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Estimating the environmental costs and benefits of demolition waste using life cycle assessment and willingness-to-pay: a case study in Shenzhen

Abstract

Construction and demolition waste is one of the largest contributors to solid waste generation. Recycling is considered an effective strategy to manage construction and demolition waste; however, the environmental costs and benefits of recycling, compared with a traditional landfill strategy, have not been fully investigated. This study uses a life cycle assessment and willingness-to-pay methodology to investigate the environmental impacts of recycling 1 tonne of demolition waste in Shenzhen. The environmental impacts are global warming, ozone depletion, acidification, eutrophication, suspended particulate matter, solid waste, and land consumption. The results show that recycling can bring an environmental benefit of ¥1.21 per tonne while direct landfill leads to an environmental cost of ¥12.04 per tonne. The environmental costs and benefits of recycling concrete, brick, steel, and mortar, which are the most commonly seen types of component from demolition waste, are also investigated. The results can be used by regulatory authorities to establish strategies and policies, such as the provision of monetary incentives, in order to encourage recycling activities. The results can also be used to establish appropriate landfill charges.

Keywords: demolition waste; recycling; landfill; life cycle assessment; willingness-to-pay; environmental cost and benefit.

1. Introduction

Construction and demolition waste (CDW) has a significant environmental impact which requires immediate action. In the European Union, 3000 million tonnes of waste are produced every year, of which 25% to 30% is generated by the construction industry (Bravo et al., 2015). Similarly, in the United States, 530 million tonnes of construction and demolition debris were generated in 2013 (US Environmental Protection Agency (EPA), 2013). Construction activities consume 25% of the virgin wood and 40% of the raw stone, gravel, and sand which are used globally every year (Kulatugna et al., 2006; Wu and Low, 2009). In addition to resource depletion, CDW has a significant impact on land degradation, global warming, and ozone depletion (Coelho and de Brito, 2012). For example, 14 million tonnes of waste are landfilled each year in Australia, 44% of which comes from construction activities (Lu and Tam, 2013). According to Bhada-Tata and Hoornweg (2016), disposal is a significant source of carbon emissions and landfill is a significant source of methane, both of which contribute to global warming.

Because of the negative impact of CDW on the environment, many studies have investigated its environmental impact. For example, in a Spanish case study, Ortiz et al. (2010) used the life cycle assessment (LCA) method to evaluate the environmental impact of construction waste. Similarly, Coelho and de Brito (2012) investigated the environmental impact of buildings by using five waste management options—complete demolition, selective demolition, deconstruction of non-structural elements, full deconstruction and recycling, as well as full deconstruction and partial recycling. Selective demolition, which is the reverse of the construction process, has been introduced for easy recycling and reuse (Lu et al., 2009; Coelho and de Brito, 2013). The method can help reduce the overall demolition cost by reducing disposal charges (Lu et al., 2009). Many studies treat CDW in a similar way because both sources of waste are generated from the construction industry. However, it should be noted that the amount of demolition waste is significantly higher than the amount of construction waste. According to the US EPA (2013), demolition activities cause more than 90% of total CDW. As such, demolition waste should have a higher priority than construction waste. It should be noted that the exact contribution of demolition activities to CDW may vary depending on country-specific characteristics and site conditions. For example, demolition activities, which contribute to 74% of annual CDW in China, only contribute to 36% of annual CDW in Norway (Statistics Norway, 2017; Lu et al., 2017). In addition, the composition of construction waste and demolition waste varies significantly, indicating that waste management strategies developed for managing construction waste may be unsuitable for demolition waste management. According to Zhao and Rotter (2008), the composition of construction waste in China includes concrete, sand, brick, and stone, while the most significant components in demolition waste are brick and tile, followed by concrete. Because such composition differs significantly, it is useful to separate the evaluation process of the environmental impact of CDW. Moreover, many scholars use case studies to investigate the environmental impact of construction and demolition activities (Wu et al., 2015; Wu et al., 2016). It should be noted that case studies can be useful to evaluate the environmental impact associated with specific buildings or processes; however, case studies rely on data from individual construction and demolition activities such as the construction of a single family home (Cuéllar-Franca and Azapagic, 2012) and a commercial building (Zhang et al., 2013). Thus, case studies may offer limited guidance and reference for local governments which rely on local and regional analysis to establish CDW policies, such as charging appropriate fees for mitigating CDW's environmental impact.

