Towards a sustainable and circular metals economy: the case of copper in China

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Chapter 5

Towards "Zero waste" management of copper in China: dematerialization and environmental impact minimization

Abstract

The electrical conductivity of copper makes it an important metal in a variety of applications. To conserve resources and enhance security of supply, China has launched the "Zero waste" concept, focused on reutilization of solid waste and recovery of materials, including copper. This Chapter explores scenarios of copper waste generation and management in China. Six types of waste sources are investigated in relation to various "Zero waste" strategies and their effect on the copper cycle and associated environmental impacts assessed. We conclude that under present Chinese policies, reuse and recycling of copper containing products will lead to a somewhat lower dependency on primary copper resources, as well as lower greenhouse gas emissions and energy demand. Maximizing such "Zero waste" options may lead to further reductions, but may also be counter-productive if applied too stringently. GHG emissions related to secondary copper production may exceed those of primary copper production despite lower per kg GHG emissions of secondary production.

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5.1 Introduction

Solid waste is a heterogeneous waste stream from a wind range of sources in the economy that unless properly managed can lead to considerable resource losses and cause serious environmental damage. To improve the efficiency of resource use, China is transitioning from a linear, “take-make-dispose” economy to a circular economy that aims to maintain products, components and materials at their highest utility and value (NDRC, 2017). To further promote the development of a circular economy, China has introduced the “Zero waste” concept and applied it to selected cities to minimize landfill and emissions to the environment through waste prevention and reuse (Song et al., 2015; State Council, 2018).

Waste management is a key element of moving towards “Zero waste” and a circular economy, and is particularly relevant for metals where supply constraints may emerge in the future. Copper is one of the crucial metals that has a high economic value and can be efficiently recycled. It is widely used in buildings, transportation and infrastructure, and is especially critical in a transition to a low-carbon energy system, which is expected to accelerate future copper use (Dong et al., 2019; Eheliyagoda et al., 2019; Watari et al., 2020). This increased use of copper will in due course also result in increased of copper waste generation. Effective and efficient management of these potentially large waste streams is a complex issue, requiring careful consideration of aspects like scarcity and security of supply of specific resources, besides costs, energy efficiency and the environmental impacts of recovery options.

There have been numerous studies on waste management that have comprehensively analyzed the technologies, costs, feasible strategies and social and environmental impacts based on the ‘reduce’, ‘reuse’ and ‘recycle’ principles (Das et al., 2019; Giusti, 2009; Kaufman et al., 2010). With respect to copper waste management, most previous studies have focused on recycling. Such research has highlighted various strategies (e.g. enhanced recycling rates and use of clean energy in the recycling process) and their implications on the availability of recycled copper (Ciacci et al., 2020; Dong et al., 2020a; Pfaff et al., 2018; Soulier et al., 2018b; Wang, J. et al., 2019; Yoshimura and Matsuno, 2018). Several studies have analyzed the environmental benefits of copper recycling or secondary copper production, usually including the steps of copper collection, mechanical processing and metallurgical processes (Hong et al., 2018; Kulczycka et al., 2016; Northey et al., 2013). Moreover, a few such studies have assessed the historical and prospective environmental impacts of producing a unit of secondary copper, taking into account such factors as the copper content of the waste, the energy efficiency of the metallurgical processes and the potential influence of an energy transition. These studies suggest that increased use of secondary copper over the next few decades may contribute to reducing greenhouse gas (GHG) emissions (Dong et al., 2020b; Kuipers et al., 2018; Northey et al., 2014; Van der Voet et al., 2018). However, other studies have found significant economic and organizational barriers to implementation of circular economy options and greater use of secondary copper (Fu et al., 2017; Rubin et al., 2014).

While such studies have provided a useful basis for exploring copper recycling and its environmental impacts, little effort has been made to distinguish between different types of copper waste and investigate circular economy strategies, especially for reuse (repair, remanufacturing or refurbishment) rather than recycling options for copper in China. Studies on specific copper-containing products (e.g. electronic products) provide important information for understanding the treatment of copper waste at product level (Fiore et al., 2019; Ruhrberg, 2006; Santini et al., 2011). However, given that copper comes from a variety of waste products and that treatment technologies for these products differ widely, a systematic analysis of various types of waste may provide a more comprehensive decision-making basis for optimizing the copper waste management system (WMS) and even the copper lifecycle as a whole.

This article aims to address this gap in the current literature and further to explore future developments in copper waste management and the associated environmental impacts. To this end we investigated copper waste management options in relation to various “Zero waste” strategies and assessed the possible effects on the copper cycle in China by addressing the following questions:

- How will the generation of copper waste develop, by type and source, in the coming decades?

- How can a waste management system be designed that minimizes copper losses under various "Zero waste" strategies?
- What environmental benefits and/or drawbacks can be expected from such an optimized waste management system for the copper cycle as a whole?

The methods used to answer these questions are discussed in Section 2, while Section 3 reports and discusses the results and presents some of the implications.

5.2 Materials and Methods

To address the above research questions, this study combines the methods of material flow analysis and life cycle assessment to calculate mass flow and environmental impacts, as depicted in Figure 5.1. MFA is a method for quantitative analysis and evaluation of the input and output of materials in a system (Brunner and Rechberger, 2003). Application of this method in the field of circular economy has increased significantly in recent years, being employed as a core method in much of the literature that has studied the copper flows and stocks in the world, in an individual country or in a regional system. Such studies generally include all life cycle stages (mining, metallurgical production, manufacturing, fabrication, use and waste management) as well as trade (Daigo et al., 2009; Deetman et al., 2018; Rechberger and Graedel, 2002; Schipper et al., 2018; Yue et al., 2009). LCA, as a method for evaluating the environmental impacts of a product or service system, has made major contributions to the formulation of policies towards "Zero waste" and circular economy (Guinée, 2002; Haupt and Zschokke, 2017). To explore the future environmental impacts of secondary copper production, the method of *prospective* LCA is being developed and will be applied to emerging technologies (which usually still improve over time owing to learning effects) in a changing system (Arvidsson et al., 2018; Villares et al., 2017).

