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Evaluating the environmental impacts of stabilization and solidification technologies for managing hazardous wastes through life cycle assessment: A case study of Hong Kong

Md. Uzzal Hossain [a], Lei Wang [b,c,*], Liang Chen [b], Daniel C.W. Tsang [b,*], S. Thomas Ng [a], Chi Sun Poon [b], Viktor Mechtcherine [c]

[a] Department of Civil Engineering, The University of Hong Kong, Pokfulam Road, Hong Kong, China
[b] Department of Civil and Environmental Engineering, The Hong Kong Polytechnic University, Hung Hom, Kowloon, Hong Kong, China
[c] Institute of Construction Materials, Technische Universität Dresden, 01062 Dresden, Germany

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ABSTRACT

Proper management of hazardous materials arouses widespread environmental concerns due to its enormous ecological and health impacts. The development of green stabilization/solidification (S/S) technology for resourceful utilization of hazardous materials, as well as the immobilization of potentially toxic elements is of great scientific interests. Cement-based S/S is often considered a low-cost and highly efficient technology, but the environmental sustainability of a broad spectrum of S/S technologies has yet to be evaluated. Therefore, this study assessed the environmental sustainability of S/S technologies for managing two common types of hazardous wastes, i.e., contaminated marine sediment and municipal solid waste incineration fly ash (MIFA) by using life cycle assessment (LCA). A total of 17 scenarios under three strategies for sediment and two strategies for MIFA S/S technologies were comprehensively evaluated. The LCA results identified the most preferable S/S technology in each strategy. In particular, Scenario 1 (mixture of sediment with a small percentage of ordinary Portland cement and incinerated sewage sludge ash) of Strategy 1 (use as fill materials) would be the preferred option, as it reduces about 54% and 70% global warming potential compared to those of Scenarios 2 and 3, respectively. This is the first initiative for evaluating the environmental impacts of a wide range of recently developed S/S technologies using green/alternative binders for diverting hazardous wastes from disposal. The results can serve as a decision support for the practical application of the environmentally friendly S/S technology for sustainable remediation.

1. Introduction

Contaminants often deposit and accumulate in the sediment via surface runoff, sewage discharge, and atmospheric deposition. Globally, a considerable amount of contaminated sediment from dredging activities is generated annually, for example, about 0.9 million m^3 in Taranto (southern coast of Italy) (Barjoveanu et al., 2018), over 150 million m^3 in the USA (Bates et al., 2015), and about 300 million m^3 in Europe each year (Snellings et al., 2018). As a harbour city, approximately 3.89 million m^3 marine sediment should be dredged annually to maintain a sufficient depth of shipping channels in Hong Kong (HK CEDD, 2016). Approximately one-third of dredged sediment was contaminated by potentially toxic elements (As, Cd, Cr, Hg, Pb, etc.) and exogenous organic matter (polychlorinated biphenyls, polycyclic aromatic hydrocarbon, etc.) (HK EPD, 2017). The off-site disposal at the landfills or offshore contaminated mud pits was a non-sustainable option for contaminated sediment, which occupies precious land resources or easily cause secondary pollution (Wang et al., 2019a). Thus, the sustainable management of contaminated dredged sediment is a huge challenge in Hong Kong. Besides, a new municipal solid waste incinerator is expected to be commissioned in 2024 in Hong Kong (HK EPD, 2018). The daily treatment capacity of the incinerator is 3000 tonnes of municipal solid waste, however, it will also generate >900 tonnes of incineration bottom ash and fly ash. The incineration fly ash is

* Corresponding authors at: Department of Civil and Environmental Engineering, The Hong Kong Polytechnic University, Hung Hom, Kowloon, Hong Kong, China (D.C.W. Tsang); Institute of Construction Materials, Technische Universität Dresden, 01062 Dresden, Germany (L. Wang).
E-mail addresses: lei.wang1@mailbox.tu-dresden.de (L. Wang), dan.tsang@polyu.edu.hk (D.C.W. Tsang).

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hazardous wastes as it contains a high concentrations of potentially toxic elements and even dioxin (Bogush et al., 2015; Tong et al., 2020). Disposal at landfill consumes limited capacity and needs continuous environmental monitoring. Thus, there is an urgent demand for the sustainable treatment of municipal solid waste incineration fly ash (MIFA).

Cement-based stabilization/solidification (S/S) is a low-cost and time-efficient treatment method for hazardous materials (Shi and Fernández-Jiménez, 2006; Roy and Stegemann, 2017). It is one of the commonly used treatment methods for hazardous wastes including MIFA and contaminated sediment (Barjoveanu et al., 2018; Huber et al., 2018; Huber and Fellner, 2018; Margallo et al., 2019). Contaminants can be chemically fixed and physically encapsuled in the cement-based matrix, and the S/S products can be used as value-added construction materials (Wang et al., 2018a). Although ordinary Portland cement (OPC) is a mature and versatile binder, the OPC is a high carbon footprint material (Wang et al., 2020a). Approximately 1 tonne of CO_2 is generated for one tonne of locally produced OPC in Hong Kong (Damineli et al., 2010; Hossain et al., 2017). Besides, the compatibility between OPC and potentially toxic elements are questionable (Senneca et al., 2020; Wang et al., 2018c; Wang et al., 2020b,c). For instance, low content of Zn (0.1 wt%) would seriously delay cement hydration and influence S/S performance (Garg and White, 2017); OPC is not effective for immobilization of As and even increase its leachability due to the elevated pH (Wang et al., 2019b).

The application of alternative cements can enhance the compatibility with contaminants and reduce the dosage of binder, thus alleviating the carbon footprint. For example, reactive magnesia cement (MC) shows high compatibility with different metals/metalloids and even organic matter (Jin and Al-Tabbaa, 2014; Wang et al., 2017; Yi et al., 2015). Calcium aluminate cement (CAC) is a rapid-hardening cement, which can rapidly gain early strength even in extreme condition (Engbert et al., 2020; Nowacka and Pacewska, 2020). Moreover, some industrial waste and by-products can be recycled as supplementary cementitious materials (SCMs) to reduce CO_2 emission. Pulverized fuel ash (PFA), a Si-rich by-product from thermal power plants, is one of the most commonly used SCMs, which can partially replace OPC and improve the mechanical properties of cement-based composites via pozzolanic reaction (Jia and Richardson, 2018). Ground-granulated blast-furnace slag (GGBS) is a type of Si- and Ca-rich SCMs that is a by-product from a blast furnace, which has high hydration and pozzolanic reactivity and can be employed to improve the strength and durability of concrete (Ganesh and Murthy, 2019). Incinerated sewage sludge ash (ISSA) is Si- and Fe-rich waste from sewage sludge incinerator, which also can be recycled as a sustainable pozzolanic material in concrete (Zhou et al., 2020). Therefore, it is a low-carbon and sustainable strategy that recycles contaminated sediment and MIFA as alternative cement or SCMs-blended cement in the S/S treatment. The technical feasibility of different types of green cement-based sediment S/S composites and MIFA S/S composites was validated in many previous studies (e.g. Chen et al., 2019, 2020; De Gisi et al., 2020; Todaro et al., 2020; Wang et al., 2017, 2018a, 2018b, 2019a, 2019b; Zhang et al., 2020). Nevertheless, the environmental merits of these novel products should be quantified.

Life cycle assessment (LCA) is a commonly used method to assess the environmental viability of treatment/manufacturing technologies in a holistic manner. Several studies evaluated the environmental impacts of sediment placement strategies, such as open water placement and upland placement (Bates et al., 2015; Sharaan and Negm, 2017), physicochemical decontamination methods (Falciglia et al., 2018), and S/S treatment (Barjoveanu et al., 2018). Based on LCA, Bates et al. (2015) found that upland placement of uncontaminated dredged sediment was associated with greater environmental impacts due largely to the use of fuel, compared to open water placement in New York, USA. Falciglia et al. (2018) comparatively evaluated the environmental sustainability of remediation techniques of marine sediments such as electrokinetic and citric acid-enhanced microwave heating remediation for decontamination of hydrocarbons using LCA. The study reported the latter approach was most sustainable with rapid and effective decontamination. Moreover, Barjoveanu et al. (2018) studied the environmental impacts of S/S of marine sediment for potential reuse or landfill disposal using LCA. The study found that *ex-situ* treatment can induce higher impacts with increasing dosage of additives in the S/S mixture (activated carbon, chemical reagents, etc.), but could reduce the potential risk of leaching.

Besides, the treatment and disposal of MIFA were also assessed in previous research. For example, the environmental impacts of different scenarios were evaluated, including underground deposition, cement-based stabilization, thermal treatment, etc. (Huber et al., 2018), whereas the potential use in the production of cement, metal and de-icing salts were assessed by Huber and Fellner (2018). Similar studies were conducted in Spain by Margallo et al. (2019) and in Taiwan by Huang and Chuieh (2015). The environmental impacts of different treatment technologies for MIFA were evaluated by Pei et al. (2020). The study concluded that plasma vitrification technology could significantly reduce the toxicity impacts, whereas water-washing treatment resulted in the highest impacts. In addition, Tang et al. (2020) proposed an integrated approach for metal recovery and decontamination of hazardous wastes, including the associated carbon reduction compared to landfilling. However, Billen et al. (2015) demonstrated that cement-based S/S mixtures for fly ash and municipal solid waste incineration residue might induce higher environmental impacts, in terms of climate change and acidification compared to alternative S/S mixtures.

