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Eco-efficiency Analysis of Sludge Treatment Scenarios in Urban Cities: the Case of Hong Kong

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Abstract

Urbanization is unrelenting due to rapid world population growth, necessitating a sustainable assessment with consideration of environmental impact to minimize resources inputs and waste outputs. An eco-efficiency analysis (EEA) framework has been developed to evaluate urban sludge handling options. Assessment of economic cost and environmental impact has revealed the suitability of the framework in urban application, as demonstrated by a case-study assessment of six sewage sludge management scenarios in Hong Kong. Land cost considerations, which are trivial in rural areas, have been revealed to be crucial in urban cities by the recognition of consequential sensitivity to high urban land costs. Furthermore, separate and detailed assessment of sludge treatment facilities based on actual transportation data are also

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highly significant because the accumulated GHG emissions associated with transportation can vary up to 187,000 tons when using single transportation distance assumptions. By the inclusive evaluation of sludge scenarios instead of individual treatment technology, comprehensive and informative results were obtained for sustainable town planning and sludge management. The EEA framework for urban sludge management developed in this study, which considers the economic and environmental aspects of the scenarios, enables informed sustainable town planning based on the priorities of the decision makers.

Keywords: Eco-efficiency, Life-Cycle Assessment, Sludge Treatment, Urban Cities

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 (91/271/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 (European Commission, 2010). In the USA, 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 USA (European Commission, 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 (MOUHUR and NDARC, 2011). Processes adopted in China for sludge treatment include drying, thickening, dewatering, anaerobic digestion, 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.

Sludge is an unavoidable by-product of water and wastewater treatment processes. According to the information provided by the Hong Kong Drainage Services Department (DSD), Hong Kong will generate nearly 30,000 m^3 of sludge per day (EPD, 2008b) 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, 2008a); 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 evaluate the appropriateness of various wastewater sludge treatment options adopted in Hong Kong, this study examines the performance of six treatment scenarios using eco-efficiency analysis (EEA).

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 (Guinee et al., 2011), such as waste management and water management, by striking a balance between economic growth and environmental conservation (Chang et al., 2014). Early in 2000, a life-cycle approach for evaluating the sustainability of sludge reuse options was suggested (Bridle & Skrypski-Mantele, 2000). 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). Foley et al. (2010) carried out a study to reveal the life-cycle inventories of wastewater treatment scenarios without assessing the environmental trade-offs, using life-cycle impact assessment (LCIA). Conventional LCA that only focused on environmental consequences was conducted to analyze the resource consumption and environmental emissions associated with sludge handling processes (Suh & Rousseaux, 2002; Houillon & Jolliet, 2005).

To provide a more practical and comprehensive urban sludge management solution, the economic cost of the treatment scenarios was included in the 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 life-cycle cost (LCC) approach (Kicherer, Schaltegger, Tschochohei, & Pozo, 2006). LCC methodology was adopted in addition to the traditional LCA in previous studies on sludge management (Hong, Hong, Otaki, & Jolliet, 2009; Lundin, Olofsson, Pettersson, & Zetterlund, 2004; Murray, Horvath, & Nelson, 2008; Uggetti, Ferrer, Molist, & García, 2011; Xu, Chen, & Hong, 2014). To provide sound evidence for strategic sludge management decisions in urban cities, an EEA framework for urban sludge handling is developed for the evaluation of both the economic and environmental aspects with the inclusion of the characteristics of urban cities.

2. Goal and Scope Definition

2.1. Goal

The primary goal of this study is to develop an EEA framework that is suitable for sludge management in urban cities. 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, in the life-cycle impact assessment (LCIA) research conducted by Suh and Rousseaux (2002) and Houillon and Jolliet (2005), “land occupation”, which has a crucial impact in compact cities, was excluded. Hong et al. (2009) and Xu et al. (2014) included the impact of land use in their studies. However, the elimination of operating costs and capital costs of infrastructures led to inadequacies in their studies. Murray et al. (2008) and Xu et al. (2014) assumed that the transportation distances between the treatment facilities were 25km and 40 km respectively. The assumptions led to inaccuracies in the estimation of atmospheric emissions associated with transportation. 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. 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 appropriate sludge treatment approach for adoption in Hong Kong. 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 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 (Munoz 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. Based on the actual conditions in Hong Kong, this study evaluated the economic and environmental consequences of six sludge treatment scenarios, with the aim of informing decision-making on sludge management in the city.