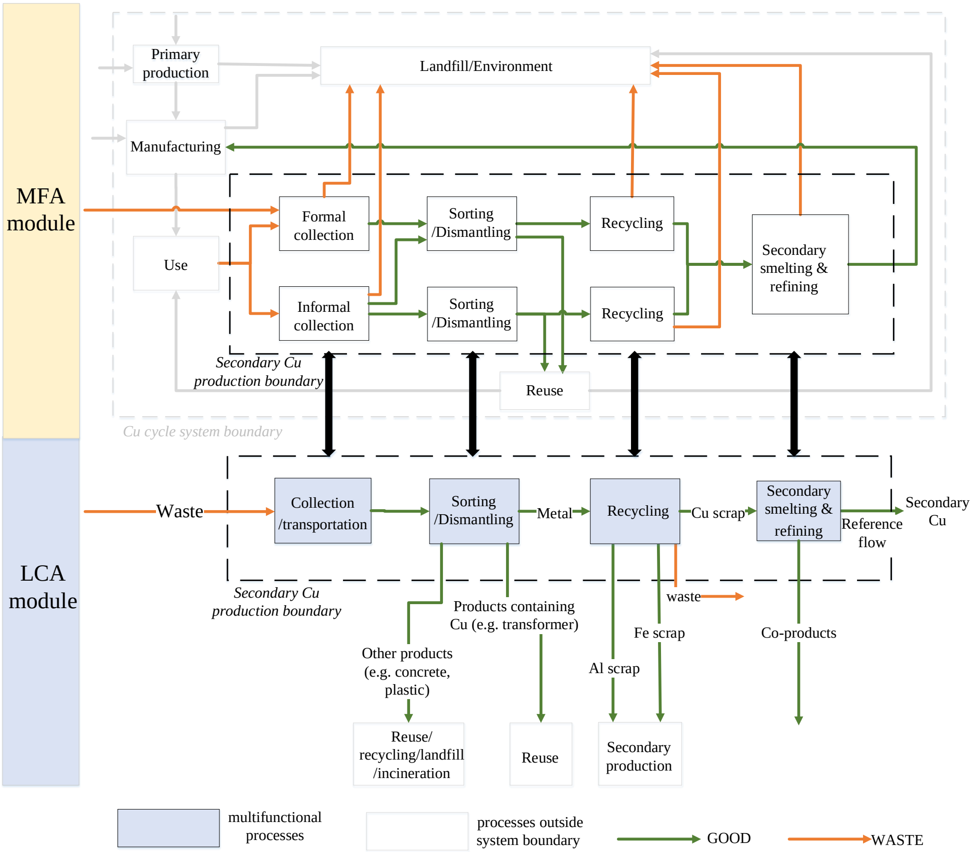


Figure 5.1 Copper cycle system boundary and definition for China: schematic representation of the MFA and LCA combination on the secondary copper production. The MFA processes (collection, sorting & dismantling, recycling, secondary smelting & refining) are represented by the LCA processes. Note: The formal and informal production processes (from collection to refining) are not distinguished in LCA module. Detailed MFA and LCA systems and definitions for each type of waste streams can be found in Figures S5.1 and S5.4 in the SI.

5.2.1 Dynamic modelling of future copper waste generation and management

Figure S5.1 depicts China's copper cycle, including primary production, manufacturing, use, waste management, reuse (repair, remanufacturing or refurbishment), secondary smelting and refining. Following the classifications described by Ruhrberg (2006) and Soulier et al. (2018b), five domestic waste streams are distinguished: C&DW, ELV, WEEE, MSW and Industrial Equipment Waste (IEW). Imported copper waste (ICW), as the main input of secondary copper production before the implementation of China's "Green Fence" policy, is also considered in this study. The "Green Fence" policy has been implemented since 2013 to restrict the import of low-quality copper scrap (State Council, 2013).

Copper waste management generally involves a sequence of stages. In China, EoL copper products are normally collected by peddlers, private companies, salvage stations or by municipal authorities and then manually or mechanically sorted, dismantled, shredded and separated for copper recycling and reuse purposes, while the non-copper containing residues are usually either incinerated or directly landfilled. China still currently has an informal waste management sector, which is particularly relevant for the recycling of WEEE and ELV (Chi et al., 2011). Large amounts of informally collected copper go to informal dismantling industries or to economically underdeveloped areas for reuse in second-hand products. Until 2019 the development of advanced reuse options like remanufacturing posed a challenge to the formal Chinese recycling sector, since the "Administrative Measures for the Recycling of Scrapped Automobiles" implemented in 2001 stipulated that certain components of scrapped cars should be recycled as raw materials by specific companies (e.g. steel producers) (State Council, 2001). The copper-containing components that can be directly re-used, remanufactured or refurbished account for only a small proportion: around 4% of the collected copper in C&DW and 10% in ELV in 2017 in China (as reported in Appendix 2, e.g. electrical accessories, transformers, engines) (Liu et al., 2020; Wang, 2012; Weber et al., 2009), for example. In certain developed countries, in contrast, this rate ranges from 45% to 80% (CELVE, 2019). For MSW, treatment usually consists of incineration and landfill. Copper recycling from bottom ash is complex and is hence relatively uncommon (Lassesson et al., 2014; Seniunaite and Vasarevicius, 2017; Šyc et al., 2020). With regard to imports, the quality of ICW has been improving, although it has been restricted in accordance with the "Green Fence" Policy, as described by Wang, J. et al. (2019) and Dong et al. (2020a).

In this study, historical copper waste generation from EoL copper products in

China is explored using a stock-driven MFA method in which in-use stocks and final domestic copper demand are estimated based on several drivers, including GDP, population, urbanization rate, Chinese government policies, copper content and lifespans, as presented by Dong et al. (2020a). The lifetime distribution of copper products is modelled based on a Normal distribution. To model the Chinese copper waste management system, we considered the following variables: collection rate (CR), formal processing rate (FPR), informal processing rate (IFPR), processing rate of incineration & treatment of ash (IPR), fractions of formal (FoFC) and informal (FoIFC) collection, fraction of reuse from formally collected copper (REoF), fraction of reuse from informally collected copper (REoIF) and fraction of formal sorting and dismantling from informal collection (FSoIC), as shown in Table 5.1. Detailed definitions of these variables are to be found in Appendix 1.

To investigate the future scope for moving towards "Zero waste" or a circular economy with respect to the copper cycle, several authors have developed scenarios to assess the effectiveness of such options as reduced copper content, lifetime extension, increased recycling rate and improved copper production and manufacturing efficiency (Ciacci et al., 2020; Pfaff et al., 2018; Rötzer and Schmidt, 2020). However, the data used in most of this literature, such as an assumed future average copper recycling rate of 85%, are theoretical values and there is no indication whether it is technically feasible to actually achieve such percentages.

In this paper we examine various sources including peer-reviewed papers, as well as technical reports and patents, to find process data on these circularity options and come up with more realistic estimates of the level of copper circularity that might be achieved in the future. Based on this information we defined two scenarios from 2017 to 2100: the Chinese Policy (CP) scenario and the Technical & Circular (TC) scenario, as shown in Table 5.1. The CP scenario assumes that the future technologies used throughout the copper life cycle remain equivalent to the practical levels of 2017; for example, the efficiency of copper waste management is assumed to remain constant, with 2017 levels of reuse and recycling. In the TC scenario there is significantly improved circular use of copper, facilitated by diffusion of novel technologies currently available at laboratory or pilot scale but with the potential for future application at industrial scale. For this scenario, future copper waste generation from EoL products was estimated by calculating annual product flows to in-use stocks, the resulting vintages of in-use stocks and waste outflows based on the projected lifetimes of products that flowed in the past to stocks. Future copper demand is based on the cohort of stocks and assumes, further, increased use of copper owing to the energy system transition and related electricity infrastructure (Figure S5.2). In the TC scenario, copper product lifetimes are assumed to be extended, while in the CP scenario these remain unchanged, as shown in Table S5.1. In the CP scenario, the aforementioned variables CR, FPR, IFPR, IPR, FoFC, FoIFC, REoF, REoIF, FSoIC are assumed to remain at 2017 levels. In the TC scenario, improved processing rates and reuse fractions of each type of waste are assumed, based on information on improved separation and processing techniques, while it is also assumed that policies will be implemented to encourage higher CRs. The collection rates of copper from C&DW and ELV are assumed to be 95% in 2100 in the TC scenario, considering that these two waste categories can be collected and managed by professional companies and can achieve very high rates in China, similar to those in other countries (Graedel et al., 2004b; Pfaff et al., 2018; Ruhrberg, 2006; Yoshimura and Matsuno, 2018). For the other waste categories, future collection rates in the TC scenario were modelled based on the relationship between historical collection rate and waste generation rate, as described by Magalini et al. (2014). In view of China's proactive policies on reuse of ELV products, spare parts and components, the fraction of collected ELV copper reused (REoF) is assumed to be 50% in 2100. For the other waste categories, these rates are assumed to be twice the current level in 2100. The data and assumptions used in our model are summarized in Table 5.1. For details we refer to Appendix 1.

