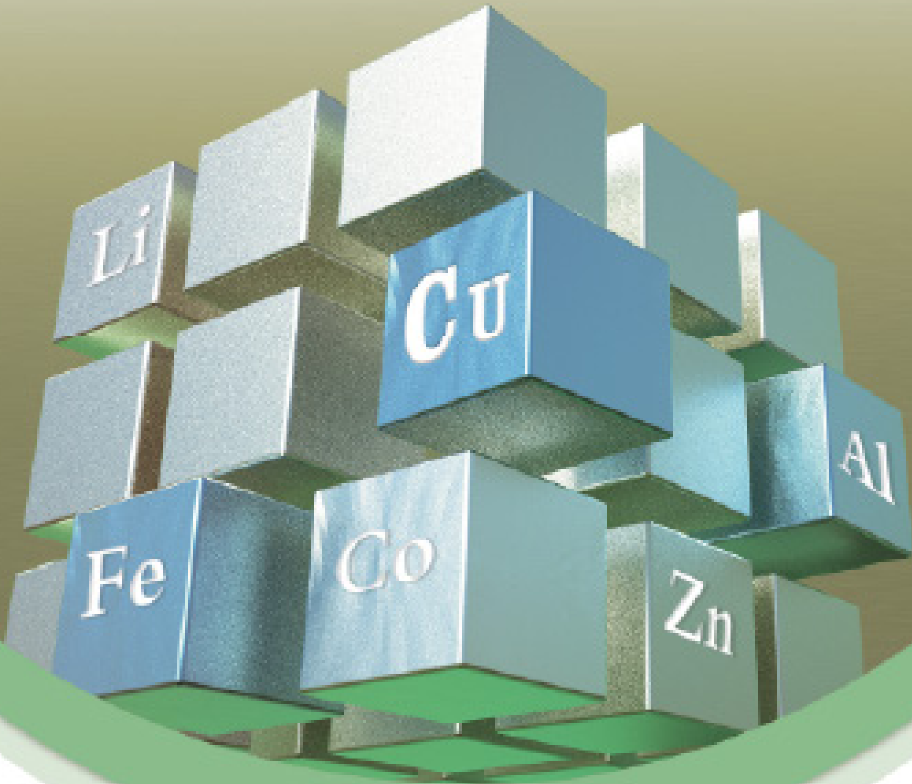
Towards a sustainable and circular metals economy: the case of copper in China

Dong, D.



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

Di Dong

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

Di DONG

董迪

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

PhD Thesis at Leiden University, The Netherlands

The research described in this thesis was conducted at the Institute of Environmental Sciences (CML), Leiden University, the Netherlands.

ISBN: 9789051915198

Cover Photograph & Design: Di Dong

Layout: Di Dong

Printing: GVO printers & designers B.V., Ede, The Netherlands

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

Proefschrift

ter verkrijging van
de graad van doctor aan de Universiteit Leiden,
op gezag van rector magnificus prof.dr.ir. H. Bijl,
volgens besluit van het college voor promoties
te verdedigen op woensdag 16 juni 2021
klokke 11.15 uur

door

Di Dong
Geboren te Jining, Shandong province, China

Promotor

Dr. E. van der Voet (Universiteit Leiden)

Copromotor

Prof. dr. A. Tukker (Universiteit Leiden)

Promotiecommissie

Prof. dr. ir. P.M. van Bodegom (Universiteit Leiden)

Prof. dr. ing. M.G. Vijver (Universiteit Leiden)

Prof. Dr. H. Rechberger (Technische Universität Wien)

Dr. Ing. L. Tercero Espinoza (Fraunhofer Institute for Systems and Innovation Research ISI)

Dr. T. Fishman (Universiteit Leiden)

Table of Contents

Chapter 1

Introduction

1.1 Material resource use and challenges

The global economy has developed rapidly in recent decades. From 1970 to 2019, the global population almost doubled, and GDP increased fourfold (World Bank, 2019; Worldometer, 2020). Global economic growth has been supported by the significant increase in the use of materials, which shows that materials use has approximately tripled during this period (OECD, 2019; Schandl et al., 2018). Materials such as fossil energy and metals are used in various fields, including infrastructure, construction, and transportation. Economic growth in these sectors will further increase the use of materials. These materials include the large and growing category of non-metallic minerals and metals in buildings and infrastructure. As reported by the OECD (2019), the global population is projected to reach more than 10 billion people by 2060, an increase of around 35% from 2017 up to that date. During this period, global GDP is again expected to more than triple. The global demand for materials is consequently expected to continue growing over the coming decades. This increased use of materials will allow developing economies to mature, but it will also bring with it a series of negative consequences.

One of the concerns in this scenarios is resource availability. Resource availability is critically important since we need to know whether the available resources will be sufficient to meet the growing demand. Many of the resources used in our society are non-renewable; some of these are abundant (e.g. sand) while some are scarce (e.g. critical metals) (Ioannidou et al., 2020; Müller et al., 2014). Resource availability depends on a range of aspects besides geological scarcity, including geopolitical developments and technology development (Asiedu, 2006; Schneider et al., 2014).

A second issue refers to waste. When materials use increases, this generally leads to more waste, and hence to the need for effective and efficient waste management to achieve sustainable development. Any society with a history of rapid economic growth faces the issue of sustainable waste management, and, given the expected increase in materials use, this issue continues to raise considerable concerns. Waste is increasingly seen as a resource to be mined through reuse, recycling and/or energy recovery (Geng et al., 2019). Previous work by Forti et al. (2020) reported that around 53.6 million tons of waste from electronics and electrical appliances were generated in 2019 worldwide, of which a mere 17.4% was recycled.

Last but not least, the extraction and use of materials cause serious damage to the environment, resulting as they do in greenhouse gas (GHG) emissions and local environmental issues related to toxicity, as well as water shortages (IRP, 2019). These negative environmental impacts occur at any stage in the life cycle of materials, including material extraction, processing, manufacturing, use and/or waste disposal. Many sources mention a decoupling of materials use from associated environmental impacts as a necessity for a sustainable materials management (Crane et al., 2011; Haberl et al., 2020; Kleijn, 2012; Schandl et al., 2016; Van der Voet et al., 2005).

These issues are even more challenging for emerging economies such as China. China accounts for 19% of the global population, but was responsible for approximately 40% of materials use in 2019 (Jiang et al., 2019; OECD, 2019). Since the turn of the century, the strong economic growth in China has driven a powerful development of urbanization and build-up of infrastructure. This in turn has resulted in a massive increase in materials use. From 1995 to 2019, China's materials use more than doubled, while the population grew by 15% and GDP by a factor of 7 (Wang, H. et al., 2019; World Bank, 2019; Worldometer, 2020). In the future, overall materials use is still expected to continue increasing due to the striving for higher living standards in China, although the increase is not expected to be as rapid as in the past 25 years (OECD, 2019). At the present time, the resource related per capita GHG emissions have nearly caught up with the global average. Since China is such a large country, it is now the country with the highest resource-related GHG emissions in the world (IRP, 2019). As the largest world player, China needs to take further steps to move toward a sustainable resources use.

1.2 Transition to a circular economy

The increasing problems related to resource extraction and use have given rise to policy initiatives promoting a transition from a linear economy to a more resource-efficient and circular economy (Figure 1.1). The linear economy normally follows the "take-make-dispose" characteristics established since the Industrial Revolution. In the linear economy, virgin materials specifically for metals are extracted from the lithosphere, smelted, refined and manufactured into various products. These products are used and discarded when they no longer serve their purpose. The linear economy has led to the present global problems related to e.g. resource depletion and GHG emissions as described earlier. By contrast, a circular economy aims to keep products, components and materials in use at their highest utility and value. By sharing or leasing, products can be used more efficiently. Repair, re-use, refurbishing and remanufacturing may increase product life spans substantially. By recycling, materials can be kept in use even after the product's lifespan finally expires. In that way, the services provided by in-use stocks of products in society can be maintained while reducing the input of virgin resources, thus alleviating the negative impacts of increasing materials use (Ellen MacArthur Foundation, 2013).

A report that issued by the Ellen MacArthur Foundation (2013) provided the following definition of the circular economy:

"Circular economy is an industrial system that is restorative or regenerative by intention and design. It replaces the End-of-life (EoL) concept with restoration, shifts towards the use of renewable energy, eliminates the use of toxic chemicals, which impair reuse, and aims for the elimination of waste through the superior design of materials, products, systems, and, within this, business models."

The concept of the circular economy is based on principles such as reducing, reusing and recycling. It reconsiders waste as input, aiming to close the loop to reduce the dependence on new materials and decouple economic growth from the extraction of natural resources (Scheel et al., 2020). Many countries are developing in line with the concept of the circular economy, including the EU's *Circular Economy Action Plan* (2015) and the *European Green Deal* (2020), and the Japan's *Basic Act on Establishing a Circular Society* (2000). These policies generally involve several sectors, contain multiple goals including enhancing resource efficiency, and indicate a series of strategies to achieve these goals.

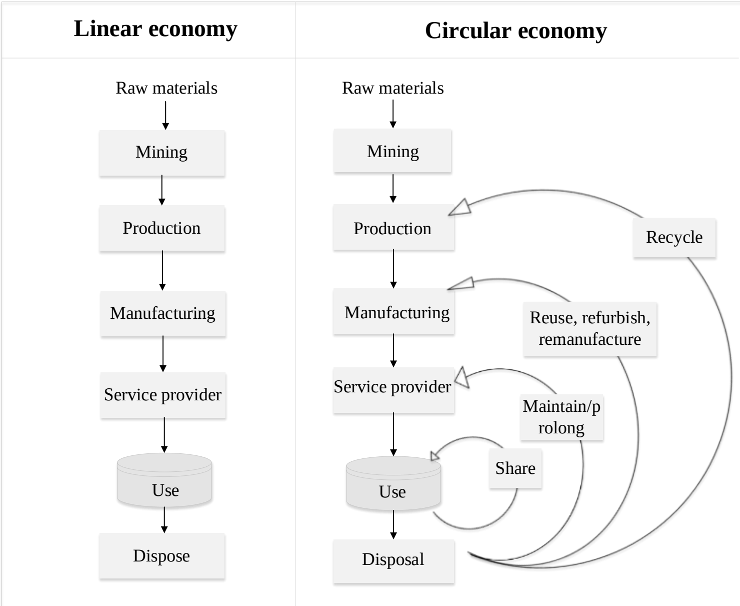


Figure 1.1 Schematic representation of the transition from a linear economy to a circular economy

China is a country that also has advanced circular economy policies (Zhu et al., 2019). The concept of the circular economy was introduced into China at the end of the last century. Since then, China has successively implemented a series of laws and regulations on the circular economy. In order to prioritize the policy initiatives in the field of environmental protection, the *Cleaner Production Promotion Law* was introduced in 2003 and updated in 2012. This law advocates improving resource utilization and reducing pollution at the source. It targets production techniques to reduce losses and emissions of pollutants in production and service sectors to reduce or eliminate health-related human and environmental impacts. From 2003 to 2008, several successful pilot projects on the circular economy were conducted to achieve higher resource efficiency and more economic benefits (Piatkowski et al., 2019; Yu et al., 2018). In 2009, China launched the *Circular Economy Promotion Law* to encourage the implementation of circular economy in industry, through investment, technical support, subsidies and other means. Following this law, China implemented the first national special plan for the development of a circular economy in 2013, *Circular Economy Development Strategy and Short-term Action Plan*. This plan focuses on waste utilization (e.g. recycling, remanufacturing), green buildings, green transportation system and green consumption. Since then, several other regulations and policies have been implemented, including circular economy measures in general regulations such as the *Five Year Plan* and *Work Plan for the Pilot Program of "Zero Waste City" Building*, and policies for specific sectors, such as the *Measures for the Management of End-of-Life Vehicles Recycling*.

Laws and regulations can be put into effect at micro, meso and macro level in China (Fan and Fang, 2020; Park et al., 2010). At the micro level, cleaner production, circular utilization of waste and eco-design are the main focal points. The *Clean Production Promotion Law* requires enterprises to increase the utilization rate of waste and achieve energy conservation and consumption reduction. Specifically, the *Regulations on the Management of the Recycling of Waste Electronic and Electrical Products* (initial version in 2011, updated version in 2019) and the *Implementation Plan of Extended Producer Responsibility System* (2017) pointed out that enterprises that produce electronic products are encouraged to put into practice the implementation of the extended producer responsibility (EPR) system. At the meso level, the priority is to facilitate the development of eco-industrial parks to achieve a goal of waste minimization with maximum resource efficiency through integrating management systems for water, material and energy (Geng and Doberstein, 2008). Finally, at the macro level, the laws and regulations provide an impetus for production and consumption activities to move towards a sustainable and circular economy.