Consequently, the current study aims to 1) investigate the environmental impact of demolition waste using the LCA approach; 2) compare the environmental costs and benefits of demolition waste from two different treatment pathways—recycling and landfill—using the willingness-to-pay (WTP) approach; and 3) investigate appropriate levels of fees and charges to mitigate the environmental impact of demolition waste. The results will be useful for regulatory authorities to understand the environmental costs and benefits of demolition waste and establish relevant strategies to reduce demolition waste. It should be noted that this study focuses on ordinary demolition waste management scenarios and excludes scenarios which include the occurrence of natural hazards, such as seismic hazard assessment and post-quake recovery (Jun et al., 2012; Faleschini et al., 2017; Zanini et al., 2017).

2. Literature review

2.1. The environmental impact of CDW

CDW is usually defined as the solid waste which arises from construction, renovation, and demolition activities (Lu et al., 2011). Shenzhen is a rapidly developing megacity in China. According to Wu et al. (2016a), approximately 14 million tonnes of demolition waste have been generated annually in Shenzhen since 2010; moreover, because of the rapid urban development, CDW generation is expected to increase in the future. Wu et al. (2016b) used a geographic information system (GIS) to investigate the waste types and volumes associated with buildings in Shenzhen. According to Wu et al. (2016b), the expected demolition waste types in Shenzhen include concrete (57.2%), brick (16%), mortar (12.4%), metal (4.9%), glass (0.2%), and others (9.3%). In order to manage the large volume of CDW, landfill and recycling have been adopted, with 10 landfill sites and six recycling facilities established over the past two decades (Wu et al., 2016a).

CDW reduction and recycling, as effective waste management strategies, have received much attention for some years. Various strategies and policies have been proposed for the effective reduction of CDW. For example, in Greece, a CDW treatment and recycling unit must be established in each city for inert waste (Fatta et al., 2003). The research on using recycled aggregates from CDW to meet mechanical and durability requirements has also received much attention globally, including in the European Union (Rodrigues et al., 2013). Thus, CDW is a significant problem for many countries because of its high volume and negative impact on the environment. As a result, some studies have investigated CDW's environmental impact. For example, Bohne et al. (2008) evaluated the eco-efficiency of waste treatment strategies for CDW at the city level, Blengini and Garbarino (2010) calculated the environmental impact related to the recycling of inert materials in Italy, and Mercante et al. (2012) assessed the environmental impact of a Spanish CDW management system. Further, according to Solís-Guzmán et al. (2009), the estimated waste volume for new construction is 0.30 m^3 per m^2, while the estimated waste volume for demolition, 1.27 m^3 per m^2, is more than four times greater. In addition, the composition of demolition waste differs significantly from new construction waste. Thus, it is useful and necessary to separate new construction waste from demolition waste when evaluating environmental impact.

2.2. The environmental impact of recycling and disposal

According to Xing and Charles (2006), a few recycling technologies are available for the treatment of demolition waste. For example, manual separation can be used for concrete recycling in order to separate recoverable materials. In addition, crushers can be employed for size reduction, and shaking tables can separate light materials from heavy materials. A recycling plant for concrete usually consists of crushers, screeners, magnetic separators, wind-sifting, and manual separation (Zhao et al., 2010). Figure 1 presents a simplified concrete recycling process which produces recycled aggregates from crushed concrete. The production of recycled aggregates usually involves primary and secondary crushing to produce aggregates with various size fractions (Li, 2008). The recycled aggregates also need to meet the requirements, such as particle density, water absorption, and fine particles, which are provided in national and international specifications; for example, the Technical Code for Application of Recycled Aggregated Concrete (in China), the Specification for Constituent Materials and Concrete (in the UK), and the Specifications for Concrete with Recycled Aggregates (SCSS, 2007; BS8500-2:2002, 2002; RILEM, 1994).