2.2. System Boundary

A number of research studies have been conducted on the application of LCA to sewage sludge handling processes. A few of them 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 set up in numerous studies. Murray et al. (2008) and Foley et al. (2010) analyzed the life-cycle inventories of the scenarios, while

LCIA was included in the studies conducted by Suh & Rousseaux (2002), Houillon & Jolliet (2005), Hong et al. (2009) and Xu et al. (2014). In this study, sludge handling scenarios, rather than individual technologies, were investigated to offer more comprehensive results. The six scenarios, which were defined based on actual practices and conditions, involved different combinations of treatment processes used in Hong Kong (Figure 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 (Supporting Information Fig. S3 and S5) according to the real practice in the four STWs studied; treatment options without AD (Supporting Information Fig. S5) 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 (Supporting Information Fig. S7 and S9). Since sludge ash utilization in cement production has been investigated in previous research studies (Houillon & Jolliet, 2005; Murray et al., 2008; Lam, Barford & McKay, 2010), 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, Barford & McKay, 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.

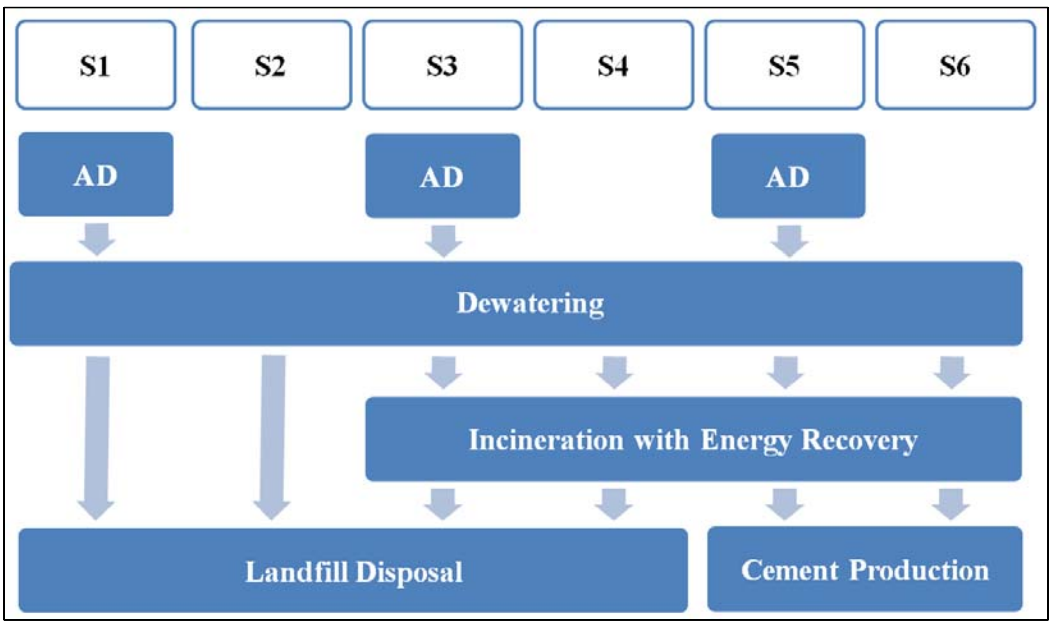

Figure 1 Six sludge treatment scenarios defined as the scope for LCA

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 ton of dry solids in raw sewage sludge, which has also been used in previous research (Suh & Rousseaux, 2002; Lundin et al., 2004; Houillon & Jolliet, 2005; Hong et al., 2009; 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. Methodology

The boundary of the LCA framework should include all of the processes that contribute significantly to the products or activities that were studied (Rebitzer et al., 2004) and an EEA framework, which includes economic analysis and environmental impact assessment, was adopted in this study. Actual operational information, emission factors from literatures and an economic input-output life-cycle assessment (EIO-LCA) tool (CMU, 2006) was used to evaluate the emissions and environmental impacts of the scenarios.