Table 5.1 Data used for modelling the copper waste management system in 2017 in the CP and TC scenarios, and efficiency improvement assumed in the TC scenario in 2100

Waste Management									
	Types	Collection	Sorting/dismantling				Recycling/incineration		
		CR	FoFC	FSoIC	REoF	REoIF	FPR	IFPR	IPR
CP scenario *(2017-2100) and TC scenario (2017)*	C&DW	81%[a]	100%		4%[b]		90%[a,c]		
	ELV	79%[a,d]	30%[e]		10%[f]	50%[e]	55%	55%[g]	
	WEEE	75%[h]	17%[i]	30%[i]	10%[i,j]	10%[i]	55%[g]	20%[k]	
	MSW	63%[g]	100%						20%[g]
	IEW	83%[g]	100%				75%[g]		
	ICW	100%[l]	100%				82%[l]		
TC scenario *(2100)*	C&DW	95%[m]	100%		8%[n]		97%[o]		
	ELV	95%[m]	100%		50%[p]	50%[e]	90%[q]		
	WEEE	88%[r]	80%[s]	100%	20%[t]		96%[u]		
	MSW	80%[r]	100%						80%[v]
	IEW	89%[r]	100%				96%[o,w]		
	ICW	100%	100%				96%[l]		

Note: Collection rate (CR), formal processing rate (FPR), informal processing rate (IFPR), processing rate of incineration (IPR), fraction of formal (FoFC) collection, fraction of reuse from formal collected copper (REoF), fraction of reuse from informal collected copper (REoIF), as well as fraction of formal sorting and dismantling from informal collection (FSoIC). a: (Soulier et al., 2018b), b: (MOHURD, 2005; Zhao and Rotter, 2008), c: (Zhao and Rotter, 2008), d: (Zhang, T. et al., 2014), e: (Chen et al., 2018; NDRC, 2008), f: (CELVE, 2019; NDRC, 2008), g: (Soulier et al., 2018b), h: (Zhang, S. et al., 2015), i: (Chi et al., 2014; Salhofer et al., 2016), j: (MEEC, 2006), k: (Liu et al., 2006), l: (Dong et al., 2020a; GACC, 2018), m: (Graedel et al., 2004b; Pfaff et al., 2018; Ruhrberg, 2006; Yoshimura and Matsuno, 2018), n: (MOHURD, 2019; NDRC, 2017), o: (Pita and Castilho, 2018), p: (CELVE, 2019) (MIIT, 2020; State Council, 2019), q: (Molteni, 2017), r: (Magalini et al., 2014), s: (Steuer et al., 2018), t: (CHARI, 2018; MEEC, 2018), u: (Meng et al., 2018; Pita and Castilho, 2018; Zhang et al., 2011), v: (Holm et al., 2018; Muchová and Rem, 2006), w: (Soulier et al., 2018a)

5.2.2 Future production of secondary and primary copper

To understand the effect of non-optimized and optimized copper waste management on the copper life cycle, production of secondary and primary copper were estimated. Secondary copper is produced from the aforementioned sources of EoL scrap (old scrap, including imported waste) and new scrap generated during the fabrication and manufacture of copper products. The present collection rate of copper (excluding new scrap) from fabrication and manufacturing is 89% and was assumed to remain constant in both scenarios in the future (Dong et al., 2020a). The current smelting and refining rate (SRR) of copper in secondary copper production is already quite high, with figures of 99%, 97%, 97%, 99%, 97%, 97% and 99% for C&DW, ELV, WEEE, IEW, MSW, ICW and new scrap, respectively. The future smelting and refining rates of all types of copper waste were assumed to remain unchanged in the CP scenario and to be 99% in 2100 with a regression analysis in the TC scenario. Historical imports and exports of semi-finished and finished products were quantified using MFA and their future development was modelled on past trends. Future primary copper production is given by the difference between inputs (secondary copper and net imports of semi-finished and finished products) and outputs (new scrap, fabrication and manufacturing losses and domestic final copper demand).

5.2.3 Modelling the environmental impacts of secondary copper production

The main goal of this assessment is to quantity the environmental impacts of secondary copper production from six types of waste in China. In this study, secondary copper production is broken down into several foreground processes including collection & transportation, mechanical processing (sorting & dismantling, recycling), secondary smelting and refining for each of the different waste streams separately, as depicted in Figure 5.1, while the LCA systems for separated waste streams can be found in Figure S5.4. Formal and informal recycling are not distinguished here for reasons of data availability. To allocate the environmental impacts in multifunctional processes, we used two different methods: mass-based allocation for collection & transportation and sorting & dismantling, and economic allocation for recycling and secondary smelting and the refining process, according to the economic value of the outputs (other than Cu, e.g. Fe, Al, Zn). The input data for the foreground processes of secondary copper production from different waste sources was scaled to 1 kg waste input from various processes, as summarized in Table 5.2. The economic data (e.g. price), materials content and processing efficiency of recycling for coproducts are reported in Tables S5.4-5.9 in Appendix 2. For recycling, it is important to note that the processing efficiency as defined here refers to material recycling and does not include any type of product (part) reuse. The energy and resource inputs associated with direct secondary production are defined as background processes, in line with the Ecoinvent database (V3.4) (Moreno Ruiz et al., 2017). The energy mix used for electricity production in background systems was set according to the share of fossil fuels and renewable energy in current Chinese electricity production, as shown in Figure S5.2. In addition, two of the CML2001 impact categories were used to conduct the analysis: GHG emissions and CED (CML, 2016; Guinée, 2002). For interpretation of the results, we conducted a contribution analysis to identify the contribution of each production process to total GHG emissions and CED. We also performed a sensitivity analysis on the choice of allocation methods and the influence of reuse fraction.

Next, to expand the assessment of 1 kg secondary copper production to include forward-looking developments in the production processes involved, processing efficiency improvements in foreground systems and changes of the electricity production mix in background systems were considered. For copper processing efficiency, changes in sorting & dismantling (refers to REoF), recycling (refers to FPR, IPR) and smelting & refining (refers to SRR) processes were assumed to be in line with the trends in Table 5.2 and descriptions in Section 5.2.2. The target processing efficiencies of co-products in recycling and smelting & refining processes in 2100 in the TC scenario were assumed to be enhanced to the same level as copper in 2100. If the 2017 level was already higher than the 2100 level in the TC scenario, however, processing efficiencies were assumed to remain unchanged. For the background systems, the future electricity production mix was assumed to be in accordance with China's electricity production roadmaps for fossil fuels and renewables resulting in lower GHG emissions over time, corresponding to the CP and TC scenarios, as shown in Figure S5.2.

Table 5.2 Input data of the secondary copper production by different waste sources, scaled to 1 kg waste input of different processes, copper content and processing efficiency can be found in Tables S5.4-5.9 in Appendix 2.

