.8	39.6

*2013 data from Drainage Services Department (DSD), HKSAR

A1.4.2.Incineration (Scenario 4)

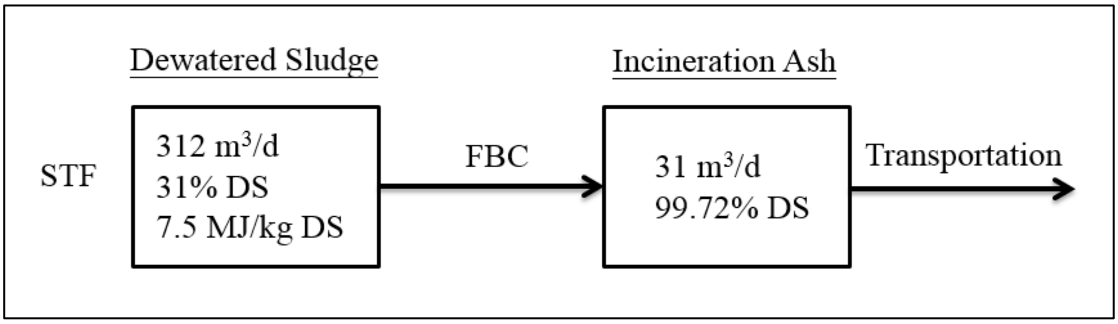

Figure A1- 9 Process Flow Diagram of Fluidized-Bed Incineration (Scenario 4)

Table A1- 21 Methodology for Incineration Cost Estimation (Scenario 4)

Capital Cost[1]	(Table A1- 15)
Operation and Maintenance (O&M) Cost[1]	(Table A1- 15)
Transportation Cost	(Table A1- 15)

Table A1- 22 Data and Assumptions for Incineration Cost Estimation (Scenario 4)

	STF
Daily volume (m^3)	312
Averaged percent DS (%)*	31.25
Averaged percent VS (%)	67.00
Operation hours per day[1]	24
Operation days per year[1]	360
Wage for labor (HK$/hr)[2]	30
Travelling distance (km)[5]	1.7
Diesel consumption (kg/km)[6]	0.168
Volume of trucks (m^3)	14
Cost of diesel (HK$/L)[7]	12.87
Heating value of sludge (MJ/kg DS)[6]	10,500

A1.4.3.Landfill Disposal (Scenario 4)

Table A1- 23 Methodology for Landfill Disposal Cost Estimation (Scenario 4)

Capital Cost[1]	(Table A1- 5)
Operation and Maintenance (O&M) Cost[1]	(Table A1- 5)

Table A1- 24 Data and Assumptions for Landfill Cost Estimation (Scenario 4)

	WENT Landfill
Daily volume (m^3)	31
Operation hours per day[13]	12
Operation days per year[13]	365
Wage for labor (HK$/hr)[2]	30
Cost of diesel (HK$/L)[7]	12.87
Industrial land cost (HK$/$ft^2$)[10]	850

A3.5. Scenario 5

Figure A1- 10 Process Diagram of Scenario 5

A1.5.1.Anaerobic Digestion (Scenario 5)

Process Flow Diagram of Anaerobic Digestion (Scenario 5): Refer to Figure A1- 2

Table A1- 25 Methodology of Anaerobic Digestion Cost Estimation (Scenario 5)

Capital Cost[1]	(Table A1- 1)
Operation and Maintenance (O&M) Cost[1]	(Table A1- 1)
Transportation Cost	(Table A1- 1)

Table A1- 26 Data and Assumptions for Anaerobic Digestion Cost Estimation (Scenario 5)

	ST	TP	YL	SWH
Daily volume (m^3)*				
Percent DS (%)*				
Percent DS after AD (%)*				
Percent VS (%)*				
Percent VS after AD (%)*	(Table A1- 2)			
Operation hours per day*				
Operation days per year*				
Wage for labor (HK$/hr)[2]				
Annual methane production (m^3)*				

*2013 data from Drainage Services Department (DSD), HKSAR

A1.5.2.Dewatering (Scenario 5)

Process Flow Diagram of Dewatering (Scenario 5): Refer to Figure A1- 3

Table A1- 27 Methodology of Dewatering Cost Estimation (Scenario 5)

Capital Cost[1]	(Table A1- 3)
Operation and Maintenance (O&M) Cost[1]	(Table A1- 3)
Transportation Cost	(Table A1- 3)

Table A1- 28 Data and Assumptions for Dewatering Cost Estimation (Scenario 5)

	ST	TP	YL	SWH
Daily volume (m^3)*	(Table A1- 4)			
Percent DS (%)*				
Percent DS in dewatered sludge (%)*				
Operation hours per day*				
Operation days per year*				
Wage for labor (HK$/hr)[2]				
Diesel consumption (kg/km)[6]				
Volume of trucks (m^3)*				
Cost of diesel (HK$/L)[7]				
Travelling distance (km)[5]	44.2	50.5	25.8	39.6

*2013 data from Drainage Services Department (DSD), HKSAR

A1.5.3.Incineration (Scenario 5)

Process Flow Diagram for Fluidized-Bed Incineration (Scenario 5): Refer to Figure A1- 7

Table A1- 29 Methodology for Incineration Cost Estimation (Scenario 5)

Capital Cost[1]	(Table A1- 15)
Operation and Maintenance (O&M) Cost[1]	(Table A1- 15)
Transportation Cost	It was assumed that the raw material for cement production requires transportation. As the replacement of the material by sludge ash was assumed to place no extra burden to transportation, the cost of transportation was not included.

Data and Assumptions for Incineration Cost Estimation (Scenario 5): Refer to Table A1- 16

A1.5.4.Cement Production

Table A1- 30 Methodology of Cement Production Cost Estimation (Scenario 5)

Economic savings	The sludge ash from incineration was assumed to be used to replace the clinker in the cement production process[14]. This final destination for sludge ash was assumed to place no extra requirement for the cement production industry, thus only the economic savings from material substitution were considered. The savings from offset of energy (coal) requirement in the original clinker production process was included.

Table A1- 31 Data and Assumptions for Cement Production Cost Estimation (Scenario 5)

Clinker price[14]	US$30.3/ton
Energy savings[14]	850 kcal/ton

[14] Lam, H. K., Barford, J. P. & Mckay, G. (2010). Utilization of Incineration Waste Ash Residues in Portland Cement Clinker. Chemical Engineering Transactions, 21 757-762.

A3.6. Scenario 6

Figure A1- 11 Process Diagram of Scenario 6

A1.6.1.Dewatering (Scenario 6)

Process Flow Diagram of Dewatering (Scenario 6): Refer to Figure A1- 5

Table A1- 32 Methodology of Dewatering Cost Estimation (Scenario 6)

Capital Cost[1]	(Table A1- 3)
Operation and Maintenance (O&M) Cost[1]	(Table A1- 3)
Transportation Cost	(Table A1- 3)

Table A1- 33 Data and Assumptions for Dewatering Cost Estimation (Scenario 6)

<table>
<tr><th></th><th>ST</th><th>TP</th><th>YL</th><th>SWH</th></tr>
<tr><td>Daily volume (m^3)*</td><td colspan="4" rowspan="9">(Table A1- 8)</td></tr>
<tr><td>Percent DS (%)*</td></tr>
<tr><td>Percent DS in dewatered sludge (%)*</td></tr>
<tr><td>Operation hours per day*</td></tr>
<tr><td>Operation days per year*</td></tr>
<tr><td>Wage for labor (HK$/hr)[2]</td></tr>
<tr><td>Diesel consumption (kg/km)[6]</td></tr>
<tr><td>Volume of trucks (m^3)*</td></tr>
<tr><td>Cost of diesel (HK$/L)[7]</td></tr>
<tr><td>Travelling distance (km)[5]</td><td>44.2</td><td>50.5</td><td>25.8</td><td>39.6</td></tr>
</table>

*2013 data from Drainage Services Department (DSD), HKSAR

A1.6.2.Incineration (Scenario 6)

Process Flow Diagram of Fluidized-Bed Incineration (Scenario 6): Refer to Figure A1- 9

Table A1- 34 Methodology for Incineration Cost Estimation (Scenario 6)

Capital Cost[1]	(Table A1- 15)
Operation and Maintenance (O&M) Cost[1]	(Table A1- 15)
Transportation Cost	(Table A1- 15)

Table A1- 35 Data and Assumptions for Incineration Cost Estimation (Scenario 6)

	STF
Daily volume (m^3)	(Table A1- 22)
Averaged percent DS (%)*	
Averaged percent VS (%)	
Operation hours per day[1]	
Operation days per year[1]	
Wage for labor (HK$/hr)[2]	
Travelling distance (km)[5]	
Diesel consumption (kg/km)[6]	
Volume of trucks (m^3)	
Cost of diesel (HK$/L)[7]	
Heating value of sludge (MJ/kg DS)[6]	

A1.6.3.Cement Production (Scenario 6)

Table A1- 36 Methodology of Cement Production Cost Estimation (Scenario 6)

Economic savings	(Table A1- 30)

Table A1- 37 Data and Assumptions for Cement Production Cost Estimation (Scenario 6)

Clinker price[15]	(Table A1- 31)
Energy savings[16]	

[15] China Resources Cement's Turnover Reaches HK$12.9 Billion Profit Up 80.4% to HK$1.15 Billion in 1H 2013 (2013). Retrieved May 6, 2015 from China Resources Cement Holdings Limited, Web site: http://www.crcement.com/home/Newscentre/Companynews/201409/t20140919_311330.html

[16] Xu, C. Q., Chen, W. & Hong, J. L. (2014). Life-cycle Environmental and Economic Assessment of Sewage Sludge Treatment in China. *Journal of Cleaner Production*, 67 79-87.

Part 2A: Environmental Impact Assessment

A2.1. Dewatering

Table A1- 38 Atmospheric Emissions Estimation of Dewatering Process

Description	Data and Assumptions
Electricity Consumption	
The components of the fuel mix for electricity generation in Hong Kong include coal (54%), natural gas (23%) and imported nuclear power (23%)[17].	Emissions from electricity generation (kg/kWh)[18]: GHGs: ▪ 9.0×10^{-1} (Coal) ▪ 7.3×10^{-1} (LNG) ▪ 1.9×10^{-2} (Nuclear) SO_2: ▪ 4.5×10^{-3} (Coal) ▪ 1.7×10^{-4} (LNG) ▪ 2.1×10^{-5} (Nuclear) NO_X: ▪ 3.0×10^{-3} (Coal) ▪ 7.5×10^{-4} (LNG) ▪ 2.5×10^{-5} (Nuclear)
Chemicals Addition	
Polymer is added to facilitate better dewatering performances in Sha Tin and Tai Po STWs, while both polymer and ferric chloride are used in Yuen Long and Shek Wu Hui STWs.	Emissions from chemical addition: Estimated by EIO-LCA tool[19]
Transportation	
Emissions from transportation were estimated by considering the diesel consumption for vehicles and total transportation distance.	Emissions from diesel consumption [20] (kg/km): $CO = 5.0 \times 10^4$ $NO_X = 4.0 \times 10^3$ GHGs[21] = 7.95 (CO_2-eq.)

[17] The Energy Scene of Hong Kong (2013). Retrieved September 17, 2014 from HKSAR, EMSD Web site: http://www.energyland.emsd.gov.hk/en/energy/energy_use/energy_scene.html

[18] Turconi, R., Boldrin, A. & Astrup, T. (2013). Life cycle assessment (LCA) of electricity generation technologies: Overview, comparability and limitations. *Renewable and Sustainable Energy Reviews, 28* 555-565.

[19] Carnegie Mellon University (CMU). Economic Input-Output LCA. 2006 Retrieved February 2, 2014 from: http://www.eiolca.net.

[20] Shen, X., Yao, Z., Zhang, Q., Wagner, D. V., Huo, H., Zhang, Y., … He, K. (2015). Development of database of real-world diesel vehicle emission factors for China. *Journal of Environmental Sciences (China)*, 31, 209–220.

[21] Rose, L., Hussain, M., Ahmed, S., Malek, K., Costanzo, R., & Kjeang, E. (2013). A comparative life cycle assessment of diesel and compressed natural gas powered refuse collection vehicles in a Canadian city. *Energy Policy*, 52, 453–461.

A2.2. Anaerobic Digestion

Table A1- 39 Atmospheric Emissions Estimation of Anaerobic Digestion Process

Description	Data and Assumptions
Electricity Consumption	
Emissions from electricity consumption and the emission offsets from methane production from the processes were estimated.	Emissions from electricity generation (kg/kWh): (Table A1- 38)
Fugitive Emissions	
Methane was produced in the AD process and 1% of the methane was assumed to be the fugitive emission from the process[22].	
Transportation	
As all the anaerobic digestion process is followed by dewatering and both the treatment process occur in the same STW, no transportation is required after AD process.	

[22] Gould, M., Tsang, R. & Bandi, R. T. (2008). A Greenhouse Gas Emissions Accounting Model for Biosolids Management Planning. *2008 NC AWWA-WEF Annual Conference*.

A2.3. Incineration

Table A1- 40 Atmospheric Emissions Estimation of Incineration Process

Description	Data and Assumptions
Electricity Consumption	
Emissions from electricity consumption and the emission offsets from heat energy recovery from incineration were estimated.	Emissions from electricity generation (kg/kWh): (Table A1- 38)
Fuel Oil Consumption	
Emissions from combustion of fuel oil as auxiliary fuel to maintain the incinerator temperature were considered. Diesel has been assumed to be the fuel oil used.	Emissions from diesel consumption (kg/km): (Table A1- 38)
Sludge combustion	
The combustion of organic substances in sewage sludge was estimated by stoichiometric calculation. The NO_2 emission was capped 6.36 kg/DT based on calculation with reference to the emission limit in Hong Kong[23].	Sludge combustion equation: $C_5H_7O_2N+\frac{27}{4}O_2 \rightarrow 5CO_2+\frac{7}{2}H_2O+NO_2$
Transportation	
The incineration ash from STF is transported to WENT landfill for disposal.	Emissions from diesel consumption (kg/km): (Table A1- 38)

[23] Guidance Note on the Best Practicable Means for Incinerators (Sewage Sludge Incineration) (2010). Retrieved November 26, 2014 from Government of HKSAR, Environmental Protection Department (EPD) Web site: http://www.epd.gov.hk/epd/sites/default/files/epd/english/environmentinhk/air/guide_ref/files/bpm12_3_2010.pdf

A2.4. Landfill Disposal

Table A1- 41 Atmospheric Emissions Estimation of Landfill Disposal

Description	Data and Assumptions
Diesel Consumption	
Atmospheric emissions from diesel consumption by excavation and earth-moving vehicles in landfills were considered.	Emissions from diesel consumption (mg/MJ)[24]: CO = 6.5×10^{-3} NO_x = 5.0×10^{2} SO_2 = 6.7×10^{1} CO_2 = 7.6×10^{3} CH_4 = 4.2 N_2O = 1.9
Landfill Gas Production	
Landfill gas that consists mainly of methane was produced in landfill sites. Emission offsets from methane in landfill gas was included.	Emissions from electricity generation (kg/kWh): (Table A1- 38)

[24] Miller, S. A., & Theis, T. L. (2006). Comparison of life cycle inventory databases. *Journal of industrial ecology*, *10*(1-2), 133-147.