3.1. Economic Analysis

Capital costs, operational and maintenance (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 DSD, as well as the price of diesel. The lifetime economic costs of the six scenarios (Supporting Information Part 1) over a 20-year time horizon and with 6.6% inflation rate (Census and Statistics Department, 2014) were presented in present values (PVs).

3.2. Environmental Impact Assessment

Emissions from material production, electricity balance and vehicles have been commonly considered in LCIA (Lundin et al., 2004; Murray et al., 2008; Xu et al., 2014). Energy consumption and atmospheric emissions were included in the LCIA. 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 (Supporting Information 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 (Life Cycle Assessment: Principles and Practice, 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 1).

Table 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) in a 20-year time horizon, and CO_2 was used as the reference GHG for the global warming potentials. Human toxicity impact, expressed in kg 1, 4-dichlorobenzene equivalent, was investigated for a 20-year time horizon and a global scale. 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. Normalization of the impact category indicators was not necessary as the inventory data used for the LCIA was already expressed in kilogram of emissions per dry ton of raw sewage sludge.

The characterized impact assessment results was normalized using a set of normalization factors (Dong & Ng, 2014) presented in **Error! Reference source not found.** 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 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

3.3. Data Source

The economic and environmental performance of the six sewage sludge treatment scenarios applied on the four major secondary sewage treatment works (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 shows the data for the STWs in 2013.

Table 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

4. Results

4.1. Life-cycle Cost Inventory

Table 4 presents the total costs of the sludge treatment processes in the four STWs studied (Supporting Information 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 $1 M USD respectively. In the scenarios with AD application, the dewatering process costs $10 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 4 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	-	3,642,000,000
S3	(52,000,000)	10,000,000	19,000,000	95,000,000	-	71,000,000
S4	-	23,000,000	104,000,000	367,000,000	-	492,000,000
S5	(52,000,000)	10,000,000	19,000,000	-	(1,000,000)	(25,000,000)
S6	-	23,000,000	104,000,000	-	(1,000,000)	126,000,000

4.2. Environmental Impact Inventory

4.2.1. Electricity Balance

The electricity consumptions (Supporting Information Table S48) of the six scenarios are presented in Figure 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.