Waste category	**Process**	**Electricity (kWh)**	**Diesel (MJ)**	**Hard coal (kg)**	**Fuel Oil (kg)**	**Distance (tkm)**
C&DW	Collection & transportation					0.2231[a,b]
	Sorting & dismantling	0.0009[a, c]	0.0010[a,c]			
	Recycling	0.0140[a,d]	0.0295[a, d]			
	Smelting & refining	0.4070[e]		0.154[e]	0.0675[e]	
ELV	Collection & transportation					0.1560[b, f]
	Sorting & dismantling	0.0169[g, h]	0.1270[g, h]			
	Recycling	0.0187[f, g]	0.1529[f, g]			
	Smelting & refining	0.3990[e]		0.151[e]	0.0662[e]	
IEW	Collection & transportation					0.1532[a, b, i]
	Mechanical processing	0.0131[j]	0.0193[j]			
	Smelting & refining	0.4070		0.154	0.0675	
MSW	Collection & transportation					0.0133[b, i]
	Sorting & dismantling	0.0040[i, k]				
	Recycling	0.1236[i, j]	0.6986[i, j]			
	Smelting & refining	0.3990		0.151	0.0662	
WEEE	Collection & transportation					0.3265[b, l]
	Sorting & dismantling	0.0150[l, m]				
	Recycling	0.0414[l, m]	0.0481[l, m]			
	Smelting & refining	0.3990		0.151	0.0662	
ICW	Collection & transportation					0.3265
	Mechanical processing	0.0148	0.0002			
	Smelting & refining	0.3990		0.151	0.0662	

a: (Wang et al., 2012), b: (Moreno Ruiz et al., 2017), c: (Li et al., 2020), d: (Mercante et al., 2011), e: (Dong et al., 2020b), f: (Chen, Y. et al., 2019), g: (Liu et al., 2020), h: (Sato et al., 2019), i: (Hong et al., 2010), j: (Hunt, 2013; Schäfer and Schmidt, 2020), k: (Allegrini et al., 2015), l: (Song et al., 2013), m: (De Meester et al., 2019)

Finally, time-series of secondary copper production and the environmental impacts of 1 kg secondary copper production from six types of wastes were explored in the CP and TC scenarios. To estimate the aggregate environmental impacts of secondary copper production, the amounts of secondary copper produced from these six types of waste were multiplied by the corresponding impacts per kg and per future year, and summed to yield a total, given by Equation 1:

$$EIS_{x,t} = \sum_{x=1}^{n}(EI_{x,t} \times M_{x,t}) \quad (5.1)$$

where x represents the copper waste types (x=1, 2, 3…n), t refers to the time period, $EI_{x,t}$ is the GHG emission (kg CO2-eq./kg) or cumulative energy demand (MJ/kg) of producing 1 kg copper by each waste type x in year t, $M_{x,t}$ is the production of secondary copper by each type of waste x in year t, and $EIS_{x,t}$ is the total GHG emission (kg CO2-eq./year) or cumulative energy demand (MJ/year) of secondary copper production.

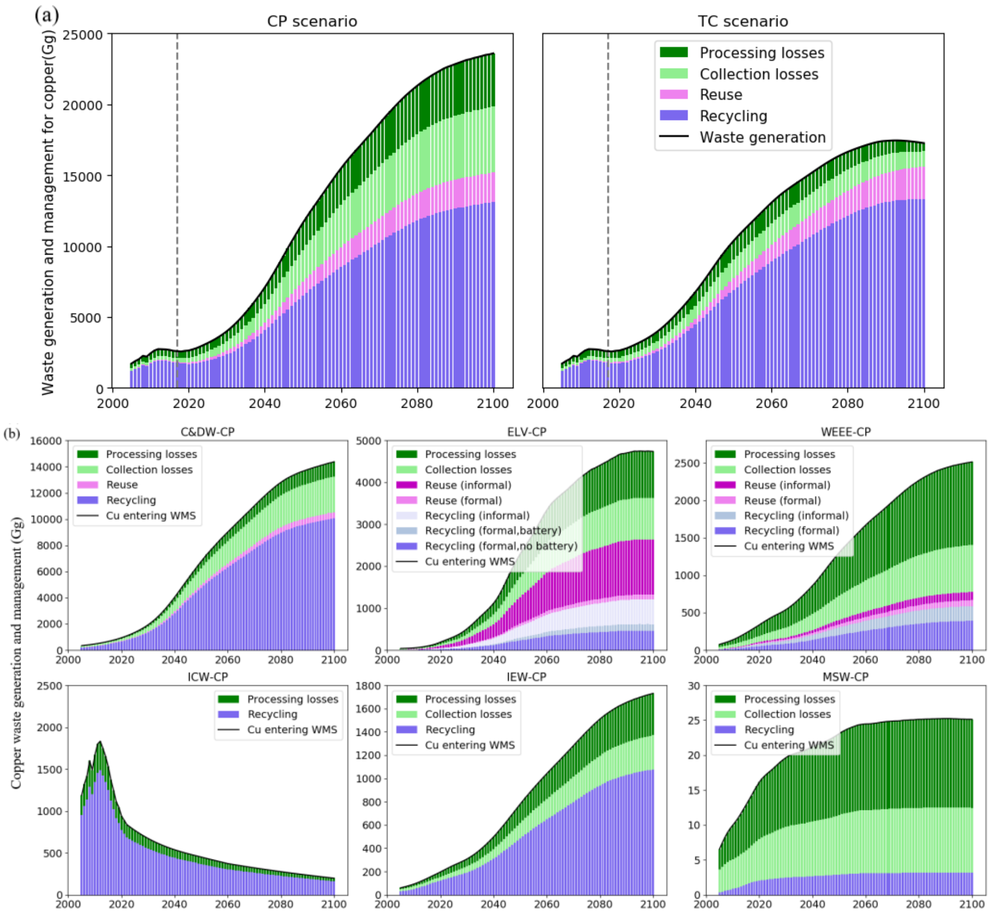
A detailed data and assumptions are available in supporting files (https://doi.org/10.6084/m9.figshare.14593737.v1).

5.3 Results and Discussion

5.3.1 The copper waste generation and management in China

Figure 5.2 depicts the historical (2005-2017) copper waste generation and management for six types of waste in China, followed by future projections up to 2100 under the CP and TC scenarios. Total copper waste generation shows an increasing trend in both scenarios, as a result of significant upward developments in socioeconomic conditions and demographics. Under the CP scenario, more waste is generated than that under the TC scenario, attributable to the assumption of the extended lifetimes of copper products in the TC scenario, which will reduce copper waste generation. Although in the paste ICW accounted for almost half the total amount of waste generated, because of present restrictions on the import of copper waste, the ICW category is expected to decline in the future. Consequently, copper in C&DW is expected to contribute most to the aggretate copper waste generation expectedly in the coming decades.

Looking beyond copper waste generation, copper waste management is expected to be considerably improved in China under the TC scenario. While the amount of copper waste deposited in landfills will increase in both scenarios from 2005 to 2100, in the TC scenario relative amount of copper waste losses are anticipated to be reduced substantially, from around 30% of total copper waste generated in 2005 to less than 10% in 2100, since more is kept in the economy by either reuse or recycling, while in the CP scenario this number will remain fairly unchanged. Moreover, with the higher recycling rates in the TC scenario, the amount of copper recycling is almost equivalent to that in the CP scenario in 2100, even though far less copper waste is generated in the TC scenario. The results also show that copper reuse in the TC scenario is even greater than in the CP scenario.