In sum, Chinese circular economy policies aim for a more efficient use of materials, thereby reducing the need for primary resources as well as reducing pressure on the environment. The ideas from the circular economy policy to copper cycle in China will be applied in this thesis since copper is one of the major metals and used in many branches of modern technologies, which has made vital contributions to sustaining and improving society but also been a great concern for negative consequences such as described in section 1.1, particularly in view of declining copper ore grades (Eheliyagoda et al., 2019). Furthermore, the development of a copper cycle has a series of challenges (e.g. huge copper demand, GHG emissions) and opportunities (e.g. improving recycling) that are typical of multiple metals cycles like iron and aluminum. A deep study of Chinese copper cycle on circularity can support its transition and may provide insights into other metals as well.

1.3 The historical development of the copper cycle at global level and in China

Copper is the third most consumed metal in the world after iron and aluminum and has some very useful qualities (USGS, 2021). Its high ductility allows it to be stretched and shaped into complex surfaces without breaking. Thus, products with a wide variety of shapes and sizes can be produced for use in wires. Its good thermal and electrical conductivity enables its use in electric and electrical applications, including power generation, power transmission and personal computers. Its high resistance to corrosion makes it suitable for use in tube for pipelines (ICSG, 2020). Along with copper alloys, such as brass and bronze, copper has also become an indispensable material in construction. In addition, the antibacterial characteristic of copper allows it to play a crucial role in pollution control. In view of these properties, copper has been instrumental for economic growth and well-being in society and will most probably remain so for the foreseeable future.

Copper is mostly extracted from natural mineral deposits. Globally, the geological copper reserves[1] were estimated at 0.87 billion tons in 2019, while the remaining resources of this important and valuable metal were estimated at 5.6 billion tons in 2015, of which around 2.1 and 3.5 billion tons of copper are contained in the identified and undiscovered resources, respectively (USGS, 2017). However, as copper demand continues to rise, more low-quality copper ores are being mined, leading to an overall decrease in ore grades (Crowson, 2012). Global copper production has been increasing over the past 30 years. Copper production consists of primary and secondary production. Primary copper production has led to a considerable transmission from the lithosphere to the anthroposphere. In 2019, global refined copper production reached 24.5 million tons, of which primary copper accounts for almost 80% (USGS, 2021). Huge copper stocks are now residing in infrastructure, buildings and vehicles and will at some point in time emerge as a source of waste for secondary copper production (Kuipers et al., 2018;

[1] The term 'reserves' and 'resources' refer to the definition from the USGS. (https://pubs.usgs.gov/periodicals/mcs2020/mcs2020.pdf)

Watari et al., 2020). However, several studies have shown that the increase in global copper demand is much higher than the growth of secondary copper supply, which indicates that the need for primary copper is increasing (Elshkaki et al., 2016; Schipper et al., 2018). Manufactured copper products can be divided into pure copper and copper alloy products, according to their copper content. Copper alloys are made of copper as the main alloying element and are fused with other auxiliary elements to further improve the function and processing performance of copper to meet different industries and products requirements for different uses and properties. Copper processing efficiency in manufacturing is very high, and its average loss rate is less than 1% (ICA, 2021).

In China, the (primary) copper reserves were estimated at 26 million tons in 2019, while the identified resources were estimated at 114.4 million tons up to 2018 (MNR, 2020). A decline in the average grade of copper ore has also been observed in China, which could be attributed to the high copper demand and improved mining technologies (CNMIA, 2019). China is the world's main producer of refined copper, accounting for about 40% of the global production in 2019. In the past two decades, copper production in China has increased by more than 7 times, with imported copper ores and concentrates accounting for a major part (MNR, 2020). In China, copper semi-finished goods and final products are far more export-oriented than Europe (OECD, 2019). As one of the emerging economies, China has also experienced a substantial growth in copper consumption and since 2002 has been the world's largest consumer of copper. Its share in the global copper demand more than doubled between 2002 and 2019, growing year by year during this period (ICGS, 2021; MNR, 2020). Copper waste generation has been increasing in recent years; however, even though the recycling rate for copper has been increasing and is somewhat higher in China than the global average, around half of the generated waste was still lost.

This trend is expected to continue in coming decade, at global level and in China, which has raised concern regarding the future availability of copper and the associated energy use and environmental pressures (Eheliyagoda et al., 2019; OECD, 2019). The increasing demand for copper indicates that copper production is expected to increase as well, in the form of primary and/or secondary copper. Primary copper supply depends on the aforementioned copper reserves and resources, as well as on copper mining technologies. Future exploration could lead to an increase in copper reserves and resources; innovative technologies may also increase the success in mining complex ores, and reduce environmental impacts. These deposits are likely to provide a part of the future copper supply.

However, energy consumption is normally more intensive in primary production than secondary production. By passing mining and beneficiation processes allows for a much lower energy use for secondary copper production (Hong et al., 2018; Northey et al., 2013). Furthermore, the declining copper ore grade will result in more energy being required for primary production (Ayres et al., 2003; Rötzer and Schmidt, 2020). Energy efficiency considerations therefore point in the direction of increasing the share of secondary copper production. This is particularly important in countries like China, where fossil fuels still account for a large part of the energy sources. This knowledge makes it all the more important that we ourselves realize that primary copper consumed today will be the source of secondary copper to be reused and recycled in due time in the future. With the foreseeable increasing copper demand on the one hand and the need to reduce environmental pollution on the other, it is likely that a combination of primary and secondary copper supply will be required in the future.

1.4 The link between circular economy policies and the copper cycle in China

In this thesis, the transition from a linear economy to a circular economy has been linked to copper in China. Table 1.1 shows China's circular economy measures, and their translation to copper. Recycling is strongly promoted as an effective way to alleviate resource scarcity and mitigate the environmental pressure associated with copper production and use. With regard to the secondary copper produced from EoL products, this consists of waste collection, sorting, dismantling, shredding, separation, secondary smelting and refining processes. The energy use for recycling copper from EoL products mainly depends on its purity and the efficiency of the recycling system, but in general the energy requirement in secondary production could be 85% less than that of primary production (Chen, J. et al., 2019). Most of the policies that propose to encourage and facilitate the construction of waste recycling system, such as the waste classification standard and measures for remanufacturing of vehicle motors, can be translated into terms of improving copper reuse and recycling. Copper recycling in China has improved considerably in recent years. However, due to the long lifetimes of copper products (an average 30 years), most of China's copper products have not entered their retirement period, resulting in a shortage of domestic copper scrap for the development of recycling industry (Wang, J. et al., 2019). Until recently, the imported copper scrap was the main source for secondary copper production in China. The implementation of the "Green Fence" policy in 2013 put a stop to this, leaving domestic scrap as the main source for secondary production. Even with the increasing copper recycling rate in China, only around 25% of the copper demand was met by recycled copper in 2019, while in Europe almost 50% of the copper demand was already being met by recycled copper (Ciacci et al., 2020; Soulier et al., 2018a). This suggests that it is necessary to further improve the recycling system and to consider other strategies.

Some policies listed in Table 1.1 focus on the extraction and processing of non-ferrous metals, aiming to enhance the mining rate, tailings utilization and energy efficiency. From 1992 to 2018, the loss rate of underground mining and open pit mining of copper decreased by 30% and 50% in China, respectively (CNMIA, 2019). Other policies aim to reduce demand by extending the lifetimes of products (e.g. electrical and electronic equipment). Promoting green design in the *Work Plan for the Pilot Program of "Zero-Waste City" Building* is another measure at an early stage of the product life cycle, which could also result in longer lifespans. The same is true of the measures aimed at improving product disassembly and recyclability to obtain more reuse and recycling of the materials at their end of life.

In some cases, policies are not specifically targeted; they are applicable to both primary and secondary copper producers. For example, the *Proposals of the Central Committee of the Communist Party of China on Formulating the Fourteenth Five-Year Plan for National Economic and Social Development and the Long-term Goals for 2035* aims to promote the clean, low-carbon, safe and efficient use of energy. Measures that serve to reduce the use of fossil energy (e.g. hard coal in electricity production) may indirectly cut down the environmental impacts of both primary and secondary copper production.

Table 1.1 Circular economy policies and implications for copper in China

Policy title	Content	Implications for copper	Year
Cleaner Production Promotion Law	• Conserve resources, reduce energy consumption and the emissions of key pollutants. • Adopt reasonable exploration and mining techniques for mineral to protect the environment, prevent pollution, and improve the level of resource utilization.	• Improvement of energy efficiency in copper production • Improvement of copper mining rate	2003, 2012
Circular Economy Promotion Law	• Non-ferrous metal enterprises will replace fuel oil with clean energy such as clean coal, petroleum coke, and natural gas. • Encourage and promote the construction of waste recycling systems, e.g. waste collection and utilization facilities. • Support the remanufacturing of parts from EoL vehicle (ELV) and engineering machinery.	• Reduction of fuel oil use in copper production • Increase of copper recycling rate • Increase of copper reuse from ELVs	2009
Circular Economy Development Strategy and Short-term Action Plan	• Accelerate the development of pressure leaching, bio-metallurgy and other technologies, processes and equipment for copper, aluminum and other minerals. • Promote the high-value utilization of recycled metals such as secondary copper and aluminum. • Reduce the comprehensive energy consumption in non-ferrous metal production. • Improve waste classification, collection, transportation and recovery. By 2015, the recycling rate of main categories including waste from metal, plastic, paper, vehicles, WEEE will reach 70%.	• Increased share of hydrometallurgical copper production • Increased share of secondary copper in copper use • Improvement of energy efficiency in copper production • Increase of copper reuse and recycling from ELVs, WEEE and engineering machinery	2013

Policy title	Content	Implications for copper	Year
	• Establish reverse logistics for recycling systems and construct remanufacturing industries.		
"Made in China 2025" strategy	• The comprehensive utilization rate of industrial solid waste will reach 73% and 79% by 2020 and 2025, respectively.	• Increase of copper recycling rate of industrial solid waste	2015
Work Plan for the Pilot Program of "Zero-Waste City" Building	• Guided by the green lifestyle, promote the reduction of domestic waste. Support the development of a sharing economy and reduce the wasting of resources. • Reduce generation and improve utilization of construction and demolition waste. • Focus on electrical and electronic products, implement the EPR system. • Promote green design to improve product disassembly and recyclability.	• Reduction of copper waste generation from buildings, infrastructure and WEEE • Increase the copper recycling rate • Increase copper reuse • Reduce of energy use in copper recycling	2018
China's Energy Development in the New Era	• Accelerate the increase in the proportion of non-fossil energy in energy supply.	• Improvement of renewable energy use in copper production and electricity production	2020
Measures for the Management of End-of-Life Vehicle Recycling	• Dismantled ELV engines, steering gears, transmissions, front and rear axles, and frames (also named "Five Assembly") that are eligible for remanufacturing can be sold to companies with remanufacturing capabilities in accordance with relevant state regulations and reused after remanufacturing. Dismantled parts (other than the "Five Assembly" of ELV) meeting the compulsory national standards and eligible for reuse can be sold as "Reused ELV Assembly".	• Increase in copper reuse and recycling from ELVs	2019

Policy title	Content	Implications for copper	Year
Catalogue for Administration of Import of Solid Wastes	• Restriction or ban of imported solid wastes.	• Reduction of imported copper waste	2018
Implementation Scheme of Municipal Solid Waste (MSW) Classification System	• By the end of 2020, laws, regulations and standards relating to waste classification shall be established in basic form. In cities implementing compulsory classification of waste the recycling rate shall exceed 35%.	• Increase of copper recycling from MSW	2017
Provisions on the Standards for Compulsory Retirement of Motor Vehicles	• The standard prescribes the compulsory retirement of motor vehicles meeting the standards for being discarded. • The standard specifies the maximum service life and mileage of motor vehicles.	• Lifetimes of copper-containing products (vehicles)	2013
Discarded Household Appliance and Electronic Product Pollution Control Technology Policy	• Establish a relatively complete recycling system for WEEE, adopt schemes conducive to recycling and reuse, and gradually increase the recycling and reuse rate. • Encourage use of modular design, lifetime convergence of components and parts, easy maintenance, easy upgrade design, etc., in order to extend product service life. • Encourage use of fewer different and more readily recyclable materials, use internationally accepted standards to mark parts (materials), adopt designs and processes that facilitate dismantling of waste products and improve recycling rate.	• Increase of copper reuse and recycling from WEEE • Lifetimes of copper-containing products	2006

1.5 Methods for analyzing the copper cycle and associated environmental impacts

Increased demand has led to an accumulation of considerable stocks of materials in the anthroposphere, and the recycling of materials from these waste streams has become more and more important (Müller et al., 2014). To analyze the activities that can help to realize a more circular use of copper, such as waste collection and mechanical processing, knowledge of material cycles with respect to geographic location, accumulated quantities, material content in products, energy requirement is needed. Analytical tools, such as material flow analysis (MFA) and life cycle assessment (LCA), are useful to quantify and evaluate these activities and the associated environmental performance, and therefore to provide potential policy measures and decisions on circular economy. Below we introduce these methods and how they can be combined to support the analysis of the potential for a circular use of copper in China.