Although OPC-based S/S of sediment composites or MIFA composites has been evaluated (e.g. Barjoveanu et al., 2018; Billen et al., 2015), there is a lack of comprehensive LCA for examining the sediment and MIFA-derived S/S products treated by low-carbon alternative cement and SCMs-blended cement. Environmental sustainability is an important indicator of newly developed technology in addition to technical feasibility. However, it is impractical to adopt the LCA results of S/S technologies from different studies due to the different technologies developed, system boundaries, experimental data, geographic locations, etc. Thus, the environmental sustainability of the developed technologies should be evaluated in accordance with the local characteristics and existing systems. This study aims to (i) evaluate the environmental impacts of three types of sediment-derived S/S products (with eleven scenarios) using different alternative cementitious materials in the context of Hong Kong, which represents a typical harbour city; and (ii) assess the environmental impacts of two types of MIFA derived S/S products (with six scenarios) in the context of Hong Kong, which represents a typical densely populated metropolis. By validating the environmental feasibility, the current study can assist the establishment of a sustainable treatment method for sediment- and MIFA-derived construction materials. The LCA results of the studied S/S technology can also serve as a reference for other regions globally.

2. Methodology

2.1. Materials, mixtures, and strategies

The materials and mixtures for the developed S/S technologies for managing contaminated sediment and MIFA are shown in Table 1. To ensure the closed-loop resource utilization, different recycled materials were used as aggregates (recycled concrete aggregates and waste glass), alternative cements (reactive magnesia cement and calcium aluminate cement), supplementary cementitious materials (PFA, GGBS, ISSA, glass powder), along with conventional materials (OPC and natural aggregates). To comparatively analyze the studied materials, 11 scenarios were developed under three strategies for sediment S/S technologies, whereas 6 scenarios under two strategies for MIFA S/S technologies.

The production methods of different sediment- and MIFA-derived S/S blocks were described and proved effective in our previous research (Chen et al., 2019, 2020, Wang et al., 2017, 2018a, 2018b, 2019a,

Table 1
Considered strategies and scenarios of S/S for the studied materials.

Strategies	Scenarios (mix-designs)	Reference	Obtained technical requirements
Sediment S/S technologies			
S1: Fill materials	Sc1: 1.7%OPC + 3.3% ISSA + 47.5%Sed + 47.5% RF	Wang et al. (2018a)	>1 MPa and pass TCLP[a]
	Sc2: 1%OPC + 4%GGBS + 47.5%Sed + 47.5%RF	Wang et al. (2018a)	
	Sc3: 5%MC + 5%ISSA + 90%Sed	Wang et al. (2019a)	
S2: Partition blocks	Sc4: 16%OPC + 4%ISSA + 16% Sed + 16%RF + 48%RC	Wang et al. (2018a)	>7 MPa and pass TCLP [a,b]
	Sc5: 12%OPC + 8%GGBS + 16% Sed + 16%RF + 48%RC	Wang et al. (2018a)	
	Sc6: 10%MC + 5% PFA + 85%Sed	Wang et al. (2019a)	
	Sc7: 13.5%MC + 1.5% GP + 85%Sed	Wang et al. (2019a)	
S3: Paving blocks	Sc8: 30%OPC + 14%Sed + 14%RG + 42%NC	Wang et al. (2018a)	>30 MPa and pass TCLP[a,c]
	Sc9: 24%OPC + 6%GGBS + 14%Sed + 14%RF + 42%NC	Wang et al. (2018a)	
	Sc10: 20%OPC + 5%PFA + 15%Sed + 60%NC	Wang et al. (2018b)	
	Sc11: 20%OPC + 5%MC + 15%Sed + 60%NC	Wang et al. (2017)	
MIFA S/S technologies			
S4: Fill materials	Sc12: 8% OPC + 2%SF + 90% MIFA	Chen et al. (2019a)	>1 MPa and pass TCLP[a]
	Sc13: 6% OPC + 4%SF + 90% MIFA	Chen et al. (2019a)	
	Sc14: 10% CAC + 90% MIFA	Chen et al. (2020)	
	Sc15: 6% CAC + 1%TSP + 93% MIFA	Chen et al. (2020)	
S5: Partition blocks	Sc16: 20% OPC + 80% MIFA	Chen et al. (2019a)	>7 MPa and pass TCLP [a,b]
	Sc17: 13.5% CAC + 1.5% TSP + 85% MIFA	Chen et al. (2020)	

[Note: Sed: sediment; MIFA: municipal solid waste incineration fly ash; OPC: ordinary Portland cement; MC: reactive MgO cement; CAC: Calcium aluminate cement; GGBS: ground granulated blast-furnace slag; GP: glass powder; ISSA: Incinerated sewage sludge ash; PFA: pulverized fly ash; SF: silica fume; RC: recycled coarse aggregate; RF: recycled fine aggregate; RG: recycled glass; NC: natural coarse aggregate; TSP: trisodium phosphate;]

[a] HK EPD (2011).
[b] BS EN 6073 (1981).
[c] HK ETWB (2004).

2019b, 2019c, 2019d). The uniaxial compressive strength of different sediment- and MIFA-derived fill material after 7-day and 28-d air curing was determined by a universal strength testing machine (Testometric CXM 500-50 KN). The compressive strength of paving blocks and partition blocks was determined by a wide range testing machine (SERVO-PLUS, 4000 KN) according to the standard method (BS EN 12390, 2009). Besides, the leaching concentrations of potentially toxic elements (e.g. As, Cr, Pb, etc.) from S/S blocks were analyzed after Toxicity Characteristic Leaching Procedure (US EPA, 1992). The compressive strength and TCLP leachability of all the S/S blocks were referred to previous papers as mentioned above, and all of them fulfilled different technical requirements (HK EPD, 2011; BS EN 6073, 1981; HK ETWB, 2004).

2.2. *Life cycle assessment of S/S technologies*

2.2.1. *Goal and scope of the study*

The aim of the study is to comparatively evaluate the environmental performance of S/S technologies for managing two types of hazardous wastes such as contaminated dredged sediment (referred to as sediment hereafter) and MIFA using LCA technique according to ISO guidelines (ISO, 2006a,b). Considering the comparative analysis for the scenarios, 'cradle-to-gate' system boundary is adopted with the functional unit of 1 tonne of final product manufacturing (e.g., fill materials, partition blocks, and paving blocks). Three most common types of functional units are used in LCA of concrete or construction products, *viz.*: (i) environmental impacts for per 1 m^3 of concrete or concrete product (Panesar et al., 2017; Visintin et al., 2020); (ii) per tonne of concrete products (Saade et al., 2015; Hossain et al., 2016b), and (iii) environmental impacts of concrete products based on its functional performance (e.g., compressive strength) (Damineli et al., 2010; Panesar et al., 2017; Visintin et al., 2020). Based on the mixtures shown in Table 1, the quantity of materials was calculated for the production of 1 tonne of products, and then LCA was conducted accordingly. It should be noted that all of the S/S mixtures accomplished the standard technical requirements, thus, the mechanical performance was also considered as functional unit. This would reduce the complexity of volume as functional unit, because the involved quantity of materials were different under different scenarios and strategies. A cradle-to-gate system boundary was adopted, which included the production, processing, collection and transportation of necessary materials to the production sites, followed by the production process of the studied products (Fig. 1).

2.2.2. *Life cycle inventory analysis and impact assessment*

As shown in Table 1, a number of materials along their processing are needed for the comprehensive analysis of all scenarios. Considering the resource-scarce city, Hong Kong mostly depends on imported construction materials from different regions or countries. The imported materials along with the respective transport distances and modes are shown in Table 2, and the energy consumption for the processing and production of the materials, and the manufacturing of products (considered strategies in this study) are shown in Table 3. For all strategies, the transport distances were calculated from materials generation sites to the block manufacturing site in Hong Kong.

Locally produced OPC and PFA were considered in this study though more than half of the OPC and PFA was imported in Hong Kong, whereas GGBS sourced from Mainland China was considered, as >65% of the GGBS were imported from Mainland China. Almost the entire amount of aggregates used in Hong Kong were sourced from Mainland China in 2017 (Hossain et al., 2019). The life cycle inventory data for aggregates production was based on Hossain et al. (2016a). The data for cement production in Hong Kong were referenced to Hossain et al. (2017) and the Chinese Life Cycle Database (CLCD, 2010a). In this study, economic allocation was adopted for upstream impacts of the commonly used industrial by-products as supplementary cementitious materials (SCMs), such as GGBS, PFA and silica fume (SF). The life cycle inventory for the economic allocation with the case-specific SCMs in Hong Kong conducted by Hossain et al. (2018) was adopted in this study. The case-specific life cycle inventory data for recycled aggregates production from construction and demolition (C&D) waste and waste glass in Hong Kong was collected from Hossain et al. (2016a), whereas the data for the glass powder production from the waste glass bottles was based on the case-specific local study conducted by Hossain et al. (2017). Considering both of waste materials disposals of at landfills or public fills, the avoided impacts were taken into account in this study.