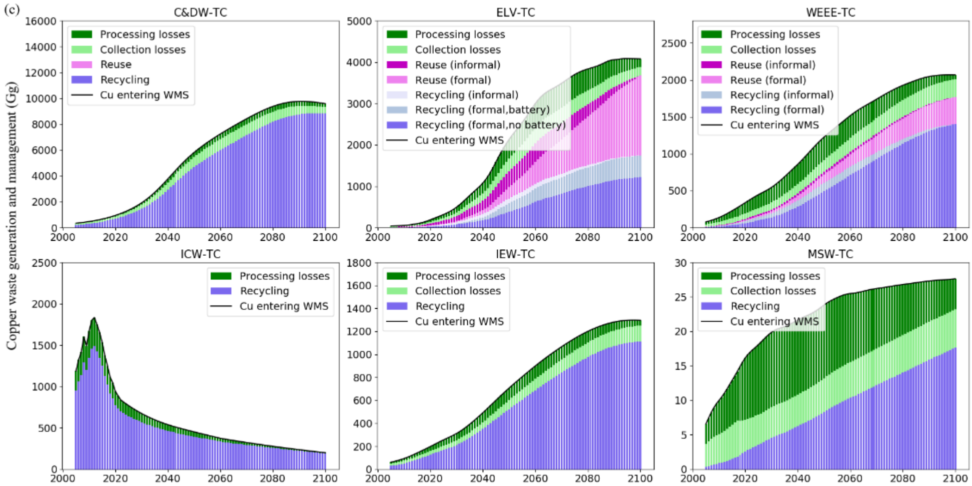
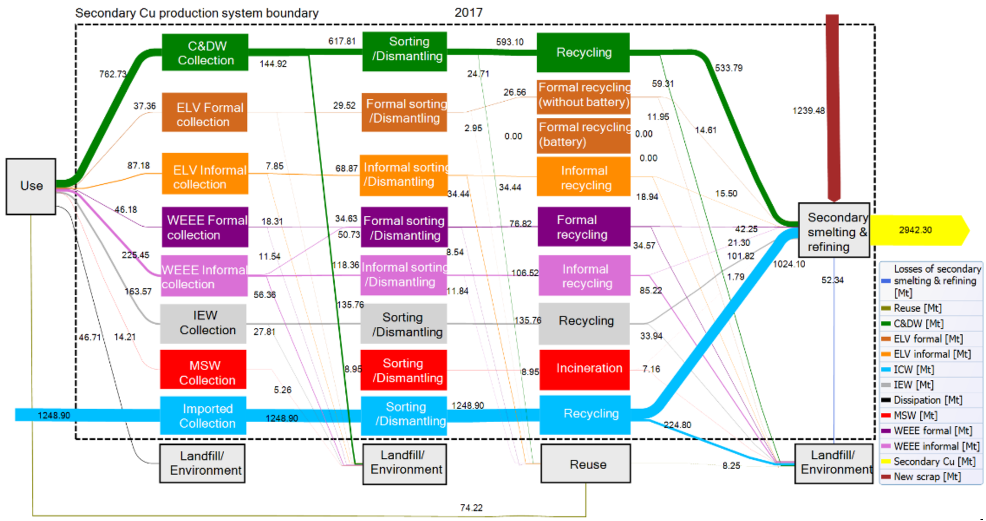
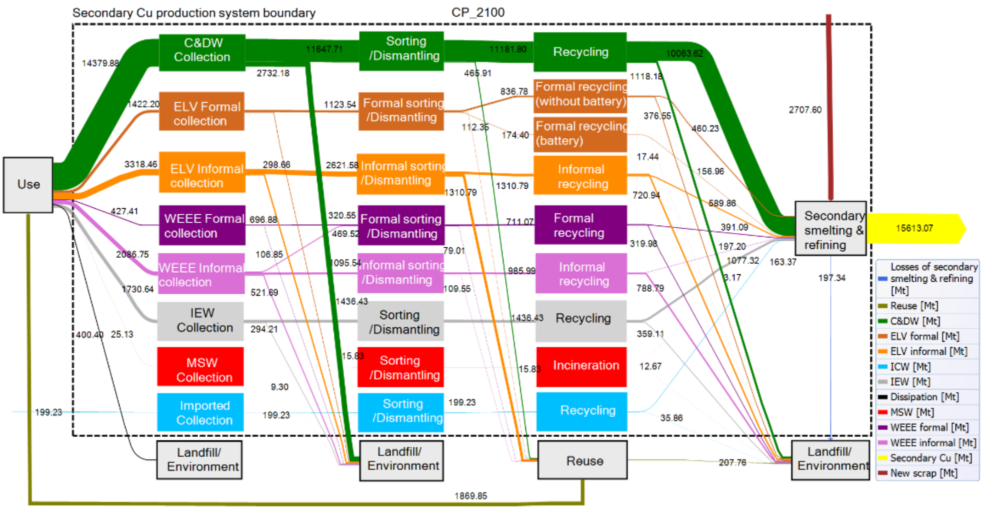


Figure 5.2 Copper waste generation and management from 2005 to 2100 in the CP and TC scenarios in China: (a) total copper waste generation and its destination, the vertical black dashed line marks the boundary between historical data and future scenarios, (b) by waste source in the CP scenario, (c) by waste source in the TC scenario.

Whether recycling is always the preferred option depends on the type of waste. Compared with other kinds of domestic waste, due to the high recycling rate of copper in C&DW and the high volume of C&DW generated, the copper recycled from this waste stream accounts for the largest proportion and will remain so in future in both scenarios. Copper recycled from ELV batteries is worthy of close attention. As shown in Figure 5.3, with the increasing uptake of electric vehicles, the share of copper recycled from ELV batteries in total copper recycling is expected to increase by 4% from 2017 to 2100 in the TC scenario. In certain sectors (ELV, WEEE) informal recycling has dominated in the past. In the TC scenario, however, professional recycling, which is much more efficient and far less polluting, is assumed to gradually take over, with informal copper recycling projected to disappear entirely by 2100. With regard to MSW, copper recycling from bottom ash, which has been challenging owing to technical limitations accounting for only 0.1% of copper waste generated in 2017, offers major scope for improvement in the future.

Reuse, including repair, remanufacturing and refurbishment, means reusing products or components again with the same function, and this can be quite an effective way of circulating copper before it finally enters the recycling process. In 2017 it accounted for only 3% of the total copper waste generated, however. There was a relatively important informal market for direct reuse of WEEE, in particular, responsible for over half copper reuse in 2017. In the TC scenario, reuse of copper components in ELV is very likely to already increase over the next few decades, with a major shift from informal to formal reuse.