A2.5. Cement Production

Table A1- 42 Atmospheric Emissions Estimation of Cement Production

Energy savings	
Emission offsets from the energy savings from material substitution in cement production were estimated.	Emissions from coal combustion (kg/MJ): CO_2 [25] = 9.3 x 10^{-2} CO [6] = 1.4 x 10^{-3} NO_x = 2.6 x 10^{-4} CH_4 = 6 x 10^{-7} SO_2 = 2.2 x 10^{-3}
Polymer Addition	
In the original cement production process, polymer is added for clinker production. The offset from clinker substitution by sludge ash was estimated.	Emissions from polymer addition: Estimated by EIO-LCA tool[19]

[25] Monni, S. (2012). From landfilling to waste incineration: Implications on GHG emissions of different actors. *International Journal of Greenhouse Gas Control*, 8, 82–89.

Part 2B: Life-cycle Impact Assessment

Table A1- 43 Methodologies and Factors in Life-cycle Impact Assessment

Description	Data and Assumptions
Land Occupation	
The Land Occupation impacts of incineration, composting and landfill disposal were evaluated, while the impacts on land caused by anaerobic digestion and dewatering were negligible.	▪ Land area requirement[1] ▪ Occupation period = 30 years (temporal boundary of this study)
Climate Change	
The global warming effects of treatment processes were presented. Greenhouse gases including CO_2, CH_4 and N_2O are contributors of this impact category.	GWP_{20} (kg CO_2-eq./kg)[26]: ▪ $CO_2 = 1$ ▪ $CH_4 = 280$ ▪ $N_2O = 56$
Human toxicity	
This impact category includes the human health impacts caused by NO_x and NH_3 emissions.	HTP_{20} (kg 1,4-DCB eq./kg)[26]: ▪ $NO_x = 1.2$ ▪ $NH_3 = 0.1$
Acidification	
Acidifying atmospheric emissions could be converted to acids when dissolved in rainwater or air moisture. The effects of acidifying emissions, including SO_2, NO_x and NH_3, were evaluated in this category.	AP (kg SO_2-eq./kg)[26]: ▪ $SO_2 = 1$ ▪ $NO_x = 0.7$ ▪ $NH_3 = 1.88$
Eutrophication	
Atmospheric deposition of nitrogen-containing emissions into water bodies may cause eutrophication. Emissions evaluated in this study include NO_x and NH_3.	EP (kg PO_4^{3-}-eq./kg)[26]: ▪ $NO_x = 0.35$ ▪ $NH_3 = 0.13$

[26] Guinee, J. B. (2002). *Handbook on Life Cycle Assessment: Operational Guide to the ISO Standards*. Netherlands: Kluwer Academic Publishers.

Part 3: Results

A3.1. Life-cycle Cost Analysis

Table A1- 44 Life-cycle Costs (Million US$) of Sludge Treatment Scenarios in Sha Tin STW

	AD	Dewatering	Incineration	Landfill	Cement Production	Total
S1	(35,363,378)	2,399,757		486,271,714		453,308,093
S2		6,578,564		2,218,794,217		2,225,372,781
S3	(35,363,378)	1,708,995	10,244,831	51,622,325		28,212,773
S4		6,869,359	63,801,663	225,347,522		296,018,544
S5	(35,363,378)	1,708,995	10,244,831		(83,770)	(23,493,322)
S6		6,869,359	63,801,663		(386,416)	70,284,606

Table A1- 45 Life-cycle Costs (Million US$) of Sludge Treatment Scenarios in Tai Po STW

	AD	Dewatering	Incineration	Landfill	Cement Production	Total
S1	(10,368,362)	1,575,117		157,600,787		148,807,542
S2		3,158,249		611,137,293		614,295,542
S3	(10,368,362)	1,709,571	3,223,312	16,241,834		10,806,354
S4		3,494,384	17,432,864	61,572,887		82,500,134
S5	(10,368,362)	1,709,571	3,223,312		(15,000)	(5,450,479)
S6		3,494,384	17,432,864		(101,845)	20,825,403

Table A1- 46 Life-cycle Costs (Million US$) of Sludge Treatment Scenarios in Yuen Long STW

	AD	Dewatering	Incineration	Landfill	Cement Production	Total
S1	(1,851,345)	2,411,452		66,006,576		66,566,683
S2		5,809,743		354,102,723		359,912,466
S3	(1,851,345)	1,085,520	1,266,549	6,381,969		6,882,693
S4		5,820,999	10,019,367	35,388,410		51,228,776
S5	(1,851,345)	1,085,520	1,266,549		(11,097)	489,627
S6		5,820,999	10,019,367		(65,021)	15,775,345

Table A1- 47 Life-cycle Costs (Million US$) of Sludge Treatment Scenarios in Shek Wu Hui

STW

	AD	Dewatering	Incineration	Landfill	Cement Production	Total
S1	4,412,542	4,412,542	4,412,542	4,412,542	4,412,542	4,412,542
S2	18,725,208	18,725,208	18,725,208	18,725,208	18,725,208	18,725,208
S3	4,412,542	4,412,542	4,412,542	4,412,542	4,412,542	4,412,542
S4	18,725,208	18,725,208	18,725,208	18,725,208	18,725,208	18,725,208
S5	4,412,542	4,412,542	4,412,542	4,412,542	4,412,542	4,412,542
S6	18,725,208	18,725,208	18,725,208	18,725,208	18,725,208	18,725,208

A3.2. Energy Balance

Table A1- 48 Energy Balance (kWh/DT) of Sludge Treatment Process in Scenarios

	AD	Dewatering	Incineration	Landfill	Cement Production	Total
S1	(858.98)	27.38		(0.03)		(831.63)
S2		60.19		(0.49)		59.71
S3	(858.98)	27.38	578.90			(252.70)
S4		60.19	1761.36			1821.56
S5	(858.98)	27.38	578.90		(6.74)	(259.44)
S6		60.19	1761.36		(29.64)	1791.92

Appendix 2

Background Information and Assumptions

Table A2- 1 Information about mock building

Number of floors	35
Apartments per floor	24
Residents per apartment	3
Total number of apartments	840
Total number of residents	2520
Height of each floor (m)	3
Total height of building (m)	105

Table A2- 2 Water consumption information

Average freshwater consumption (m^3/person-day)	0.13
Daily freshwater consumption (m^3/day)	327.6
Average seawater consumption for flushing (m^3/person-day)	0.0838
Daily water consumption for flushing (m^3/day)	211.2
Water consumption for kitchen sink (% of freshwater consumption)	15
Daily blackwater quantity (for kitchen and toilet) (m^3/day)	292.0
Daily greywater quantity (consumption except kitchen and toilet) (m^3/day)	246.8

Table A2- 3 Parameters and assumptions for engineering system design

Parameters	
Hourly change parameter (K_h)	2.8
Building daily water usage time (hours)	24
Sanitary wares and equivalent (N)	
Kitchen basin	1
Toilet	0.5
Wash basin	0.75
Shower	0.75
Washing machine	1

Engineering water system designs

Water supply systems

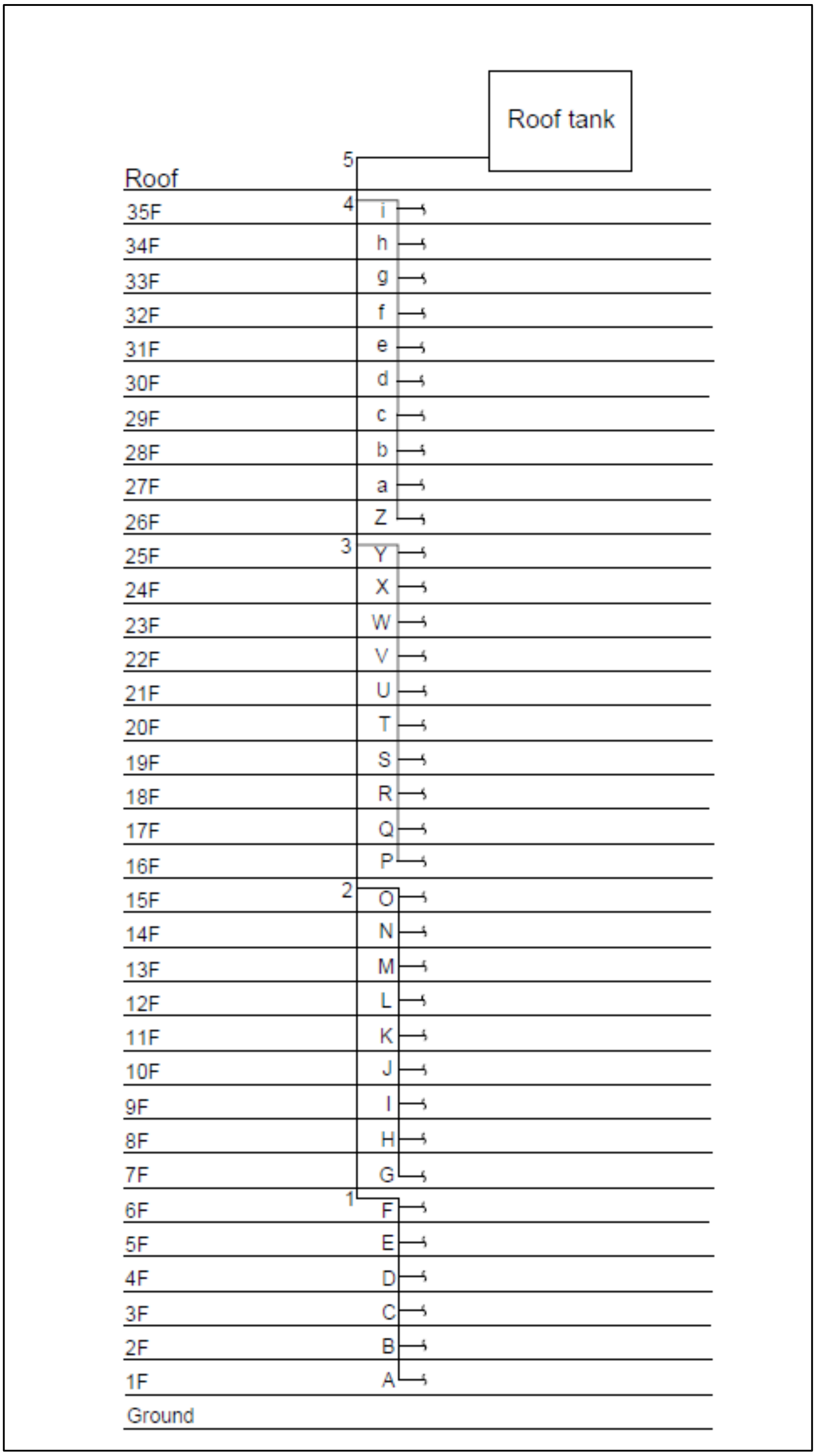

Figure A2- 0-1 Design of one set of pipeline in water supply systems

Scenario 1: Freshwater flushing

Table A2- 4 Hydraulic calculation for freshwater system in Scenario 1

Maximum daily freshwater consumption (m^3/day)	538.78
Average hourly freshwater consumption (m^3/hr)	22.45
Maximum hourly freshwater consumption (m^3/hr)	62.86
Average discharge probability of water supply equivalent of sanitary wares at maximum daily fresh water consumption (%)	2.60
a_c	0.0160

Table A2- 5 Design of pipeline system for freshwater supply for Scenario 1

Target pipe line	Water supply equivalent of this section	Sum of equivalents	Meanwhile discharge probability of water supply equivalent of sanitary wares	Flowrate(L/s)	Pipe diameter DN(mm)	Flow velocity (m/s)	Hydraulic gradient (mH20/m)	Length of pipeline (m)	Head loss (m H20)	Accumulated head loss (m)
A-B	24	24	0.2193	1.0525	40	0.9280	0.0287	2.75	0.0789	0.0789
B-C	24	48	0.1595	1.5315	50	0.7800	0.0151	2.75	0.0415	0.1205
C-D	24	72	0.1330	1.9157	50	0.9760	0.0229	2.75	0.0630	0.1834
D-E	24	96	0.1172	2.2508	50	1.1460	0.0308	2.75	0.0847	0.2681
E-F	24	120	0.1064	2.5545	50	1.3010	0.0389	2.75	0.1070	0.3751

Table A2- 5 Design of pipeline system for freshwater supply for Scenario 1 (Cont'd)

Target pipe line	Water supply equivalent of this section	Sum of equivalents	Meanwhile discharge probability of water supply equivalent of sanitary wares	Flowrate(L/s)	Pipe diameter DN(mm)	Flow velocity (m/s)	Hydraulic gradient (mH20/m)	Length of pipeline (m)	Head loss (m H20)	Accumulated head loss (m)
1-2				2.5545	50	1.3010	0.0389	24.75	0.9628	-
G-H	24	24	0.2193	1.0525	40	0.9280	0.0287	2.75	0.0789	0.0789
H-I	24	48	0.1595	1.5315	50	0.7800	0.0151	2.75	0.0415	0.1205
I-J	24	72	0.1330	1.9157	50	0.9760	0.0229	2.75	0.0630	0.1834
J-K	24	96	0.1172	2.2508	50	1.1460	0.0308	2.75	0.0847	0.2681
K-L	24	120	0.1064	2.5545	50	1.3010	0.0389	2.75	0.1070	0.3751
L-M	24	144	0.0985	2.8358	50	1.4440	0.0472	2.75	0.1298	0.5049
M-N	24	168	0.0923	3.1003	65	0.9340	0.0155	2.75	0.0426	0.5475
N-O	24	192	0.0873	3.3512	65	1.0100	0.0179	2.75	0.0492	0.5968
2-3				5.9058	80	1.2850	0.0231	27.5	0.6353	-
P-Q	24	24	0.2193	1.0525	40	0.9280	0.0287	2.75	0.0789	0.0789
Q-R	24	48	0.1595	1.5315	50	0.7800	0.0151	2.75	0.0415	0.1205
R-S	24	72	0.1330	1.9157	50	0.9760	0.0229	2.75	0.0630	0.1834

Table A2- 5 Design of pipeline system for freshwater supply for Scenario 1 (Cont'd)

Target pipe line	Water supply equivalent of this section	Sum of equivalents	Meanwhile discharge probability of water supply equivalent of sanitary wares	Flowrate(L/s)	Pipe diameter DN(mm)	Flow velocity (m/s)	Hydraulic gradient (mH20/m)	Length of pipeline (m)	Head loss (m H20)	Accumulated head loss (m)
S-T	24	96	0.1172	2.2508	50	1.1460	0.0308	2.75	0.0847	0.2681
T-U	24	120	0.1064	2.5545	50	1.3010	0.0389	2.75	0.1070	0.3751
U-V	24	144	0.0985	2.8358	50	1.4440	0.0472	2.75	0.1298	0.5049
V-W	24	168	0.0923	3.1003	65	0.9340	0.0155	2.75	0.0426	0.5475
W-X	24	192	0.0873	3.3512	65	1.0100	0.0179	2.75	0.0492	0.5968
X-Y	24	216	0.0831	3.5913	65	1.0820	0.0204	2.75	0.0561	0.6529
3-4				9.4970	100	1.1620	0.0137	27.5	0.3768	-
Z-a	24	24	0.2193	1.0525	40	0.9280	0.0287	2.75	0.0789	0.0789
a-b	24	48	0.1595	1.5315	50	0.7800	0.0151	2.75	0.0415	0.1205
b-c	24	72	0.1330	1.9157	50	0.9760	0.0229	2.75	0.0630	0.1834
c-d	24	96	0.1172	2.2508	50	1.1460	0.0308	2.75	0.0847	0.2681
d-e	24	120	0.1064	2.5545	50	1.3010	0.0389	2.75	0.1070	0.3751
e-f	24	144	0.0985	2.8358	50	1.4440	0.0472	2.75	0.1298	0.5049