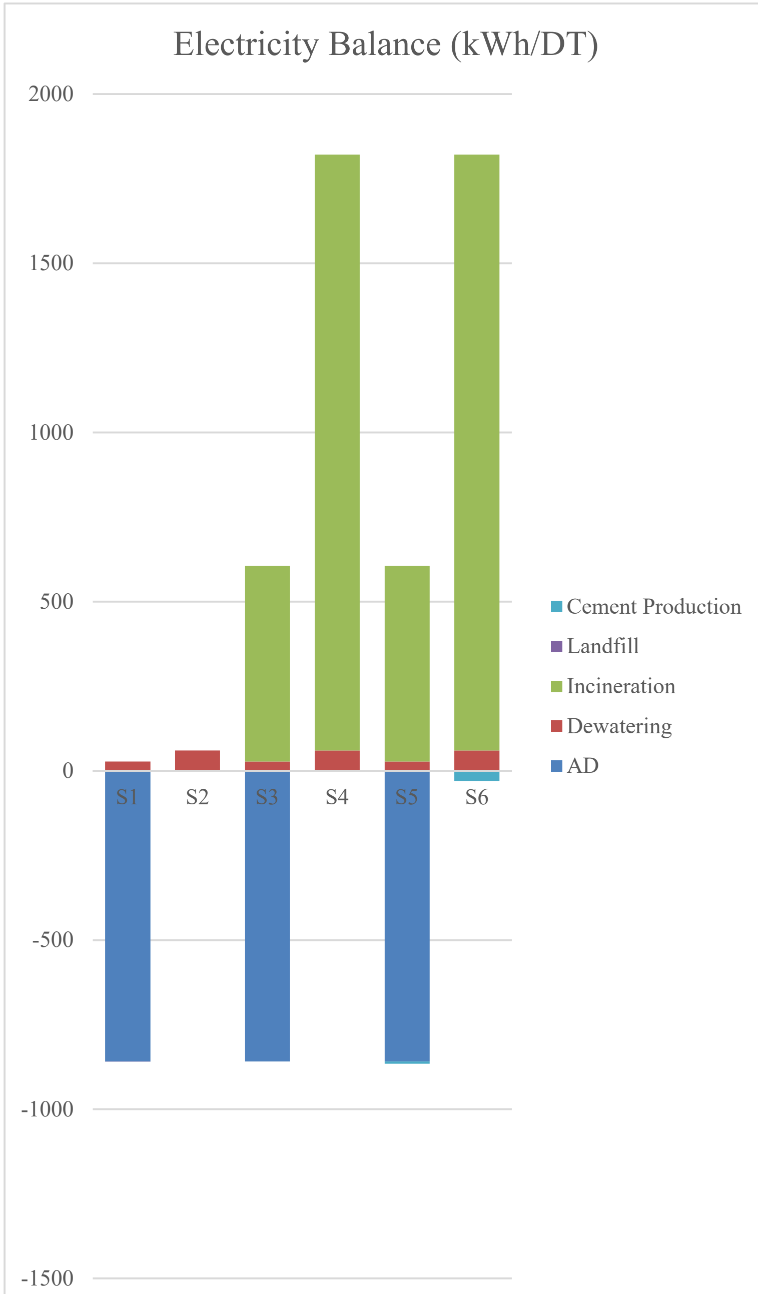

Figure 2 Electricity consumption (kWh/DT) of sludge treatment processes

4.2.2. Atmospheric Emissions

Air emissions from the sewage sludge treatment processes are listed in Table 5 (Supporting Information 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.

4.2.3. Life-cycle Impact Assessment (LCIA)

Five life-cycle environmental impact categories (land occupation, global warming potential, human toxicity, acidification and eutrophication) were assessed (Supporting Information Part 2B), and the results are presented in this section (Table 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 7. The ascending order of the normalized environmental impact of the scenarios is S5 < S3 < S1 < S6 < S4 < S2.

Table 5 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 6 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 7 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

5. Discussion

5.1. Economic Cost Analysis

The total life-cycle costs (Supporting Information Table S44 to S47) of the six sludge treatment scenarios in the Sha Tin, Tai Po, Yuen Long and Shek Wu Hui STWs are presented in Figure 3.