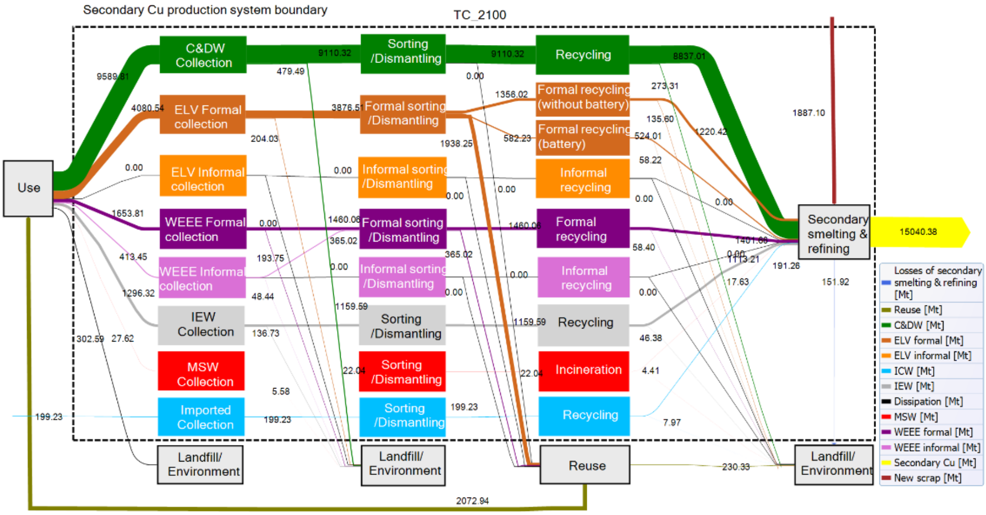


Figure 5.3 Sankey diagram of copper waste management and secondary copper production in China in 2017 and 2100 in the CP and TC scenarios

5.3.2 GHG emissions and energy demand related to per-kg secondary copper production

Figure 5.4 shows the GHG emissions and cumulative energy demand of 1 kg secondary copper produced from different waste sources in China in 2017. By waste type, copper production from MSW has the highest GHG emissions and cumulative energy demand. In terms of contributing processes, secondary smelting and refining account for the bulk of GHG emissions and cumulative energy demand for all types of waste. The differences of GHG emissions and cumulative energy demand among the six types of waste are due mainly to differences in impacts of mechanical processing and collection & transportation, which depend on the purity of the waste purity and the copper grade. Our analysis shows, furthermore, that the environmental impacts of producing 1 kg secondary copper from any waste stream are much lower than those of primary copper production, which obviously indicates the potential benefits of copper recycling. However, it is worth noting that this finding may not always hold, and due consideration will always need to be given to several key variables (e.g. waste quality, recycling technology, energy sources, geographical location) as well as modeling assumptions (e.g. allocation method) and analysis made on a case-by-case basis. For example, the cumulative energy demand of 1 kg secondary copper production from C&DW

is almost the same as that of primary copper production in Germany in 2014 (Schäfer and Schmidt, 2020).

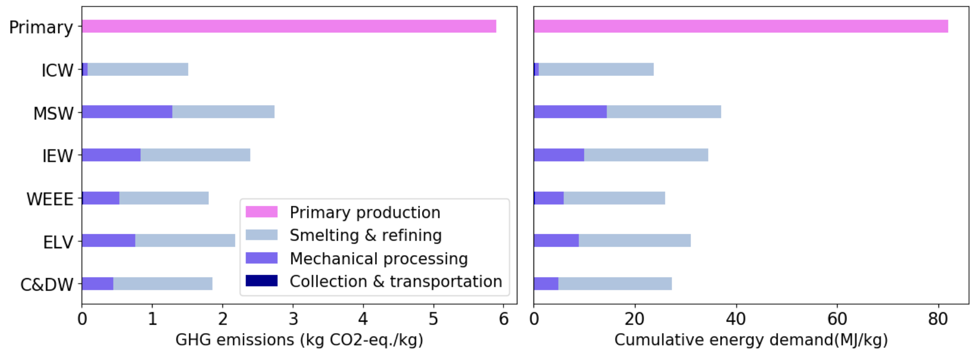


Figure 5.4 GHG emissions and cumulative energy demand of 1 kg secondary copper produced from different waste types in China in 2017, broken down into constituent processes, and comparison with production of 1 kg primary copper in China in 2017. The primary copper data is derived from Dong et al.(2020b) and represents an average value based on pyrometallurgical and hydrometallurgical production.

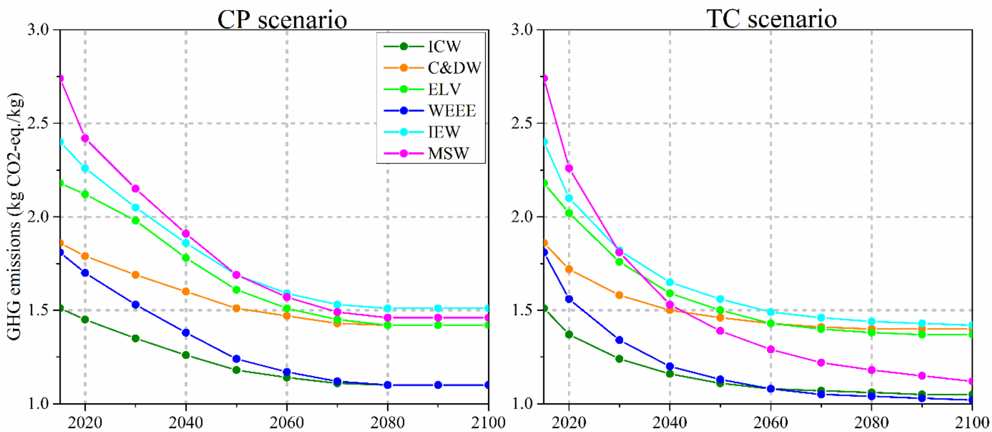


Figure 5.5 GHG emissions of production of 1 kg secondary copper from different waste types in the CP and TC scenarios (Figure S5.5 shows the results for CED).

Future GHG emissions and cumulative energy demand of producing 1 kg secondary copper from different waste types in China were also projected. The results are shown in Figure 5.5 for GHG emissions and Figure S5.5 for cumulative energy demand. In both the CP and TC scenario the GHG emissions and cumulative energy demand are expected to decline for all types of waste, with an unsurprising sharp decreasing through to mid-century and a relatively gradual decrease thereafter. This is especially true for MSW, a clear reflection of the decoupling of energy consumption and environmental impacts resulting from the energy transition (Ciacci et al., 2020; Guan et al., 2018). In the case of secondary copper production from C&DW, future potential reduction of GHG emissions and cumulative energy demand is modest in both scenarios, implying that the low-carbon transition (specifically electricity) and improved processing efficiency play a smaller role for this waste stream than in the other cases. Moreover, C&DW will almost certainly become the main contributor to the environmental impacts of aggregate secondary copper production from all types of waste, because it is the single largest source of secondary copper. The environmental impacts of secondary copper production from ELV are likely to decrease significantly in both scenarios, though the difference between the two scenarios is only minor.

Figure 5.6 compares the projected GHG emissions and cumulative energy demand of production of 1 kg primary, secondary and reused copper in China in the two scenarios. As can be seen, the future environmental impacts of 1 kg secondary copper production are still expected to be much lower than those of primary copper production, even when secondary production in the CP scenario is compared with primary production in the TC scenario. This finding holds not only for aggregate secondary production but also for secondary production from each type of waste considered in this study. It should be noted that in determining the environmental impacts of secondary copper production no distinction was made between formal and informal recycling. Although previous studies have generally posited that informal recycling with suboptimal treatment can cause a variety of environmental and human health issues, it is hard to argue that formal recycling outperforms informal recycling environmentally (Foelster et al., 2016; Hong et al., 2015; Vergara et al., 2016). With respect to reuse, this study has quantified the environmental impacts of production of 1 kg Cu-containing products for reuse, as reported in Table S5.10. Production of 1 kg metal for recycling and 1 kg Cu-containing products for reuse have the same cumulative energy demand based on the mass allocation method in sorting & dismantling process. However, the environmental impacts of re-used 1 kg copper is not assessed in this study since the re-used process (remanufacturing or refurbishment) is out of the secondary production system boundary. While assessing the environmental impacts of one specific material (e.g. copper) in a remanufactured products may be challenging, it is common knowledge that reuse, especially direct re-use as a second-hand product, is more environmentally friendly as compared to recycling since no new materials have to be processed (Zhang et al., 2020). The energy use embodied in a remanufactured product could range from 15% to 85% of that for a new product (Ardente et al., 2018; ICA, 2013). Xu (2013) has even pointed out that this is the best option for disposing of waste and reducing environmental impacts. There may sometimes be a trade-off, however, when products are reused but newer products are more energy-efficient, although this trade-off will become smaller as more renewables are used.