Table A2- 5 Design of pipeline system for freshwater supply for Scenario 1 (Cont'd)

Target pipe line	Water supply equivalent of this section	Sum of equivalents	Meanwhile discharge probability of water supply equivalent of sanitary wares	Flowrate(L/s)	Pipe diameter DN(mm)	Flow velocity (m/s)	Hydraulic gradient (mH20/m)	Length of pipeline (m)	Head loss (m H20)	Accumulated head loss (m)
f-g	24	168	0.0923	3.1003	65	0.9340	0.0155	2.75	0.0426	0.5475
g-h	24	192	0.0873	3.3512	65	1.0100	0.0179	2.75	0.0492	0.5968
h-i	24	216	0.0831	3.5913	65	1.0820	0.0204	2.75	0.0561	0.6529
4-5				13.0883	100	1.6020	0.0248	2.75	0.0682	-
$Tank_{roof}$				6.24	100	0.76		100	0.63	

Table A2- 6 Freshwater tanks for Scenario 1

Roof tank		
Working volume	m^3	31.43
Length	m	5.00
Width	m	2.50
Height	m	3.01
1. working height	m	2.51

Table A2- 6 Freshwater tanks for Scenario 1 (Cont'd)

2. protection height	m	0.30
3. Ineffective height in the bottom	m	0.20
Lowest water level of roof freshwater tank (Z_s)	m	107.85
H_1	m	96.25
H_2	m	0.33
H_3	m	10.00
H_B	m	1.27
Required water pressure at the inlet of roof tank (H)	m	113.64
Height between inlet pipe of roof tank and highest water level	m	0.15
Pipe central level of starting point of sunction pipe of pump	m	-2.00
Headloss between sunction pipe of pump and inlet pipe of roof tank	m	0.63
Required flow head at the inlet pipe of roof tank	m	0.50
Sump tank		
Working volume	m3	107.76
Length	m	8.00
Width	m	6.00
Height	m	2.74
1. working height	m	2.24
2. protection height	m	0.30
3. Ineffective height in the bottom	m	0.20

Table A2- 7 Freshwater pump for Scenario 1

Flow	m^3/h	22.45
Pumping head	m	125.01

Scenario 2: Seawater flushing

Table A2- 8 Hydraulic calculation for freshwater system in Scenario 2

Maximum daily freshwater consumption (m^3/day)	327.6
Maximum daily seawater consumption (m^3/day)	211.2
Average hourly freshwater consumption (m^3/hr)	13.65
Average hourly seawater consumption (m^3/hr)	8.80
Maximum hourly freshwater consumption (m^3/hr)	38.22
Maximum hourly seawater consumption (m^3/hr)	24.64
Sum of equivalent of sanitary wares per household (N_g)	3.5
Average discharge probability of water supply equivalent of sanitary wares at maximum daily fresh water consumption (%)	1.81
a_c	0.0094

Table A2- 9 Design of pipeline system for freshwater supply for Scenario 2

Target pipe line	Water supply equivalent of this section	Sum of equivalents	Meanwhile discharge probability of water supply equivalent of sanitary wares	Flowrate(L/s)	Pipe diameter DN(mm)	Flow velocity (m/s)	Hydraulic gradient (mH20/m)	Length of pipeline (m)	Head loss (m H20)	Accumulated head loss (m)
A-B	21	21	0.2271	0.9540	32	1.1290	0.0491	2.75	0.1349	0.1349
B-C	21	42	0.1633	1.3714	50	0.6980	0.0123	2.75	0.0339	0.1688
C-D	21	63	0.1349	1.7004	50	0.8660	0.0183	2.75	0.0504	0.2192
D-E	21	84	0.1181	1.9835	50	1.0100	0.0244	2.75	0.0671	0.2863
E-F	21	105	0.1065	2.2372	50	1.1390	0.0305	2.75	0.0838	0.3701
1-2				2.2372	50	1.1390	0.0305	24.75	0.7549	-
G-H	21	21	0.2271	0.9540	32	1.1290	0.0491	2.75	0.1349	0.1349
H-I	21	42	0.1633	1.3714	50	0.6980	0.0123	2.75	0.0339	0.1688
I-J	21	63	0.1349	1.7004	50	0.8660	0.0183	2.75	0.0504	0.2192
J-K	21	84	0.1181	1.9835	50	1.0100	0.0244	2.75	0.0671	0.2863
K-L	21	105	0.1065	2.2372	50	1.1390	0.0305	2.75	0.0838	0.3701
L-M	21	126	0.0980	2.4702	50	1.2580	0.0366	2.75	0.1006	0.4707
M-N	21	147	0.0914	2.6873	50	1.3690	0.0428	2.75	0.1176	0.5883

Table A2- 9 Design of pipeline system for freshwater supply for Scenario 2 (Cont'd)

Target pipe line	Water supply equivalent of this section	Sum of equivalents	Meanwhile discharge probability of water supply equivalent of sanitary wares	Flowrate(L/s)	Pipe diameter DN(mm)	Flow velocity (m/s)	Hydraulic gradient (mH20/m)	Length of pipeline (m)	Head loss (m H20)	Accumulated head loss (m)
N-O	21	168	0.0861	2.8919	50	1.4730	0.0490	2.75	0.1347	0.7230
2-3				2.8919	50	1.4730	0.0490	27.5	1.3475	-
P-Q	21	21	0.2271	0.9540	32	1.1290	0.0491	2.75	0.1349	0.1349
Q-R	21	42	0.1633	1.3714	50	0.6980	0.0123	2.75	0.0339	0.1688
R-S	21	63	0.1349	1.7004	50	0.8660	0.0183	2.75	0.0504	0.2192
S-T	21	84	0.1181	1.9835	50	1.0100	0.0244	2.75	0.0671	0.2863
T-U	21	105	0.1065	2.2372	50	1.1390	0.0305	2.75	0.0838	0.3701
U-V	21	126	0.0980	2.4702	50	1.2580	0.0366	2.75	0.1006	0.4707
V-W	21	147	0.0914	2.6873	50	1.3690	0.0428	2.75	0.1176	0.5883
W-X	21	168	0.0861	2.8919	50	1.4730	0.0490	2.75	0.1347	0.7230
X-Y	21	189	0.0816	3.0864	65	0.9300	0.0154	2.75	0.0424	0.7654
3-4				3.0864	65	0.9300	0.0154	27.5	0.4235	-
Z-a	21	21	0.2271	0.9540	32	1.1290	0.0491	2.75	0.1349	0.1349

Table A2- 9 Design of pipeline system for freshwater supply for Scenario 2 (Cont'd)

Target pipe line	Water supply equivalent of this section	Sum of equivalents	Meanwhile discharge probability of water supply equivalent of sanitary wares	Flowrate(L/s)	Pipe diameter DN(mm)	Flow velocity (m/s)	Hydraulic gradient (mH20/m)	Length of pipeline (m)	Head loss (m H20)	Accumulated head loss (m)
a-b	21	42	0.1633	1.3714	50	0.6980	0.0123	2.75	0.0339	0.1688
b-c	21	63	0.1349	1.7004	50	0.8660	0.0183	2.75	0.0504	0.2192
c-d	21	84	0.1181	1.9835	50	1.0100	0.0244	2.75	0.0671	0.2863
d-e	21	105	0.1065	2.2372	50	1.1390	0.0305	2.75	0.0838	0.3701
e-f	21	126	0.0980	2.4702	50	1.2580	0.0366	2.75	0.1006	0.4707
f-g	21	147	0.0914	2.6873	50	1.3690	0.0428	2.75	0.1176	0.5883
g-h	21	168	0.0861	2.8919	50	1.4730	0.0490	2.75	0.1347	0.7230
h-i	21	189	0.0816	3.0864	65	0.9300	0.0154	2.75	0.0423	0.7653
4-5				3.0864	65	0.9300	0.0154	2.75	0.0424	-
$Tank_{roof}$				3.79	100	0.83		100	1.02	

Table A2- 10 Freshwater tanks for Scenario 2

Roof tank		
Working volume	m^3	19.11
Length	m	4
Width	m	2
Height	m	2.89
1. working height	m	2.39
2. protection height	m	0.30
3. Ineffective height in the bottom	m	0.20
Lowest water level of roof freshwater tank (Z_s)	m	107.82
H_1	m	96.25
H_2	m	0.30
H_3	m	10.00
H_B	m	1.27
Required water pressure at the inlet of roof tank (H)	m	113.88
Height between inlet pipe of roof tank and highest water level	m	0.15
Pipe central level of starting point of sunction pipe of pump	m	-2.00
Headloss between sunction pipe of pump and inlet pipe of roof tank	m	1.02
Required flow head at the inlet pipe of roof tank	m	0.50
Sump tank		
Working volume	m3	65.52
Length	m	7

Table A2- 10 Freshwater tanks for Scenario 2 (Cont'd)

Width	m	4
Height	m	2.84
1. working height	m	2.34
2. protection height	m	0.3
3. Ineffective height in the bottom	m	0.2

Table A2- 11 Freshwater pump for Scenario 2

Flow	m^3/h	13.65
Pumping head	m	125.27

Table A2- 12 Hydraulic calculation for seawater system in Scenario 2

Sum of equivalent of sanitary wares per household (N_g)	0.5
Average discharge probability of water supply equivalent of sanitary wares at maximum daily fresh water consumption (%)	8.15
a_c	0.0649

Table A2- 13 Design of pipeline system for seawater supply for Scenario 2

Target pipe line	Water supply equivalent of this section	Sum of equivalents	Meanwhile discharge probability of water supply equivalent of sanitary wares	Flowrate(L/s)	Pipe diameter DN(mm)	Flow velocity (m/s)	Hydraulic gradient (mH20/m)	Length of pipeline (m)	Head loss (m H20)	Accumulated head loss (m)
A-B	3	3	0.6300	0.3780	25	0.8360	0.0405	2.75	0.1114	0.1114
B-C	3	6	0.4665	0.5598	32	0.6630	0.0183	2.75	0.0503	0.1617
C-D	3	9	0.3933	0.7079	32	0.8380	0.0282	2.75	0.0776	0.2393
D-E	3	12	0.3493	0.8384	32	0.9920	0.0386	2.75	0.1062	0.3454
E-F	3	15	0.3193	0.9578	40	0.8450	0.0241	2.75	0.0663	0.4117
1-2				0.9578	40	0.8450	0.0241	24.75	0.5965	-
G-H	3	3	0.6300	0.3780	25	0.8360	0.0405	2.75	0.1114	0.1114
H-I	3	6	0.4665	0.5598	32	0.6630	0.0183	2.75	0.0503	0.1617
I-J	3	9	0.3933	0.7079	32	0.8380	0.0282	2.75	0.0776	0.2393
J-K	3	12	0.3493	0.8384	32	0.9920	0.0386	2.75	0.1062	0.3454
K-L	3	15	0.3193	0.9578	40	0.8450	0.0241	2.75	0.0663	0.4117
L-M	3	18	0.2970	1.0692	40	0.9430	0.0296	2.75	0.0814	0.4931
M-N	3	21	0.2797	1.1746	40	1.0360	0.0352	2.75	0.0968	0.5899

Table A2- 13 Design of pipeline system for seawater supply for Scenario 2 (Cont'd)

Target pipe line	Water supply equivalent of this section	Sum of equivalents	Meanwhile discharge probability of water supply equivalent of sanitary wares	Flowrate(L/s)	Pipe diameter DN(mm)	Flow velocity (m/s)	Hydraulic gradient (mH20/m)	Length of pipeline (m)	Head loss (m H20)	Accumulated head loss (m)
N-O	3	24	0.2657	1.2753	40	1.1240	0.0410	2.75	0.1128	0.7026
2-3				1.2753	40	1.1240	0.0410	27.5	1.1275	-
P-Q	3	3	0.6300	0.3780	25	0.8360	0.0405	2.75	0.1114	0.1114
Q-R	3	6	0.4665	0.5598	32	0.6630	0.0183	2.75	0.0503	0.1617
R-S	3	9	0.3933	0.7079	32	0.8380	0.0282	2.75	0.0776	0.2393
S-T	3	12	0.3493	0.8384	32	0.9920	0.0386	2.75	0.1062	0.3454
T-U	3	15	0.3193	0.9578	40	0.8450	0.0241	2.75	0.0663	0.4117
U-V	3	18	0.2970	1.0692	40	0.9430	0.0296	2.75	0.0814	0.4931
V-W	3	21	0.2797	1.1746	40	1.0360	0.0352	2.75	0.0968	0.5899
W-X	3	24	0.2657	1.2753	40	1.1240	0.0410	2.75	0.1128	0.7026
X-Y	3	27	0.2541	1.3721	50	0.6990	0.0123	2.75	0.0338	0.7365
3-4				1.3721	50	0.6990	0.0123	27.5	0.3383	-
Z-a	3	3	0.6300	0.3780	25	0.8360	0.0405	2.75	0.1114	0.1114

Table A2- 13 Design of pipeline system for seawater supply for Scenario 2 (Cont'd)

Target pipe line	Water supply equivalent of this section	Sum of equivalents	Meanwhile discharge probability of water supply equivalent of sanitary wares	Flowrate(L/s)	Pipe diameter DN(mm)	Flow velocity (m/s)	Hydraulic gradient (mH20/m)	Length of pipeline (m)	Head loss (m H20)	Accumulated head loss (m)
a-b	3	6	0.4665	0.5598	32	0.6630	0.0183	2.75	0.0503	0.1617
b-c	3	9	0.3933	0.7079	32	0.8380	0.0282	2.75	0.0776	0.2393
c-d	3	12	0.3493	0.8384	32	0.9920	0.0386	2.75	0.1062	0.3454
d-e	3	15	0.3193	0.9578	40	0.8450	0.0241	2.75	0.0663	0.4117
e-f	3	18	0.2970	1.0692	40	0.9430	0.0296	2.75	0.0814	0.4931
f-g	3	21	0.2797	1.1746	40	1.0360	0.0352	2.75	0.0968	0.5899
g-h	3	24	0.2657	1.2753	40	1.1240	0.0410	2.75	0.1128	0.7026
h-i	3	27	0.2541	1.3721	50	0.6990	0.0123	2.75	0.0338	0.7365
4-5				1.3721	50	0.6990	0.0123	2.75	0.0338	-
$Tank_{roof}$				2.44	50	1.24		100	3.58	