1.5.1 Material flow analysis

MFA is a method for the quantitative analysis and evaluation of the input and output of materials in a system based on the main principle of mass balance (Brunner and Rechberger, 2003). The applications of this method in the field of sustainability and the circular economy have increased significantly in recent years, which provides a comprehensive and systematic account of a defined physical system to support decision makers (Eriksen et al., 2020; Geng and Doberstein, 2008; Virtanen et al., 2019). This approach evaluates different systems in terms of the system spatial scale (e.g. regional economy, national economy, eco-industrial park), materials (goods or substances), system temporal scale (e.g. static or dynamic) and inclusion of processes (e.g. the whole life cycle or a specific process).

Significant research has been conducted on metals cycles in the anthroposphere based on a MFA approach. Chen and Graedel (2012) reviewed the anthropogenic cycles of 59 elements at different scale levels, including the major engineering metals iron, aluminum, copper and lead. Specifically for copper, some research constructed a comprehensive technological copper cycle treating a series of life stages (e.g. mining and processing, manufacturing, use, and waste management) at different spatial and time levels (Bertram et al., 2003; Guo and Song, 2008; Spatari et al., 2002). With the development of emerging technologies, the MFAs of other metals, such as lithium and rare earth elements, were also investigated (Geng et al., 2021; Sun et al., 2017). Most of these MFAs refer to a “snapshot” of flows at a specific point in time. This provides some insight in the metabolisms of these metals but does not address the dynamics of resource use and resulting changes over time in stocks and flows. Estimations of past and future flows can be used to assess the influence of drivers for resource use and related environmental problems, and they can support investment planning in infrastructures for mining, production, and waste management (Müller, 2006). The methodology of dynamic MFA was developed by Baccini and Bader (1996), afterwards Baccini and Bader (1996) and Melo (1999) constructed a copper cycle system in the United States and an aluminum cycle system in Germany, respectively. Since then, a number of dynamic MFAs on retrospective and prospective analysis of metals have been established, which provide information on reservoir stocks and on the evolution of stocks and flows over time (Elshkaki and Graedel, 2013; Elshkaki et al., 2005; Eriksen et al., 2020; Glöser et al., 2013; Yan et al., 2013; Yoshimura and Matsuno, 2018).

1.5.2 Life cycle assessment

LCA is an analytical tool specifically designed to assess the environmental impacts relating to the whole life cycle of production of a material, including processes of virgin ore extraction, refining, manufacturing, use and waste management for metal, for example (Guinée, 2002; Tukker, 2000). The ISO framework of LCA provided a definition for LCA as “*The collection and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle*” (ISO, 2006a, b). The European Commission’s Integrated Product Policy Communication even identified LCA as the best framework for assessing the potential environmental impacts of products (European Commission, 2003). Although the ISO standard defines LCA and provides a general framework for conducting an assessment, it leaves much to interpretation by the practitioner. As a result, LCA studies have been criticized for producing different results for seemingly the same product, which can be attributed to methodological choices such as the inclusion of processes, the use of different databases, the use of different allocation methods, and the use of different impact categories. Nevertheless, when applied transparently it can provide very relevant information about a product or material over its life cycle.

LCA has been used to assess the environmental performance of metals, both for primary and secondary production (Farjana et al., 2019b; Tan and Khoo, 2005; Zamagni et al., 2013). The environmental impacts per unit (e.g. kilogram) metal production, such as steel, copper and aluminum, have been assessed by several scholars (Davidson et al., 2016; Hong et al., 2010). Tan and Khoo (2005) and Nunez and Jones (2016) studied the environmental impacts of primary aluminum production in a cradle-to-gate analysis. Others specified environmental impacts of secondary metal production (Chen, Y. et al., 2019; Mercante et al., 2011; Song et al., 2013). In several of these publications, a contribution analysis was added to identify hotpots in the production chain (Khoo et al., 2017; Weng et al., 2016). From such studies, it appears that closing the cycle will be highly beneficial, not only from the perspective of resource conservation, but also from an environmental point of view (Kuipers et al., 2018; Van der Voet et al., 2018).

1.5.3 Integration of MFA and LCA

The integration of MFA and LCA is advantageous and has been used in many studies, especially for complex systems that consist of multiple products and services. As stated by Brunner and Rechberger (2003), it is necessary to include environmental impact assessment in MFA. Using integrated combined LCA and MFA method, a more synergetic analysis of the system and its material flows can be provided. This is also relevant for the assessment of circularity options.

Combining MFA and LCA has been applied on waste management systems in a fair number of studies, focusing on materials and environmental impacts (Nakem et al., 2016; Padeyanda et al., 2016). This combination has particularly been used to study WEEE, estimating the environmental impacts of collection, mechanical processing and recovery processes, and identifying the potential strategies to optimize waste treatment (Assefa et al., 2005; Kiddee et al., 2013; Wäger et al., 2011). Some studies, such as Venkatesh et al. (2009), Rincón et al. (2013) and Rochat et al. (2013), have evaluated material requirements in systems such as infrastructure, construction and educational services, and at the same time assessed energy use or GHG emissions. A few studies analyzed the environmental impacts of metal recycling, with emphasis on GHG emissions, to support for shifting toward circular development (Farjana and Li, 2021; Sevigné-Itoiz et al., 2014). Even though many studies have been conducted with this approach in recent years, several challenges including system boundaries and allocation principles still need to be overcome.

1.6 Research aims and questions

Taking into account that most modern technologies rely on copper, it is of crucial importance to secure future supply by closing copper cycles, thereby also reducing environmental pressure. The motivation behind this research is to understand how the concept of the circular economy can be used to foster the sustainable development of the copper cycle in China. The aim of this research is therefore to explore *how the copper cycle in China can be transformed into a sustainable and circular economy*. This overall aim will be supported by answering the following research questions:

1. How are copper demand, in-use stocks and waste generation expected to develop under the current Chinese policies related to general economic development, the energy transition and ambitions with regard to circular economy? (Chapter 2)
2. How could China meet its future copper demand in the context of moving towards a circular economy, and how may this be affected by the import restrictions of copper scrap? (Chapter 3)
3. What are environmental benefits and drawbacks related to present and future copper production in China, and how could the environmental performance be improved in the future? (Chapters 4 and 5)
4. What is the potential to close the copper cycle in China? (Chapters 2, 3 and 5)

1.7 Thesis outline

The thesis consists of 6 chapters (see Figure 1.2).

Chapter 1 starts with the resources use and challenges on resources availability, waste generation and management and environmental impacts of resources production, and the need for resource efficiency and the transition to a circular economy. The Chinese circular economy policies are presented, and the idea of the transition is introduced to copper cycle, firstly describing the historical development of the copper cycle at global level and in China, and then linking the Chinese circular economy policies to the copper cycle. This chapter also presents the methods used in this thesis for analyzing the copper cycle, together with the research questions and the organization of the thesis.

Chapter 2 develops a dynamic stock model and business-as-usual scenario involving a bottom-up approach to analyze copper demand in China from 2005 to 2050 based on government and related sectoral policies. The results show that per capita and total copper demand are both set to increase substantially, especially in infrastructure, transportation and buildings. It will not be possible to close the copper cycle, even if all copper waste is recycled, in the period up to 2050.

Chapter 3 explores a circular economy scenario of copper demand from 2005 to 2100 in China based on a dynamic stock model, and compares this scenario to the business-as-usual scenario as in Chapter 2. This Chapter also assesses the influence of the "Green Fence" policy restricting the import of copper scrap. Based on the analysis, the additional measures to achieve a circular economy beyond the current Chinese policies could be to prolong the lifetimes of copper products and increase the processing efficiency of copper production. In combination with these measures and the establishment of a state-of-the-art and efficient copper recycling industry, secondary copper could satisfy the bulk of Chinese copper demand, which could be an opportunity for Chinese copper cycle to transition to a more circular economy.

Chapter 4 assesses the environmental impacts of copper production from 2010 to 2050 in China using a LCA approach. The technology routes of copper production considered in this chapter are pyrometallurgical, hydrometallurgical and secondary production. Several variables are considered to model future changes in the foreground and background systems of copper production, including declining copper ore grade, energy efficiency improvements of production processes and a changing background system as a result of the ongoing energy transition. Moreover, potential options are identified for improving the environmental performance of Chinese copper production.

Chapter 5 continues to investigate how to minimize the copper losses and environmental impacts of copper production in the future by designing an optimum copper waste management system based on various “Zero waste” strategies including waste prevention, reuse (repair, remanufacturing or refurbishment) and recycling. Copper waste streams are divided into six types in this chapter, including 5 types of domestic waste streams and an imported waste stream. These “Zero waste” strategies as well as the transition from informal recycling to formal recycling are considered as possible models of the waste management system. The environmental benefits and drawbacks of such an optimized waste management system for the copper cycle are also discussed.

In Chapter 6, all results are combined to present the answers to the research questions. Discussion and recommendations are provided for future research based on the findings of Chapters 2 through 5.

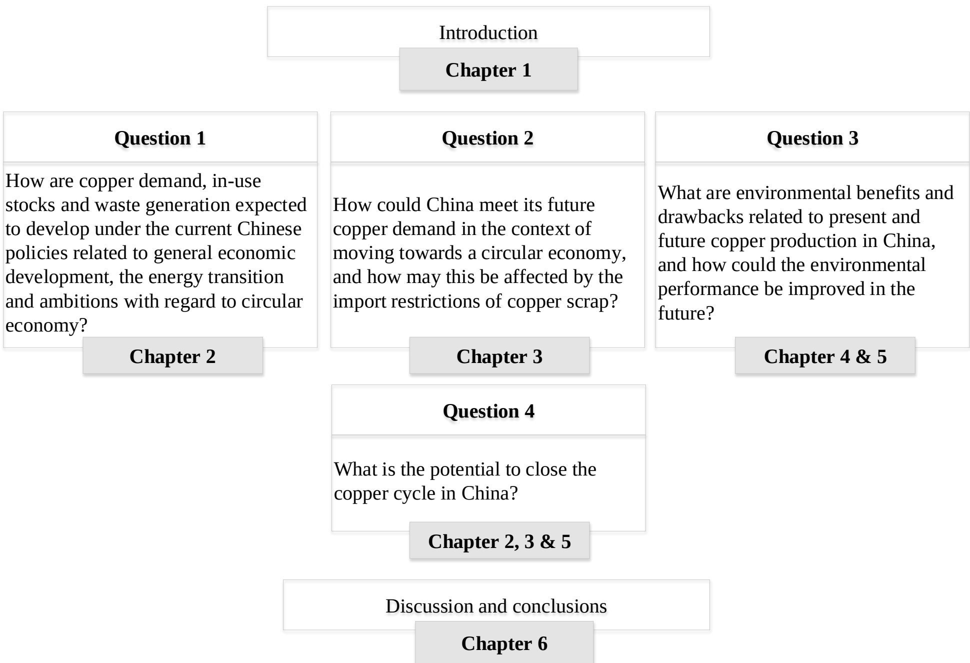


Figure 1.2 Conceptual scheme of the thesis

Chapter 2

Modeling copper demand in China up to 2050: a business-as-usual scenario based on dynamic stock and flow analysis

Abstract

A dynamic stock model and scenario analysis involving a bottom-up approach to analyze copper demand in China from 2005 to 2050 was developed based on government and related sectoral policies. In the short-term China's copper industry cannot achieve a completely circular economy without additional measures. Aggregate and per capita copper demand are both set to increase substantially, especially in infrastructure, transportation and buildings. Between 2016 and 2050 total copper demand will increase almost threefold. Copper use in buildings will stabilize before 2050, but the copper stock in infrastructure and transportation will not yet have reached saturation in 2050. The continuous growth of copper stock implies that secondary copper will be able to cover just over 50% of demand in 2050, at best, even with an assumed recycling rate of 90%. Finally, future copper demand depends largely on the lifetime of applications. There is therefore an urgent need to prolong the service life of end-use products to reduce the amount of materials used, especially in large-scale applications in buildings and infrastructure.

Published as: Dong, D., Tukker, A., & van der Voet, E. (2019). Modeling copper demand in China up to 2050: A business-as-usual scenario based on dynamic stock and flow analysis. Journal of Industrial Ecology, 23(6), 1363-1380. https://doi.org/10.1111/jiec.12926.

2.1 Introduction

In today's world, copper is an essential resource that is used in a wide range of applications. Because of its unique conductive properties, it is difficult to replace. With its rapid economic development in recent decades, China has experienced pronounced growth in copper production and consumption, becoming the world's largest copper consumer in 2002. Its share in global copper demand increased from 20% in 2006 to 46% in 2016, growing year on year during that period (Schipper et al., 2018; Yang et al., 2017).

This rapidly rising demand is not expected to slow down in the coming decades. This may cause future supply problems and contribute to environmental issues. The systemic solution to these issues is to use copper more efficiently and keep copper in closed loops wherever possible. Thus, an essential first step in this direction is to understand the country's copper flows and stocks.