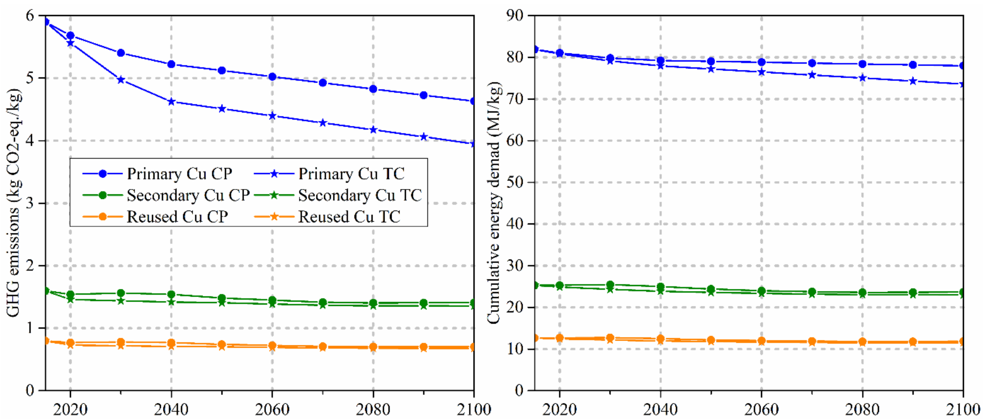


Figure 5.6 GHG emissions and cumulative energy demand of 1 kg primary and secondary copper production and comparison with copper reuse in CP and TC scenarios. The primary copper is derived from Dong et al. (2020b) and represents an average value based on pyrometallurgical and hydrometallurgical production in the respective CP scenario (Stated Policies scenario) and TC (Below 2 Degree scenario). Data for reused copper is assumed to be roughly 50% of secondary production (Ardente et al., 2018; ICA, 2013).

5.3.3 Impacts on optimization of copper waste management on copper cycle

Figure 5.7 shows projections of total secondary copper production in China and associated total GHG emissions and cumulative energy demand through to 2100. From the perspective of entire copper life cycle, overall copper demand and associated environmental impacts are obviously much lower in the TC scenario than in the CP scenario in 2100. This means optimized copper waste management system (TC scenario) is expected to not only mitigate the environmental impacts associated with copper ore extraction and processing but also those associated with copper waste disposal, which would lead simultaneously to dematerialization and improved environmental sustainability of the copper cycle in China. In the CP scenario, furthermore, total cumulative GHG emissions are very likely to increase approximately linearly, while in the TC scenario they are expected to gradually decline over the years, potentially leading to about 25% lower cumulative GHG emissions in the TC scenario in 2100 compared with the CP scenario (Figure S5.6).

Another interesting finding is that the GHG emissions and CED of copper production are expected to peak between 2040 and 2050 in both scenarios, attributable to a number of factors including copper demand, changes in the recycling system (e.g. recycling rate) and the Chinese energy transition, and probably also related to the climate target of carbon emissions peaking in China around 2030, indicating further net improvements in the decades thereafter.

As a result of the steadily improving recycling rates in the TC scenario, even though the volume of copper waste generated is far lower, aggregate secondary copper production is not that different compared with the CP scenario and even higher relative to the total copper supply. As a result, the dynamics of secondary copper production combined with the scenario projections of environmental impacts per kg of secondary copper produced result in similar outcomes in the CP and TC scenarios or in other words, the environmental impact reduction in the TC scenario is a result of the reduced demand for primary copper. Moreover, GHG emissions related to secondary copper production may in fact come to exceed those of primary copper production despite lower per kg GHG emissions of secondary production.

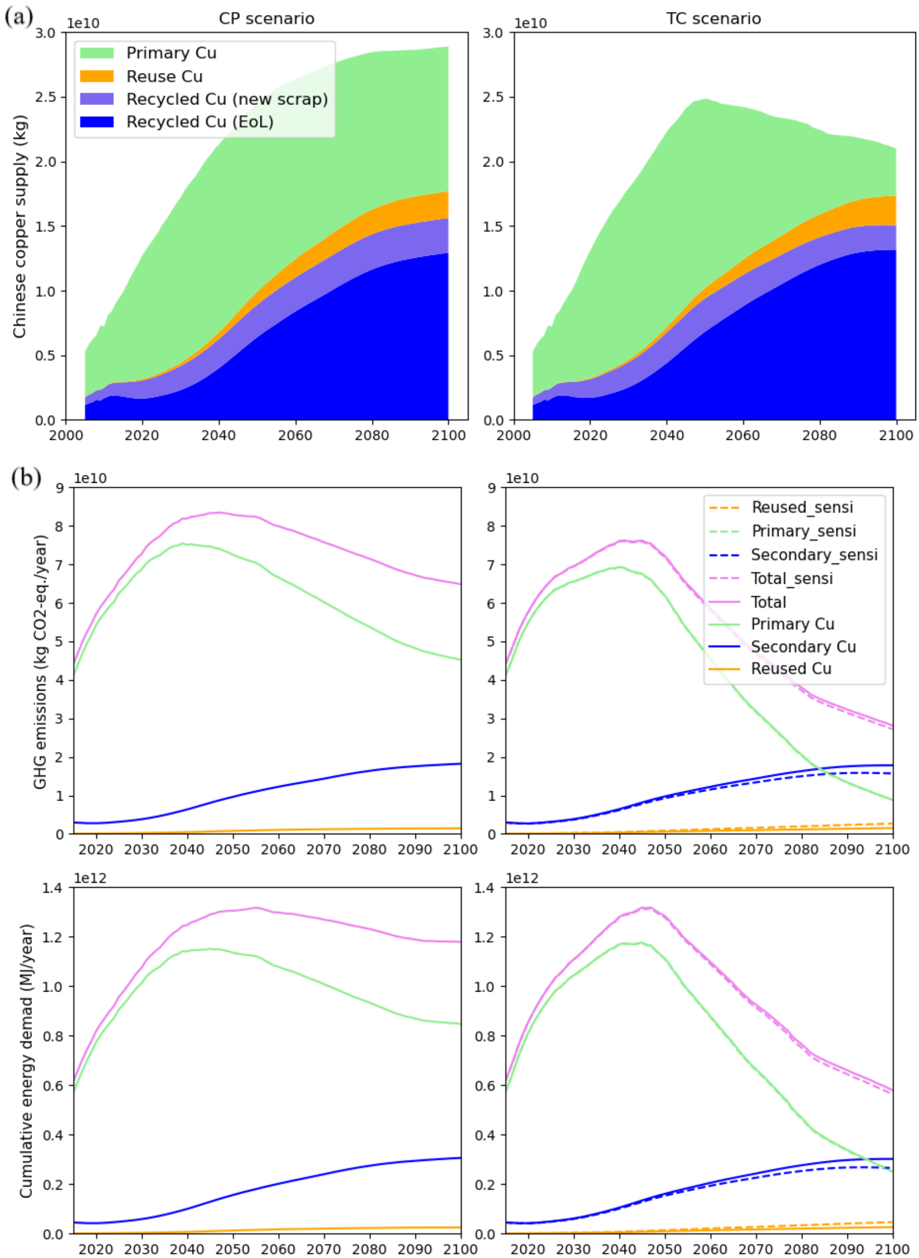


Figure 5.7 (a) Sources of Chinese copper supply in CP and TC scenarios, (b) GHG emissions and CED of primary, secondary and reused copper production. The dash lines in Figure (b) indicate the sensitivity analysis on reused copper that modeled based on the assumptions of increasing 60% of reuse copper (REoF) based on the level of TC scenario in 2100.