Table A2- 14 Seawater tanks for Scenario 2

Roof tank		
Working volume	m^3	12.3186
Length	m	3
Width	m	1.5
Height	m	3.24
1. working height	m	2.74
2. protection height	m	0.30
3. Ineffective height in the bottom	m	0.20
Lowest water level of roof freshwater tank (Z_s)	m	107.81
H_1	m	96.25
H_2	m	0.29
H_3	m	10.00
H_B	m	1.27
Required water pressure at the inlet of roof tank (H)	m	116.78
Height between inlet pipe of roof tank and highest water level	m	0.15
Pipe central level of starting point of sunction pipe of pump	m	-2.00
Headloss between sunction pipe of pump and inlet pipe of roof tank	m	3.58
Required flow head at the inlet pipe of roof tank	m	0.50
Sump tank		
Working volume	m3	42.2352
Length	m	6

Table A2- 14 Seawater tanks for Scenario 2 (Cont'd)

Width	m	4
Height	m	2.2598
1. working height	m	1.7598
2. protection height	m	0.3
3. Ineffective height in the bottom	m	0.2

Table A2- 15 Seawater pump for Scenario 2

Flow	m^3/h	8.80
Pumping head	m	128.46

Scenario 3 and 4: Greywater flushing (Aerobic and Anaerobic)

Table A2- 16 Hydraulic calculation for freshwater system in Scenario 3 and 4

Maximum daily freshwater consumption (m^3/day)	327.6
Maximum daily greywater consumption (m^3/day)	211.176
Average hourly freshwater consumption (m^3/hr)	13.65
Average hourly greywater consumption (m^3/hr)	8.799
Maximum hourly freshwater consumption (m^3/hr)	38.22
Maximum hourly greywater consumption (m^3/hr)	24.6372
Sum of equivalent of sanitary wares per household (N_g)	3.5
Average discharge probability of water supply equivalent of sanitary wares at maximum daily fresh water consumption (%)	1.8056
a_c	0.0094

Table A2- 17 Design of pipeline system for freshwater supply for Scenario 3 and 4

Target pipe line	Water supply equivalent of this section	Sum of equivalents	Meanwhile discharge probability of water supply equivalent of sanitary wares	Flowrate(L/s)	Pipe diameter DN(mm)	Flow velocity (m/s)	Hydraulic gradient (mH20/m)	Length of pipeline (m)	Head loss (m H20)	Accumulated head loss (m)
A-B	21	21	0.2271	0.9540	32	1.1290	0.0491	2.75	0.1349	0.1349
B-C	21	42	0.1633	1.3714	50	0.6980	0.0123	2.75	0.0339	0.1688
C-D	21	63	0.1349	1.7004	50	0.8660	0.0183	2.75	0.0504	0.2192
D-E	21	84	0.1181	1.9835	50	1.0100	0.0244	2.75	0.0671	0.2863
E-F	21	105	0.1065	2.2372	50	1.1390	0.0305	2.75	0.0838	0.3701
1-2				2.2372	50	1.1390	0.0305	24.75	0.7549	-
G-H	21	21	0.2271	0.9540	32	1.1290	0.0491	2.75	0.1349	0.1349
H-I	21	42	0.1633	1.3714	50	0.6980	0.0123	2.75	0.0339	0.1688
I-J	21	63	0.1349	1.7004	50	0.8660	0.0183	2.75	0.0504	0.2192
J-K	21	84	0.1181	1.9835	50	1.0100	0.0244	2.75	0.0671	0.2863
K-L	21	105	0.1065	2.2372	50	1.1390	0.0305	2.75	0.0838	0.3701
L-M	21	126	0.0980	2.4702	50	1.2580	0.0366	2.75	0.1006	0.4707
M-N	21	147	0.0914	2.6873	50	1.3690	0.0428	2.75	0.1176	0.5883

Table A2- 17 Design of pipeline system for freshwater supply for Scenario 3 and 4 (Cont'd)

Target pipe line	Water supply equivalent of this section	Sum of equivalents	Meanwhile discharge probability of water supply equivalent of sanitary wares	Flowrate(L/s)	Pipe diameter DN(mm)	Flow velocity (m/s)	Hydraulic gradient (mH20/m)	Length of pipeline (m)	Head loss (m H20)	Accumulated head loss (m)
N-O	21	168	0.0861	2.8919	50	1.4730	0.0490	2.75	0.1347	0.7230
2-3				2.8919	50	1.4730	0.0490	27.5	1.3475	-
P-Q	21	21	0.2271	0.9540	32	1.1290	0.0491	2.75	0.1349	0.1349
Q-R	21	42	0.1633	1.3714	50	0.6980	0.0123	2.75	0.0339	0.1688
R-S	21	63	0.1349	1.7004	50	0.8660	0.0183	2.75	0.0504	0.2192
S-T	21	84	0.1181	1.9835	50	1.0100	0.0244	2.75	0.0671	0.2863
T-U	21	105	0.1065	2.2372	50	1.1390	0.0305	2.75	0.0838	0.3701
U-V	21	126	0.0980	2.4702	50	1.2580	0.0366	2.75	0.1006	0.4707
V-W	21	147	0.0914	2.6873	50	1.3690	0.0428	2.75	0.1176	0.5883
W-X	21	168	0.0861	2.8919	50	1.4730	0.0490	2.75	0.1347	0.7230
X-Y	21	189	0.0816	3.0864	65	0.9300	0.0154	2.75	0.0424	0.7654
3-4				3.0864	65	0.9300	0.0154	27.5	0.4235	-
Z-a	21	21	0.2271	0.9540	32	1.1290	0.0491	2.75	0.1349	0.1349

Table A2- 17 Design of pipeline system for freshwater supply for Scenario 3 and 4 (Cont'd)

Target pipe line	Water supply equivalent of this section	Sum of equivalents	Meanwhile discharge probability of water supply equivalent of sanitary wares	Flowrate(L/s)	Pipe diameter DN(mm)	Flow velocity (m/s)	Hydraulic gradient (mH20/m)	Length of pipeline (m)	Head loss (m H20)	Accumulated head loss (m)
a-b	21	42	0.1633	1.3714	50	0.6980	0.0123	2.75	0.0339	0.1688
b-c	21	63	0.1349	1.7004	50	0.8660	0.0183	2.75	0.0504	0.2192
c-d	21	84	0.1181	1.9835	50	1.0100	0.0244	2.75	0.0671	0.2863
d-e	21	105	0.1065	2.2372	50	1.1390	0.0305	2.75	0.0838	0.3701
e-f	21	126	0.0980	2.4702	50	1.2580	0.0366	2.75	0.1006	0.4707
f-g	21	147	0.0914	2.6873	50	1.3690	0.0428	2.75	0.1176	0.5883
g-h	21	168	0.0861	2.8919	50	1.4730	0.0490	2.75	0.1347	0.7230
h-i	21	189	0.0816	3.0864	65	0.9300	0.0154	2.75	0.0423	0.7653
4-5				3.0864	65	0.9300	0.0154	2.75	0.0424	-
$Tank_{roof}$				3.79	100	0.83		100	1.02	

Table A2- 18 Freshwater tanks for Scenario 3 and 4

Roof tank		
Working volume	m^3	19.11
Length	m	4
Width	m	2
Height	m	2.89
1. working height	m	2.39
2. protection height	m	0.30
3. Ineffective height in the bottom	m	0.20
Lowest water level of roof freshwater tank (Z_s)	m	107.82
H_1	m	96.25
H_2	m	0.30
H_3	m	10.00
H_B	m	1.27
Required water pressure at the inlet of roof tank (H)	m	113.88
Height between inlet pipe of roof tank and highest water level	m	0.15
Pipe central level of starting point of sunction pipe of pump	m	-2.00
Headloss between sunction pipe of pump and inlet pipe of roof tank	m	1.02
Required flow head at the inlet pipe of roof tank	m	0.50
Sump tank		
Working volume	m3	65.52
Length	m	7

Table A2- 18 Freshwater tanks for Scenario 3 and 4 (Cont'd)

Width	m	4
Height	m	2.84
1. working height	m	2.34
2. protection height	m	0.3
3. Ineffective height in the bottom	m	0.2

Table A2- 19 Freshwater pump for Scenario 3 and 4

Flow	m^3/h	13.65
Pumping head	m	125.27

Table A2- 20 Hydraulic calculation for greywater system in Scenario 3 and 4

Sum of equivalent of sanitary wares per household (N_g)	0.5
Average discharge probability of water supply equivalent of sanitary wares at maximum daily fresh water consumption (%)	8.1472
a_c	0.0649

Table A2- 21 Design of pipeline system for greywater supply for Scenario 3 and 4

Target pipe line	Water supply equivalent of this section	Sum of equivalents	Meanwhile discharge probability of water supply equivalent of sanitary wares	Flowrate(L/s)	Pipe diameter DN(mm)	Flow velocity (m/s)	Hydraulic gradient (mH20/m)	Length of pipeline (m)	Head loss (m H20)	Accumulated head loss (m)
A-B	3	3	0.6300	0.3780	25	0.8360	0.0405	2.75	0.1114	0.1114
B-C	3	6	0.4665	0.5598	32	0.6630	0.0183	2.75	0.0503	0.1617
C-D	3	9	0.3933	0.7079	32	0.8380	0.0282	2.75	0.0776	0.2393
D-E	3	12	0.3493	0.8384	32	0.9920	0.0386	2.75	0.1062	0.3454
E-F	3	15	0.3193	0.9578	40	0.8450	0.0241	2.75	0.0663	0.4117
1-2				0.9578	40	0.8450	0.0241	24.75	0.5965	-
G-H	3	3	0.6300	0.3780	25	0.8360	0.0405	2.75	0.1114	0.1114
H-I	3	6	0.4665	0.5598	32	0.6630	0.0183	2.75	0.0503	0.1617
I-J	3	9	0.3933	0.7079	32	0.8380	0.0282	2.75	0.0776	0.2393
J-K	3	12	0.3493	0.8384	32	0.9920	0.0386	2.75	0.1062	0.3454
K-L	3	15	0.3193	0.9578	40	0.8450	0.0241	2.75	0.0663	0.4117
L-M	3	18	0.2970	1.0692	40	0.9430	0.0296	2.75	0.0814	0.4931
M-N	3	21	0.2797	1.1746	40	1.0360	0.0352	2.75	0.0968	0.5899

Table A2- 21 Design of pipeline system for greywater supply for Scenario 3 and 4 (Cont'd)

Target pipe line	Water supply equivalent of this section	Sum of equivalents	Meanwhile discharge probability of water supply equivalent of sanitary wares	Flowrate(L/s)	Pipe diameter DN(mm)	Flow velocity (m/s)	Hydraulic gradient (mH20/m)	Length of pipeline (m)	Head loss (m H20)	Accumulated head loss (m)
N-O	3	24	0.2657	1.2753	40	1.1240	0.0410	2.75	0.1128	0.7026
2-3				1.2753	40	1.1240	0.0410	27.5	1.1275	-
P-Q	3	3	0.6300	0.3780	25	0.8360	0.0405	2.75	0.1114	0.1114
Q-R	3	6	0.4665	0.5598	32	0.6630	0.0183	2.75	0.0503	0.1617
R-S	3	9	0.3933	0.7079	32	0.8380	0.0282	2.75	0.0776	0.2393
S-T	3	12	0.3493	0.8384	32	0.9920	0.0386	2.75	0.1062	0.3454
T-U	3	15	0.3193	0.9578	40	0.8450	0.0241	2.75	0.0663	0.4117
U-V	3	18	0.2970	1.0692	40	0.9430	0.0296	2.75	0.0814	0.4931
V-W	3	21	0.2797	1.1746	40	1.0360	0.0352	2.75	0.0968	0.5899
W-X	3	24	0.2657	1.2753	40	1.1240	0.0410	2.75	0.1128	0.7026
X-Y	3	27	0.2541	1.3721	50	0.6990	0.0123	2.75	0.0338	0.7365
3-4				1.3721	50	0.6990	0.0123	27.5	0.3383	-
Z-a	3	3	0.6300	0.3780	25	0.8360	0.0405	2.75	0.1114	0.1114

Table A2- 21 Design of pipeline system for greywater supply for Scenario 3 and 4 (Cont'd)

Target pipe line	Water supply equivalent of this section	Sum of equivalents	Meanwhile discharge probability of water supply equivalent of sanitary wares	Flowrate(L/s)	Pipe diameter DN(mm)	Flow velocity (m/s)	Hydraulic gradient (mH20/m)	Length of pipeline (m)	Head loss (m H20)	Accumulated head loss (m)
a-b	3	6	0.4665	0.5598	32	0.6630	0.0183	2.75	0.0503	0.1617
b-c	3	9	0.3933	0.7079	32	0.8380	0.0282	2.75	0.0776	0.2393
c-d	3	12	0.3493	0.8384	32	0.9920	0.0386	2.75	0.1062	0.3454
d-e	3	15	0.3193	0.9578	40	0.8450	0.0241	2.75	0.0663	0.4117
e-f	3	18	0.2970	1.0692	40	0.9430	0.0296	2.75	0.0814	0.4931
f-g	3	21	0.2797	1.1746	40	1.0360	0.0352	2.75	0.0968	0.5899
g-h	3	24	0.2657	1.2753	40	1.1240	0.0410	2.75	0.1128	0.7026
h-i	3	27	0.2541	1.3721	50	0.6990	0.0123	2.75	0.0338	0.7365
4-5				1.3721	50	0.6990	0.0123	2.75	0.0338	-
$Tank_{roof}$				2.44	50	1.24		100	3.58	

Table A2- 22 Greywater tanks for Scenario 3 and 4

Roof tank		
Working volume	m^3	12.3186
Length	m	3
Width	m	1.5
Height	m	3.24
1. working height	m	2.74
2. protection height	m	0.30
3. Ineffective height in the bottom	m	0.20
Lowest water level of roof freshwater tank (Z_s)	m	107.81
H_1	m	96.25
H_2	m	0.29
H_3	m	10.00
H_B	m	1.27
Required water pressure at the inlet of roof tank (H)	m	116.78
Height between inlet pipe of roof tank and highest water level	m	0.15
Pipe central level of starting point of sunction pipe of pump	m	-2.00
Headloss between sunction pipe of pump and inlet pipe of roof tank	m	3.58
Required flow head at the inlet pipe of roof tank	m	0.50
Greywater storage tank		
Flow rate of inlet	m^3/day	246.78
Working volume of tank	m^3	400.00
Renewable Period	hours	38.90

Table A2- 23 Greywater pump for Scenario 3 and 4

Flow	m^3/h	8.80
Pumping head	m	128.46

Drainage systems

Scenario 1 and 2: Freshwater and seawater flushing

Table A2- 24 Drainage system for Scenario 1 and 2 (Pipeline set 1)

Target pipe line	Flowrate(L/s)	Accumulated drainage equivalents of this section	Pipe diameter DN(mm)	Hydraulic gradient (mH20/m)	Flow velocity (m/s)	Depth ratio	
A-B	0.0000	0	50	-	-	-	Roof-35F
B-C	1.9297	5.7	100	-	-	-	35F-34F
C-D	2.1077	11.4	100	-	-	-	34F-33F
D-E	2.2443	17.1	100	-	-	-	33F-32F
E-F	2.3595	22.8	100	-	-	-	32F-31F
F-G	2.4609	28.5	100	-	-	-	31F-30F
G-H	2.5527	34.2	100	-	-	-	30F-29F
H-I	2.6370	39.9	100	-	-	-	29F-28F