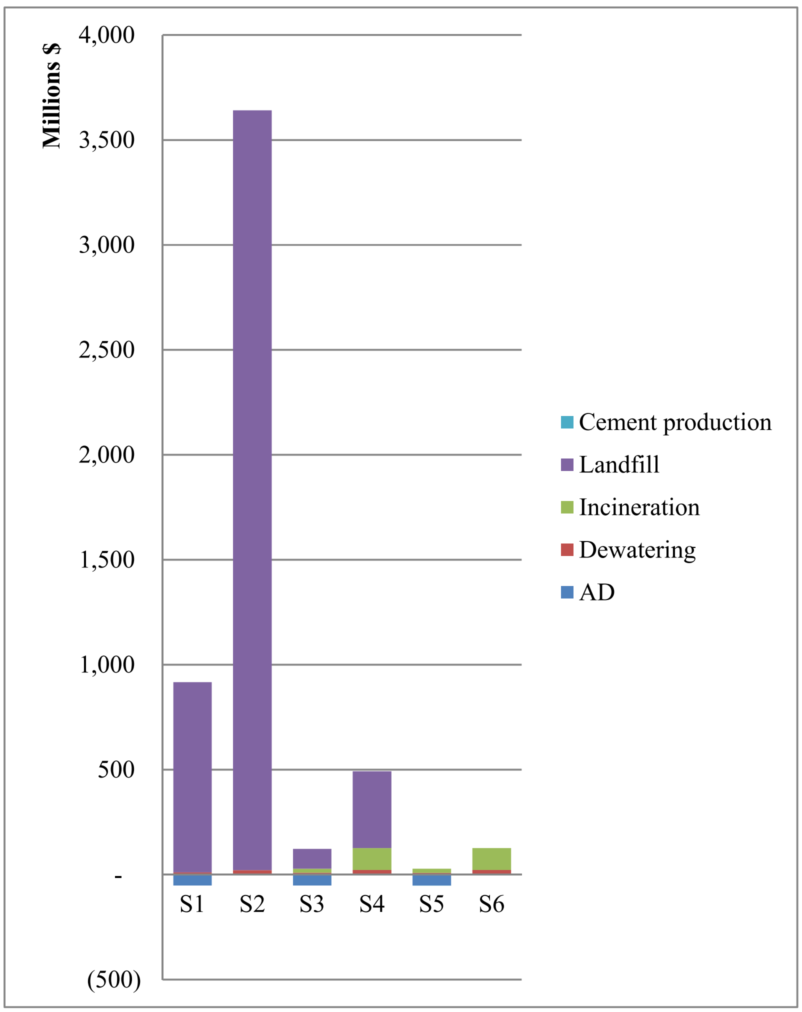

Figure 3 Total life-cycle costs of sludge treatment scenarios

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 1620 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$ 2165/ton 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$1200 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 8.

Table 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.

5.2. Life-Cycle Impact Assessment

5.2.1. Land Occupation

The impact on land occupation that resulted from the defined sewage sludge treatment scenarios, operating for 30 years, is presented in Figure 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.

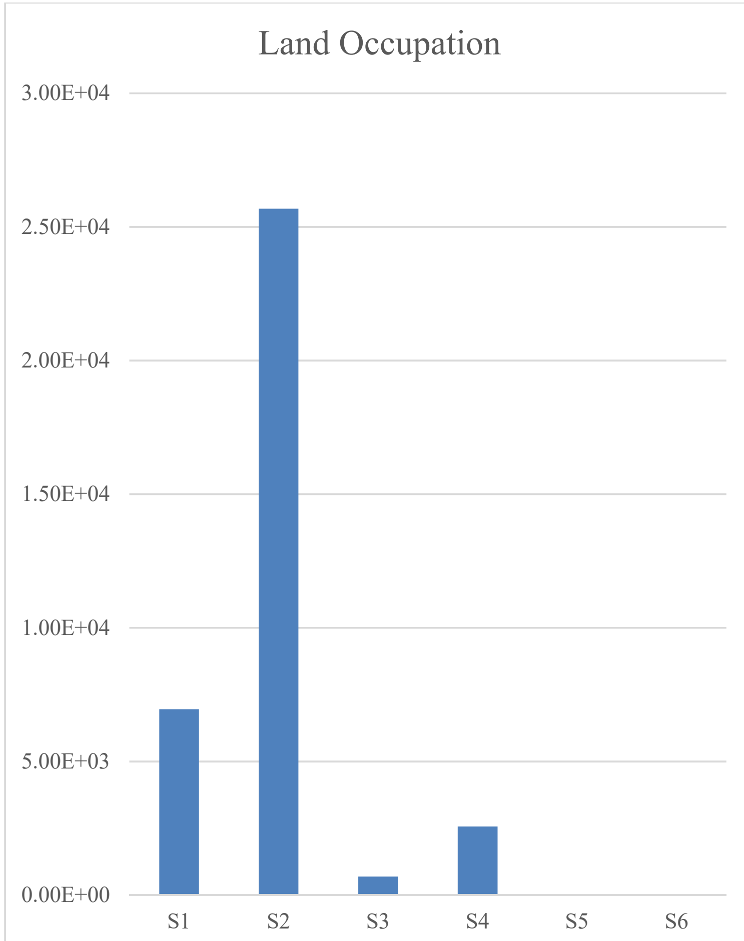

Figure 4 Life-cycle Impact on Land Occupation (acre•yr/DT) of the Sludge Treatment Scenarios