The upstream data for electricity and fuel consumption for most of the processes and transportation were based on the local and regional databases and references, such as CLCD and the China Light and Power (CLP) in Hong Kong. Due to the technological, temporal and spatial representation, the local and regional data/databases provide more reliable results in the LCA study (Hossain and Ng, 2020). In Hong Kong, some of the data related to materials and processes was not available in CLCD or even in case specific studies. Therefore, the European Reference Life Cycle Database (ELCD), Ecoinvent and other scientific literature

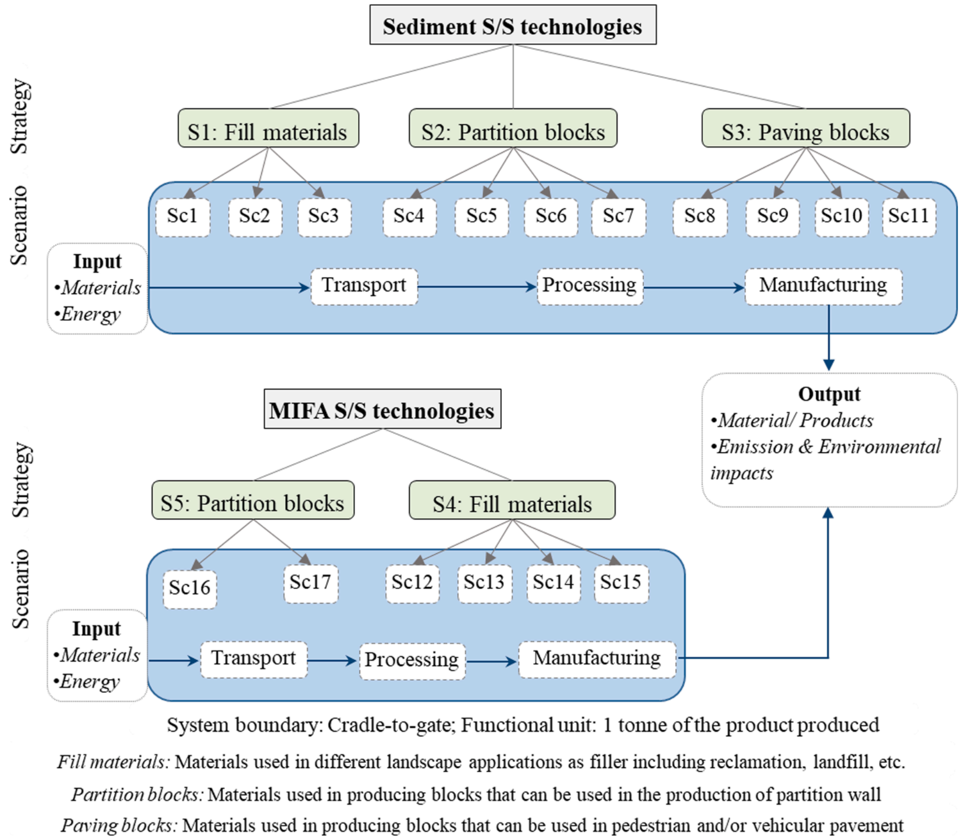

Fig. 1. System boundary of the studied S/S technologies.

were also used.

The upstream data for reactive magnesia cement were collected from Ruan and Unluer (2016). The transport distance of reactive magnesia cement from Liaoning province to the manufacturing site were calculated (about 2750 km). Similarly, the transport distances for other materials such as MIFA, calcium aluminate cement, and trisodium phosphate were calculated (Table 2), whereas incinerated sewage sludge ash from the local sewage sludge incineration plant in Hong Kong to the manufacturing sites was modelled in this study. The upstream data for the production of calcium aluminate cement and trisodium phosphate production were collected from the IBU (2015) and Ecoinvent (2016b), respectively. Contaminated sediment was collected from the top 0.5 m of the Victoria Harbor in Hong Kong as dredging sludge, and several processes including oven-dry, crush and sieve (2.36 mm) and then and re-saturate to a moisture content of 60% (dry basis) for optimal water-binder ratio (Wang et al., 2015a). The avoided impacts due to the transportation and landfill disposal for incinerated sewage sludge ash and MIFA were included with Ecoinvent database (treatment of average incineration residue, residual material landfill) (Ecoinvent, 2016c). Finally, the energy consumption for the block production process was collected from the Hossain et al. (2016b). All materials including their processes and transportation in each scenario were modeled using SimaPro 9.1.5 software, and the three environmental impact indicators (mid-point) such as acidification potential as SO_2 eq, global warming potential as CO_2 eq, and non-renewable energy consumption as MJ were comparatively evaluated by the IMPACT 2002 + impact method (Jolliet et al., 2003). The results were then comprehensively analyzed to select the best options from the available S/S technologies for the studied hazardous wastes.

2.3. Assumptions

For conducting comprehensive LCA study for such materials under different scenarios, some assumptions are inevitable. Some of them are highlighted as:

- For most of the cases, the shortest transport distance was considered in this study. For example, Guangdong was taken as a sourced location if imported from Mainland China.
- No impacts were considered for both ISSA and MIFA generation (rather than associated with transportation), as it is assumed that these materials should be produced whether these can be valorized or not. Moreover, this applies to all scenarios and does not affect the comparative analysis.
- Due to unavailability of industrial processing data for contaminated sediment, the impacts for the sediment dredging/processing were not considered in this study. This material should go through a series of treatment (biological and chemical) before disposal if not recycled. Thus, it is assumed that the energy consumption for both processing options (recycling and disposal) would be balanced. Similarly, this applies to all scenarios and does not affect the comparative analysis.
- For trisodium phosphate, the data for the sodium phosphate production were used due to unavailability of trisodium phosphate data. As the main ingredients and/or the major production processes are accountable for most of the impacts of a product, thus, such assumption can be adopted in LCA study to fill the data gap (Huijbregts et al., 2001).
- The production process for the fill materials (strategies 1 and 4) was not considered due to the unavailability of data. Similarly, it does not affect the comparative analysis as it applies to all scenarios.
- The use phase and end-of-life phase of the designed products were not considered in this study due to model complexity for such 17 scenarios considering the unavailability of data. Similar phenomena are assumed for all scenarios in each strategy.

Table 2
Raw materials including the associated transport distances used in this study.

Materials	Source locations	Distance (km) and transport type	Sources of data	Upstream data/ database
Ordinary Portland cement (OPC)	Local (average distance from local cement manufacturer to manufacturing site)	28 km by 30 t trucks	Estimated	CLCD (2010b)
Crushed stone (both fine and coarse aggregates)	Dongguan in Guangdong Province (China) to Hong Kong Port	50 km by 30 t trucks, and 128 km by inland barge	Hossain et al., (2016a)	CLCD (2010b); CLCD (2010c)
	To manufacturing sites (averaged)	30 km by 30 t trucks		CLCD (2010b)
Recycled fine concrete aggregate (from C&D waste)	Production and block manufacturing site (variables; averaged)	45 km by 30 t trucks	Hossain et al., (2016a)	CLCD (2010b)
	Generation to fill sites (variables; averaged)	35 km by 30 t trucks		CLCD (2010b)
Recycled glass aggregate (from waste glass bottle)	Waste glass bottles (locally generated) to aggregates production site	50 km by 18 t trucks	Hossain et al., (2016a)	CLCD (2010b)
	Generation to fill sites (variables; averaged)	6 km by 6 t truck, and 35 km by 30 t tucks		CLCD (2010b)
Glass powder from waste glass bottles	Waste glass bottles (locally generated) to glass powder production and manufacturing site	55 km by 18 t trucks	Hossain et al., (2017)	CLCD (2010b)
	Waste glass bottles (locally generated) to landfill site	6 km by 6 t truck, and 35 km by 30 t tucks		CLCD (2010b)
Fly ash (FA)	Local coal-fired power plant to manufacturing site (averaged)	28 km by 30 t trucks	Hossain et al. (2018)	CLCD (2010b)
Ground granulated blast-furnace slag (GGBS)	Guangdong to Hong Kong Port	128 km by inland barge	Hossain et al. (2018)	CLCD (2010c)
	To manufacturing site (averaged)	30 km by 30 t trucks		CLCD (2010b)
Silica fume (SF)	Guangdong to Hong Kong Port	128 km by inland barge	Hossain et al. (2018)	CLCD (2010c)
	To manufacturing site (averaged)	30 km by 30 t trucks		CLCD (2010b)
Reactive MgO cement (MC)	From Liaoning province to Guangdong and then to manufacturing site	2750 km by rail freight and 180 km 30 t trucks	Estimated	Ecoinvent (2016a); CLCD (2010b)
Incinerated sewage sludge ash (ISSA)	From T-Park (Sewage sludge inclination plant) to manufacturing site	10 km by 15 t trucks	Estimated	CLCD (2010b)
	From T-Park (Sewage sludge inclination plant) to landfill site	5 km by 15 t trucks	Estimated	CLCD (2010b)
Contaminated sediment	Kai Tak Approach Channel to manufacturing site	42 km by 30 t trucks	Estimated	CLCD (2010b)
			Estimated	
MSWI fly ash (MIFA)	Guangdong to Hong Kong Port	128 km by inland barge		CLCD (2010c)
	To manufacturing site (averaged)	30 km by 30 t trucks	Estimated	CLCD (2010b)
Calcium aluminate cement (CAC)	Guangdong to Hong Kong (manufacturing site)	180 km 30 t trucks	Estimated	CLCD (2010c)
Trisodium phosphate (TSP)	Guangdong to Hong Kong (manufacturing site)	180 km 30 t trucks	Estimated	CLCD (2010c)

Table 3
Sources of energy for different materials/processes.