<Insert Figure 1 here>

Figure 1. A simplified process map for recycling concrete waste (Source: Hiete, 2013)

According to Tam and Tam (2006), as a typical source of waste, bricks from demolition are often contaminated with mortar and plaster. Thus, on-site sorting is usually needed before bricks are transported to a waste treatment plant. Tam and Tam (2006) also found that in Hong Kong, the most common practice for brick recycling is crushing in order to create filling materials and hard core. Figure 2 presents a typical brick recycling process which produces filling materials from crushed bricks. A similar process to concrete recycling is also adopted for brick recycling, where bricks are crushed, screened, and grouped in accordance with different sizes (Aliabdo et al., 2014). The requirements listed in the aforementioned specifications must also be met if the crushed bricks are to be used as recycled aggregates. Moreover, recycled brick powder can replace cement in mortar; however, the requirements for the mechanical properties of the material, including flow, comprehensive strength, and flexural strength, must be met (Zheng et al., 2011).

<Insert Figure 2 here>

Figure 2. A simplified process map for recycling bricks as recycled aggregates and brick powder

According to Tam et al. (2007a), ferrous metal accounts for almost 3% of CDW in Shenzhen, while non-ferrous metal accounts for less than 0.5%. Ferrous metal, such as steel, has the highest recycling rates from demolition waste because of its magnetic properties and high market value (Kartam et al., 2004). For example, reinforcement bars from crushed concrete can be collected by scrap dealers who then sell the bars for re-melting. Figure 3 shows a typical production process for steel. As can be seen from Figure 3, the production process is related to the use of an electric arc furnace (EAF) in order to recycle steel scrap into finished steel products through charging, melting, and decarburization (Burchart-Korol, 2013).

<Insert Figure 3 here>

Figure 3. A simplified steel recycling process

The recycling process for cement mortar is usually not separated from the recycling process for concrete. However, the value of cement mortar is relatively low when compared with other constituents; indeed, the value of recycled mortar is normally not considered. According to Tam et al. (2007b), cement mortar can have high porosity and water absorption rates which weaken the strength and mechanical performance of concrete made from recycled aggregates. As such, many new techniques have been developed, such as the pre-soaking approach (Tam et al., 2007b), to remove cement mortar from recycling activities. In addition, new methods, such as the equivalent mortar volume (EMV) method, have been developed to ensure that concrete made with recycled aggregate concrete satisfies mechanical and durability requirements (Abbas et al., 2009). Because these technologies have yet to be implemented in Shenzhen, the recycling activities for mortar can be represented by Figure 1.

2.3. CDW disposal charge

A CDW disposal charge is considered one of the most effective strategies to help minimise disposal. According to Hao et al. (2008), because waste producers are charged when disposing of CDW, they are encouraged to recycle in order to reduce their disposal costs. However, it should be noted that the implementation of a disposal charge in China is restricted to municipal solid waste, not CDW. The difficulty in establishing an appropriate disposal charge for CDW is partly related to the need for an effective mechanism to quantify its impact (Yuan and Wang, 2014). Consequently, a few studies have investigated the establishment of such a charge. For example, Yuan et al. (2011) used simulation to investigate the effectiveness of high and low disposal charges, while Nunes et al. (2007) investigated the financial performance of recycling centres in order to establish an appropriate disposal charge. However, as Martin and Scott (2003) pointed out, the current disposal charges are ineffective at changing the behaviour of the main waste producers. These disposal charges are mostly related to municipal solid waste instead of CDW and may be too low to force a change towards recycling and reusing. In addition, the disposal charges are not quantitatively related to the impact of CDW, including its environmental impact. This failure to link the charges to CDW's impact is a research gap which needs to be filled.