There has been significant research into metals flows and stocks at both the global and national level using MFA. MFA studies in the past have focused mostly on flows, ignoring stocks. More recent approaches acknowledge the importance of the material stocks as a driver for flows. The inflow arises when applications discarded from the use phase must be replaced by new ones. Especially for long lifetime applications, the stocks as a driver are found to be very important (Chen et al., 2016; Guo and Song, 2008; Soulier et al., 2018b; Spatari et al., 2005; Zhang, L. et al., 2015a). In general, two different types of methods have been used to quantify flows and stocks of the major engineering metals, such as steel, copper, lead, zinc and aluminum (Chen et al., 2010; Davis et al., 2007; Graedel et al., 2004b; Igarashi et al., 2008; Liu et al., 2013; Yan et al., 2013). The first is the top-down approach. In studies using the top-down approach, the material cycle is assumed to be the driven by external drivers such as GDP, population and per capita income projections (Kapur, 2006; Soulier et al., 2018b). The approach starts from the material itself: amounts mined, and distributed over various categories of uses. Stocks in such an approach are often not included. If they are, they are mostly estimated as the difference between inflows and outflows accumulated over time. The other is the bottom-up approach, which quantifies both flows and stocks directly by identifying all the products that contain the material within the system boundary at a given time, quantifying the number of products in use, and arriving at material flows using data on the material content of these products (Müller et al., 2014). To date, most studies adopted the top-down approach (Ayres et al., 2003; Daigo et al., 2009; Glöser et al., 2013; Hedbrant, 2001; Ruhrberg, 2006; Zeltner et al., 1999). Due to the data intensity and the limited availability of the data required, only a few studies adopted the bottom-up approach to examine specific end-use sectors like buildings, transportation and infrastructure (Bader et al., 2011; Gerst, 2009; Ling et al., 2012; Zhang, L. et al., 2014). To compensate this part of research, it is essential to perform a more accurate estimation of copper stock by the bottom-up method. Meanwhile, several scholars have studied future generation of copper scrap and concluded that at some time in the next 30 years, China will face an explosion of copper-containing waste generated by the socio-economic system (Wang et al., 2017). This scrap can be used to meet rising copper demand, at least in part, which is an issue not covered by simple scenario studies of future demand.

Against this background, this paper therefore explores copper stock, demand and the potential of scrap copper for closing cycles in China, investigating whether China may be able to improve copper industry by combining increased copper demand with scrap recycling and a reduction of primary production. In this study, we address three questions:

1) How great will future copper demand be in different end-use categories in China?
2) How large is the in-use stock and how much copper waste will be generated in different end-use categories in China?
3) What is the potential to close cycles, or more specifically: to what extent can copper demand be met by scrap recycling?

To address these questions, we use dynamic stock modelling to estimate the in-use stock of copper-containing products through to 2050 based on a bottom-up approach. This is translated into copper demand under a baseline scenario representing developments as indicated by either extrapolating driving forces or applying assumptions based on Chinese government policy. In this paper, we thus work with a scenario that can be characterized as business-as-usual. With this scenario it can be assessed how the policies set by the Chinese government will affect the country's copper demand. In this scenario, no specifically copper-related policies are assumed. In the follow-up research, this scenario can be used as a baseline for comparing other scenarios that do contain specific resource-related policies such as circular economy policies.

2.2 Methodology and data

2.2.1 Dynamic stock analysis

The core aim of this paper is to predict copper demand in mainland China through to 2050. To this end, we distinguish six main categories of copper end-use: infrastructure, transportation, buildings, consumer durables, commercial durables and agricultural and industrial durables. These were further sub-divided into 29 product categories that include novel applications such as battery charging stations and "new energy vehicles"(see Table 2.1). In our bottom-up approach, these 29 product categories are used to estimate future copper demand. This detailed subdivision of products and inclusion of new applications differentiates our work from earlier studies by e.g. (Zhang, L. et al., 2015b).

In principle, calculating copper demand using the bottom-up method needs to factor in two important issues. The first is the change in in-use stock caused by changes in demographics, economic welfare and government policies. The second is EoL replacement of products. Taking into account these two factors, we use the following formulae to estimate the quantities of interest:

$$CS_{I,t} = \sum_{i=1}^{I}(P_{i,t} \times m_{i,t}) \tag{2.1}$$

$$CF_{i,t-l}^{in} = F_{i,t-l}^{in} \times m_{i,t-l} \tag{2.2}$$

$$CF_{I,t}^{out} = \sum_{i=1}^{I} CF_{i,t-l}^{in} \tag{2.3}$$

$$CD_{I,t} = \left(CS_{I,t} - CS_{I,t-1}\right) + \mathrm{C}F_{I,t}^{out} \tag{2.4}$$

Here, i represents the product of each sub-category, such as residential buildings, household appliances or vehicles. I is the total number of product categories. t is the model year. $\mathrm{C}S_t$ is the total copper stock in all products. $P_{i,t}$ is the physical quantity of each product used, such as the number of cars or mileage of railway. $m_{i,t}$ is the copper intensity of each product. $CF_{i,t}^{out}$ is the outflow of discarded or obsolete products i in year t, which equals to the sum of $CF_{i,t-l}^{in}$, the inflow of products use i in the year $t-l$, l is the average lifetime of each product i. $CS_{I,t}$ and $CS_{I,t-1}$ are the total in-use stock of copper in year t and year t–1, respectively. $CD_{I,t}$ is the total copper demand of all products I. An example was given in the Appendix 3.

Owing to a paucity of data, however, for certain product categories such as power generation and transmission and agricultural and industrial durables, equations (1, 2, 3, 4) were not used to calculate copper demand, this being estimated directly based on demand for copper products, as formulated in equation (5). In essence, future copper demand was estimated by taking future activity levels in a specific sector (e.g. electricity generation) from Chinese policy plans and multiplying this by figure for copper intensity. We use the method of (Schipper et al., 2018) to calculate the copper outflow.

$$CD_t = D_t \times m_t \tag{2.5}$$

where CD_t is the copper demand for power generation. D_t is the installed capacity of power. The detailed calculation methods applied for the each of the 29 products can be found in Appendix 3.

2.2.2 Scenario analysis

In this article only a business-as-usual scenario is developed. Future demand is modelled based on one or more of the following six drivers: GDP, population, urbanization rate, copper intensity, product lifespan and future activity levels under Chinese government policy, briefly discussed in a generic sense below. The method used to calculate future copper demand in each of the 29 categories is described in detail in the SI and summarized in Table 2.1 below.

Population. Population is a key driver of consumption. Because of slow population growth, in 2014 the Chinese government cancelled its "One Child Policy" and implemented a "separate two-child" and "two-child policy" to prevent populous decline. China is still the most populous country in the world, and will continue to maintain steady growth in the coming years. Researchers have undertaken projections of China's future population and explained related changes in the number of households (Han et al., 2011; Zalmon et al., 1998). Here we use the population figures projected by the United Nations Population Division, which indicates that the population of China will start decreasing from around 2030 onwards, as shown in Table S2.1 (Appendix 1).

GDP. In recent years, China's economic growth has slowed. Moreover, the Chinese government is shifting its sights from high economic growth to more sustainable economic development. China's future GDP has been enumerated in the study (Li and Qi, 2011), which cited from IEA (International Energy Agency, 2010) and UNDP (United Nations Development Program, 2009). The Chinese government aims to achieve a growth rate of 6.56% during the 13th Five Year Plan. In this study, figures for future GDP have also been taken from the United Nations Development Program. Past and future trends in per capita GDP are shown in Figure S2.1 and Table S2.1 (Appendix 1).

Urbanization rate. The urbanization rate is a crucial variable for the estimated in-use stock of products, as per capita product ownership is quite different in urban and rural areas of China. Under the 13^{th} Five Year Plan, the Chinese government seeks to achieve an urbanization rate of 60%. For the long term we compared the urbanization rate cited in previous research (Li and Qi, 2011) with the Chinese government's goal and worked with figures of 60% (2016 to 2020), 65% (2021-2030), 70% (2031-2040) and 75% (2041-2050). Past and future trends are shown in Figure S2.1 and Table S2.1 (SI Appendix 1).

Copper intensity. Copper intensity affects the copper demand of all end-use products. Although copper products have already been studied by many scholars, because of the huge variety of products and different manufacturing standards in various countries, very few data are in fact available on the exact copper intensity of each product. At the same time, owing to technological improvement, lifestyle changes and policy requirements, copper intensity is also changing. It is difficult to forecast how the copper intensity of individual products will change in the future. With the exception of buildings, copper intensity has therefore been assumed to remain unchanged. Detailed data on the copper intensity of each end-use product are reported in Table 2.1 and in the SI.

Lifespan. Product lifespan has a direct impact on outflows. Extending product lifespan can reduce waste generation and demand for primary copper. In China, many products have a very short average lifespan. Some scholars have estimated that urban buildings in this country have a life expectancy of approximately 30-40 years (Song, 2005). One study (Hu et al., 2010a) of housing stock assumed that its service life follows a normal distribution, while other scholars, focusing on vehicles, took their service life to be consistent with the Weibull distribution. Normal or Weibull distributions are often assumed for product lifespans. Which of these is adopted will affect the copper outflow projected for a particular time. However, compared with using the average lifespans of all copper-containing products, assuming a distributed lifespan serves mainly to make the calculation results smoother and has no great impact on values when long time series are involved. A reflection on the differences can be found from Figure S2.2 in the SI. This is also reflected in other studies (Maung et al., 2017; Spatari et al., 2005). While acknowledging that life span distributions are often used, it is our conviction that no major errors are introduced by merely using averages, as shown in Table 2.1.

Policy plans. The data used for forecasting future use of copper-containing products originate from the 13th Five-Year Plan, statistical data and the mid- and long-term plans for each sector of industry (Made in China 2025), reports by consultancy organizations and several other publications (Elshkaki and Graedel, 2013; Krausmann et al., 2017; Wiedenhofer et al., 2015). These policies and data serve as the basis for our business-as-usual scenario projection. We refer to Table 2.1, with detailed assumptions provided in the SI (Appendix 2).

Data source and detailed assumptions used for analysis can be found, in the online version, at https://doi.org/10.1111/jiec.12926.

Table 2.1 Categories of copper demand and drivers used for calculating future copper demand (for further details, see the online SI)

Category	Sub-category		Driver					
			GDP	POP	UR	CI	LP	PP
Infrastructure	Electricity generation		-	-	-	1-2 kg/kW	30	National Energy Administration of China; Entri Company
	Electricity transmission		-	-	-	0.00041 kg/kWh	30	National Energy Administration of China
	Electronic communication		√	√	√	890 kg/m³	20	-
	Charging Infrastructure		-	-	-	20 kg (point), 15,000 kg (station)	15	Electric Vehicle Charging Infrastructure Development Guidelines; Mid- and Long-term Development Plan for Automotive Industry
	Street and traffic lights		-	√	-	890 kg/m³	12 (traffic) 25 (street)	-
	Rail lines and metro systems	ordinary-speed rail	-	-	-	6.06 tonne/km	30	Mid-and-long-term railway network plan; 13th Five-Year plan; China National North Car company
		high-speed rail				8.07 tonne/km		
Transportation	Conventional cars		√	√	-	15 kg/unit	11	Traffic Management Bureau of Public Security Ministry
	Conventional buses		√	√	-	109 kg/unit	12	
	Trucks		√	-	-	20 kg/unit	11	-
	Motorcycles		√	√	-	1.55 kg/unit	12	China Association of Automobile Manufacturers
	Trains	freight-rail cars	√	-	-	115 kg/unit	26	

Category	Sub-category		Driver					
			GDP	POP	UR	CI	LP	PP
		passenger	√	-	-	1299 kg/unit	26	Mid- and Long-Term Railway Network Plan
		locomotive	√	-	-	2960 kg/unit	26	
	New energy vehicles	car	-	-	-	75 kg/unit	13	Technology Roadmap for Energy-Saving and New Energy Vehicles
		bus				250 kg/unit		
Buildings	Residential buildings		√	√	√	0.35-1.01 kg/m^2	20 (rural) 35 (urban)	Rapid Development of Urban and Rural Development
	Service buildings		√	√	√	-	40	-
Consumer durables	Air conditioners		√	√	√	8.19 kg/unit	13	China Household Electrical Appliances Association
	Refrigerators		√	√	√	2 kg/unit	12	
	Washing machines		√	√	√	1.8 kg/unit	10	
	TVs		√	√	√	0.52 kg/unit	9	-
	Microwaves		√	√	√	0.9 kg/unit	9	-
	Heaters		√	√	√	0.01 kg/unit	8	-
	Cell phones		√	√	√	0.0003 kg/unit	4	-
	Landlines		√	√	√	0.10 kg/unit	7	-
	Computers		√	√	√	0.08 kg/unit	7	-
	Range hoods		√	√	√	0.2 kg/unit	14	-
Commercial durables	Printers		-	√	√	0.18 kg/unit	7	-
	Landlines		-	√	√	0.1 kg/unit	7	-
	Fax machines		-	√	√	0.1 kg/unit	8	-
Agricultural, Industrial durables	Agricultural durables		-	√	-	16-30 kg/unit	15-22	Made in China 2025; 13th Five-Year Plan
	Industrial durables		√	-	-	23-53 kg/unit	15	

Note: The "-" in this table means that the driver is not used to projections for this product. UR: urbanization rate, POP: population, LP: lifespan (years), CI: copper intensity, PP: Future activity levels as projected in policy plans.