Materials/processes	Energy consumption	Upstream data/ databases
Natural aggregates (crushed stone)	6.07 kWh/t (electricity) & 1.37 L/t (diesel) [a]	CLCD (2010d, 2010e)
Recycled fine aggregates (C&D waste)	3.06 kWh/t (electricity) & 1.89 L/t (diesel) [a]	CLCD (2010d, 2010e)
Recycled coarse aggregates (C&D waste)	1.11 kWh/t (electricity) & 1.89 L/t (diesel) [a]	CLCD (2010d, 2010e)
Cement production	R*	Hossain et al. (2017); CLCD (2010a)
PFA	9.3 kWh/t [b]	CLP (2014); CLCD (2010d)
GGBS	72.15 kWh/t [c]	CLP (2014); CLCD (2010d)
SF	10 kWh/t [b]	
Glass powder	25 kWh/t glass powder [d]	CLP (2014); CLCD (2010d)
Glass/C&D waste landfill	R*	ELCD (2013)
ISSA and MIFA disposal into landfill	R*	Ecoinvent (2016c)
Reactive magnesia cement	R*	Ruan and Unluer (2016)
Calcium aluminate cement	R*	IBU (2015)
Trisodium phosphate	R*	Ecoinvent (2016b)
Block production process	30.28 kWh/t blocks (electricity) [e]	CLP (2014); CLCD (2010d)

* Referred to the database/references in the right column.
[a] Hossain et al. (2016a).
[b] MPA (2009).
[c] Dunlap (2003).
[d] Hossain et al. (2017).;
[e] Hossain et al. (2016b).

3. Results and discussion

Based on the considered system boundary and assumptions, the selected environmental impact indicators for managing contaminated sediment through the studied S/S technologies are shown in Table 4. For utilizing the sediment as fill materials (Strategy 1), Scenario 1 (e.g. mixture with recycled fine aggregates and a small percentage of OPC and ISSA) would be the preferred option. Because Sc1 is associated with about 54% and 70% lower CO_2 eq emission global warming potential (GWP) compared to that of Sc2 and Sc3, respectively. From the comparative analysis presented in Fig. 2, it can be seen that Sc2 has higher non-renewable energy (NRE) consumption compared to Sc3 (about 11%), but Sc1 is associated with 40–46% lower NRE consumption compared to Sc2 and Sc3 (Fig. 2). Similarly, Sc2 is associated with higher acidification potential (AP) compared to Sc1 and Sc3. For the per

Table 4
Selected impact indicators for the sediment S/S technologies.

Strategy	Scenario	Selected impact categories		
		Global warming potential (kg CO_2 eq/t)	Non-renewable energy consumption (MJ eq/t)	Acidification potential (kg SO_2 eq/t)
S1: Fill materials	Sc1	15.90	211	0.82
	Sc2	34.50	392	1.18
	Sc3	53.70	354	0.92
S2: Partition block production	Sc4	176.57	1372	3.12
	Sc5	178.05	1524	3.38
	Sc6	179.26	1398	3.43
	Sc7	196.21	1381	2.89
S3: Paving blocks production	Sc8	340.63	2369	5.13
	Sc9	300.97	2251	4.84
	Sc10	270.27	2105	5.08
	Sc11	305.93	2179	4.74

tonne production of fill materials containing contaminated sediment, about 0.82 kg SO_2 eq AP is associated with Sc1, whereas it is 1.18 kg SO_2 eq for Sc2 and 0.92 kg SO_2 eq for Sc3, indicating that about 31% and 11% lower AP is observed for Sc1 than Sc2 and Sc3, respectively (Fig. 2). Although 90% sediment is used in Sc3 along with 5% ISSA, the impacts are higher because of the high dosage of reactive MgO cement. The results (Sc2) are also consistent with Barjoveanu et al. (2018), where higher impacts were associated due to the use of OPC.

This study considered four scenarios for sediment S/S technologies under Strategy 2 (reutilization through partition block production), where 84–90% of the total materials were recycled materials. The LCA results demonstrated that compared to Sc7 (partition blocks containing sediment, glass powder and reactive magnesia cement), about 9–10% lower GWP is observed for partition block production through Sc4-Sc6 (Table 4). However, Sc5 is associated with 8–10% higher energy consumption than those of Sc4, Sc6 and Sc7, as the total NRE is 1372, 1524, 1398, and 1381 MJ eq for per tonne of partition block production through Sc4-Sc7, respectively. In the category of AP, Sc6 is associated with 9% and 16% higher than Sc4 and Sc7, but similar to Sc5 (Fig. 2). It is noted that a relatively low percentage of sediment (only 16%) is used in Sc4 and Sc5, whereas this is about 85% in Sc6 and Sc7. Considering the environmental impacts (lower GWP and NRE) and the percentage of sediment used, Sc6 would be the preferred sustainable option among the other scenarios (Sc4, Sc5, and Sc7).

The LCA results for sediment S/S technology for paving blocks production with different scenarios (Strategy 3) are presented in Table 4 and Fig. 2. The results show that about 340, 301, 270 and 306 kg CO_2 eq GWP is associated with Sc8, Sc9, Sc10, and Sc11, respectively, per tonne of paving blocks production. About 12%, 21%, and 10% higher GWP is associated with Sc8 compared to Sc9, Sc10 and Sc11, respectively. Similarly, about 5%, 11% and 8% higher NRE as well as 6%, 1% and 8% higher AP are associated with Sc8 compared to Sc9, Sc10 and Sc11, respectively. For Strategy 3, comparatively higher environmental impacts are observed due to the requirement of higher percentage of natural materials and OPC. As a higher percentage of OPC is used in Sc8 (compared to other scenarios for paving block production), relatively high impacts are observed (Table 4). Considering the corresponding environmental impacts, the use of sediment for the paving block production through Sc10 would be the preferred option among the four

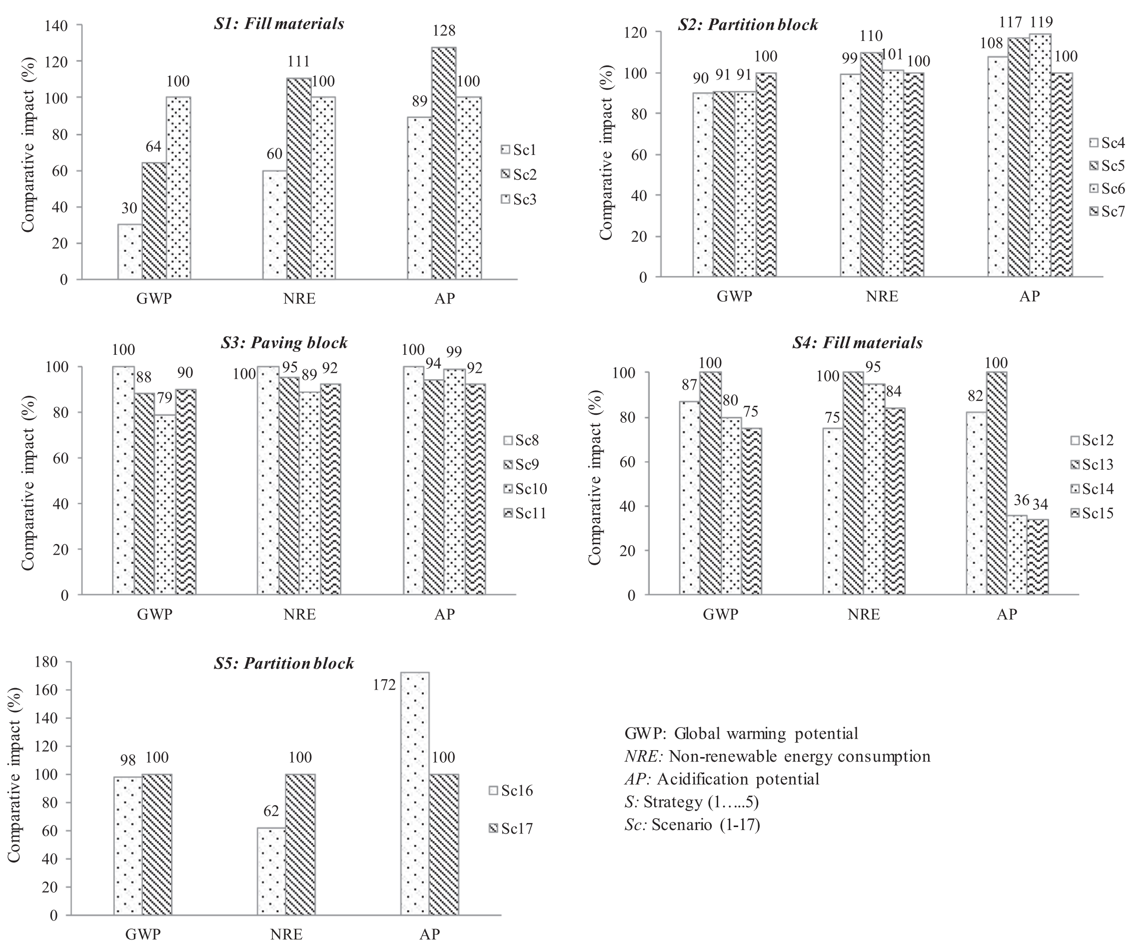

Fig. 2. Comparative impacts for different strategies and scenarios for S/S technologies.

scenarios.