2.4. LCA

LCA has been widely adopted to evaluate the environmental impact of products and processes in the construction sector. The overall environmental impact of a product/process is calculated by evaluating the environmental impact associated with the input and output (Zhang et al., 2013; Wu et al., 2014; Wu and Feng, 2012). LCA can help to evaluate many types of environmental impact. For example, the method of the environmental design of industrial products (EDIP) developed in Denmark uses resource consumption, environmental pollution, and occupational health as the main assessment categories (Danish Ministry of the Environment, 2005). In China, the building environmental performance analysis system (BEPAS) is commonly adopted (Zhang et al., 2006). BEPAS has two major impact categories: ecosystem damage and resource depletion. In a similar way to other environmental analysis systems, ecosystem damage considers global warming, ozone depletion, acidification, eutrophication, and solid waste. Based on BEPAS, a total of seven impact categories are selected in this study. These categories, the main pollutants, and the units of measurement are presented in Table 1.

<Insert Table 1 here>

3. Research method

This study uses the LCA approach. The system boundaries, functional unit, and other estimation assumptions are explained in the following subsections.

3.1 System boundaries

In accordance with the aim of this study, two system boundaries are selected. Figure 4 represents the system boundary when demolition waste is transported to waste treatment plants for recycling. Figure 5 represents the system boundary when demolition waste is directly disposed of as landfill.

<Insert Figure 4 here>

Figure 4. The system boundary of the recycling practice of demolition waste

<Insert Figure 5 here>

Figure 5. The system boundary of the landfill practice of demolition waste

3.2 Functional unit

The functional unit of this study is 1 tonne of demolition waste originating from demolished buildings. Table 2 presents the composition of demolition waste in Shenzhen, based on Wu et al. (2016b).

<Insert Table 2 here>

3.3 Other assumptions

A few other assumptions are also made in order to calculate the environmental impacts associated with the input.

Transportation

The average transportation distance from a demolition site to a waste treatment plant is calculated as follows:

$$D = \sum S_i * L_i \quad \text{(Eq. 1)}$$

where D refers to the average transportation distance; i represents a district in Shenzhen; S_i refers to the percentage of district i's demolition projects in Shenzhen; and L_i refers to the average transportation distance in district i. In total, seven districts are considered. The overall average transportation distance is estimated to be 19.47 km (see Table 3). Because of the emissions regulation, steel recycling plants are located in Shenzhen's rural areas. Thus, based on the plants' geographical locations, the transportation distance for steel recycling is assumed to be 50.00 km. In addition, Shenzhen has four major landfill sites: Baoan, Longhua, Longgang, and Luohu. Following a similar method to that of recycling, the overall average transportation distance for general disposal is calculated to be 14.71 km.

<Insert Table 3 here>

Environmental impact factors

Table 4 presents the environmental impact factors used in this study. It should be noted that the environmental impact factors for concrete, brick, and mortar recycling activities include the direct environmental impact of the activities and the indirect impact of diesel and electricity production. This approach is in accordance with many international standards, such as ISO (International Organization for Standardization) 14067 (2013) and PAS (Publicly Available Specification) 2050 (British Standards Institution, 2008), which state that the emissions associated with electricity use shall include the emissions from the generation of electricity, such as the combustion of fuels. In this context, given the large amount of data required in LCA studies, data uncertainty is an inherent problem. Further, according to Finnveden et al. (2005), the uncertainties of the LCA methodology, such as the selection of system boundaries, are often greater than uncertainties in the data. As can be seen from Table 4, this study focuses mainly on the use of China-specific emission factors and is based on a few highly cited studies, such as those of Yang (2002) and Gu (2009) who proposed the LCA approach and environmental impact factors, in order to mitigate data uncertainty.