Table A2- 24 Drainage system for Scenario 1 and 2 (Pipeline set 1) (Cont'd)

Target pipe line	Flowrate(L/s)	Accumulated drainage equivalents of this section	Pipe diameter DN(mm)	Hydraulic gradient (mH20/m)	Flow velocity (m/s)	Depth ratio	
I-J	2.7155	45.6	100	-	-	-	28F-27F
J-K	2.7892	51.3	100	-	-	-	27F-26F
K-L	2.8590	57	100	-	-	-	26F-25F
L-M	2.9253	62.7	100	-	-	-	25F-24F
M-N	2.9887	68.4	100	-	-	-	24F-23F
N-O	3.0495	74.1	100	-	-	-	23F-22F
O-P	3.1080	79.8	100	-	-	-	22F-21F
P-Q	3.1644	85.5	100	-	-	-	21F-20F
Q-R	3.2190	91.2	125	-	-	-	20F-19F
R-S	3.2719	96.9	125	-	-	-	19F-18F
S-T	3.3232	102.6	125	-	-	-	18F-17F
T-U	3.3732	108.3	125	-	-	-	17F-16F
U-V	3.4219	114	125	-	-	-	16F-15F
V-W	3.4693	119.7	125	-	-	-	15F-14F
W-X	3.5157	125.4	125	-	-	-	14F-13F
X-Y	3.5610	131.1	125	-	-	-	13F-12F
Y-Z	3.6053	136.8	125	-	-	-	12F-11F

Table A2- 24 Drainage system for Scenario 1 and 2 (Pipeline set 1) (Cont'd)

Target pipe line	Flowrate(L/s)	Accumulated drainage equivalents of this section	Pipe diameter DN(mm)	Hydraulic gradient (mH20/m)	Flow velocity (m/s)	Depth ratio	
Z-a	3.6487	142.5	125	-	-	-	11F-10F
a-b	3.6913	148.2	125	-	-	-	10F-9F
b-c	3.7330	153.9	125	-	-	-	9F-8F
c-d	3.7740	159.6	125	-	-	-	8F-7F
d-e	3.8142	165.3	125	-	-	-	7F-6F
e-f	3.8538	171	125	-	-	-	6F-5F
f-g	3.8927	176.7	125	-	-	-	5F-4F
g-h	3.9310	182.4	125	-	-	-	4F-3F
h-i	3.9687	188.1	125	-	-	-	3F-2F
i-j	4.0058	193.8	150	-	-	-	2F-1F
j-k	4.0424	199.5	150	-	-	-	1F-G
k-l	4.0424	199.5	150	0.0260	1.4186	0.25	G1-G2
m-B	1.9297	5.7	100	0.0260	1.5300	0.27	

Table A2- 25 Drainage system for Scenario 1 and 2 (Pipeline set 2)

Target pipe line	Flowrate(L/s)	Accumulated drainage equivalents of this section	Pipe diameter DN(mm)	Hydraulic gradient (mH20/m)	Flow velocity (m/s)	Depth ratio	
A-B	0.0000	0.0	50	-	-	-	Roof-35F
B-C	1.3818	4.5	100	-	-	-	35F-34F
C-D	1.5400	9.0	100	-	-	-	34F-33F
D-E	1.6614	13.5	100	-	-	-	33F-32F
E-F	1.7637	18.0	100	-	-	-	32F-31F
F-G	1.8538	22.5	100	-	-	-	31F-30F
G-H	1.9353	27.0	100	-	-	-	30F-29F
H-I	2.0102	31.5	100	-	-	-	29F-28F
I-J	2.0800	36.0	100	-	-	-	28F-27F
J-K	2.1455	40.5	100	-	-	-	27F-26F
K-L	2.2075	45.0	100	-	-	-	26F-25F
L-M	2.2664	49.5	100	-	-	-	25F-24F
M-N	2.3227	54.0	100	-	-	-	24F-23F
N-O	2.3767	58.5	100	-	-	-	23F-22F
O-P	2.4287	63.0	100	-	-	-	22F-21F
P-Q	2.4789	67.5	100	-	-	-	21F-20F
Q-R	2.5274	72.0	100	-	-	-	20F-19F

Table A2- 25 Drainage system for Scenario 1 and 2 (Pipeline set 2) (Cont'd)

Target pipe line	Flowrate(L/s)	Accumulated drainage equivalents of this section	Pipe diameter DN(mm)	Hydraulic gradient (mH20/m)	Flow velocity (m/s)	Depth ratio	
R-S	2.5744	76.5	100	-	-	-	19F-18F
S-T	2.6200	81.0	100	-	-	-	18F-17F
T-U	2.6644	85.5	100	-	-	-	17F-16F
U-V	2.7076	90.0	100	-	-	-	16F-15F
V-W	2.7498	94.5	100	-	-	-	15F-14F
W-X	2.7910	99.0	100	-	-	-	14F-13F
X-Y	2.8312	103.5	100	-	-	-	13F-12F
Y-Z	2.8706	108.0	100	-	-	-	12F-11F
Z-a	2.9092	112.5	100	-	-	-	11F-10F
a-b	2.9470	117.0	100	-	-	-	10F-9F
b-c	2.9841	121.5	100	-	-	-	9F-8F
c-d	3.0205	126.0	100	-	-	-	8F-7F
d-e	3.0563	130.5	100	-	-	-	7F-6F
e-f	3.0914	135.0	100	-	-	-	6F-5F
f-g	3.1260	139.5	100	-	-	-	5F-4F
g-h	3.1600	144.0	100	-	-	-	4F-3F
h-i	3.1935	148.5	100	-	-	-	3F-2F

Table A2- 25 Drainage system for Scenario 1 and 2 (Pipeline set 2) (Cont'd)

Target pipe line	Flowrate(L/s)	Accumulated drainage equivalents of this section	Pipe diameter DN(mm)	Hydraulic gradient (mH20/m)	Flow velocity (m/s)	Depth ratio	
i-j	3.2265	153.0	125	-	-	-	2F-1F
j-k	3.2590	157.5	125	-	-	-	1F-G
k-l	3.2910	162.0	125	0.0260	1.3334	0.3000	G1-G2
m-B	1.3818	4.5	100	0.0260	1.4153	0.4	

Table A2- 26 Septic tanks for Scenario 1 and 2

Parameters		
Percentage of population who use the sanitary wares (α)	%	100
Population to be served by the septic tank (N)	Number of individuals	2520
Daily sewage production per capita (q)	L/person-day	213.8
Hydraulic retention time (t)	hour	24
Daily sludge production per capita (a)	L/person-day	0.7
Number of days between sludge emptying (T)	day	180
Moisture content of raw sludge (b)	%	95
Sludge yield coefficient (k)		0.8
Residual sludge (after emptying) yield coefficient (m)		1.2
Moisture of digested sludge (c)	%	90
Design of septic tanks		
Effective volume required (V)	m^3	691.1856
Number of septic tanks		6
Effective volume of each septic tank (V')	m^3	115.1976
Actual effective volume of septic tank (V")	m^3	130
Length	m	15
Width	m	5
Height	m	1.73

Scenario 3 and 4: Greywater flushing (aerobic and anaerobic)

Table A2- 27 Drainage system for Scenario 3 and 4 (Pipeline set 1)

Target pipe line	Flowrate(L/s)	Accumulated drainage equivalents of this section	Pipe diameter DN(mm)	Hydraulic gradient (mH20/m)	Flow velocity (m/s)	Depth ratio	
A-B	0.0000	0	50	-	-	-	Roof-35F
B-C	0.7958	2.7	75	-	-	-	35F-34F
C-D	0.9183	5.4	75	-	-	-	34F-33F
D-E	1.0123	8.1	75	-	-	-	33F-32F
E-F	1.0915	10.8	75	-	-	-	32F-31F
F-G	1.1614	13.5	75	-	-	-	31F-30F
G-H	1.2245	16.2	75	-	-	-	30F-29F
H-I	1.2825	18.9	75	-	-	-	29F-28F
I-J	1.3366	21.6	100	-	-	-	28F-27F
J-K	1.3873	24.3	100	-	-	-	27F-26F
K-L	1.4353	27	100	-	-	-	26F-25F
L-M	1.4810	29.7	100	-	-	-	25F-24F
M-N	1.5246	32.4	100	-	-	-	24F-23F
N-O	1.5664	35.1	100	-	-	-	23F-22F
O-P	1.6067	37.8	100	-	-	-	22F-21F
P-Q	1.6455	40.5	100	-	-	-	21F-20F

Table A2- 27 Drainage system for Scenario 3 and 4 (Pipeline set 1) (Cont'd)

Target pipe line	Flowrate(L/s)	Accumulated drainage equivalents of this section	Pipe diameter DN(mm)	Hydraulic gradient (mH20/m)	Flow velocity (m/s)	Depth ratio	
Q-R	1.6831	43.2	100	-	-	-	20F-19F
R-S	1.7195	45.9	100	-	-	-	19F-18F
S-T	1.7548	48.6	100	-	-	-	18F-17F
T-U	1.7892	51.3	100	-	-	-	17F-16F
U-V	1.8227	54	100	-	-	-	16F-15F
V-W	1.8554	56.7	100	-	-	-	15F-14F
W-X	1.8873	59.4	100	-	-	-	14F-13F
X-Y	1.9185	62.1	100	-	-	-	13F-12F
Y-Z	1.9490	64.8	100	-	-	-	12F-11F
Z-a	1.9789	67.5	100	-	-	-	11F-10F
a-b	2.0081	70.2	100	-	-	-	10F-9F
b-c	2.0369	72.9	100	-	-	-	9F-8F
c-d	2.0651	75.6	100	-	-	-	8F-7F
d-e	2.0928	78.3	100	-	-	-	7F-6F
e-f	2.1200	81	100	-	-	-	6F-5F
f-g	2.1468	83.7	100	-	-	-	5F-4F
g-h	2.1731	86.4	100	-	-	-	4F-3F

Table A2- 27 Drainage system for Scenario 3 and 4 (Pipeline set 1) (Cont'd)

Target pipe line	Flowrate(L/s)	Accumulated drainage equivalents of this section	Pipe diameter DN(mm)	Hydraulic gradient (mH20/m)	Flow velocity (m/s)	Depth ratio	
h-i	2.1991	89.1	100	-	-	-	3F-2F
i-j	2.2246	91.8	100	-	-	-	2F-1F
j-k	2.2498	94.5	100	-	-	-	1F-G
k-l	2.2498	94.5	100	0.026	1.2184	0.3	G1-G2
m-B	0.7958	2.7	75	0.026	0.9418	0.3	

Table A2- 28 Drainage system for Scenario 3 and 4 (Pipeline set 2)

Target pipe line	Flowrate(L/s)	Accumulated drainage equivalents of this section	Pipe diameter DN(mm)	Hydraulic gradient (mH20/m)	Flow velocity (m/s)	Depth ratio	
A-B	0.0000	0	50	-	-	-	Roof-35F
B-C	1.8818	4.5	100	-	-	-	35F-34F
C-D	2.0400	9	100	-	-	-	34F-33F
D-E	2.1614	13.5	100	-	-	-	33F-32F
E-F	2.2637	18	100	-	-	-	32F-31F
F-G	2.3538	22.5	100	-	-	-	31F-30F

Table A2- 28 Drainage system for Scenario 3 and 4 (Pipeline set 2) (Cont'd)

Target pipe line	Flowrate(L/s)	Accumulated drainage equivalents of this section	Pipe diameter DN(mm)	Hydraulic gradient (mH20/m)	Flow velocity (m/s)	Depth ratio	
G-H	2.4353	27	100	-	-	-	30F-29F
H-I	2.5102	31.5	100	-	-	-	29F-28F
I-J	2.5800	36	100	-	-	-	28F-27F
J-K	2.6455	40.5	100	-	-	-	27F-26F
K-L	2.7075	45	100	-	-	-	26F-25F
L-M	2.7664	49.5	100	-	-	-	25F-24F
M-N	2.8227	54	100	-	-	-	24F-23F
N-O	2.8767	58.5	100	-	-	-	23F-22F
O-P	2.9287	63	100	-	-	-	22F-21F
P-Q	2.9789	67.5	100	-	-	-	21F-20F
Q-R	3.0274	72	100	-	-	-	20F-19F
R-S	3.0744	76.5	100	-	-	-	19F-18F
S-T	3.1200	81	100	-	-	-	18F-17F
T-U	3.1644	85.5	100	-	-	-	17F-16F
U-V	3.2076	90	125	-	-	-	16F-15F
V-W	3.2498	94.5	125	-	-	-	15F-14F
W-X	3.2910	99	125	-	-	-	14F-13F

Table A2- 28 Drainage system for Scenario 3 and 4 (Pipeline set 2) (Cont'd)

Target pipe line	Flowrate(L/s)	Accumulated drainage equivalents of this section	Pipe diameter DN(mm)	Hydraulic gradient (mH20/m)	Flow velocity (m/s)	Depth ratio	
X-Y	3.3312	103.5	125	-	-	-	13F-12F
Y-Z	3.3706	108	125	-	-	-	12F-11F
Z-a	3.4092	112.5	125	-	-	-	11F-10F
a-b	3.4470	117	125	-	-	-	10F-9F
b-c	3.4841	121.5	125	-	-	-	9F-8F
c-d	3.5205	126	125	-	-	-	8F-7F
d-e	3.5563	130.5	125	-	-	-	7F-6F
e-f	3.5914	135	125	-	-	-	6F-5F
f-g	3.6260	139.5	125	-	-	-	5F-4F
g-h	3.6600	144	125	-	-	-	4F-3F
h-i	3.6935	148.5	125	-	-	-	3F-2F
i-j	3.7265	153	125	-	-	-	2F-1F
j-k	3.7590	157.5	125	-	-	-	1F-G
k-l	3.7590	157.5	125	0.026	1.4466	0.35	G1-G2
m-B	1.8818	4.5	100	0.026	1.2184	0.3	

Table A2- 29 Drainage system for Scenario 3 and 4 (Pipeline set 3)

Target pipe line	Flowrate(L/s)	Accumulated drainage equivalents of this section	Pipe diameter DN(mm)	Hydraulic gradient (mH20/m)	Flow velocity (m/s)	Depth ratio	
A-B	0.0000	0	50	-	-	-	Roof-35F
B-C	1.3118	3	100	-	-	-	35F-34F
C-D	1.4409	6	100	-	-	-	34F-33F
D-E	1.5400	9	100	-	-	-	33F-32F
E-F	1.6235	12	100	-	-	-	32F-31F
F-G	1.6971	15	100	-	-	-	31F-30F
G-H	1.7637	18	100	-	-	-	30F-29F
H-I	1.8249	21	100	-	-	-	29F-28F
I-J	1.8818	24	100	-	-	-	28F-27F
J-K	1.9353	27	100	-	-	-	27F-26F
K-L	1.9859	30	100	-	-	-	26F-25F
L-M	2.0340	33	100	-	-	-	25F-24F
M-N	2.0800	36	100	-	-	-	24F-23F
N-O	2.1241	39	100	-	-	-	23F-22F
O-P	2.1665	42	100	-	-	-	22F-21F
P-Q	2.2075	45	100	-	-	-	21F-20F
Q-R	2.2471	48	100	-	-	-	20F-19F