Considering the life cycle inventory data, necessary assumptions and system boundary, the LCA results for MIFA S/S technologies through different strategies with different scenarios are presented in Table 5 and Fig. 2. In the studies scenarios, MIFA consists of >90% of the total materials used. For the per tonne production of fill materials through valorizing MIFA, about 133, 154, 124 and 116 kg CO_2 eq GWP is associated with Sc12, Sc13, Sc14 and Sc15, respectively. The results indicate that Sc13 (90% MIFA, 4% SF and 6% OPC) is associated with 13%, 20% and 25% higher GWP compared to Sc12, Sc14 and Sc15, respectively. The corresponding NRE consumption is 25%, 5% and 16% higher in Sc13. Similarly, Sc13 is associated with 2.56 kg SO_2 eq AP which is 18% higher than that of Sc12 (2.09 kg SO_2 eq). The value is significantly higher than Sc14 (about 64%) and Sc15 (66%), of which the emission is only 0.93 and 0.87 kg SO_2 eq, respectively (Fig. 2). This is mainly due to the required high dosage of OPC, as AP for OPC production is considerably higher than the calcium aluminate cement production (with the life cycle inventory data for OPC and calcium aluminate cement used in this study). Another reason is attributed to the use of SF, and its corresponding upstream emission from the main product production (considering the economic allocation). In view of the overall environmental impacts, Sc15 would be the preferred option for using MIFA as fill materials, as it can reduce 25% GWP, 16% NRE, and 66% AP, respectively, compared to Sc14, and these impacts are considerably lower than those of the other scenarios (Table 5). The results are consistent with Margallo et al. (2019), as the use of OPC incurs high impacts.

The production of partition blocks with MIFA (Strategy 5), the LCA results of two different scenarios are shown in Table 5. In the category of GWP, the emission of CO_2 eq for both scenarios (Sc16 and Sc17) is almost similar, as the emission is 232 kg CO_2 eq for Sc16 and 238 kg CO_2 eq for Sc17 (only 2% higher in Sc17 compared to Sc16), although higher amount of OPC is used in Sc16. The results also show that total NRE consumption is considerably higher in Sc17, as about 2.68 GJ of NRE is needed for the production of one tonne of partition block for Sc17, which is 38% lower in Sc16 (1.67 GJ/t). However, about 42% higher AP is associated with Sc16 than that of Sc17, as the SO_2 eq emission is 3.38 kg/t for Sc16 compared to 1.96 kg SO_2 eq/t for Sc17 (Table 5), due to significantly higher SO_2 eq emission for OPC production than the calcium aluminate cement production. Comparing the two scenarios, Sc17 would be the preferred option for partition block production utilizing MIFA. The results are also consistent with other studies (Huber et al. 2018), particularly for higher impacts of OPC-based MIFA stabilization (Sc12 and Sc13). Thus, the reduced consumption of OPC and the increased use of alternative cement (e.g., calcium aluminate cement) would give better LCA results (Sc14 and Sc15).

As an example, the contribution of total CO_2 eq emission for different materials and processes is given in Fig. 3. For Sc1, OPC is responsible for about 63% of the total emission, whereas 17% is for sediment collection and 20% is for recycled fine aggregates. The avoided emission for ISSA

Table 5
Selected impact indicators for the MIFA S/S technologies.

Strategy	Scenario	Selected impact categories		
		Global warming potential (kg CO_2 eq/t)	Non-renewable energy consumption (MJ eq/t)	Acidification potential (kg SO_2 eq/t)
S4: Fill materials	Sc12	133.19	1092	2.09
	Sc13	153.85	1450	2.56
	Sc14	123.74	1381	0.93
	Sc15	115.57	1218	0.87
S5: Partition block production	Sc16	232.18	1666	3.38
	Sc17	238.00	2680	1.96

(transport and landfilling) is higher than its transport to the valorization site, and thus about 11.72 kg CO_2 eq emission is subtracted from the total emission for Sc1 (Fig. 2). Compared to Sc1, considerably higher emission is associated with Sc2 (Table 4), because GGBS induces about 14 kg eq CO_2 emission (about 41%) to the total emission (considering the upstream impacts for GGBS due to economic allocation), whereas 29%, 13% and 17% are from OPC, sediment and recycled fine aggregate, respectively. For Sc3, OPC is responsible for about 88% of the total emission, whereas 12% is from sediment. Even though 17.78 kg CO_2 eq is avoided due to ISSA in Sc3, the total emission is still significantly higher than the values in the other two scenarios. This is mainly due to the use of higher percentage of reactive magnesia cement, as environmental impacts are considerable due to high CO_2 emission during reactive magnesia cement production (Ruan and Unluer, 2016). Although reactive magnesia cement has very high carbon capture potential (CO_2 uptake by the product during its use phase), it is not considered in this study (as cradle-to-gate system boundary).

For partition blocks production through Sc4, >85% of the total emission is attributed to OPC, 1% by sediment, 1% by recycled fine aggregate, 3% by recycled coarse aggregate, and 10% by the production process (Fig. 3). The emission is similar to Sc5 due to the high use of OPC, although some are replaced by the use of ISSA in Sc4. About 68%, 16%, 1%, 1%, 3% and 11% of the total emission is contributed by OPC, GGBS, sediment, recycled fine aggregate, recycled coarse aggregate, and production process, respectively. Even though lower amount of reactive magnesia cement is used in Sc6, its contribution is relatively high (70%), whereas 15% is from PFA (higher due economic allocation), 4% is from sediment, and 11% is from the production process. For Sc7, reactive magnesia cement is responsible for 86% of the total emission, although glass powder contributes to negligible emission (due to avoided impacts for waste glass landfilling).

For Sc8, the total GHGs emission is considerably higher than the other scenarios for paving blocks production due to the use of higher amount of OPC (contribute about 90% of the total emission). For Sc9, the contribution is 81%, 7%, 1%, 5% and 6% by OPC, GGBS, sediment and recycled fine aggregate, natural coarse aggregate and production process, respectively. Similarly, about 75%, 10%, 1%, 7% and 7% of the total emission was contributed by the OPC, PFA, sediment, natural coarse aggregate and production process, respectively. Due to the use of 5% reactive magnesia cement in Sc11, total carbon emission is considerably higher in Sc11 (even though lower amount of OPC is used) compared to the other scenarios (e.g., Sc 9 and Sc10). The total contribution is 66%, 20%, 1%, 6% and 6% by OPC, reactive magnesia cement, sediment, natural coarse aggregate and production process, respectively. The contribution analysis shows that the use of SCMs is often the better option to reduce the carbon emission compared to use of reactive magnesia cement.

In Sc12, about 61%, 31% and 8% of the total emission are attributed to OPC, SF and MIFA, respectively. The use of SF induces considerably higher emission due to the upstream impacts (considered economic allocation). Consequently, about 25% higher carbon emission is observed even with 35% lower binder is used in S13 (Fig. 3 and Table 1). The total contributions are 40%, 53% and 7% by OPC, SF and MIFA, respectively, for Sc13. The use of calcium aluminate cement as a substitute of OPC and SF significantly reduces the carbon emission (Table 5). Calcium aluminate cement and MIFA contribute to 91% and 9% of the total emission for Sc14, whereas it is 61%, 30% and 9% by calcium aluminate cement, trisodium phosphate and MIFA, respectively, for Sc15. Although trisodium phosphate has a higher emission, the use of less amount of calcium aluminate cement helps to decrease the total emission for Sc15 (Fig. 3). For Sc16, about 88%, 4% and 8% of the total emission are attributed to OPC, MIFA and production process, respectively. Although alternative binder is used in C17, the total emission remains largely similar. The contributions are 66%, 22%, 4% and 8% by calcium aluminate cement, trisodium phosphate, MIFA and production process, respectively (Fig. 3).