<Insert Table 4 here>

3.4 Weights of environmental impact categories

The WTP method is adopted to determine the weight of each environmental impact category. According to Li et al. (2005), the WTP method can help evaluate the weight of each impact category by considering the 'green tax', the impact potential, and the total emission volumes. The weight of each main environmental impact category, i, is calculated with Eq. 2:

$$W_i = \sum e_{ij} * c_{ij} \quad \text{(Eq. 2)}$$

where W_i refers to the weight of the main environmental impact category (e.g. global warming); c_{ij} refers to the pollution discharge fee for impact category j (e.g. carbon emissions) in the main impact category i; and e_{ij} refers to the coefficient of impact category j.

In addition, e_{ij} is calculated with Eq. 3:

$$e_{ij}=f_j*a_j/\sum f_j*a_j \qquad \text{(Eq. 3)}$$

where f_j is the pollution equivalency factor of impact category j and a_j refers to the annual pollution volume of impact category j.

It should be noted that there is no discharge fee for carbon emissions in Shenzhen. As such, a questionnaire survey is conducted by this study to investigate the WTP for carbon emissions. The survey uses 14 payment cards (13 cards with values from ¥0 to ¥100 and one card which the participants need to complete if they are willing to pay more than ¥100) to identify the WTP for each tonne of carbon emissions equivalent generated. Before the payment cards are presented, the participants are briefly introduced to the environmental impact of global climate change with some examples. For instance, the carbon emissions coefficient of the coal-energy chain in China is 875 g per kWh (Yu et al., 2014). In addition, some demographic information of the participants, including their ages, salaries, and occupations, is recorded. A simple random sampling method is adopted to ensure that each participant has an equal chance of being included (Sweis et al., 2008). In total, 600 questionnaires are distributed to randomly selected participants in the seven districts in Shenzhen. After excluding incomplete responses, 399 valid responses are recorded, representing a response rate of 66.5%.

4. Results

4.1 Weights of environmental impact categories

Table 5 shows the demographic information of all respondents. According to Burns and Bush (2009), the sample size, N, can be calculated using the following equation:

$$N = \frac{Z^2 \; x \; p \; x \; (1-p)}{e^2} \quad \text{(Eq. 4)}$$

where z refers to the standard error with a confidence level of 95%; e refers to the accepted error, which is 5% in this study; and p refers to the estimated variance of the population and is 0.5 if the survey contains both continuous and categorical variables. Based on the above calculation, a sample size of 385 is required. Thus, the survey meets the sample size requirement. The average respondent is 26–35 years old, has a diploma, and earns a monthly income of ¥3501–5000. As can be seen from Table 5, the monthly income levels of most respondents are ¥3501–5000 (30.85%), ¥5001–6500 (22.39%), and ¥6501–8000 (13.42%). According to the Shenzhen Human Resources and Social Security Bureau, the average salary of jobs in Shenzhen is ¥4711. Because the WTP for environmental improvement is affected by the income level (Husted et al., 2014), the sample can be considered representative.

<Insert Table 5 here>

Table 6 shows the distribution of the WTP for each tonne of carbon emissions equivalent in Shenzhen. The WTP of carbon emissions is estimated to be ¥37.96/tonne CO_2-e (or ¥0.04/kg CO_2-e).

<Insert Table 6 here>

In addition, the weights of other impact categories are calculated using the WTP model developed in this study (see Table 7). It should be noted that there is no CFC tax in China at the time of the study. The Multilateral Fund (MLF) for the implementation of the Montreal Protocol is provided to China to phase out ozone-depleting substances (ODS). This study uses the cost-effectiveness of the MLF-funded agreement to phase out ODS as the WTP to mitigate environmental impact. Such an approach has been adopted in prior studies such as that of Zhang et al. (2006). According to the Global Environment Facility (GEF) (2010), the average cost-effectiveness for the MLF is US$12.58 for each ODP kilogram, which is phased out. The