Table A2- 29 Drainage system for Scenario 3 and 4 (Pipeline set 3) (Cont'd)

Target pipe line	Flowrate(L/s)	Accumulated drainage equivalents of this section	Pipe diameter DN(mm)	Hydraulic gradient (mH20/m)	Flow velocity (m/s)	Depth ratio	
R-S	2.2855	51	100	-	-	-	19F-18F
S-T	2.3227	54	100	-	-	-	18F-17F
T-U	2.3590	57	100	-	-	-	17F-16F
U-V	2.3943	60	100	-	-	-	16F-15F
V-W	2.4287	63	100	-	-	-	15F-14F
W-X	2.4623	66	100	-	-	-	14F-13F
X-Y	2.4952	69	100	-	-	-	13F-12F
Y-Z	2.5274	72	100	-	-	-	12F-11F
Z-a	2.5588	75	100	-	-	-	11F-10F
a-b	2.5897	78	100	-	-	-	10F-9F
b-c	2.6200	81	100	-	-	-	9F-8F
c-d	2.6497	84	100	-	-	-	8F-7F
d-e	2.6789	87	100	-	-	-	7F-6F
e-f	2.7076	90	100	-	-	-	6F-5F
f-g	2.7359	93	100	-	-	-	5F-4F
g-h	2.7636	96	100	-	-	-	4F-3F
h-i	2.7910	99	100	-	-	-	3F-2F

Table A2- 29 Drainage system for Scenario 3 and 4 (Pipeline set 3) (Cont'd)

Target pipe line	Flowrate(L/s)	Accumulated drainage equivalents of this section	Pipe diameter DN(mm)	Hydraulic gradient (mH20/m)	Flow velocity (m/s)	Depth ratio	
i-j	2.8179	102	100	-	-	-	2F-1F
j-k	2.8445	105	100	-	-	-	1F-G
k-l	2.8445	105	100	0.026	1.3219	0.35	G1-G2
m-B	1.3118	3	100	0.026	1.0987	0.25	

Table A2- 30 Septic tanks and greywater equalization basins for Scenario 1 and 2

Parameters		
Percentage of population who use the sanitary wares (α)	%	100
Population to be served by the septic tank (N)	Number of individuals	2520
Daily sewage production per capita (q)	L/person-day	115.87
Hydraulic retention time (t)	hour	24
Daily sludge production per capita (a)	L/person-day	0.7
Number of days between sludge emptying (T)	day	180
Moisture content of raw sludge (b)	%	95
Sludge yield coefficient (k)		0.8
Residual sludge (after emptying) yield coefficient (m)		1.2
Moisture of digested sludge (c)	%	90

Table A2- 30 Septic tanks and greywater equalization basins for Scenario 1 and 2 (Cont'd)

Design of septic tanks		
Effective volume required (V)	m^3	444.402
Number of septic tanks		4
Effective volume of each septic tank (V')	m^3	111.1005
Actual effective volume of septic tank (V")	m^3	120
Length	m	12
Width	m	6
Height	m	1.67
Design of greywater equalization basins		
Effective volume required (V)	m^3	246.7836
Number of septic tanks		2
Effective volume of each equalization basin (V')	m^3	123.3918
Actual effective volume of equalization basin (V")	m^3	130
Length	m	8
Width	m	6
Height	m	2.71

Economic Cost Analysis

Capital costs

All costs listed in the section are presented in present value of US dollars (US$1 = HK$7.76). A discount rate of 4% is used according to the common practice in cost estimation for public works projects in Hong Kong.

Scenario 1: Freshwater flushing

Table A2- 31 Capital cost of Scenario 1

Quantity	Description	Unit	Mat. O&P	Labor O&P	Equip. O&P	Total O&P	Ext. Total O&P
1	Underground storage tank, fiberglass, single wall, U.L. listed, 30,000 gal. cap, excl. manway or hold-down strap	Ea.	$55,000.00	$2,450.00	$-	$57,450.00	$57,450.00
1	Storage tank, horizontal, steel, above ground, double wall, 10,000 gallons, incl. cradles, coating & fittings, excl. foundation, pumps or piping	Ea.	$33,200.00	$1,550.00	$-	$34,750.00	$34,750.00
144.36	Pipe, plastic, PVC, 1-1/2" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$7.70	$19.85	$-	$27.55	$3,977.12
1010.5	Pipe, plastic, PVC, 2" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$9.10	$22.00	$-	$31.10	$31,426.55
288.71	Pipe, plastic, PVC, 2-1/2" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$11.75	$23.00	$-	$34.75	$10,032.67
360.89	Pipe, plastic, PVC, 3" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$14.30	$24.50	$-	$38.80	$14,002.53

Table A2- 31 Capital cost of Scenario 1 (Cont'd)

Quantity	Description	Unit	Mat. O&P	Labor O&P	Equip. O&P	Total O&P	Ext. Total O&P
396.98	Pipe, plastic, PVC, 4" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$18.25	$27.00	$-	$45.25	$17,963.35
1	Public water supply wells, wells domestic water, pumps, 30 H.P., 100 to 300 GPM, installed in wells, 6" submersible, 25' to 500' deep	Ea.	$4,725.00	$1,750.00	$990.00	$7,465.00	$7,465.00
9.02	Pipe, plastic, PVC, 2" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$9.10	$22.00	$-	$31.10	$280.52
144.36	Pipe, plastic, PVC, 4" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$18.25	$27.00	$-	$45.25	$6,532.29
162.4	Pipe, plastic, PVC, 5" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$30.00	$30.00	$-	$60.00	$9,744.00
27.07	Pipe, plastic, PVC, 6" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$31.50	$33.00	$-	$64.50	$1,746.02
9.02	Pipe, plastic, PVC, 2" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$9.10	$22.00	$-	$31.10	$280.52
306.76	Pipe, plastic, PVC, 4" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$18.25	$27.00	$-	$45.25	$13,880.89
27.07	Pipe, plastic, PVC, 5" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$30.00	$30.00	$-	$60.00	$1,624.20

Table A2- 31 Capital cost of Scenario 1 (Cont'd)

Quantity	Description	Unit	Mat. O&P	Labor O&P	Equip. O&P	Total O&P	Ext. Total O&P
Total						$202,304.65	**$211,155.66**

Scenario 2: Seawater flushing

Table A2- 32 Capital cost of Scenario 2

Quantity	Description	Unit	Mat. O&P	Labor O&P	Equip. O&P	Total O&P	Ext. Total O&P
1	Underground storage tank, fiberglass, single wall, U.L. listed, 20,000 gal. cap, excl. manway or hold-down strap	Ea.	$30,400.00	$1,925.00	$-	$32,325.00	$32,325.00
1	Storage tank, horizontal, steel, above ground, double wall, 6,000 gallons, incl. cradles, coating & fittings, excl. foundation, pumps or piping	Ea.	$23,100.00	$1,175.00	$-	$24,275.00	$24,275.00
1	Underground storage tank, fiberglass, single wall, U.L. listed, 12,000 gal. cap, excl. manway or hold-down strap	Ea.	$16,800.00	$1,475.00	$-	$18,275.00	$18,275.00
1	Storage tank, horizontal, steel, above ground, double wall, 4,000 gallons, incl. cradles, coating & fittings, excl. foundation, pumps or piping	Ea.	$19,600.00	$775.00	$-	$20,375.00	$20,375.00

Table A2- 32 Capital cost of Scenario 2 (Cont'd)

Quantity	Description	Unit	Mat. O&P	Labor O&P	Equip. O&P	Total O&P	Ext. Total O&P
144.36	Pipe, plastic, PVC, 1-1/4" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$7.40	$17.00	$-	$24.40	$3,522.38
1587.93	Pipe, plastic, PVC, 2" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$9.10	$22.00	$-	$31.10	$49,384.62
469.16	Pipe, plastic, PVC, 2-1/2" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$11.75	$23.00	$-	$34.75	$16,303.31
144.36	Pipe, plastic, PVC, 1" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$6.55	$15.55	$-	$22.10	$3,190.36
433.07	Pipe, plastic, PVC, 1-1/4" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$7.40	$17.00	$-	$24.40	$10,566.91
1154.86	Pipe, plastic, PVC, 1-1/2" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$7.70	$19.85	$-	$27.55	$31,816.39
469.16	Pipe, plastic, PVC, 2" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$9.10	$22.00	$-	$31.10	$14,590.88
328.08	Pipe, plastic, PVC, 4" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$18.25	$27.00	$-	$45.25	$14,845.62
328.08	Pipe, plastic, PVC, 2" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$9.10	$22.00	$-	$31.10	$10,203.29

Table A2- 32 Capital cost of Scenario 2 (Cont'd)

Quantity	Description	Unit	Mat. O&P	Labor O&P	Equip. O&P	Total O&P	Ext. Total O&P
2	Public water supply wells, wells domestic water, pumps, 30 H.P., 100 to 300 GPM, installed in wells, 6" submersible, 25' to 500' deep	Ea.	$4,725.00	$1,750.00	$990.00	$7,465.00	$14,930.00
9.02	Pipe, plastic, PVC, 2" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$9.10	$22.00	$-	$31.10	$280.52
144.38	Pipe, plastic, PVC, 4" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$18.25	$27.00	$-	$45.25	$6,533.20
162.42	Pipe, plastic, PVC, 5" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$30.00	$30.00	$-	$60.00	$9,745.20
27.07	Pipe, plastic, PVC, 6" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$31.50	$33.00	$-	$64.50	$1,746.02
9.02	Pipe, plastic, PVC, 2" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$9.10	$22.00	$-	$31.10	$280.52
306.76	Pipe, plastic, PVC, 4" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$18.25	$27.00	$-	$45.25	$13,880.89
27.07	Pipe, plastic, PVC, 5" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$30.00	$30.00	$-	$60.00	$1,624.20
Total						$205,448.95	**$298,694.31**

Scenario 3: Greywater flushing (Aerobic)

Table A2- 33 Capital cost of Scenario 3

Quantity	Description	Unit	Mat. O&P	Labor O&P	Equip. O&P	Total O&P	Ext. Total O&P
1	Underground storage tank, fiberglass, single wall, U.L. listed, 20,000 gal. cap, excl. manway or hold-down strap	Ea.	$30,400.00	$1,925.00	$-	$32,325.00	$32,325.00
1	Storage tank, horizontal, steel, above ground, double wall, 6,000 gallons, incl. cradles, coating & fittings, excl. foundation, pumps or piping	Ea.	$23,100.00	$1,175.00	$-	$24,275.00	$24,275.00
1	Storage tank, horizontal, steel, above ground, double wall, 4,000 gallons, incl. cradles, coating & fittings, excl. foundation, pumps or piping	Ea.	$19,600.00	$775.00	$-	$20,375.00	$20,375.00
144.36	Pipe, plastic, PVC, 1-1/4" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$7.40	$17.00	$-	$24.40	$3,522.38
1587.93	Pipe, plastic, PVC, 2" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$9.10	$22.00	$-	$31.10	$49,384.62
469.16	Pipe, plastic, PVC, 2-1/2" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$11.75	$23.00	$-	$34.75	$16,303.31
144.36	Pipe, plastic, PVC, 1" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$6.55	$15.55	$-	$22.10	$3,190.36

Table A2- 33 Capital cost of Scenario 3 (Cont'd)

Quantity	Description	Unit	Mat. O&P	Labor O&P	Equip. O&P	Total O&P	Ext. Total O&P
433.07	Pipe, plastic, PVC, 1-1/4" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$7.40	$17.00	$-	$24.40	$10,566.91
1154.86	Pipe, plastic, PVC, 1-1/2" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$7.70	$19.85	$-	$27.55	$31,816.39
469.16	Pipe, plastic, PVC, 2" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$9.10	$22.00	$-	$31.10	$14,590.88
328.08	Pipe, plastic, PVC, 4" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$18.25	$27.00	$-	$45.25	$14,845.62
328.08	Pipe, plastic, PVC, 2" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$9.10	$22.00	$-	$31.10	$10,203.29
2	Public water supply wells, wells domestic water, pumps, 30 H.P., 100 to 300 GPM, installed in wells, 6" submersible, 25' to 500' deep	Ea.	$4,725.00	$1,750.00	$990.00	$7,465.00	$14,930.00
9.02	Pipe, plastic, PVC, 2" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$9.10	$22.00	$-	$31.10	$280.52
72.18	Pipe, plastic, PVC, 3" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$14.30	$24.50	$-	$38.80	$2,800.58
261.65	Pipe, plastic, PVC, 4" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$18.25	$27.00	$-	$45.25	$11,839.66

Table A2- 33 Capital cost of Scenario 3 (Cont'd)

Quantity	Description	Unit	Mat. O&P	Labor O&P	Equip. O&P	Total O&P	Ext. Total O&P
9.02	Pipe, plastic, PVC, 2" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$9.10	$22.00	$-	$31.10	$280.52
180.45	Pipe, plastic, PVC, 4" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$18.25	$27.00	$-	$45.25	$8,165.36
153.38	Pipe, plastic, PVC, 5" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$30.00	$30.00	$-	$60.00	$9,202.80
9.02	Pipe, plastic, PVC, 2" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$9.10	$22.00	$-	$31.10	$280.52
333.83	Pipe, plastic, PVC, 4" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$18.25	$27.00	$-	$45.25	$15,105.81
1	Aerobic membrane bioreactor (MBR)	Ea.				$129,475.84	$129,475.84
Total						$214,515.44	**$423,760.37**

Scenario 4: Greywater flushing (Anaerobic)

Table A2- 34 Capital cost of Scenario 4

Quantity	Description	Unit	Mat. O&P	Labor O&P	Equip. O&P	Total O&P	Ext. Total O&P
1	Underground storage tank, fiberglass, single wall, U.L. listed, 20,000 gal. cap, excl. manway or hold-down strap	Ea.	$30,400.00	$1,925.00	$-	$32,325.00	$32,325.00
1	Storage tank, horizontal, steel, above ground, double wall, 6,000 gallons, incl. cradles, coating & fittings, excl. foundation, pumps or piping	Ea.	$23,100.00	$1,175.00	$-	$24,275.00	$24,275.00
1	Storage tank, horizontal, steel, above ground, double wall, 4,000 gallons, incl. cradles, coating & fittings, excl. foundation, pumps or piping	Ea.	$19,600.00	$775.00	$-	$20,375.00	$20,375.00
144.36	Pipe, plastic, PVC, 1-1/4" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$7.40	$17.00	$-	$24.40	$3,522.38
1587.93	Pipe, plastic, PVC, 2" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$9.10	$22.00	$-	$31.10	$49,384.62
469.16	Pipe, plastic, PVC, 2-1/2" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$11.75	$23.00	$-	$34.75	$16,303.31
144.36	Pipe, plastic, PVC, 1" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$6.55	$15.55	$-	$22.10	$3,190.36

Table A2- 34 Capital cost of Scenario 4 (Cont'd)