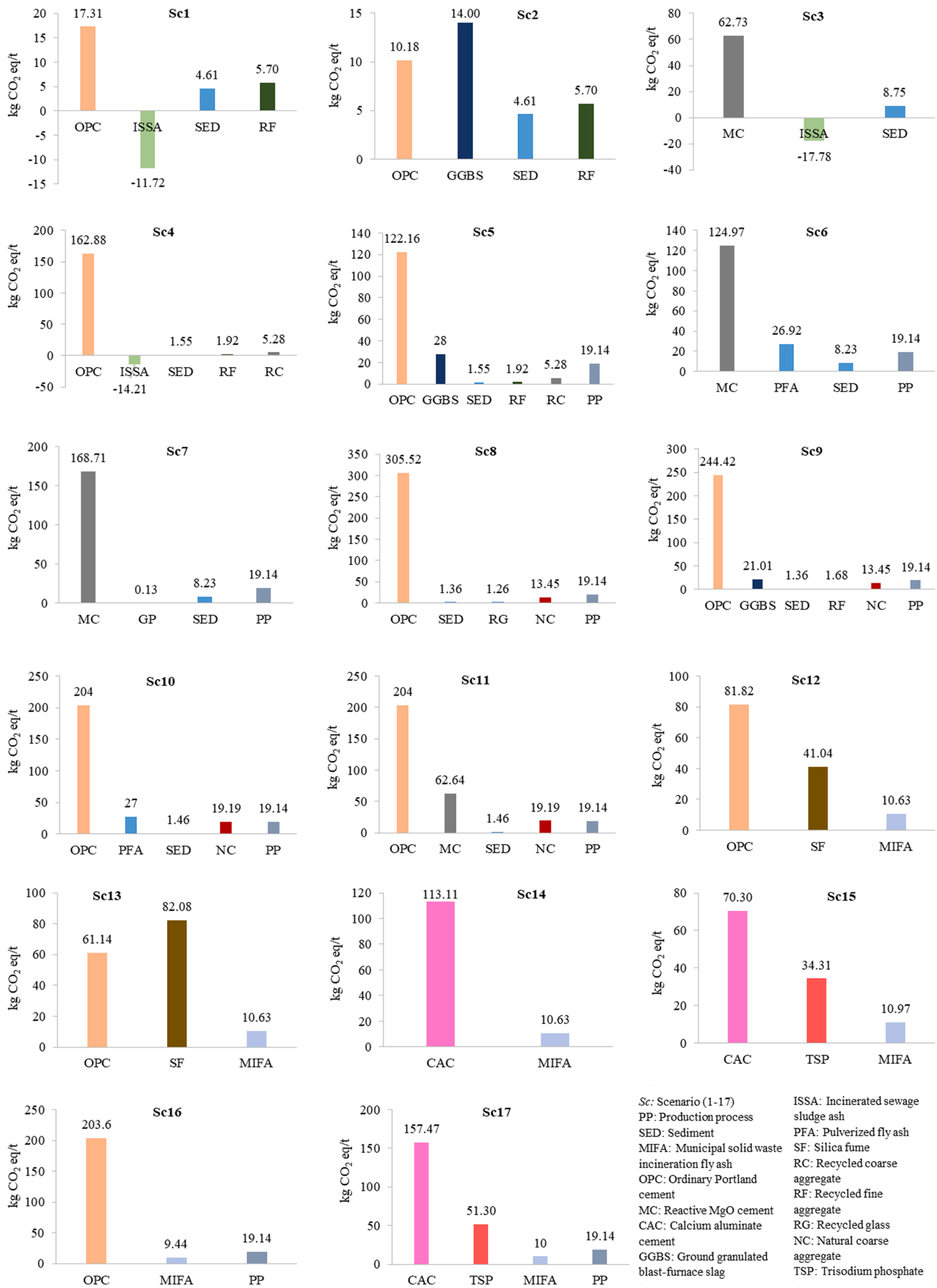

Fig. 3. Contribution analysis for different S/S technologies of the studied materials.

In view of the similar ratio of sediment and MIFA used in Sc6, Sc7, Sc16 and Sc17 for partition blocks production, the comparison of the selected impact indicators is given in Fig. 4. It can be seen that comparatively higher GWP (18–25%) is observed for Sc16 and Sc17 than Sc6 and Sc7, due to longer transport distance of MIFA materials (compared to sediment) and the use of different binders (particularly for OPC in Sc16). Similarly, about 16–48% higher NRE is found for Sc16 and Sc17 compared to Sc7 and Sc8 due to higher amount of OPC (Sc16) and higher NRE associated with trisodium phosphate (Sc17). However, Sc17 has 43% and 32% lower AP than the Sc6 and Sc7, because lower SO_2 eq is associated with calcium aluminate cement compared to reactive magnesia cement (based on the collected life cycle inventory data).

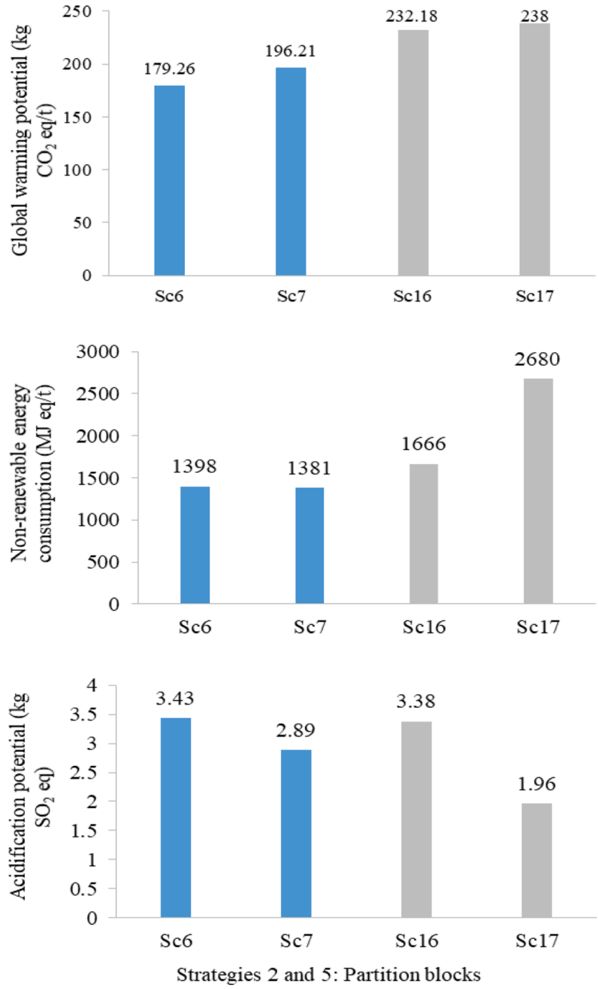

Fig. 4. Comparison of strategies for S/S technologies of the studied materials.

In this study, there are some unavoidable limitations that should be addressed in the future. Currently, the processing of sediment is not considered due to the unavailability of energy data, and MIFA is not locally generated in Hong Kong but collected from the neighbouring city. The production process for the fill materials is also not considered in this study due to the unavailability of data. This study considered cradle-to-gate system boundary for scenarios, because the required data were unavailable for the use phase and end-of-life of the designed products. It would be practical to consider these aspects, for example, the difference in carbon uptake by reactive magnesia cement and OPC during the use phase, and the potential ecological and health impacts due to contaminant leaching at the end-of-life phase (Barjoveanu et al., 2018). It should be noted that the results of LCA studies are heavily dependent on the specific location, technology considered, transportation of different materials, etc. Thus, this LCA study is more applicable to the selection of specific technology for the studied hazardous wastes based on their environmental performance in Hong Kong. Future studies could evaluate the specific S/S technologies for managing different hazardous wastes in various locations or regions for representativeness. In addition, LCA results are often sensitive to different considerations and factors including the upstream data, transportation distance, etc. Therefore, sensitivity analysis of the key factors would be helpful for decision-makers to understand the reliability and possible deviation of the final recommendations.

Moreover, the current environmental legislations are available for managing and disposing of dredged sediment and MIFA in Hong Kong (Wang et al., 2015b), and it is possible to use S/S-treated sediment as fill materials (S1) (HK EPD, 2020). However, there are currently no guidelines and regulations on how to use such materials as partition blocks (S2) and paving blocks (S3). Similarly, there is a lack of regulation for the potential recycling of MIFA as construction materials in Hong Kong. Thus, the relevant environmental regulations and material specifications for treating and recycling these waste materials under different strategies shall be developed and enacted in the near future for promoting sustainable development and circular economy.

4. Conclusions

This study evaluated the environmental sustainability of S/S technologies for contaminated marine sediment and MIFA for resourceful utilization through 17 scenarios under five strategies. Key conclusions can be drawn on the basis of collected life cycle inventory data, system boundary considered, assumptions made, and the selected impact indicators. In Strategy 1 (recycling sediment as fill materials), Sc1 would be the preferred option, as it is associated with about 54–70%, 40–46%, and 11–31% lower GWP, NRE and AP impacts compared to other options. In Strategy 2 (recycling sediment for partition block production), considering the lower GWP and NRE impacts (despite 1–16% high AP) and the high utilization percentage of sediment (85%), Sc6 would be the preferred option over Sc4, Sc5 and Sc7. In Strategy 3 (recycling sediment for paving block production), Sc10 would be the preferred option among the four scenarios, for its 12–21% and 3–11% lower GWP and NRE impacts than other options. Strategy 4 (recycling MIFA as fill materials), Sc15 would be the preferred option for as it uses 93% MIFA and reduces 7–25% GWP, 12–16% NRE, and 6–66% AP compared to Sc12-Sc14. In Strategy 5 (recycling MIFA for partition block production), Sc17 would be the preferred option for its 42% lower AP compared to Sc16. The overall results indicate that the use of calcium aluminate cement and SCMs as binders instead of OPC and reactive magnesia cement can significantly reduce the environmental impacts. These results can facilitate the selection of sustainable technologies for S/S treatment of hazardous wastes, while the identified limitations need to be addressed by future studies.