5.2.2. Climate Change, Acidification, Human Toxicity and Eutrophication

The life-cycle environmental impact of sludge management scenarios is presented in Figure 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.

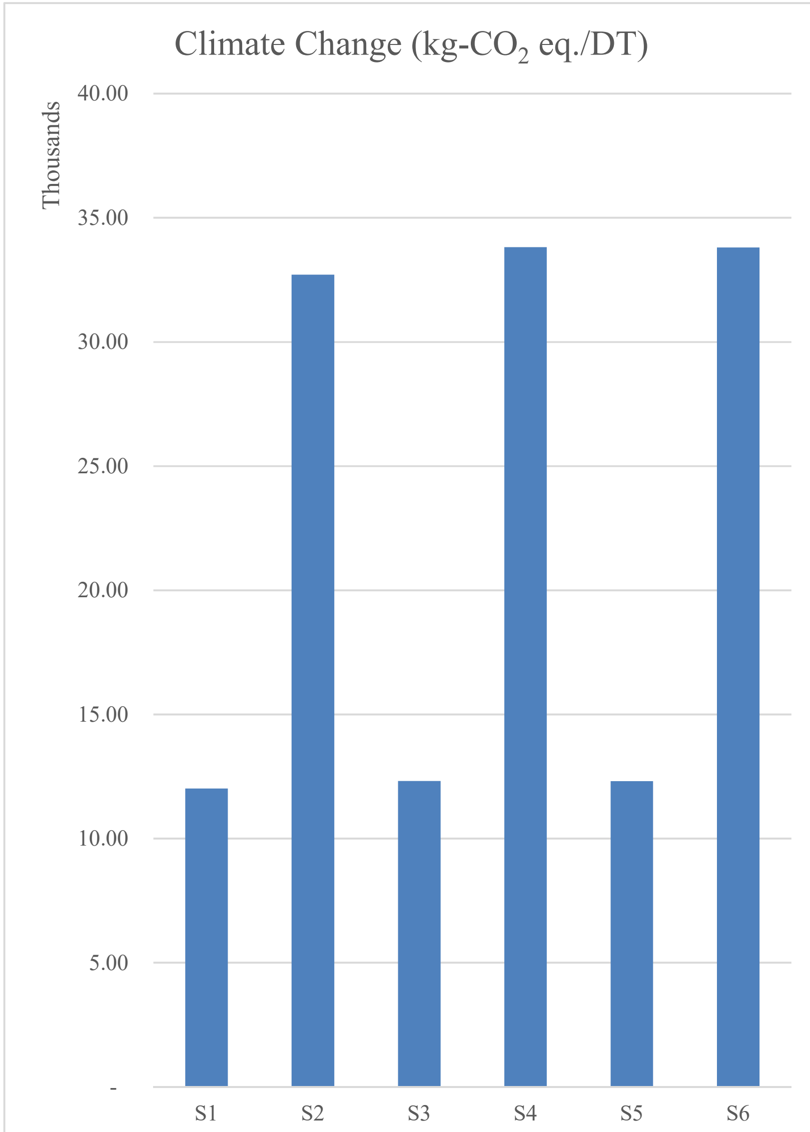

Figure 5 Life-cycle impact on climate change (kg-CO_2 eq./DT) of the sludge treatment scenarios

5.2.3. 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 was analyzed and presented in Figure 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.

5.3. 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). Most of the European countries have abandoned landfilling except three countries, namely Italy, Denmark and Estonia, where there is a slight increase in the use of landfill disposal (Kelessidis & Stasinakis, 2012). In the USA, 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.