2.3 Results and discussion

2.3.1 Past trend of copper demand

Chinese copper demand has grown rapidly since 2004 as shown in Figure 2.1. Estimates based on dynamic material flow analysis show that it was almost 4 million tonnes (Mt) in 2005, almost 20% of global copper demand in that year, rising very significantly to more than 8 Mt in 2015, which is nearly 45% of global copper demand. In addition, the per capita copper demand stood at 6 kg in that year. Prior to the 1980s, China focused mainly on industrial and agricultural development. With the urbanization drive initiated in 2004, however, the consumption of copper in infrastructure and buildings increased rapidly, gradually becoming a major source of copper consumption. We compared our results on copper demand with the figures published by the China Non-Ferrous Metals Industry Association. For 2005 and 2015 these were approximately 4 million tons and 10 million tons, respectively (Bo Zhao, 2011; Non-ferrous metal industry operation report, 2015), slightly higher than our estimates in 2015. As explained in the Appendix 1, the estimation in this paper do not encompass all copper products, whereas it already contains most of the copper products, which determines the future development trend of copper demand.

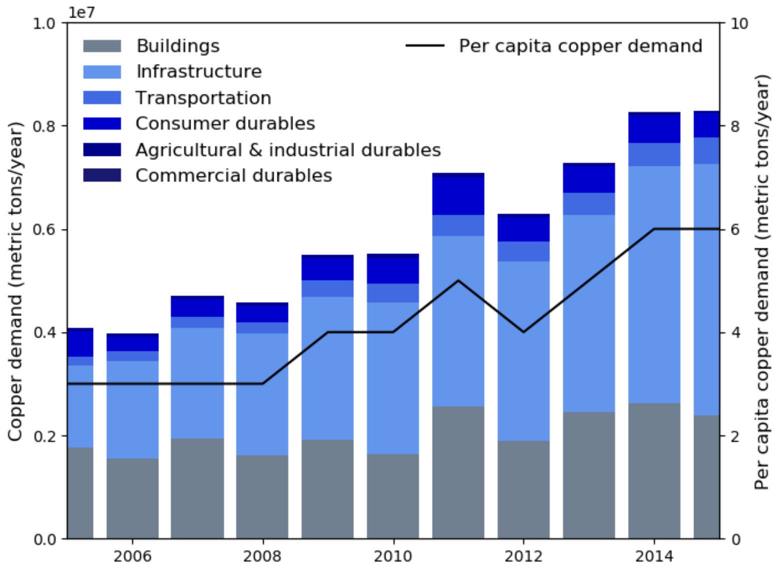


Figure 2.1 Chinese copper demand from 2005-2015

2.3.2 Future trend of copper demand

Figure 2.2(a) reports total demand for copper, demand per main category and per capita demand from 2005 to 2050. Total copper demand is expected to increase significantly over time, becoming about 6 times higher in 2050 than in 2005. The main end-use sector is infrastructure, accounting for around 50% of total copper demand by 2050, followed by transportation (25-30%), buildings (5–10%), consumer durables (5–10%), agricultural and industrial durables and commercial durables (both less than 1%).

This means the amount of copper used in infrastructure, transportation and buildings is expected to increase most. Given economic growth, national investment in infrastructure will continue to grow rapidly, especially in power facilities, railway construction (high-speed rail), and urban rail transit. The consumption intensity of copper in these applications is significantly higher than in other uses. In particular, the copper content of new power-sector equipment, including solar and wind, is very high. In addition, there is will be a pronounced surge in copper demand from the use of new charging piles and charging stations as use of new energy vehicles grows (Figure 2.2(b)).

In contrast to the substantial increase in copper use in infrastructure, demand for copper in the building sector is expected to maintain a steady but slow increase up to 2030, subsequently decreasing somewhat before slowly rising again from 2045 to 2050. This increased demand might be the results of four different developments:

- Rapid urbanization, requiring construction of large numbers of city dwellings.
- Increased per capita living space as a result of rising prosperity.
- Increased copper intensity of buildings due to safety measures, quality assurance and flexibility measures.
- Shantytown renovation: the Chinese government has adopted a three-year renovation programme for 2018-2020 to reconstruct 15 million sets of shantytown buildings.

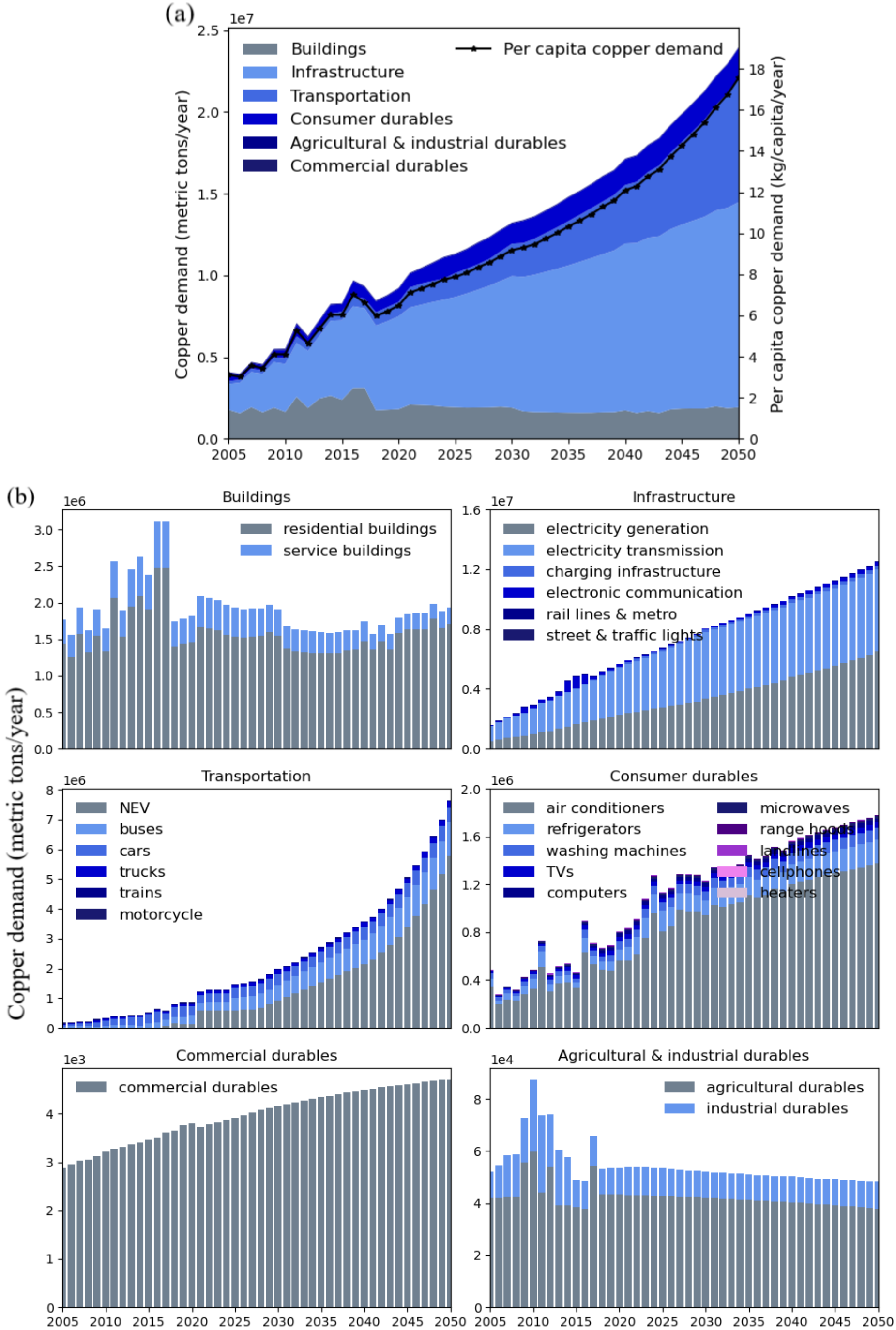


Figure 2.2 Copper demand of China from 2005 to 2050. (a) aggregate and per capita copper demand by end-use category; (b) detailed copper demand by product group within each end-use category

In other sectors, growth of copper demand is also evident in transportation, due to a projected rapid increase in use of new energy vehicles (NEV) with their relatively high copper content. While China has not implemented a ban on the sale of traditional-fuel vehicles, it has formulated a series of credit and subsidy policies to actively and systematically promote development of NEV.

Besides total copper demand, due consideration also needs to be given to per capita demand (Figure 2.2(a)). At present, this is less than 10 kg in China. As discussed in more detail in Section 4, this is about half that of other industrialized countries like Japan and South Korea, which indicates there is still substantial scope for growth in Chinese copper consumption. Our projections in Figure 2.2 indeed suggest that Chinese copper consumption will rise to about 18 kg per capita by 2050.

2.3.3 In-use stock and copper waste

From approximately 26 Mt in 2005, China's copper stocks are increasing very rapidly and are projected to reach more than 400 Mt in 2050. As shown in Figure 2.3(a), this increase is due mainly to the long-term growth of copper use in infrastructure, followed by buildings and transportation. In most of the sub-categories, such as electricity generation, cars and air conditioners, the copper stock is likewise set to increase until 2050 (Figure 2.3(b)). In that year buildings will still have the second largest copper stocks; while the number of service buildings will continue to increase until 2050, the residential building stock will likely stabilize from 2045 onwards. Infrastructure will hold the greatest copper stocks in 2050, with power generation and transmission the largest contributor. Our assessment suggests that the amount of copper used in this area will not yet have reached saturation in 2050.

There will also be major changes in per capita copper stocks. The total copper stock of 30.5 Mt in 2005 translates to about 23.3 kg per capita, but this will increase to 290 kg per capita by 2050. Literature studies on Chinese copper stocks show a similar trend (Qiang et al., 2012; Terakado et al., 2009b; Zhang, L. et al., 2015b). As discussed in more detail in Section 4 (see Table 2.3), all studies show that this value for China is much lower than in other developed countries.

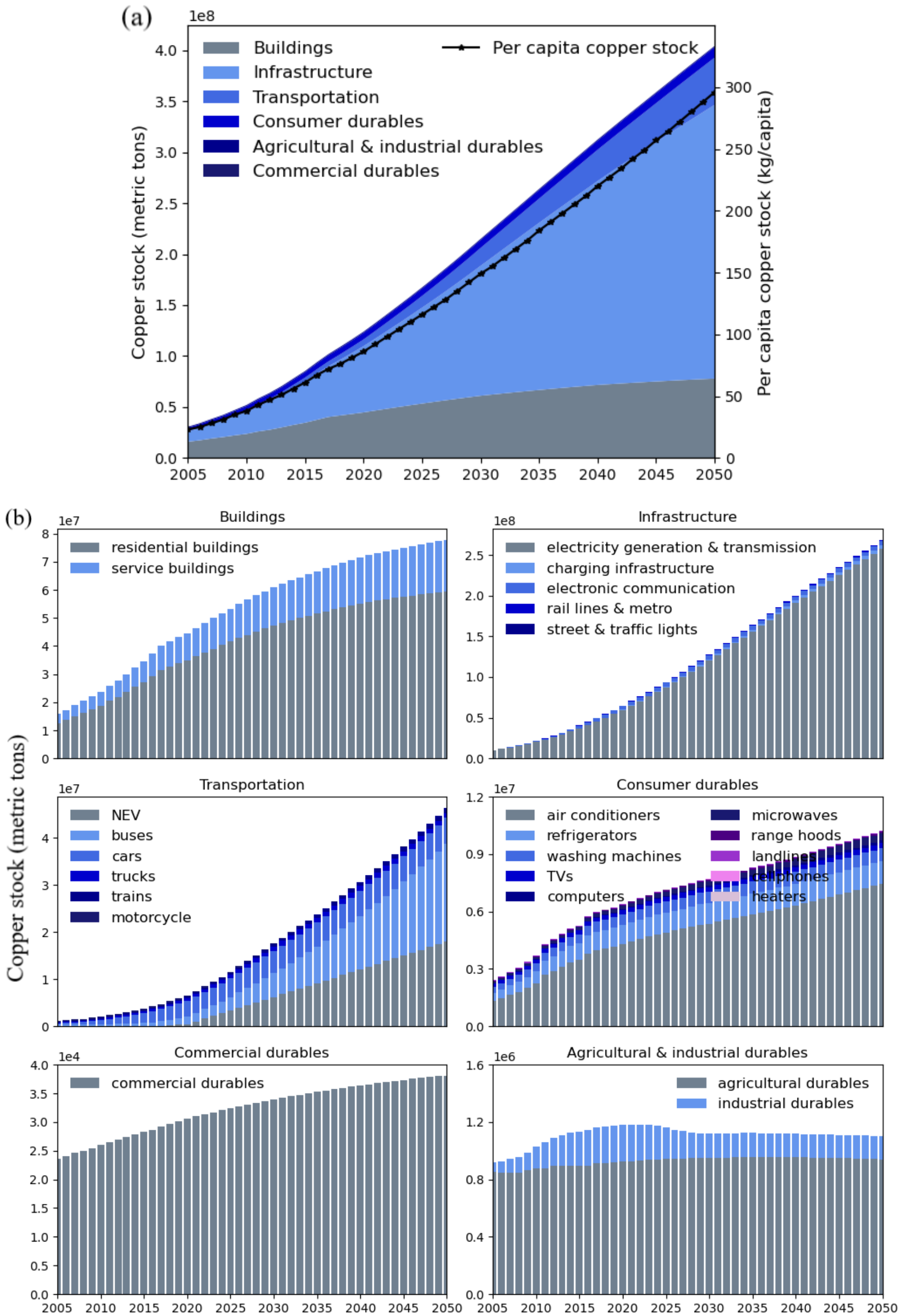


Figure 2.3 Copper stock of China from 2005 to 2050. (a) aggregate and per capita copper stock by end-use category; (b) detailed copper stock by product group