Quantity	Description	Unit	Mat. O&P	Labor O&P	Equip. O&P	Total O&P	Ext. Total O&P
433.07	Pipe, plastic, PVC, 1-1/4" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$7.40	$17.00	$-	$24.40	$10,566.91
1154.86	Pipe, plastic, PVC, 1-1/2" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$7.70	$19.85	$-	$27.55	$31,816.39
469.16	Pipe, plastic, PVC, 2" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$9.10	$22.00	$-	$31.10	$14,590.88
328.08	Pipe, plastic, PVC, 4" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$18.25	$27.00	$-	$45.25	$14,845.62
328.08	Pipe, plastic, PVC, 2" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$9.10	$22.00	$-	$31.10	$10,203.29
2	Public water supply wells, wells domestic water, pumps, 30 H.P., 100 to 300 GPM, installed in wells, 6" submersible, 25' to 500' deep	Ea.	$4,725.00	$1,750.00	$990.00	$7,465.00	$14,930.00
9.02	Pipe, plastic, PVC, 2" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$9.10	$22.00	$-	$31.10	$280.52
72.18	Pipe, plastic, PVC, 3" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$14.30	$24.50	$-	$38.80	$2,800.58
261.65	Pipe, plastic, PVC, 4" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$18.25	$27.00	$-	$45.25	$11,839.66

Table A2- 34 Capital cost of Scenario 4 (Cont'd)

Quantity	Description	Unit	Mat. O&P	Labor O&P	Equip. O&P	Total O&P	Ext. Total O&P
9.02	Pipe, plastic, PVC, 2" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$9.10	$22.00	$-	$31.10	$280.52
180.45	Pipe, plastic, PVC, 4" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$18.25	$27.00	$-	$45.25	$8,165.36
153.38	Pipe, plastic, PVC, 5" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$30.00	$30.00	$-	$60.00	$9,202.80
9.02	Pipe, plastic, PVC, 2" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$9.10	$22.00	$-	$31.10	$280.52
333.83	Pipe, plastic, PVC, 4" diameter, schedule 40, includes couplings 10' OC, and hangers 3 per 10'	L.F.	$18.25	$27.00	$-	$45.25	$15,105.81
1	Anaerobic fluidized bed membrane bioreactor (AFMBR)	Ea.				$161,973.76	$161,973.76
Total						$247,013.36	**$456,258.29**

Operation costs

All costs listed in the section are presented in present value of US dollars (US$1 = HK$7.76). A discount rate of 4% is used according to the common practice in cost estimation for public works projects in Hong Kong.

Table A2- 35 Assumptions in estimation of operation costs

% of Dongjiang water in freshwater supplied	%	75%
% of local yield in freshwater supplied	%	25%
Price of Dongjiang water	$/m^3	1.2576
Treatment cost of local yield of freshwater	$/m^3	0.5849
Treatment cost of seawater	$/m^3	0.4972
Finance provision to drainage services	HK$/yr	1,902,100,000
Sewage treated	m^3/yr	997,000,000
Average sewage treatment cost	$/m^3	0.2579
Unit cost electricity	$/kWh	0.1544

<u>Scenario 1: Freshwater flushing</u>

Table A2- 36 Costs for water supply in Scenario 1

Freshwater requirement	m^3/d	538.78
Freshwater requirement – Dongjiang water	m^3/d	405.55
Annual cost of Dongjiang water	$/yr	186,157.52
Freshwater requirement – local yield	m^3/d	133.22
Annual cost of freshwater from local yield	$/yr	28,443.35
Total freshwater costs	$/yr	214,600.86

Table A2- 37 Cost for sewage treatment in Scenario 1

Daily sewage generation	m^3/d	538.78
Total sewage treatment cost	$/yr	50,724.29

Table A2- 38 Operation costs of pumping system in Scenario 1

Volumetric flow of fluid (Q)	m^3/hr	22.45
Density of fluid (ρ)	kg/m^3	1000
Gravity (g)	m/s^2	9.81
Head produced by the pump (h)	m	125.01
Hydraulic power of the pump (P_h)	kW	7.65
Pump efficiency (η_p)	%	60
Shaft power of the pump (P_s)	kW	12.75
Motor efficiency (η_m)	%	85
Required power to the motor (P_m)	kW	15.00
Working hours of the pump	hr/yr	4,380
Annual electricity consumption	kWh/yr	65,679.76
Annual electricity cost	$/yr	10,140.85

Scenario 2: Seawater flushing

Table A2- 39 Costs for water supply in Scenario 2

Freshwater requirement	m^3/d	327.60
Freshwater requirement – Dongjiang water	m^3/d	246.59
Annual cost of Dongjiang water	$/yr	113,192.13
Freshwater requirement – local yield	m^3/d	81.01
Annual cost of freshwater from local yield	$/yr	17,294.83
Total freshwater costs	$/yr	130,486.96
Seawater requirement	m^3/d	211.18
Annual of seawater treatment	$/yr	38,323.02

Table A2- 40 Cost for sewage treatment in Scenario 2

Daily sewage generation	m^3/d	538.78
Total sewage treatment cost	$/yr	50,724.29

Table A2- 41 Operation costs of pumping system in Scenario 2

Pump 1 (freshwater)		
Volumetric flow of fluid (Q)	m^3/hr	13.65
Density of fluid (ρ)	kg/m^3	1000
Gravity (g)	m/s^2	9.81
Head produced by the pump (h)	m	125.27
Hydraulic power of the pump (P_h)	kW	4.66
Pump efficiency (η_p)	%	60
Shaft power of the pump (P_s)	kW	7.77
Motor efficiency (η_m)	%	85
Required power to the motor (P_m)	kW	9.14
Working hours of the pump	hr/yr	4380
Annual electricity consumption	kWh/yr	40,017.52
Annual electricity cost	$/yr	6,178.64
Pump 2 (seawater)		
Volumetric flow of fluid (Q)	m^3/hr	8.8
Density of fluid (ρ)	kg/m^3	1000
Gravity (g)	m/s^2	9.81
Head produced by the pump (h)	m	128.46
Hydraulic power of the pump (P_h)	kW	3.08
Pump efficiency (η_p)	%	60
Shaft power of the pump (P_s)	kW	5.13
Motor efficiency (η_m)	%	85
Required power to the motor (P_m)	kW	6.04
Working hours of the pump	hr/yr	4380
Annual electricity consumption	kWh/yr	26,455.81
Annual electricity cost	$/yr	4,084.73

Scenarios 3: Greywater flushing (Aerobic)

Table A2- 42 Costs for water supply in Scenario 3

Freshwater requirement	m^3/d	327.60
Freshwater requirement – Dongjiang water	m^3/d	246.59
Annual cost of Dongjiang water	$/yr	113,192.13
Freshwater requirement – local yield	m^3/d	81.01
Annual cost of freshwater from local yield	$/yr	17,294.83
Total freshwater costs	$/yr	130,486.96

Table A2- 43 Cost for sewage treatment in Scenario 3

Daily sewage generation	m^3/d	327.60
Total sewage treatment cost	$/yr	27,490.29

Table A2- 44 Operation cost of aerobic MBR

Daily greywater flow	m^3/d	246.78
Operation cost	$\$/m^3$	0.40
Annual operation cost	\$/yr	35,936.85

Table A2- 45 Operation costs of pumping system in Scenario 3

Pump 1 (freshwater)		
Refer to Table A2- 41		
Pump 2 (greywater)		
Refer to Table A2- 41		

Table A2- 46 Cost for chlorine disinfection in Scenario 3

Chlorine dosage	mg/L	5
Chlorination cost	\$/d	0.5777
Annual chlorination cost	\$/yr	210.87

Table A2- 47 Cost for sludge treatment in Scenario 3

Sludge yield	g/m^3	75
Sludge production	kg/d	18.51
Annual sludge production	kg/yr	6,755.70
Sludge treatment cost (incineration and landfill disposal)	\$/ton	25.74
Annual sludge treatment cost	\$/yr	173.89

Scenario 4: Greywater flushing (Anaerobic)

Table A2- 48 Costs for water supply in Scenario 4

Refer to Table A2- 42

Table A2- 49 Cost for sewage treatment in Scenario 4

Refer to Table A2- 43

Table A2- 50 Operation cost of AFMBR

Daily greywater flow	m^3/d	246.78
Operation cost	$\$/m^3$	0.07
Annual operation cost	$/yr	6,305.32

Table A2- 51 Operation costs of pumping system in Scenario 4

Pump 1 (freshwater)		
Refer to Table A2- 41		
Pump 2 (greywater)		
Refer to Table A2- 41		

Table A2- 52 Cost for chlorine disinfection in Scenario 4

Refer to Table A2- 46

Table A2- 53 Cost for sludge treatment in Scenario 4

Sludge yield	g/m^3	35
Sludge production	kg/d	8.64
Annual sludge production	kg/yr	3,512.66
Sludge treatment cost (incineration and landfill disposal)	$/ton	25.74
Annual sludge treatment cost	$/yr	81.15

Table A2- 54 Energy recovery by AFMBR in Scenario 4

Chemical oxygen demand (COD) of influent	mg/L	250
Methane yield	%	50
Methane production	mg/L	125
	g/yr	11,259,501.75
Volume of methane produced	m^3/yr	15,773.20
Methane energy content	MJ/m^3	38.8
Conversion efficiency	%	38
Unit conversion	MJ/kWh	3.6
Electricity generation	kWh/yr	64,600.03
Electricity cost savings	$/yr	-9,974.14

Economic cost analysis results

Table A2- 55 Summary of the economic cost analysis results

	Capital cost ($)	Annual operation cost ($/yr)	Operation cost of 20-year lifetime (PV$)	Total
S1	211,155.66	231,335.87	3,143,929.98	3,355,085.64
S2	298,694.31	152,326.49	2,070,166.67	2,368,860.98
S3	423,760.37	180,645.69	2,455,033.83	2,878,794.20
S4	456,258.29	140,947.27	1,915,519.38	2,371,777.67

Table A2- 56 Relationship between total life-cycle costs and lifetime

Years	S1	S2	S3	S4
1	442,491.53	451,020.80	604,406.06	597,205.56
2	647,477.01	585,996.49	764,475.24	722,098.18
3	853,133.76	721,414.18	925,068.59	847,399.79
4	1,050,880.63	851,623.50	1,079,485.28	967,882.11
5	1,241,021.86	976,824.77	1,227,962.87	1,083,730.49
6	1,423,849.96	1,097,210.60	1,370,729.78	1,195,123.16
7	1,599,646.21	1,212,966.21	1,508,005.65	1,302,231.50
8	1,768,681.06	1,324,269.68	1,640,001.69	1,405,220.29
9	1,931,214.58	1,431,292.25	1,766,920.95	1,504,247.97
10	2,087,496.80	1,534,198.57	1,888,958.70	1,599,466.90
11	2,237,768.17	1,633,146.95	2,006,302.70	1,691,023.56
12	2,382,259.87	1,728,289.63	2,119,133.46	1,779,058.81
13	2,521,194.20	1,819,772.97	2,227,624.58	1,863,708.08
14	2,654,784.90	1,907,737.72	2,331,942.96	1,945,101.62
15	2,783,237.50	1,992,319.21	2,432,249.10	2,023,364.63
16	2,906,749.62	2,073,647.56	2,528,697.31	2,098,617.53
17	3,025,511.26	2,151,847.91	2,621,435.97	2,170,976.09
18	3,139,705.15	2,227,040.54	2,710,607.76	2,240,551.62
19	3,249,506.97	2,299,341.16	2,796,349.86	2,307,451.18
20	3,355,085.64	2,368,860.98	2,878,794.20	2,371,777.67
21	3,456,603.60	2,435,706.96	2,958,067.60	2,433,630.07
22	3,554,217.01	2,499,981.94	3,034,292.02	2,493,103.53
23	3,648,076.07	2,561,784.81	3,107,584.73	2,550,289.55
24	3,738,325.16	2,621,210.64	3,178,058.49	2,605,276.10
25	3,825,103.13	2,678,350.87	3,245,821.72	2,658,147.79

Table A2- 57 Relationship between total life-cycle costs and lifetime (Dongjiang water price = HK$4/m^3)

Years	S1	S2	S3	S4
1	328,609.51	381,326.81	533,622.86	527,949.23
2	441,545.91	460,781.13	639,259.88	596,882.83
3	550,138.60	537,179.51	740,833.93	663,165.13
4	654,554.65	610,639.50	838,501.29	726,898.11
5	754,954.69	681,274.10	932,412.21	788,179.83
6	851,493.20	749,191.99	1,022,711.17	847,104.55
7	944,318.68	814,497.65	1,109,537.10	903,762.95
8	1,033,573.96	877,291.56	1,193,023.56	958,242.17
9	1,119,396.34	937,670.31	1,273,299.01	1,010,626.03
10	1,201,917.86	995,726.81	1,350,486.94	1,060,995.14
11	1,281,265.47	1,051,550.36	1,424,706.11	1,109,426.97
12	1,357,561.26	1,105,226.86	1,496,070.69	1,155,996.03
13	1,430,922.59	1,156,838.87	1,564,690.48	1,200,773.98
14	1,501,462.33	1,206,465.80	1,630,671.05	1,243,829.70
15	1,569,289.00	1,254,184.01	1,694,113.90	1,285,229.44
16	1,634,506.96	1,300,066.90	1,755,116.64	1,325,036.87
17	1,697,216.53	1,344,185.07	1,813,773.13	1,363,313.25
18	1,757,514.19	1,386,606.38	1,870,173.59	1,400,117.46
19	1,815,492.72	1,427,396.10	1,924,404.81	1,435,506.12
20	1,871,241.30	1,466,616.99	1,976,550.21	1,469,533.68
21	1,924,845.71	1,504,329.38	2,026,690.02	1,502,252.49
22	1,976,388.40	1,540,591.30	2,074,901.37	1,533,712.89
23	2,025,948.69	1,575,458.52	2,121,258.44	1,563,963.26
24	2,073,602.81	1,608,984.70	2,165,832.55	1,593,050.17
25	2,119,424.08	1,641,221.41	2,208,692.27	1,621,018.34

Table A2- 58 Relationship between total life-cycle costs and lifetime (Dongjiang water price = HK$17.2/m^3)

Years	S1	S2	S3	S4
1	572,852.84	529,837.71	682,133.77	676,460.14
2	920,638.58	752,090.99	930,569.74	888,192.68
3	1,255,047.96	965,796.06	1,169,450.47	1,091,781.67
4	1,576,595.43	1,171,281.70	1,399,143.49	1,287,540.31
5	1,885,775.69	1,368,864.05	1,620,002.15	1,475,769.77
6	2,183,064.41	1,558,847.08	1,832,366.26	1,656,759.64
7	2,468,918.94	1,741,523.07	2,036,562.51	1,830,788.36
8	2,743,779.07	1,917,173.06	2,232,905.06	1,998,123.66
9	3,008,067.65	2,086,067.28	2,421,695.97	2,159,023.00
10	3,262,191.29	2,248,465.57	2,603,225.70	2,313,733.89
11	3,506,540.94	2,404,617.77	2,777,773.51	2,462,494.37
12	3,741,492.53	2,554,764.11	2,945,607.95	2,605,533.29
13	3,967,407.52	2,699,135.60	3,106,987.21	2,743,070.72
14	4,184,633.47	2,837,954.34	3,262,159.58	2,875,318.24
15	4,393,504.58	2,971,433.90	3,411,363.79	3,002,479.32
16	4,594,342.18	3,099,779.62	3,554,829.36	3,124,749.59
17	4,787,455.26	3,223,188.98	3,692,777.04	3,242,317.16
18	4,973,140.92	3,341,851.81	3,825,419.03	3,355,362.89
19	5,151,684.82	3,455,950.70	3,952,959.41	3,464,060.72
20	5,323,361.64	3,565,661.16	4,075,594.38	3,568,577.86
21	5,488,435.51	3,671,152.00	4,193,512.63	3,669,075.11
22	5,647,160.39	3,772,585.49	4,306,895.56	3,765,707.08
23	5,799,780.46	3,870,117.69	4,415,917.61	3,858,622.43
24	5,946,530.53	3,963,898.66	4,520,746.50	3,947,964.12
25	6,087,636.36	4,054,072.66	4,621,543.52	4,033,869.59