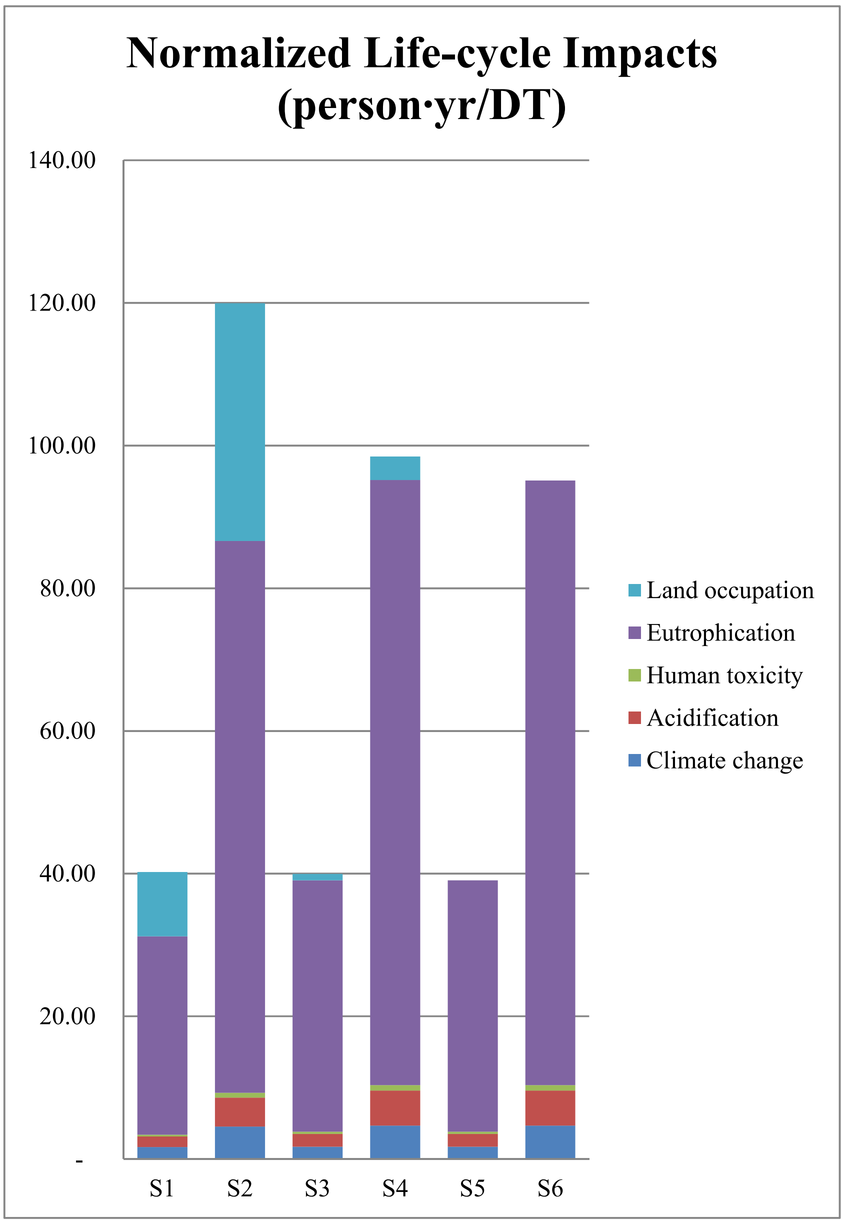

Figure 6 Normalized life-cycle environmental impacts of the sludge treatment scenarios

6. Sensitivity Analysis

6.1. 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 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 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 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 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.

6.2. 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 11).

Table 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 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 tons and 137,000 tons 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 tons 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.