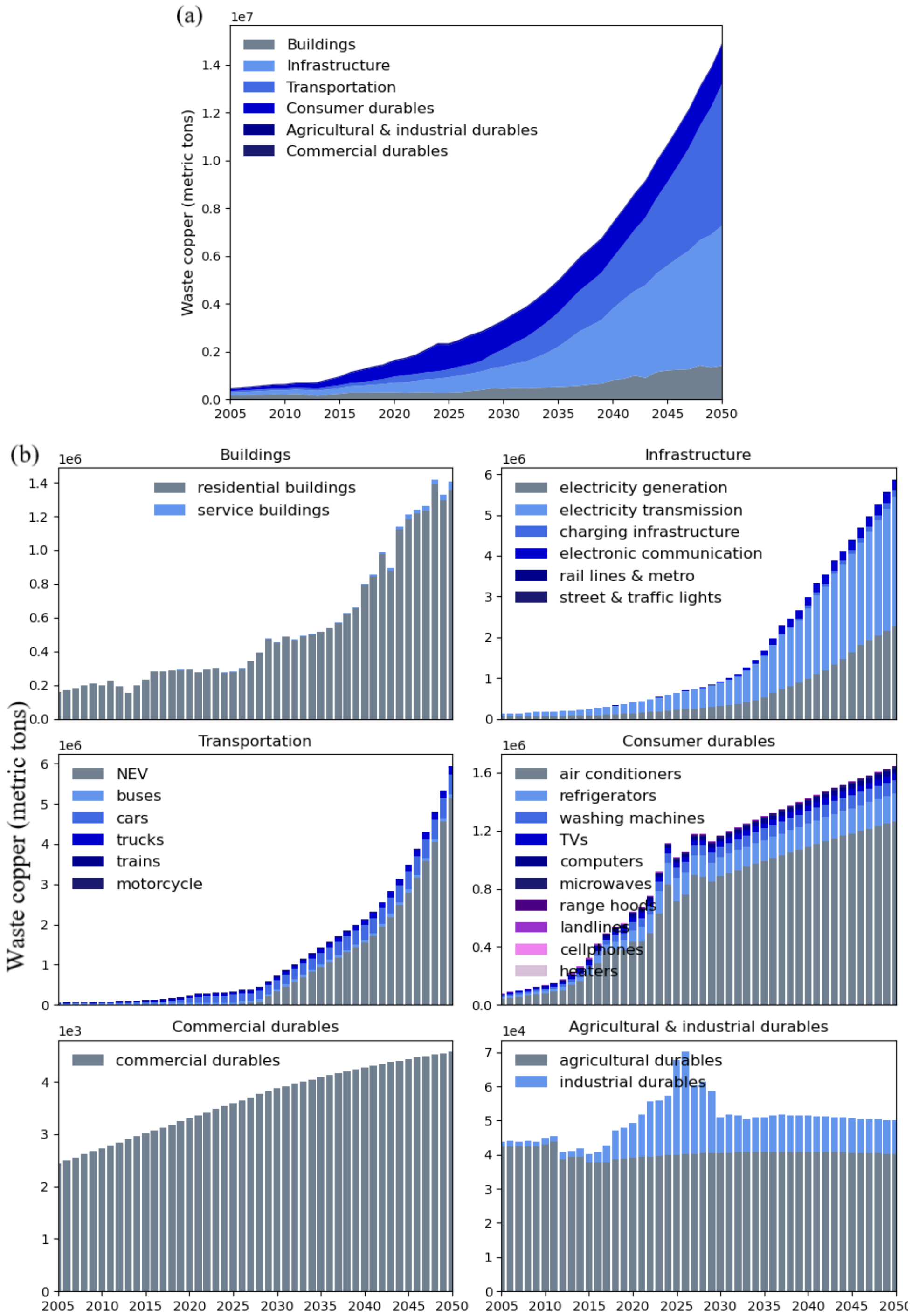


Figure 2.4 Waste copper of China from 2005 to 2050 (a) aggregate copper waste by end-use category; (b) detailed copper waste by product group

The period prior to 2030 is one of material accumulation, with relatively little generation of waste. Subsequently, an increasing amount of scrap will be generated from long-lived applications in buildings, infrastructure and durable products. This means the outflow will increase from 0.5 Mt in 2005 to about 15 Mt in 2050. At the same time, though, certain new applications, such as charging stations and NEV, will not yet have begun moving in to the waste phase.

Six types of copper scrap from end-use sectors are generally distinguished: construction and demolition waste (C&DW), MSW, ELV, WEEE, industrial electrical waste (IEW) and industrial non-electrical waste (INEW) (Soulier et al., 2018b). Much of this scrap is collected and separated for recovery, which is of relevance for development of the circular economy. Not all EoL products are indeed reprocessed, though, with some fraction being landfilled or left in the environment as "hibernating" stocks: the underground cables used for power transmission, for example.

2.3.4 Potential for circularity: fraction of new demand covered by outflow/waste

China is undergoing a period of rapid development and consuming huge amounts of copper as a key material for its infrastructure. At the same time, though, the country lacks sufficient copper resources, which means copper recycling is very important. As is well-known, there is no difference in the quality of copper from secondary and primary production and although certain applications involve irrecoverable losses, such as buried cables and copper compounds used as animal food supplements, for most copper applications a significant degree of recycling is possible. At the same time, the environmental pollution associated with the mining of copper ores required to meet the country's huge copper demand can also promote development of a circular economy. Compared with mined copper concentrates, reuse of scrap copper has the advantages of high recovery rates, low energy consumption and reduced pollution. Not only does copper recycling require up to 85% less energy than primary production (ICA, 2013); it is also a highly eco-efficient way of reintroducing a valuable material into the economy. In assessing the potential for a circular economy with respect to copper scrap, recycling the first question that needs to be answered is how much recyclable scrap is produced relative to overall demand. Detailed information on estimating the amount of recyclable copper is provided in the Appendix 4.

Figure 2.5 shows China's copper recycling potential for various end-use applications from 2005 to 2050. Over this period the amount of recoverable copper increases from 0.25 Mt to 11 Mt, driven by the increasing copper outflow from stock-in-use. This increase holds across all copper applications, but to varying degrees. In particular, copper recovered from applications in the transportation sector will increase very significantly, from about 37 thousand tonnes in 2005 to over 5 Mt in 2050. After 2030, the amount of copper recovered from the infrastructure sector will also increase rapidly.

In our analysis, we used waste copper recovery rates varying from 25% to 89% for the various waste categories in different years (see Table S2.22 in the SI). Under these assumptions, copper recovery will be able to meet only about 40% of Chinese copper demand by 2050. The question, therefore, is whether this percentage can be boosted, and to what extent. The scrap recovery rates assumed here are based on average European data and are higher than those currently seen in China are. While scientific and technological progress will mean China's copper recovery rate will in all likelihood continue to rise in the future it is still an open question whether even with an enhanced recovery rate recycling can meet the country's enormous demand for this metal. To explore whether enhanced levels of copper recovery can indeed cover aggregate demand we therefore assumed a recovery rate of 90% for every product and industry, a similar figure to that presently achieved in Western Europe (Ruhrberg, 2006), Under this assumption copper recovery in China rises to approximately 13 Mt in 2050. Compared with the total copper demand of around 24 Mt projected for that year, recycled copper will still not even come close to meeting demand (Figure 2.5). The main reason for this is that in 2050 China's economy will not yet have attained a steady state, but will still be increasing the copper stocks bound up in infrastructure and other sectors. Primary copper will therefore still be a crucial source of copper in China for decades to come.

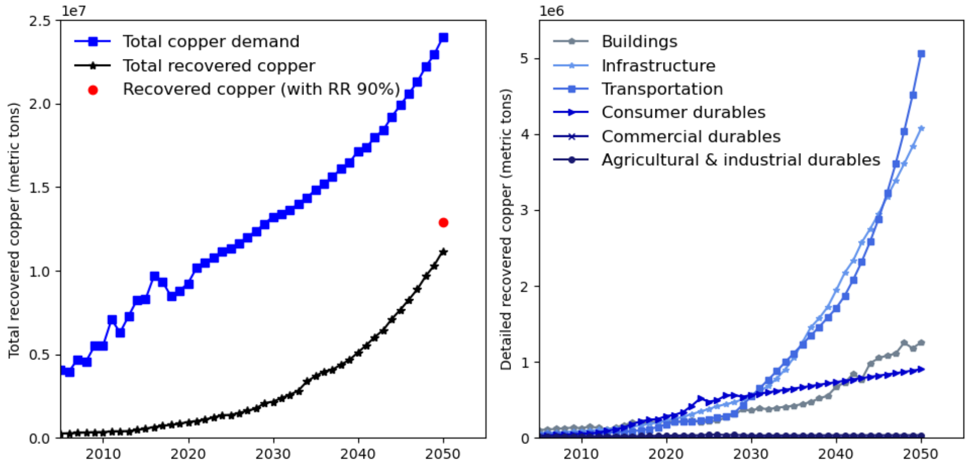


Figure 2.5 Estimated copper recovery in China from 2005 to 2050 based on current European recycling rates as shown in Table S2.22; Note: the point "recovered copper (with RR 90%)" indicates copper recovery based on an assumption of 90% of EoL recycling rate for each product.

2.3.5 Sensitivity analysis

We now assess the influence of variations in product lifetime on the results obtained in this study. The shorter the lifetime, the more waste copper will be generated and the greater demand for copper will be. When it comes to the largest copper applications, in buildings and infrastructure, long and varying lifetimes are reported (Buchner et al., 2015; Huang et al., 2013). For China in particular, however, the literature suggests that building lifetime of is between 20 and 40 years (residential and non-residential building). We found no quantitative evidence for longer building lifetimes over the past few decades. Owing to changes in building materials and maintenance, the lifetime of buildings constructed in recent years and in the future will be longer. To compare with the business-as-usual scenario, we performed a sensitivity analysis, increasing residential building lifetime to varying degrees, as shown in Figure S2.4 in the SI. This indicates that the longer the lifetime of residential buildings is, the less waste copper is generated. Over the next 10 years, the change is not large, but on the longer term, the reduction in copper waste generation is considerable.

A second sensitivity analysis was carried out for the transportation sector, varying the assumed lifetime of conventional cars, conventional buses and new energy vehicles between 8 and 18 years to assess the impact on copper waste generation; for further details see the Appendix 3. When the lifetime of these three types of vehicle is boosted to 18 years, copper waste is reduced by over half after only 8 years (see Figure S2.5). As the sensitivity analysis shows, future copper waste is very sensitive to product lifetime. In order to reduce copper demand, one key option is therefore to extend the service life of end-use products, especially the applications with long lifetime. This is anticipated to have a considerable impact on both demand and waste streams, and is an issue we may want to take up in our future research on circular economy scenarios for China.