From the perspective of optimizing the use of copper (or other materials), reuse is better than recycling, since this extends copper lifetimes in original products, parts or components and results in reduced volumes of waste requiring treatment. At the same time, the environmental impacts of the total volume copper reused are far lower than those associated with secondary copper production, as calculated in this study. Given that the crucial significance of the modeling assumption with respect to copper reuse, a sensitivity analysis of the impacts of reuse fraction on copper production and associated environmental impacts was conducted. As Figure 5.7 shows, increased copper reuse could result in a reduction of GHG emissions and CED for secondary copper production and a slight reduction of those for total copper production.

5.3.4 Uncertainties and limitations

Scenario analyses and forward-looking perspectives can provide guidance and maintain progress of the copper cycle in terms of resilience and environmental sustainability, and anticipate related changes in waste management dynamics, thereby providing a basis for long-term critical assessment. At the same time, though, they involve significant uncertainties, the principal being the insurmountable limits on statistical data availability, especially with respect to copper reuse. Several key variables for modelling dynamic copper projections, such as demographics and economic drivers (e.g. GDP, population, copper content, urbanization rate) and other drivers related to production efficiency, have been discussed in previous studies (Dong et al., 2020a; Eheliyagoda et al., 2019; Soulier et al., 2018b; Wang, J. et al., 2019).

In particular, a change in the method used for allocating the multifunctional processes involved in copper production might result in very different environmental outcomes. A sensitivity analysis on mass allocation was therefore conducted, employing the mass allocation method for all multifunctional processes. As Figures S5.7 and S5.8 show, the environmental impacts of mass allocation are in line with the trends yielded by economic allocation. Because of the substantial spread in the price of recycled copper and co-products, however, economic allocation yields a greater spread of environmental impacts for certain waste streams compared with mass allocation.

5.3.5 Discussion and conclusions

This study has dynamically modelled copper waste generation and management, explored the environmental impacts of secondary copper production from different waste sources and investigated the impacts of these on the copper cycle in China. reducing copper waste and improving copper management require actions across the full product lifecycle, not merely the EoL stage. The TC scenario, as an optimized system, reflects a transition towards minimum waste generation, maximum copper recycling and improved environmental sustainability. To reap the full benefits in terms of resource efficiency and reduced environmental impacts, the challenge will be to manage this optimization appropriately.

Waste prevention should be the first priority. Extending the lifetimes of copper products is the prime direction to be considered, given the wealth of research indicating that this can reduce waste generation significantly, in line with the guiding principle of the "Zero waste" concept (Gharfalkar et al., 2015; State Council, 2018). Such a transition is not straightforward in China, however, especially for copper products with already long lifetimes, as in buildings and infrastructure. For products with shorter lifetimes, whether to extend the lifetime of the integral product (i.e. reuse) or only parts thereof (i.e. remanufacturing, refurbishment) or undertake recycling to keep the materials circulating longer than the product itself depends on the remaining qualities and function of the product concerned.

Reuse is preferable to recycling, but might be hard to implement in all China's industries. On the one hand, its success will depend very much on government policy and consumer acceptance of reused (including remanufactured) products. On the other hand, the increasing complexity of materials and product functions requires appropriate technologies to effectively and efficiently dismantle and remanufacture, which will undoubtedly become a huge challenge over time (Chang et al., 2017; Vanegas et al., 2018). Furthermore, high spare-part costs make remanufacturing of certain products unprofitable as well (Seliger et al., 2006). However, supporting the organization of reuse (second-hand markets, remanufacturing plant) centers and networks, including through enabling technologies, could motivate this important contributor to the successful implementation of "Zero waste" and the circular economy.

Recycling is the main option for utilizing EoL copper products in China at present, with informal recycling playing a major role in the ELV and WEEE sectors (Figure 5.3). An optimized waste management system, represented in this study by the TC scenario, aims to maximize the flow of copper to the formal recycling sector and then to dismantle and separate uniformly, leading to maximum recycled material and environmental benefits. As mentioned before, however, formal recycling is not always necessarily more environmentally beneficial than informal recycling. Furthermore, decisions to formalize recycling procedures need to consider not only resources and the environment, but also social and economic impacts. Several studies have demonstrated that in addition to the challenge of implementing policies to combat informal recycling, the employment afforded to informal workers and the profits accruing from recycled products are factors that also need to be considered, potentially complicating this transition (Chi et al., 2011; Linzner and Salhofer, 2014; Steuer et al., 2018). With regard to increasing the copper recycling rate, enhancing the collection rate is probably the most important strategy for maximizing recyclables. A waste collection system needs to be construed as a socio-technical system, aligning people's decision-making to policy goals. Collection rates are a function of consumer behavior. Troschinetz and Mihelcic (2009) have suggested that the willingness of consumers to collaborate to the collection process depends on their level of environmental awareness. In addition, given the projected benefits of reduced pollution (e.g. toxic gases, slag), hydrometallurgical technologies for recycling waste circuit boards and lithium-ion batteries deserve greater attention, particularly as these have not yet been applied on any major scale in China (Liu et al., 2019; Perez et al., 2019; Wu et al., 2017).

Early-stage design plays a major role in determining whether EoL products (or parts, components or materials) are amenable to direct reuse, remanufacturing, refurbishment or recycling, from where whether they are reusable or which parts/components/materials should be removed needs to be think over holistically to anticipate minimum waste at their end of life (Ciacci et al., 2020; Ghisellini et al., 2016; Mendoza et al., 2017). Beyond the practical feasibility of recycling, the environmental impacts of different materials should also be considered in the design phase. It should be noted, though, that because the lifetime of certain copper products may be as long as decades, new designs will have no direct and immediate impact on waste management, although they will facilitate circularity in the future.

In an ideal "Zero waste" management system, copper from EoL products will be optimally re-utilized as input materials to minimize copper losses and environmental pollutions. In this study, the methods of dynamic MFA and prospective LCA are combined to model the copper stock-flow dynamics, indicate which copper recyclables from different waste sources can be reused or recycled under the present Chinese policies and more circular economy strategies, and investigate the primary copper savings and associated environmental trade-offs based on such optimization. Under the present Chinese policies, reuse (including repair, remanufacturing or refurbishment) and recycling of copper containing products will lead to a somewhat lower dependency on primary copper resources, as well as to lower total GHG emissions and energy demand. Maximizing such "Zero waste" options may lead to a further reduction, but GHG emissions related to secondary copper production may become larger than those of primary copper production despite lower per kg GHG emissions of secondary production. The notions of "Zero waste" or "circular economy" highlight the importance of secondary resources, and this study, while limited in scope, provides insights into future opportunities for improved waste management in China as well as some of the challenges involved.