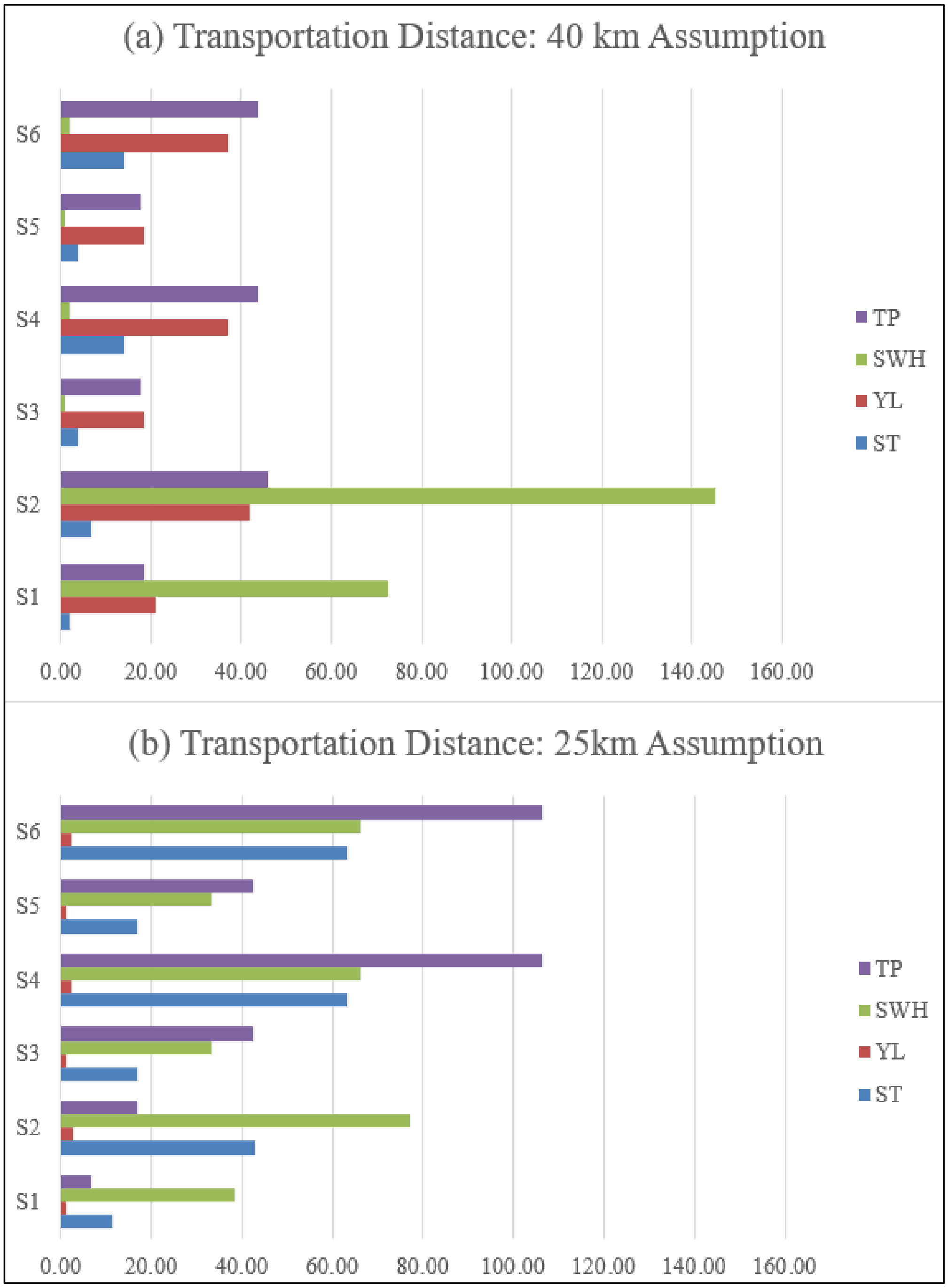

Figure 7 Absolute values of the inaccuracies with the assumptions of <u>(a) 40 km</u> and <u>(b) 25 km</u> transportation distances

7. Conclusions

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 represents 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.

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