United Nations Environment Program (UNEP) (2016) provides an average of US$6.1 (¥40.69 equivalent) through the MLF to help China phase out ODS. This amount is adopted here as the WTP for ozone depletion. In addition, the compensatory usage fee of land is used as the WTP for land consumption. According to Yuan (2013), more habitats will be occupied when pristine land is used for new and expanded landfill sites. According to the Shenzhen Municipal Office (2015), the current compensatory usage fee is ¥32.00 per m^2. The pollution equivalency factors represent the acid formation potential of SO_2, and the eutrophication potential of NO^{3-}e (European Commission, 2006).

<Insert Table 7 here>

4.2 The environmental costs and benefits of recycling practices

The environmental impacts of the recycling activities related to concrete, brick, steel, and mortar are listed in Table 8. As can be seen from Table 8, the environmental performance of each type of demolition waste differs significantly. For example, recycling 1 tonne of steel leads to an environmental credit of 1811 kg CO_2-e, because recycling activities help reduce the CO_2 generated from the production process by almost 30%. This finding accords with studies such as that of Johnson et al. (2008). Recycling 1 tonne of brick and concrete, however, causes 32.22 kg and 4.83 kg CO_2-e respectively.

When the WTP for each source of waste is considered, the environmental costs for the waste are -¥1.18 (per tonne of concrete), ¥2.86 (per tonne of brick), -¥82.78 (per tonne of steel), and ¥3.52 (per tonne of mortar). The results show that steel and concrete recycling generate environmental benefits. However, brick and mortar recycling have a negative environmental impact.

In terms of environmental costs, concrete recycling has a high positive value (¥0.92) on acidification, indicating its high negative impact. However, it has a relatively low negative value (-¥1.92) on land consumption. It seems that reducing land consumption is the most obvious environmental benefit of concrete recycling.

Unlike concrete recycling, brick recycling has a high positive value (¥1.29) on global climate change, followed by acidification (¥1.15) and solid waste (¥0.40). The only environmental benefit of brick recycling is associated with suspended particulate matter, with a marginal value of ¥0.02. Assumptions about how the recycled materials will be used may cause the difference. In this regard, bricks are crushed in order to create filling materials. The environmental benefits of these filling materials may be less than the direct use of recycled bricks.

Steel recycling has significant environmental benefit. As can be seen from Table 8, recycling 1 tonne of steel has an environmental benefit of ¥72.44 in terms of global warming, followed by ¥3.99 in terms of acidification and ¥3.59 in terms of solid waste. The only source of environmental cost is ozone depletion; however, this is at a minimal level (¥9.27×10^{-5}).

With regard to mortar, recycling credits are not considered because recycled mortar is usually not reused in Shenzhen. Thus, recycling 1 tonne of mortar normally causes an environmental cost instead of a benefit. Solid waste and acidification are the most important sources of environmental costs (¥2.25 and ¥0.95 respectively).

It should be noted that the accuracy of the environmental costs and benefits identified in Table 8 depends on the system boundaries, which in this study exclude the extraction of raw materials. As pointed out by Faleschini et al. (2016), CDW recycling can help reduce the extraction rate of raw materials, leading to the preservation of materials such as virgin aggregates. The environmental benefit of such preservation is not investigated in this study although it may affect the WTP of environmental impacts such as land use. Thus, it is recommended that upstream activities are included in future studies in order to investigate the environmental costs and benefits of CDW recycling systematically.

<Insert Table 8 here>

4.3 The environmental costs and benefits of demolition waste in Shenzhen

Based on the WTP identified in this study, the environmental impact of 1 tonne of demolition waste in Shenzhen is shown in Table 9. As can be seen from Table 9, the environmental benefit of recycling 1 tonne of demolition waste is ¥1.21, which is the aggregated value of ¥0.70 (the environmental benefit from concrete recycling), ¥0.84 (the environmental cost of brick recycling), -¥1.69 (the environmental benefit from steel recycling), and ¥0.34 (the environmental cost of mortar recycling). The environmental benefits and costs of each environmental impact category are detailed in Figure 6. As can be seen from Figure 6, land consumption and global warming represent the two largest sources of environmental benefit from recycling 1 tonne of demolition waste with estimated values of ¥1.12 and ¥0.97 respectively. However, acidification is the most important source of environmental cost with an estimated value of ¥0.89 per tonne.