2.4 Reflection and discussion

2.4.1 Comparison of global and Chinese copper demand and stocks

To put our results into a broader perspective, we now compare them with the results of earlier studies on copper flows and in-use stocks, as summarized in Table 2.2. For China, the most detailed databases of copper stocks and flows have been compiled by Zhang et al. (Zhang, L. et al., 2015a; Zhang, L. et al., 2014, 2015b) and Yang et al. (2017). These studies had a very similar aim to our own: to provide estimates of present and future copper stocks as well as future demand in China. Zhang, L. et al. (2015b), Zhang, L. et al. (2015a), Maung et al. (2017) and Soulier et al. (2018b) concluded that China's copper stocks were around 50-70 Mt in 2010, Zhang, L. et al. (2015a) also estimated that the copper stocks may gradually increase to a peak of 163-171 Mt by 2050 based on a stock-driven model. Another study cited in Zhang's research estimates that in-use stock will peak at 190-220 Mt in 2060 (Zongguo and Xiaoli, 2013). In addition, Terakado et al. (2009a) used a dynamic MFA approach to estimate in-use stocks for copper in China and reported a value of 25 Mt in 2005.

As Table 2.2 shows, the results of our study are similar to those of other studies for 2010 and 2015, but differ for future years. In the present study, both demand and in-use stock are estimated higher in 2050. We attribute this difference to two factors. First, we used a slightly different categorization of copper applications, distinguishing 29 sub-categories and including new energy applications such as new energy vehicles and charging infrastructure, both of which will require huge amounts of copper. We also modelled demand at a more detailed level than most other studies, leading to slightly different outcomes overall. Second, we worked with different assumptions regarding future developments. In particular, our simulation is based on current government policies rather than assuming that the level of development in 2050 will mirror that of last year, which makes a great deal of difference. For example, the addition of new energy applications according to already existing government policy leads to quite different results for infrastructure compared with the other studies. While for some categories, such as the built environment, copper stocks are expected to stabilize in our study as well (see Figure 2.3), we find that in 2050 China's total copper stocks and demand will still not yet have peaked.

Table 2.2 Studies on Chinese copper demand and in-use stock, 2005, 2010, 2030 and 2050

Source	**Copper demand (Mt/year)**				**In-use stock (Mt)**			
	2005	2010	2030	2050	2005	2010	2030	2050
Zhang, L. et al. (2014)	5.3	7.9	-	-	-	-	-	-
Zhang, L. et al. (2015a)	-	-	10.1	10.9	-	-	140	163-171
Zhang, L. et al. (2015b)	-	-	-	-	30	48	-	-
Soulier et al. (2018b)	-	8.6	-	-	-	50	-	-
Yang et al. (2017)	3.6	7.6	15.4	-	-	-	-	-
OECD (2019)	-	-	~18	~24	-	-	~200	~450
Terakado et al. (2009a)	-	-	-	-	25	-	-	-
Maung et al. (2017)	-	-	-	-	35-40	60-70	-	-
This paper	4.1	5.5	13.2	23.9	30.5	51.9	215	404

Note: The "-" in this table means that there is no available data from the corresponding reference for this year.

Table 2.3 Studies on copper demand and in-use stock at different scale levels, 2005, 2010 and 2050

Scale	**Copper demand**			**Per capita copper demand**			**In-use stock**			**In-use stock per capita**		
	Mt/year			kg/capita/year			Mt			kg/capita		
	2005	2010	2050	2005	2010	2050	2005	2010	2050	2005	2010	2050
Global	16.6[a]		60 [a]	2.5[a]		-		-	-		-	-
Global		22[b]	62[b]		3.2[b]	6.4 [b]		-	-		-	-
Global		-	-		-	-	330[c]		-	50[c]		-
Global		-	-		-	-		-	381-588[i]		-	-
OECD		8.7[b]	9.33[b]		9.1[b]	9.6[b]		-	-		-	-
REF[h]		1.7[b]	3.45[b]		4.2[b]	8.5[b]		-	-		-	-
ASIA[h]		8.7[b]	31.33[b]		2.4[b]	6.7[b]		-	-		-	-
ALM[h]		3[b]	16.12[b]		1.6[b]	4.9[b]		-	-		-	-
Developing		-	-		-	-		200[d]	700[d]		30[d]	75[d]
Industrialized		-	-		-	-		200[d]	300[d]		160[d]	225[d]
United States		-	-		-	-		70[j]	-		~230[j]	-
Italy		-	-		-	-		~20[j]	-		~350[j]	-
Germany		-	-		-	-		~20[j]	-		~230[j]	-
India		0.3[e]	4-9[e]		0.3[e]	2.9-6.3[e]		-	-		-	-
Japan		-	-		-	-	18.7[f]	25[j]	-	146[f]	170[j]	-
Switzerland		-	-		8[g]	9[g]		-	-		-	-
China		7.6-8.6	11-24		-	-		48-70	163-450		~50	-
This paper		5.5	23.9		4.1	17.5		51.9	404		38.7	296

Note: The "-"means that there is no available data from the corresponding reference for this year. Unless otherwise specified, retrospective data are for 2010. Some data cited from the literature are only approximate. a: (Schipper et al., 2018); b: (Ayres et al., 2003); c: (Glöser et al., 2013); d:(Gerst, 2009); e: (Kapur, 2006); f: (Daigo et al., 2009); g: (Bader et al., 2011); h: Supporting Information Appendix 2.4; i: (Yoshimura and Matsuno, 2018); j: (Maung et al., 2017). Developing/ Industrialized countries.

Table 2.3 compares the results of our study with those of studies on other countries and regions as well as the world as a whole. Copper stocks in China will increase from 23-30 kg per capita in 2005 to around 300 kg per capita in 2050. This is faster growth than the average for industrialized countries as well as the average for developing countries. By 2050, Chinese copper stocks are expected to be close to the present level of the developed world. China is presently at the stage of rapidly increasing its in-use stocks, a stage that has already occurred in the developed countries and is expected to occur in certain developing countries later, or at a slower pace than in China. The very high per capita demand, associated with the build-up of China's stock could decline after 2050 to the level needed to maintain the stock.

When it comes to per capita copper demand, China's was similar to the global average in 2010, but lower than that of the OECD countries. For the future, our calculations suggest that by 2050 China's per capita copper demand will be significantly higher than the world average, OECD countries and certain developed countries, such as Switzerland. This is probably due mainly to the fact that in 2050 China will still be catching up with reaching the stock levels of OECD countries, as explained further below. In that year China's per capita copper demand will be more than 4 times higher than in 2010 and about 15 times higher that of India in 2010. In 2050, aggregate copper stocks in developing countries are expected to be more than three times what they were in 2010 and more than twice as in-use stock per capita in 2010 (Gerst, 2009). This indicates that India will undoubtedly become a major player in terms of copper demand. While there is extensive informal recycling in developing countries, appropriate recycling infrastructure for end-of-life management of complex products is presently lacking. As waste streams will undoubtedly increase, greater efforts to recover and manage scrap copper in these countries would be very beneficial.

2.4.2 Limitations

There are several issues of relevance for forecasting copper demand that were not taken into account in this study. Below we discuss three of them.

1) Copper intensity

While copper intensity is a key factor determining both copper stocks and future copper demand, there are very few accurate data available on many products, especially when it comes to possible changes in the years ahead. In this study, the copper intensity of most products except buildings was taken to remain fixed. However, previous studies have shown that the amount of copper used in products has changed over time. For some products, such as buildings, cars and electrical and electronic equipment, the amount of copper is increasing (Charles et al., 2017). In some cases, it seems likely that copper use will decline as it is replaced by other materials. For lack of quantitative information, however, in this study we made no allowance for possible future changes in copper intensity.

2) Copper substitution

Copper demand may be reduced as a result of substitution, at either the material or product level. Graedel et al. (2015) have shown it is quite difficult to find suitable alternatives for copper quickly, though, one reason being that such alternatives will have to be produced in large quantities. Nevertheless, in some of its applications copper has already been replaced or is expected to be so by aluminum, titanium or optical fiber. This could reduce copper demand to some extent. At the same time, though, novel technologies requiring copper are also being developed at present, leading to a potential increase in copper demand. For example, recent studies have pointed to the possibility of replacing silver by copper in silicon-based photovoltaic solar technology (García-Olivares, 2015). Given the rather speculative nature of this broad topic is, we have not attempted to modify our demand projections by introducing assumptions on substitution.

3) Other applications of copper in China

With all its appealing properties, copper it gradually being applied in new applications in addition to those considered in this paper. Since these new applications are only just on the market, we considered their development too speculative and so ignored them in our study. However, some of them may be relevant for future copper demand in China.

One example is the use of copper in non-polluting heating equipment. Copper is used in the heat exchangers for air-source heat pumps, for example, requiring an average of over ten kilograms per unit. To address the problem of smog the Chinese government has decreed that 28 cities must substantially reduce the burning of bulk coal. In these cities scattered coal burning is an important winter heating source. Using electrical heating or air-source heat pumps instead of coal would improve local air quality substantially. Use of air-source heat pumps in fact increased about 28 times in just a single year, from 2015 to 2016. If the heat pump market continues to develop according to the current trend, this would increase copper demand significantly.

A second example is provided by the copper cages used in mariculture (a new type of aquaculture). One of the tasks set out in the Chinese government's 13th Five Year Plan, is to build 10,000 deep-sea cages for mariculture. Each cage uses 20 kg of copper with even more required for the associated fencing. While this is still relatively little compared to total Chinese copper demand, this sector may expand further, becoming relevant in the future.

2.5 Conclusions and outlook

In this study we adopted a dynamic stock model and scenario analysis involving a bottom-up approach to analyze copper demand in China from 2005 to 2050, based on a series of drivers including GDP, population growth, urbanization level and government policy plans. In the business-as-usual scenario, both aggregate and per capita copper demand are set to increase significantly over time, especially in infrastructure, transportation and buildings. Between now and 2050 total demand for copper will rise by a factor of almost 3. It should be noted, though, that our estimates make no allowance for a possible increase in copper content in certain applications nor for any new applications, which means the copper demand in the baseline scenario for 2050 may be in fact be an underestimate.

The copper stock in infrastructure and transportation will not yet have reached saturation by 2050 but will then still be growing. Stock in buildings will have virtually stabilized, though, which means per capita demand in this segment will no longer be significantly rising. Our calculations show that aggregate demand in 2050 cannot be met through use of secondary copper, even if a recycling rate of 90% is assumed In the baseline scenario, then, there would appear to be no way China's copper industry can achieve a completely circular economy in the short term.

Given the projected increase in Chinese copper demand through to 2050, we can only conclude that a major quantity of raw materials will need to be mined and processed, which will have a pronounced environmental impact, particularly in light of the challenges involved in improving the environmental footprint of copper production and processing. As demand continues to rise, copper will become even more important as a source of greenhouse gas emissions than it is at present (Van der Voet et al., 2018).

One development not explicitly addressed in this study is the coming energy transition. While renewable energy technologies will reduce GHG emissions, they will also mean increased demand for copper, leading to increased GHG emissions from this industry (Deetman et al., 2018). In this respect the balance between reducing the environmental impact of new energy technologies and increased copper use in new energy devices is an issue requiring due attention.

Copper recovery is another aspect of major relevance for environmental impacts, given that secondary production via recycling requires up to 85% less energy than primary copper production. Based on the present research, we can only conclude that under baseline assumptions a very large inflow of primary copper will still be needed up until 2050. For China at least, increasing the scrap recovery rate will not be sufficient to meet copper demand in 2050, when aggregate new demand will be almost twice as high as the total copper outflow from all sectors combined.

Our results show that China's economy will be reliant on an increasingly large supply of primary copper. It is therefore important to develop alternative development scenarios in which copper demand is sustainably fulfilled. A circular economy scenario could be an effective way to reduce primary production and therefore also reduce the environmental impacts associated with copper production. In future research, we intend to focus on developing alternative scenarios, exploring different options for improving the sustainability of China's copper supply.

Chapter 3

Scenarios for anthropogenic copper demand and supply in China: implications of a scrap import ban and a circular economy transition

Abstract

From 2013 on, China has implemented the "Green Fence" policy to restrict copper scrap imports, which have affected and will continue to affect its future copper supply. To explore how China's copper demand can be met in the future, including the effects of the "Green Fence" policy change, a stock-driven approach is combined with a scenario analysis. We compare two scenarios (Continuity Policy, Circular Economy) and assess the influence of the "Green Fence" policy on each. Effective measures to prolong product lifetime could lead to a significant reduction in copper demand. Given the limited scope for domestic mining, China will still have to depend largely on imports of primary material in the form of concentrates and refined copper or, otherwise, put major emphasis on its recycling industry and continue to import high-quality copper scrap. In combination with the establishment of a state-of-the-art, efficient and environmentally friendly recycling industry, secondary copper could satisfy the bulk of Chinese copper demand and this could be an opportunity for China to transition to a more circular economy with regard to copper.

Published as: Dong, D., Tercero Espinoza, Luis A., Loibl, A., Pfaff, M., Tukker, A., & van der Voet, E. (2020). Scenarios for anthropogenic copper demand and supply in China: implications of a scrap import ban and a circular economy transition. Resources, Conservation and Recycling, 161, 104943.
https://doi.org/10.1016/j.resconrec.2020.104943.