CRediT authorship contribution statement

Md. Uzzal Hossain: Conceptualization, Methodology, Data curation, Investigation, Writing - original draft. **Lei Wang:** Conceptualization, Methodology, Validation, Funding acquisition, Writing - original draft. **Liang Chen:** Methodology, Data curation, Investigation. **Daniel C.W. Tsang:** Conceptualization, Supervision, Project administration, Funding acquisition, Resources, Writing - review & editing. **S. Thomas Ng:** Resources, Supervision, Validation, Writing - review & editing. **Chi Sun Poon:** Resources, Validation, Writing - review & editing. **Viktor Mechtcherine:** Resources, Funding acquisition, Writing - review & editing.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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References

Barjoveanu, G., De Gisi, S., Casale, R., Todaro, F., Notarnicola, M., Teodosiu, C., 2018. A life cycle assessment study on the stabilization/solidification treatment processes for contaminated marine sediments. J. Clean. Prod. 201, 391–402.

Bates, M.E., Fox-Lent, C., Seymour, L., Wender, B.A., Linkov, I., 2015. Life cycle assessment for dredged sediment placement strategies. Sci. Total Environ. 511, 309–318.

Billen, P., Verbinnen, B., De Smet, M., Dockx, G., Ronsse, S., Villani, K., De Greef, J., Van Caneghem, J., Vandecasteele, C., 2015. Comparison of solidification/stabilization of fly ash and air pollution control residues from municipal solid waste incinerators with and without cement addition. J. Mater. Cycles Waste Manag. 17, 229–236.

Bogush, A., Stegemann, J.A., Wood, I., Roy, A., 2015. Element composition and mineralogical characterization of air pollution control residue from UK energy-from-waste facilities. Waste Manag. 36, 119–129.

BS EN 6073, 1981. Precast Concrete Masonry Units. Specification for Precast Concrete Masonry Units. British Standards Institution, London, United Kingdom.

Chen, L., Wang, L., Cho, D.W., Tsang, D.C.W., Tong, L.Z., Zhou, Y.Y., Yang, J., Hu, Q., Poon, C.S., 2019. Sustainable stabilisation/solidification of municipal solid waste incinerator fly ash by incorporation of green materials. J. Clean. Prod. 222, 335–343.

Chen, L., Wang, Y.S., Wang, L., Guo, B., Tong, L., Tsang, D.C.W., Wu, Z., Dai, J.G., Hu, Q., 2020. Stabilisation/solidification of municipal solid waste incinerator fly ash by chemical enhanced calcium aluminate cement. J. Hazard. Mater. under review.

CLCD, 2010a. Cement production (dry method)–CN–AP. Chinese Life Cycle Database Version 0.8. Sichuan University; IKE Environmental Technology CO. Ltd.

CLCD, 2010b. Heavy-duty diesel truck transportation (6t, 18t, 30t)–CN–AP. Chinese Life Cycle Database Version 0.8, Sichuan University; IKE Environmental Technology CO. Ltd.

CLCD, 2010c. Bulk cargo shipping (2500 t)–CN–AP. Chinese Life Cycle Database Version 0.8. Sichuan University; IKE Environmental Technology CO. Ltd.

CLCD, 2010d. Electricity generation, China Southern Grid. Chinese Life Cycle Database Version 0.8. Sichuan University; IKE Environmental Technology Co., Ltd.

CLCD, 2010e. Fuel combustion-Diesel–CN–AP. Chinese Life Cycle Database Version 0.8. Sichuan University; IKE Environmental Technology CO. Ltd.

CLP, 2014. Sustainability Report 2014, the China Light and Power (CLP), Hong Kong, 2014.

Damineli, B.L., Kemeid, F.M., Aguiar, P.S., John, V.M., 2010. Measuring the eco-efficiency of cement use. Cem. Conc. Com. 32, 555–562.

De Gisi, S., Todaro, F., Mesto, E., Schingaro, E., Notarnicola, M., 2020. Recycling contaminated marine sediments as filling materials by pilot scale stabilization/solidification with lime, organoclay and activated carbon. J. Clean. Prod. 269, 122416.

Dunlap, R., 2003. Life cycle inventory of slag cement manufacturing process: project CTL. No. 312012. Illinois: Construction Technology Laboratories, 2003, 13p.

Ecoinvent, 2016a. Transport, freight train, diesel CN, Alloc Def, U. Swiss Centre for Life Cycle Inventories.

Ecoinvent, 2016b. Sodium phosphate production, RoW, Conseq, U. Swiss Centre for Life Cycle Inventories.

Ecoinvent, 2016c. Treatment of average incineration residue, residual material landfill (RoW), Alloc Def, U. Swiss Centre for Life Cycle Inventories.

ELCD, 2013. Landfill of glass/inert waste EU-27. ELCD (European reference Life Cycle Database), Joint Research Centre (JRC) of the European Commission, ELCDDATA 13292600296.

Engbert, A., Gruber, S., Plank, J., 2020. The effect of alginates on the hydration of calcium aluminate cement. Carbohydr. Polym. 236, 116038.

Falciglia, P.P., Ingrao, C., De Guidi, G., Catalfo, A., Finocchiaro, G., Farina, M., Liali, M., Lorenzano, G., Valastro, G., Vagliasindi, F.G.A., 2018. Environmental life cycle assessment of marine sediment decontamination by citric acid enhanced-microwave heating. Sci. Total Environ. 619–620, 72–82.

Ganesh, P., Murthy, A.R., 2019. Tensile behaviour and durability aspects of sustainable ultra-high performance concrete incorporated with GGBS as cementitious material. Constr. Build. Mater. 197, 667–680.

Garg, N., White, C.E., 2017. Mechanism of zinc oxide retardation in alkali-activated materials: an in situ X-ray pair distribution function investigation. J. Mater. Chem. A 5, 11794.

HK CEDD, 2016. Dredging, management and capping of contaminated sediment disposal facility to the south of the brothers. Civil Engineering and Development Department, Hong Kong SAR Government.

HK EPD, 2011. Practice Guide for Investigation and Remediation of Contaminated Land. Environmental Protection Department, Hong Kong SAR Government, Hong Kong.

HK EPD, 2017. Marine Water Quality in Hong Kong in 2016. Environmental Protection Department, Hong Kong SAR Government.

HK EPD, 2018. Integrated Waste Management Facilities Environmental Protection Department, Hong Kong SAR Government.

HK EPD, 2020. Waste management implication and land contamination, Hong Kong SAR Government. https://www.epd.gov.hk/eia/register/report/eiareport/eia_2282015/HTML_June%202015/EIA/Ch%206%20Waste%20%20Land%20Contamination.htm (accessed on 15.07.2020).

HK ETWB, 2004. Specification Facilitating the Use of Concrete Paving Units Made of Recycled Aggregates. Technical Circular (Works) No. 24/2004. Environment, Transport and Works Bureau, Hong Kong SAR Government, Hong Kong.

Hossain, M.U., Ng, S.T., 2020. Strategies for enhancing the accuracy of evaluation and sustainability performance of building. J. Environ. Manage. 261, 110230.

Hossain, M.U., Poon, C.S., Dong, Y.H., Xuan, D., 2018. Environmental impact distribution methods for supplementary cementitious materials. Renew. Sustain. Energy Rev. 82, 597–608.

Hossain, M.U., Poon, C.S., Lo, I.M.C., Cheng, J.C.P., 2016a. Comparative environmental evaluation of aggregate production from recycled waste materials and virgin sources by LCA. Resour. Conserv. Recyc. 109, 67–77.

Hossain, M.U., Poon, C.S., Lo, I.M.C., Cheng, J.C.P., 2016b. Evaluation of environmental friendliness of concrete paving eco-blocks using LCA approach. Intl. J. Life Cycle Assess. 21 (1), 70–84.

Hossain, M.U., Poon, C.S., Lo, I.M.C., Cheng, J.C.P., 2017. Sustainable strategies for the reduction of environmental impacts in the cement industry. Resour. Conser. Recyc. 120, 199–208.

Hossain, M.U., Sohail, A., Ng, T.S.T., 2019. Developing a GHG-based methodological approach to support the sourcing of sustainable construction materials and products. Resour. Conserv. Recyc. 145, 160–169.

Huang, T.Y., Chuieh, P.T., 2015. Life cycle assessment of reusing fly ash from municipal solid waste incineration. Proc. Eng. 118, 984–991.

Huber, F., Fellner, J., 2018. Integration of life cycle assessment with monetary valuation for resource classification: the case of municipal solid waste incineration fly ash. Resour. Conserv. Recyc. 139, 17–26.

Huber, F., Laner, D., Fellner, J., 2018. Comparative life cycle assessment of MSWI fly ash treatment and disposal. Waste Manage. 73, 392–403.

Huijbregts, M.A.J., Norris, G., Bretz, R., Ciroth, A., Maurice, B., Bahr, B.V., Weidema, B., Beaufort, A.S.H., 2001. Framework for modelling data uncertainty in life cycle inventories. Int. J. Life Cycle Assess. 6 (3), 127–132.

IBU, 2015. Environmenntal Product Declaration- Calcium aluminate cement. Institut Bauen und Umwelt (IBU), Germany.