<Insert Table 9 here>

<Insert Figure 6 here>

Figure 6. The environmental benefits and costs of recycling 1 tonne of demolition waste in Shenzhen

In addition, environmental benefits and costs of each type of waste are detailed in Figure 7. As can be seen from Figure 7, when recycling 1 tonne of demolition waste, concrete recycling has a significant environmental benefit for land consumption and steel recycling has a significant environmental benefit for global warming. These benefits can offset the significant environmental costs of acidification (from concrete and brick recycling), global warming (from concrete and brick recycling), and solid waste (from mortar recycling).

<Insert Figure 7 here>

Figure 7. The environmental benefits and costs of recycling concrete, brick, steel, and mortar from 1 tonne of demolition waste in Shenzhen

5. Discussion

Recycling demolition waste can have environmental benefits. In order to obtain the comparative benefits of recycling, it is useful to compare the environmental benefits with traditional landfill practices.

Two assumptions are made when calculating the environmental cost of landfilling. The first is related to the diesel consumption associated with landfill activities. This study selects one landfill site in Shenzhen and uses the daily landfill volume and the daily diesel consumption to calculate the diesel usage for the disposal of 1 tonne of demolition waste. The result is 4.09 l/tonne, based on the daily landfill volume of 5200 tonnes and the daily diesel consumption of 21,268 litres. In addition, the depth of the landfill site is assumed to be 5 m, with 1 tonne of demolition waste occupying $0.34m^2$. The latter figure is relatively less than the land use of CDW proposed by Faleschini et al. (2016), who used LCA to investigate the emissions and land use of natural and recycled aggregates. As pointed out by Faleschini et al. (2016), the lack of a consistent LCA framework to evaluate land use may contribute to any differences because land use evaluation is highly site-specific and depends on a landfill's depth and the waste's density. Similarly, Habert et al. (2010) argued that the LCA approach is not well suited to assess resource consumption. Instead, they proposed a new assessment related to the stock of resources in order to assess resource depletion at a regional scale. Compared with this study, which relies on a single landfill site to evaluate land consumption, the indicator of abiotic depletion potential (ADP) developed in Habert et al. (2010) is a more accurate representation of resource depletion at a regional scale and can be investigated in future studies to assess land consumption more accurately.

The environmental impacts of direct landfill in terms of each tonne of demolition waste are shown in Table 10. As can be seen from Table 10, the most significant environmental impact from direct landfill is land consumption, with an estimated environmental cost of ¥10.88, followed by acidification (¥0.67), suspended particulate matter (¥0.32), and global warming (¥0.14). When compared with the environmental benefit of ¥1.21 from recycling, there is a difference of ¥13.24 per tonne, demonstrating the significant environmental benefit of recycling activities.

<Insert Table 10 here>

A landfill charge has been widely recognised as an effective way to manage CDW (Hao et al., 2008). As such, establishing an appropriate landfill charge is critically important. For example, Poon et al. (2001) argued that the landfill charging system should be based on the 'polluter pays principle'. Moreover, at the initial stage, waste producers need to pay at least 50% of landfill costs; then, the costs should be increased to cover the full construction and operational costs of landfill sites. In this context, low landfill charges will not encourage contractors to implement waste management policies and practices (Yuan et al., 2011). The landfill charge in Shenzhen is ¥5.88 (Wang and Yuan, 2009), which is significantly lower than the environmental cost of direct landfill (¥12.04). As a result, such a low landfill charge provides no incentive to contractors to invest in waste management activities. The charge is also not sufficient to offset the environmental impact caused by direct landfill activities, not to mention the construction and operational costs of the landfill sites.