Life-cycle Assessment

Construction phase

Scenario 1: Freshwater flushing

Table A2- 59 Conventional air emissions (tons) from construction phase in Scenario 1

	CO	NH_3	NO_x	PM_{10}	$PM_{2.5}$	SO_2	VOC
Total for all sectors	**0.433**	**0.013**	**0.298**	**0.089**	**0.042**	**0.297**	**0.325**
Iron and steel mills	0.125	0	0.019	0.005	0.004	0.014	0.004
Alumina refining and primary aluminum production	0.025	0	0.001	0	0	0.008	0
Truck transportation	0.022	0	0.023	0.007	0.001	0	0.002
Oil and gas extraction	0.021	0	0.015	0	0	0.001	0.022
Natural gas distribution	0.02	0	0.001	0	0	0	0.001
Plastics Pipe and Pipe Fitting Manufacturing	0.018	0.002	0.013	0.015	0.008	0.007	0.22
Iron, steel pipe and tube manufacturing from purchased steel	0.018	0	0.003	0	0	0.002	0.001
Other basic organic chemical manufacturing	0.016	0	0.024	0.003	0.002	0.019	0.013
Plastics material and resin manufacturing	0.015	0	0.013	0.002	0.001	0.009	0.009
Commercial and industrial machinery and equipment rental and leasing	0.012	0	0	0	0	0	0.001

Table A2- 60 Greenhouse gas (GHG) emissions (ton CO_2-eq.) from construction phase in Scenario 1

	Total GHG
Total for all sectors	**144**
Power generation and supply	40.1
Iron and steel mills	25
Plastics material and resin manufacturing	14.2
Petrochemical manufacturing	10.3
Other basic organic chemical manufacturing	10.1
Oil and gas extraction	8.36
Petroleum refineries	5.67
Truck transportation	2.65
Coal mining	2.19
Metal tank, heavy gauge, manufacturing	2.15

Scenario 2: Seawater flushing

Table A2- 61 Conventional air emissions (tons) from construction phase in Scenario 2

	CO	NH_3	NO_x	PM_{10}	$PM_{2.5}$	SO_2	VOC
Total for all sectors	**0.587**	**0.02**	**0.442**	**0.133**	**0.061**	**0.44**	**0.526**
Iron and steel mills	0.135	0	0.02	0.006	0.005	0.015	0.005
Oil and gas extraction	0.034	0	0.025	0	0	0.002	0.035
Truck transportation	0.031	0	0.033	0.009	0.002	0	0.004
Alumina refining and primary aluminum production	0.031	0	0.001	0.001	0	0.01	0
Plastics Pipe and Pipe Fitting Manufacturing	0.031	0.004	0.022	0.025	0.013	0.012	0.372
Natural gas distribution	0.03	0	0.001	0	0	0	0.001
Other basic organic chemical manufacturing	0.027	0.001	0.04	0.005	0.004	0.033	0.022
Plastics material and resin manufacturing	0.026	0	0.022	0.003	0.002	0.016	0.016
Iron, steel pipe and tube manufacturing from purchased steel	0.019	0	0.003	0.001	0	0.002	0.001
Commercial and industrial machinery and equipment rental and leasing	0.017	0	0	0	0	0	0.001

Table A2- 62 Greenhouse gas (GHG) emissions (ton CO_2-eq.) from construction phase in Scenario 2

	Total GHG
Total for all sectors	**211**
Power generation and supply	59.9
Iron and steel mills	27.1
Plastics material and resin manufacturing	23.9
Petrochemical manufacturing	17.4
Other basic organic chemical manufacturing	16.8
Oil and gas extraction	13.4
Petroleum refineries	9.23
Truck transportation	3.76
Coal mining	2.9
Pipeline transportation	2.67

Scenario 3: Greywater flushing (Aerobic)

Table A2- 63 Conventional air emissions (tons) from construction phase in Scenario 3

	CO	NH_3	NO_x	PM_{10}	$PM_{2.5}$	SO_2	VOC
Total for all sectors	**0.893**	**0.026**	**0.612**	**0.184**	**0.085**	**0.649**	**0.758**
Iron and steel mills	0.172	0	0.026	0.007	0.006	0.019	0.006
Alumina refining and primary aluminum production	0.141	0	0.006	0.004	0.003	0.045	0.002
Truck transportation	0.046	0	0.048	0.014	0.002	0.001	0.005
Oil and gas extraction	0.045	0	0.033	0	0	0.002	0.045
Natural gas distribution	0.042	0	0.002	0	0	0	0.002
Other basic organic chemical manufacturing	0.035	0.001	0.051	0.006	0.005	0.042	0.028
Plastics Pipe and Pipe Fitting Manufacturing	0.033	0.004	0.024	0.027	0.014	0.013	0.4
Plastics material and resin manufacturing	0.033	0	0.028	0.004	0.002	0.02	0.02
Iron, steel pipe and tube manufacturing from purchased steel	0.024	0	0.004	0.001	0.001	0.003	0.002
Commercial and industrial machinery and equipment rental and leasing	0.023	0	0	0	0	0	0.002

Table A2- 64 Greenhouse gas (GHG) emissions (ton CO_2-eq.) from construction phase in Scenario 3

	Total GHG
Total for all sectors	**289**
Power generation and supply	85.1
Iron and steel mills	34.5
Plastics material and resin manufacturing	30
Other basic organic chemical manufacturing	21.6
Petrochemical manufacturing	21.4
Oil and gas extraction	17.6
Petroleum refineries	12
Truck transportation	5.52
Alumina refining and primary aluminum production	5.18
Coal mining	4.04

Scenario 4: Greywater flushing (Anaerobic)

Table A2- 65 Conventional air emissions (tons) from construction phase in Scenario 4

	CO	NH_3	NO_x	PM_{10}	$PM_{2.5}$	SO_2	VOC
Total for all sectors	**0.975**	**0.028**	**0.653**	**0.196**	**0.09**	**0.7**	**0.809**
Iron and steel mills	0.187	0	0.028	0.008	0.006	0.021	0.006
Alumina refining and primary aluminum production	0.169	0	0.007	0.005	0.003	0.054	0.002
Truck transportation	0.05	0	0.052	0.015	0.003	0.001	0.006
Oil and gas extraction	0.047	0	0.034	0	0	0.002	0.048
Natural gas distribution	0.045	0	0.002	0	0	0	0.002
Other basic organic chemical manufacturing	0.036	0.001	0.053	0.007	0.005	0.044	0.029
Plastics material and resin manufacturing	0.034	0	0.029	0.004	0.003	0.02	0.021
Plastics Pipe and Pipe Fitting Manufacturing	0.033	0.004	0.024	0.027	0.014	0.013	0.4
Iron, steel pipe and tube manufacturing from purchased steel	0.026	0	0.004	0.001	0.001	0.003	0.002
Commercial and industrial machinery and equipment rental and leasing	0.024	0	0	0	0	0	0.002

Table A2- 66 Greenhouse gas (GHG) emissions (ton CO_2-eq.) from construction phase in Scenario 4

	Total GHG
Total for all sectors	**309**
Power generation and supply	91.2
Iron and steel mills	37.5
Plastics material and resin manufacturing	31.1
Other basic organic chemical manufacturing	22.6
Petrochemical manufacturing	22.1
Oil and gas extraction	18.5
Petroleum refineries	12.5
Alumina refining and primary aluminum production	6.22
Truck transportation	5.97
Coal mining	4.35

Life-cycle impact assessment results of construction phase

Table A2- 67 Characterization results of the construction phase (%)

	S1	S2	S3	S4
Climate change Human Health	46.6019	68.2848	93.5275	100
Photochemical oxidant formation	45.3792	67.3016	93.6408	100
Particulate matter formation	44.9552	66.6482	93.6297	100
Climate change Ecosystems	46.6019	68.2848	93.5275	100
Terrestrial acidification	43.7046	64.933	93.0477	100

Table A2- 68 Single score results of the construction phase (kPt)

	S1	S2	S3	S4
Climate change Human Health	5.9762	8.7568	11.994	12.824
Photochemical oxidant formation	0.0004	0.0006	0.0008	0.0008
Particulate matter formation	2.0049	2.9723	4.1756	4.4597
Climate change Ecosystems	0.5317	0.779	1.067	1.1409
Terrestrial acidification	0.0013	0.002	0.0029	0.003
Total	**8.5145**	**12.5107**	**17.2402**	**18.4284**

Operation phase

<u>Life-cycle impact assessment results of operation phase</u>

Table A2- 69 Characterization results of the operation phase (%)

	S1	S2	S3	S4
Climate change Human Health	100	81.6353	79.6324	50.0138
Ozone depletion	100	93.0232	55.9506	55.8194
Human toxicity	100	99.8523	63.8467	52.286
Photochemical oxidant formation	100	88.1915	80.633	48.2292
Particulate matter formation	100	76.3705	73.6936	53.0704
Ionising radiation	100	98.0745	55.2152	55.156
Climate change Ecosystems	100	81.6343	79.6338	50.0136
Terrestrial acidification	100	92.9904	95.7317	41.969
Freshwater eutrophication	99.8986	100	54.5281	54.4046
Terrestrial ecotoxicity	100	98.4525	57.0754	53.9359
Freshwater ecotoxicity	99.7165	100	56.3755	54.0307
Marine ecotoxicity	99.9697	100	56.7774	54.1592
Agricultural land occupation	98.9364	100	55.2106	55.0714
Urban land occupation	99.8171	100	54.5253	54.3297
Natural land transformation	88.2978	100	63.7456	62.1361
Metal depletion	100	98.1764	54.8787	54.742
Fossil depletion	97.9698	89.1289	100	39.6397

Table A2- 70 Single score results of the operation phase (kPt)

	S1	S2	S3	S4
Climate change Human Health	10.473	8.5496	8.3399	5.2379
Ozone depletion	0.0007	0.0007	0.0004	0.0004
Human toxicity	1.2559	1.254	0.8019	0.6567
Photochemical oxidant formation	0.0009	0.0008	0.0008	0.0004
Particulate matter formation	6.4839	4.9518	4.7782	3.441
Ionising radiation	0.0211	0.0207	0.0117	0.0117
Climate change Ecosystems	0.9316	0.7605	0.7419	0.4659
Terrestrial acidification	0.0038	0.0035	0.0036	0.0016
Freshwater eutrophication	0.0043	0.0043	0.0023	0.0023
Terrestrial ecotoxicity	0.0019	0.0019	0.001	0.001
Freshwater ecotoxicity	0.0001	0.0001	0.0001	0.0001
Marine ecotoxicity	0	0	0	0
Agricultural land occupation	0.005	0.005	0.0028	0.0028
Urban land occupation	0.0124	0.0124	0.0068	0.0068
Natural land transformation	0.0003	0.0003	0.0002	0.0002
Metal depletion	0.014	0.0138	0.0077	0.0077
Fossil depletion	7.2617	6.6064	7.4121	2.9382
Total	**26.4706**	**22.1859**	**22.1113**	**12.7746**

Life-cycle assessment results (construction + operation phases)

Table A2- 71 Final characterization results (%)

	S1	S2	S3	S4
Climate change Human Health	100	83.4355	82.9907	54.579
Ozone depletion	100	93.0232	55.9506	55.8194
Human toxicity	100	99.8523	63.8467	52.286
Photochemical oxidant formation	100	89.3687	83.0949	51.6
Particulate matter formation	100	77.465	75.7425	55.649
Ionising radiation	100	98.0745	55.2152	55.156
Climate change Ecosystems	100	83.4346	82.9924	54.5793
Terrestrial acidification	100	93.9574	97.7701	45.2195
Freshwater eutrophication	99.8986	100	54.5281	54.4046
Terrestrial ecotoxicity	100	98.4525	57.0754	53.9359
Freshwater ecotoxicity	99.7165	100	56.3755	54.0307
Marine ecotoxicity	99.9697	100	56.7774	54.1592
Agricultural land occupation	98.9364	100	55.2106	55.0714
Urban land occupation	99.8171	100	54.5253	54.3297
Natural land transformation	88.2978	100	63.7456	62.1361
Metal depletion	100	98.1764	54.8787	54.742
Fossil depletion	97.9698	89.1289	100	39.6397

Table A2- 72 Total single score results (kPt)

	S1	S2	S3	S4
Climate change Human Health	215.4353	179.7495	178.7912	117.5825
Ozone depletion	0.0141	0.0132	0.0079	0.0079
Human toxicity	25.119	25.0819	16.0377	13.1337
Photochemical oxidant formation	0.019	0.017	0.0158	0.0098
Particulate matter formation	131.682	102.0075	99.7393	73.2798
Ionising radiation	0.4228	0.4147	0.2335	0.2332
Climate change Ecosystems	19.1644	15.9897	15.905	10.4598
Terrestrial acidification	0.077	0.0723	0.0752	0.0348
Freshwater eutrophication	0.0852	0.0853	0.0465	0.0464
Terrestrial ecotoxicity	0.038	0.0375	0.0217	0.0205
Freshwater ecotoxicity	0.0029	0.0029	0.0016	0.0016
Marine ecotoxicity	9.05E-06	9.05E-06	5.14E-06	4.9E-06
Agricultural land occupation	0.099	0.1	0.0553	0.0551
Urban land occupation	0.2481	0.2486	0.1355	0.135
Natural land transformation	0.0051	0.0058	0.0037	0.0036
Metal depletion	0.2804	0.2753	0.1539	0.1535
Fossil depletion	145.2331	132.1272	148.2428	58.763
Total	**537.9257**	**456.2284**	**459.4667**	**273.9205**

Eco-efficiency Analysis

Table A2- 73 Eco-efficiency profile positions of scenarios with different R_{EC} ratios

	S1	S2	S3	S4
R_{EC}=1				
PP_e	1.25E+00	1.06E+00	1.06E+00	6.34E-01
PP_c	1.22E+00	8.64E-01	1.05E+00	8.64E-01
R_{EC}=100				
PP_e	3.46E+00	1.56E+00	1.64E+00	-2.66E+00
PP_c	1.02E+00	9.86E-01	1.00E+00	9.86E-01
R_{EC}=0.01				
PP_e	1.02E+00	1.01E+00	1.01E+00	9.63E-01
PP_c	3.23E+00	-3.63E-01	1.49E+00	-3.57E-01

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