ISO, 2006a. ISO 14040 International Standard. Environmental management–Life Cycle Assessment- Principles and Framework. International Organisation for Standardization, Geneva, Switzerland.

ISO, 2006b. ISO 14044 International Standard. Environmental Management–Life Cycle Assessment- Requirements and Guidelines. International Organisation for Standardization, Geneva, Switzerland.

Jia, S., Richardson, I.G., 2018. Micro- and nano-structural evolutions in white Portland cement/pulverized fuel ash cement pastes due to deionized-water leaching. Cem. Concr. Res. 103, 191–203.

Jin, F., Al-Tabbaa, A., 2014. Evaluation of novel reactive MgO activated slag binder for the immobilisation of lead and zinc. Chemosphere 117, 285–294.

Jolliet, O., Margni, M., Charles, R., Humbert, S., Payet, J., Rebitzer, G., 2003. IMPACT 2002+: a new life cycle impact assessment methodology. Int. J. Life Cycle Assess. 8 (6), 324–330.

Margallo, M., Cobo, S., Laso, J., Fernandez, A., Munoz, E., Santos, E., Aldaco, R., Irabien, A., 2019. Environmental performance of alternatives to treat fly ash from a waste to energy plant. J. Clean. Prod. 231, 1016–1026.

MPA, 2009. The concrete industry sustainability performance report 1st Report, 2009. Mineral Products Association (MPA), The Concrete Centre, pp. 24.

Nowacka, M., Pacewska, B., 2020. Effect of structurally different aluminosilicates on early-age hydration of calcium aluminate cement depending on temperature. Constr. Build. Mater. 235, 117404.

Panesar, D.K., Seto, K.E., Churchill, C.J., 2017. Impact of the selection of functional unit on the life cycle assessment of green concrete. Int. J. Life Cycle Assess. 22, 1969–1986.

Pei, S.L., Chen, T.L., Pan, S.Y., Yang, Y.L., Sun, Z.H., Li, Y.J., 2020. Addressing environmental sustainability of plasma vitrification technology for stabilization of municipal solid waste incineration fly ash. J. Hazard. Mater. 122959 https://doi.org/10.1016/j.jhazmat.2020.122959.

Roy, A., Stegemann, J.A., 2017. Nickel speciation in cement-stabilized/solidified metal treatment filtercakes. J. Hazard. Mater. 321, 353–361.

Ruan, S., Unluer, C., 2016. Comparative life cycle assessment of reactive MgO and Portland cement production. J. Clean. Prod. 137, 258–273.

Senneca, O., Cortese, L., Martino, R.D., Fabbricino, M., Ferraro, A., Race, M., Scopino, A., 2020. Mechanisms affecting the delayed efficiency of cement-based stabilization/solidification processes. J. Clean. Prod. 261, 121230.

Saade, M.R.M., da Silva, M.G., Gomesa, V., 2015. Appropriateness of environmental impact distribution methods to model blast furnace slag recycling in cement making. Resour. Conserv. Recyc. 99, 40–47.

Sharaam, M., Negm, A., 2017. Life cycle assessment of dredged materials placement strategies: case study, Damietta Port. Egypt. Proc. Eng. 181, 102–108.

Shi, C., Fernández-Jiménez, A., 2006. Stabilization/solidification of hazardous and radioactive wastes with alkali-activated cements. J. Hazard. Mater. 137, 1656–1663.

Snellings, R., Horckmansa, L., Van Bunderen, C., Vandewalle, L., Vandekeybus, J., Cizer, Ö., 2018. Use of calcined dredging sediments as supplementary cementitious materials, in: Wascon 2018-10th International Conference on the Environmental and Technical Implications of Construction with Alternative Materials, Date: 2018/06/06–2018/06/08, Location: Tampere, Finland. Mol.

Tang, J., Su, M., Wei, L., Wei, Y., Liang, J., Liu, Y., Luo, Y., 2020. Comprehensive evaluation of the effectiveness on metals recovery and decontamination from MSWI fly ash by an integrating hydrometallurgical process in Guangzhou. Sci. Total Environ. 728, 138809.

Tong, L., He, J., Wang, F., Wang, Y., Wang, L., Tsang, D.C.W., Hu, Q., Hu, B., Tang, Y., 2020. Evaluation of the BCR sequential extraction scheme for trace metal fractionation of alkaline municipal solid waste incineration fly ash. Chemosphere 249, 126115.

Todaro, F., De Gisi, S., Notarnicola, M., 2020. Contaminated marine sediment stabilization/solidification treatment with cement/lime: leaching behaviour investigation. Environ. Sci. Poll. Res. 27, 21407–21415.

Visintin, P., Xie, T., Bennett, B., 2020. A large-scale life-cycle assessment of recycled aggregate concrete: the influence of functional unit, emissions allocation and carbon dioxide uptake. J. Clean. Prod. 248, 119243.

Wang, L., Chen, L., Cho, D.W., Tsang, D.C.W., Yang, J., Hou, D.Y., Baek, K., Kua, H.W., Poon, C.S., 2019a. Novel synergy of Si-rich minerals and reactive MgO for stabilisation/solidification of contaminated sediment. J. Hazard. Mater. 365, 695–706.

Wang, L., Chen, L., Guo, B., Tsang, D.C.W., Huang, L., Ok, Y.S., Mechtcherine, V., 2020a. Red mud-enhanced magnesium phosphate cement for remediation of Pb and As contaminated soil. J. Hazard Mater. 400, 123317.

Wang, L., Chen, L., Provis, J.L., Tsang, D.C.W., Poon, C.S., 2020b. Accelerated carbonation of reactive MgO and Portland cement blends under flowing CO_2 gas. Cem. Conc. Comp. 106, 103489.

Wang, L., Chen, L., Tsang, D.C.W., Kua, H.W., Yang, J., Ok, Y.S., Ding, S.M., Hou, D.Y., Poon, C.S., 2019c. The roles of biochar as green admixture for sediment-based construction products. Cem. Conc. Comp. 104, 103348.

Wang, L., Chen, L., Tsang, D.C.W., Li, J., Poon, C.S., Baek, K., Hou, D.Y., Ding, S.M., 2018a. Recycling dredged sediment into fill materials, partition blocks, and paving blocks: technical and economic assessment. J. Clean. Prod. 199, 69–76.

Wang, L., Chen, L., Tsang, D.C.W., Yeung, T.Y.K., Li, J.S., Ding, S.M., Poon, C.S., 2018b. Green remediation of contaminated sediment by stabilisation/solidification with industrial by-products and CO_2 utilisation. Sci. Total Environ. 631–632, 1321–1327.

Wang, L., Cho, D.W., Tsang, D.C.W., Cao, X., Hou, D., Shen, Z., Alessi, D.S., Ok, Y.S., Poon, C.S., 2019d. Green remediation of As and Pb contaminated soil using cement-free claybased stabilization/solidification. Environ. Int. 126, 336–345.

Wang, L., Geddes, D.A., Walkley, B., Provis, J.L., Mechtcherine, V., Tsang, D.C.W., 2020c. The role of zinc in metakaolin-based geopolymers. Cement Concrete Res. 136, 106194.

Wang, L., Kwok, J.S.H., Tsang, D.C.W., Poon, C.S., 2015a. Mixture design and treatment methods for recycling contaminated sediment. J. Hazard. Mater. 283, 623–632.

Wang, L., Tsang, D.C.W., Poon, C.S., 2015b. Green remediation and recycling of contaminated sediment by waste-incorporated stabilisation/solidification. Chemosphere 122, 257–264.

Wang, L., Tsang, D.C.W., Zhou, Y.Y., Rinklebe, J., Song, H., Kown, E.E., Baek, K., Ok, Y. S., 2019b. Mechanistic insights into red mud, blast furnace slag, or metakaolin-assisted stabilisation/solidification of arsenic-contaminated sediment. Environ. Int. 133, 105247.

Wang, L., Yeung, T.L.K., Tsang, D.C.W., Lau, A.Y.T., Poon, C.S., 2017. Recycling contaminated sediment into eco-friendly paving blocks by a combination of binary cement and carbon dioxide curing. J. Clean. Prod. 164, 1279–1288.

Wang, Y.S., Dai, J.G., Wang, L., Tsang, D.C.W., Poon, C.S., 2018c. Influence of lead on stabilization/solidification using ordinary Portland cement and magnesium phosphate cement. Chemosphere 190, 90–96.

Yi, Y., Zheng, X., Liu, S., Al-Tabbaa, A., 2015. Comparison of reactive magnesia- and carbide slag-activated ground granulated blast furnace slag and Portland cement for stabilisation of a natural soil. Appl. Clay Sci. 111, 21–26.

Zhang, W., Zhao, L.Y., McCabe, B.A., Chen, Y.H., Morrison, L., 2020. Dredged marine sediments stabilized/solidified with cement and GGBS: Factors affecting mechanical behaviour and leachability. Sci. Total Environ. 733, 138551.

Zhou, Y., Li, J., Lu, J., Cheeseman, C., Poon, C.S., 2020. Recycling incinerated sewage sludge ash (ISSA) as a cementitious binder by lime activation. J. Clean. Prod. 244, 118856.