In order to establish relevant strategies to manage the environmental impact of demolition waste, it is also useful to understand the environmental cost and benefit of each activity (see Table 11). As can be seen from Table 11, recycling 1 tonne of demolition waste has a relatively low environmental impact on ozone depletion and eutrophication. With regard to global warming, the recycling activities of steel (¥0.75) are the most significant contributors, followed by the recycling activities of brick (¥0.33). It seems that in order to manage and reduce the carbon emissions from recycling demolition waste, steel and brick are the two sources that should be focused on. Similarly, the most significant contributors to acidification are transportation activities, with a total value of ¥0.95. In addition, the recycling activities of concrete and mortar are the two most important contributors to solid waste (¥1.32 and ¥0.22 respectively). In terms of environmental benefits, the most significant sources are the recycling credits of steel with regard to global warming (¥2.24) and the recycling credits of concrete with regard to solid waste and land consumption (¥1.47 and ¥1.13 respectively).

These values provide useful references for the establishment of incentive schemes to promote recycling activities. Tam and Tam (2006) argued that the lack of financial incentive schemes is the reason behind the low recycling rate for construction debris. Indeed, financial incentives are found to have a positive effect on the reduction of the total waste produced (Bucciol et al., 2015). Based on the results of the current study, ¥1.21 per tonne should be provided as an incentive to encourage recycling activities.

However, the results of this study need cautious interpretation when applied to other countries which have significantly different WTP values. For example, Japan has a carbon tax of US$2.89 (equivalent to ¥19.83) per tonne of CO2-e. This is significantly less than the WTP identified in this study (¥38.65). In Europe, the charge for sulphur emissions is €1600/tonne (equivalent to ¥11.79/kg) in Sweden and €1300/tonne in Denmark (equivalent to ¥9.58/kg) (Millock et al., 2004). These values are significantly higher than the discharge fee in Shenzhen, which is only ¥0.95/kg. However, the method presented in this study will be useful to help investigate the environmental costs and benefits of demolition waste in other countries and thereby establish appropriate levels of charges. In addition, transportation distance can affect the accuracy of an LCA study (Faleschini et al., 2016). It is recommended that future studies investigate landfill charges using various transportation scenarios.

<Insert Table 11 here>

6. Conclusions

Managing demolition waste is important for sustainable urban development. Because of the rapid urban development of China, a significant amount of demolition waste is generated each year. As such, investing in waste management activities, such as recycling and reusing, seems imperative. This study employs LCA and WTP approaches to investigate the environmental impact of recycling and direct landfill activities, thereby providing guidance for policy improvements such as the establishment of appropriate landfill disposal charges and incentive-based schemes to encourage recycling.

The results show that recycling 1 tonne of demolition waste in Shenzhen leads to an environmental benefit of ¥1.21. Further, the most significant sources of environmental benefit are land consumption in terms of concrete recycling (¥1.13) and global warming in terms of steel recycling (¥1.49). These sources of environmental benefit offset the environmental costs of acidification from concrete and brick recycling (¥0.54 and ¥0.34 respectively), global warming from brick and concrete recycling (¥0.38 and ¥0.11 respectively), and solid waste from mortar recycling (¥0.22). The results also show that the current landfill charge of ¥5.88 is not sufficient to offset the environmental cost of landfill (¥12.04), the construction and operational costs of landfill sites, and the potential landfill penalty charge.

This study has some limitations. The assumptions, such as the end-of-life treatment of recycled products, are closely related to the accuracy of the results. If bricks are recycled and reused directly (compared with this study's assumption that they are crushed as filling materials) and mortar is reused as recycled aggregate, the environmental benefit of recycling activities will be higher. Future studies could focus on various recycling scenarios in order to understand the environmental benefit of recycling when compared with traditional landfill.

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