3.1 Introduction

Copper is an indispensable resource for a wide range of applications. In recent years, with China's vigorous development of their electricity infrastructure and the rise of electric vehicles, copper consumption has increased enormously. Since 2002, China has become the world's largest copper consumer. The proportion of Chinese copper consumption in global copper consumption has increased from 17% (2.6 million tons) in 2002 to 46% (8.4 million tons) in 2016 (Dong et al., 2019; MNR, 2018). Under the current policy plan, this fast-growing demand is expected to continue in the near future, which might cause future supply problems and environmental issues related to copper production (Elshkaki et al., 2016; OECD, 2019; Van der Voet et al., 2018).

Copper demand is met by both primary and secondary copper. Primary copper is the predominant resource for Chinese copper production until now, accounting for more than 65%. Secondary copper production has seen a significant development in the past few years. As a result of low generation of domestic copper scrap, the main source of secondary copper is still imported scrap. Large amounts of copper scrap have been imported since the early 1990s to meet the country's growing copper demand. Today, however, China is flooded with discarded electronic products from all over the world. At the same time, the technology and equipment for dismantling and recycling such products is relatively unsophisticated in China. Some of the dismantling plants incinerate copper scrap like motors and wires in the open air, and dump dismantling residuals, leading to serious pollution problems (Duan et al., 2011; Gradin et al., 2013). Therefore, China has implemented several policies to improve the environmental performance of its copper industry since 2013, one of which is to increase the number of inspections and restrict the import of low-quality copper scrap: the so-called "Green Fence" policy (State Council, 2013). The decision on whether or not to continue to import copper scrap will undoubtedly affect the future development of China's copper industry, and because of its sheer size consequently the global picture as well. It is therefore important to explore what impact this "Green Fence" policy will have on Chinese copper supply and how Chinese copper demand will be met.

A multitude of historical studies have been undertaken in the realm of copper supply and demand. To date, most studies have utilized MFA based on a top-down approach to calculate copper demand, which is projected by several external drivers like GDP, population and income (Babaei et al., 2015; Choi et al., 2008; Daigo et al., 2009; Deetman et al., 2018; Elshkaki et al., 2018; Glöser et al., 2013; Kapur, 2006; Singer, 2017). Some studies have adopted a bottom-up method to directly quantify copper demand, multiplying copper content by all the products that consumed within the system boundary in a given research period (Bader et al., 2011; Dong et al., 2019; Schipper et al., 2018; Zhang, L. et al., 2014). These studies quantify the number of copper products in use and arrive at copper flows using data on the copper content of these products. To explore how to meet copper demand, several scholars have investigated the broader copper cycle comprising copper production, fabrication and manufacturing, use and waste management (Bertram et al., 2003; Glöser et al., 2013; Graedel et al., 2004a; Graedel et al., 2004b; Soulier et al., 2018a; Soulier et al., 2018b; Spatari et al., 2002; Tanimoto et al., 2010). Specifically, Elshkaki et al. (2016) modelled four scenarios of global primary and secondary copper supply. Furthermore, some research has estimated potential copper scrap and analyzed the copper recycling industry (Brahmst, 2006; Daigo et al., 2007; Reijnders, 2003; Tatsumi et al., 2008; Wang et al., 2017). Wormser (1921), Tercero Espinoza and Soulier (2016), Zhang et al. (2017) and Dong et al. (2018) have focused on copper trade flows and identify the relative role of various countries in global copper-relevant value chains. With respect to copper supply, the recent literature has paid particular attention to copper ore grades and studied the environmental impacts of primary and/or secondary copper production, such as energy use, greenhouse gas (GHG) emissions and acidification issues, and proposed several strategies for reducing these impacts (Dong et al., 2017; Elshkaki et al., 2016; Giurco and Petrie, 2007; Kuipers et al., 2018; Norgate et al., 2007; Van der Voet et al., 2018).

So far, however, there have been no publications on the impact of implementing the Chinese ”Green Fence” policy on long-term copper supply and on the question of whether or not to continue to import copper scrap or even increase such imports. Based on the aforementioned special position of imported copper scrap in the Chinese copper supply chain, the purpose of this study is to explore the questions of whether or not China should continue to import scrap copper and how the country should meet its future copper demand. To this end, two copper demand scenarios (Continuity Policy (CP) scenario and Circular Economy (CE) scenario) were developed in accordance with the development path set by the Chinese government from 2005 to 2100 using a stock-driven approach in line with Dong et al. (2019). Copper supply scenarios matching this demand were then defined

based on the “Green Fence” policy. Finally, considering the many benefits of secondary copper production proposed by scholars (Alvarado et al., 2002; Elshkaki et al., 2016; Giurco and Petrie, 2007; Pfaff et al., 2018), two hypotheses are proposed that reduce primary copper production and import more copper scrap, based on two scenarios (CP and CE), explained below. The results can be used to identify possible measures and policy options in response to a future copper supply challenge in China.

3.2 Methodology and data

3.2.1 System definition of Chinese copper cycle

The Chinese copper cycle was analyzed using a system definition based on the structure presented by Soulier et al. (2018a), as shown in Figure 3.1. The model consists of mining, smelting & refining, semi-finished goods production, fabrication of end-use products, use, waste management and recycling processes. Compared with most studies on the copper cycle, we provide considerable detail on the EoL phase: obsolescence, collection, dissipation, separation and recycling. Table 3.1 shows the definition of items used for the Chinese copper cycle.

3.2.2 Modelling future copper in-use stock and copper demand

MFA has been widely used to analyze metal flows and stocks on the global and national scale. Most such studies explicitly or implicitly assume that substance flows emerge from economic activity and ultimately determine the levels of stocks (Kapur, 2006; Soulier et al., 2018b). Other approaches have acknowledged the importance of the in-use stock as the driver of flows, especially for applications with a long life span (Müller, 2006; Pauliuk et al., 2012). This method uses historical data of the in-use stock as the provider of services to society and corresponding life spans to calculate product demand

and waste generation. The difficulty with this method concerns the simulation of the future in-use stock. In this study, therefore, we first introduced a detailed analysis for modelling the future copper in-use stock based on a stock-driven method and then determined copper demand and waste generation.

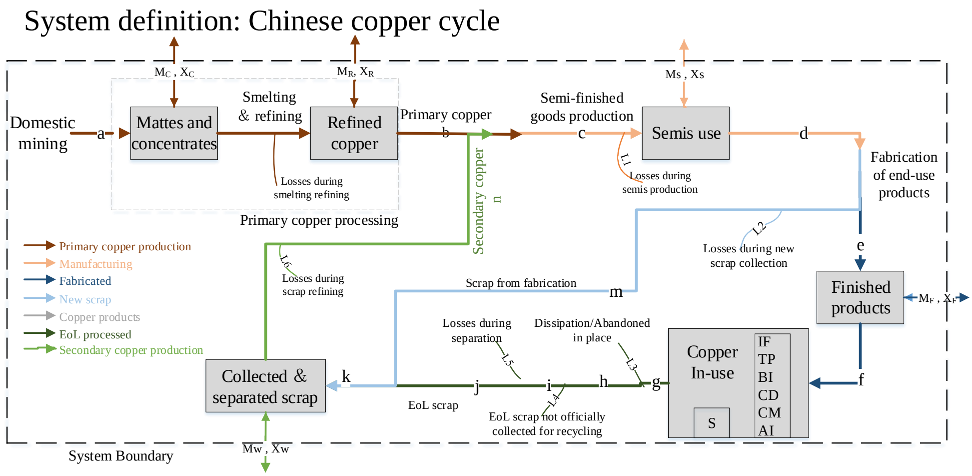


Figure 3.1 Chinese copper cycle: estimated future primary and secondary copper supply

3.2.2.1 Modelling future copper in-use stock

To estimate the in-use stock of copper, we refer to the method and data described in Dong et al. (2019). End-use copper products were divided into six main categories: infrastructure, buildings, transportation, consumer products, commercial products and agricultural & industrial durables. These consist of 29 sub-categories that include new energy applications, for instance, battery charging stations and new energy vehicles. To calculate the in-use stock of each copper product from 2005 to 2100, one or more drivers in demographics, product features, economic welfare and government policies are used, such as population, urbanization rate, copper content, GDP and Chinese government policy related to future activity levels. In order to better observe long-term waste copper generation and copper supply, we extended the study period further than Dong et al. (2019), through to 2100. The various drivers and the data used to generate the time series from 2050 to 2100 are discussed below. Drivers and data for the period up to 2050 are described in Dong et al. (2019).

Population. We used the population figures projected by the United Nations Population Division (United Nations, 2017). The population of China is expected to start decreasing from around 2030 onwards and reach around 1.02 billion in 2100 (Figure S3.1).

GDP. We used the future GDP of China projected by OECD and the University of Denver, expressed in 2010 USD Purchasing Power Parities, as shown in Figure S3.1 (IFs, 2017; OECD, 2018).

Urbanization. Urbanization rate is a key variable in estimating the in use-stock of products, because the per capita product ownership in urban and rural areas in China varies greatly. We assumed it follows a logistic function and to be 60% in 2020, 75% in 2050 and 80% in 2100 (Hu et al., 2010a; Li and Qi, 2011; United Nations, 2018).

Copper intensity. The copper intensity of each copper product was derived from Dong et al. (2019) and is shown in Table S3.1. Future trends in copper intensities are difficult to forecast since technology development could increase or decrease copper intensity in some applications, e.g. development in alternative materials and substitutes for copper. However, due to the superior performance of copper, substitution is minimal so far (ICA, 2017). Therefore, with the exception of buildings, copper intensities were therefore assumed to remain unchanged. A sensitivity analysis in Section 3.5 briefly addresses the implications of different copper intensities.

Policy plans. For estimations from 2005 to 2050, the 11th, 12th and 13th Five-Year Plans and the medium and long-term plans for each industry sector and reports by consultancy organizations (CNREC, 2017; ICA, 2013; Macquarie, 2015) were used, as reported in Dong et al. (2019). For the period from 2050 to 2100 there is no relevant government policy for all the categories, however. The in-use stock of copper applications from 2050 to 2100 was therefore estimated based on one or more of the above five drivers, with no allowance made for possible additional policies. Detailed assumptions are provided in the Supporting Information.

Table 3.1 Definition of items used for the Chinese copper cycle

Process	**Symbol**	**Name**
Use	S	In-use stock
	IF	Infrastructure
	TP	Buildings
	BI	Transportation
	CD	Consumer durables
	CM	Commercial durables
	AI	Agricultural & industrial durables
	LT	Lifetimes of copper products
	f	Domestic final demand
Trade of finished products	M_F	Import of finished products
	X_F	Export of finished products
Fabrication	e	Production of finished products
	Ex	Fabrication efficiency
	m	Scrap from fabrication (new scrap)
	L2	Loss during new scrap collection
	SR	Loss rate during new scrap collection
Trade of semi-finished goods	M_S	Import of semi-finished goods
	X_S	Export of semi-finished goods
Manufacturing	c	Production of semi-finished goods
	d	Semi-finished goods to fabrication
	L1	Loss during manufacturing
	FR	Loss rate during manufacturing
Smelting & refining	b	Primary copper supply to manufacturing
Trade of primary copper	M_c	Import of copper concentrates
	X_c	Export of copper concentrates
	M_r	Import of refined copper
	X_r	Export of refined copper
Domestic mining	a	Domestic extraction
Waste management and recycling	L3	Dissipation/abandoned copper in place
	DR	Dissipation/abandoned rate in place
	h	Total copper content in EoL scrap
	g	Generation of EoL scrap
	CR	EoL scrap collection rate
	i	EoL copper collected
	PR	EoL separation rate
	L5	Losses during separation
	j	EoL copper recycled (old scrap)
	k	Collected & separated domestic scrap
	L6	Loss during scrap refining

Process	Symbol	Name
	SRR	Loss rate during scrap refining
	n	Secondary copper supply to manufacturing
Trade of copper scrap	M_W	Import of copper scrap
	X_W	Export of copper scrap

3.2.2.2 Modelling future copper demand

To estimate copper demand and waste, we adopted a stock-driven approach, as originally presented by Müller (2006), Pauliuk et al. (2012) and Pauliuk (2014). We calculated the copper flows into the stock in year t_0 that have survived for t years using Normal distribution of lifetimes. The Normal distribution is commonly used for general reliability analysis (Ayres et al., 2003; Bader et al., 2011; Glöser et al., 2013; Hu et al., 2010b). The lifetime distribution *f(t)* in the form of a probability density function (PDF), the likelihood of failure *F(t)* in the form of a cumulative distribution function (CDF) and the survival function (*SF(t)*) of the normal distribution are given by:

$$\mathrm{f(t)} = \frac{1}{\sigma\sqrt{2\pi}} e^{-\frac{1}{2}(\frac{t-\mu}{\sigma})^2} \quad (3.1)$$

$$\mathrm{F(t)} = \frac{1}{\sqrt{2\pi}} \int_{-\infty}^{x} e^{\frac{-t^2}{2}} dt \quad (3.2)$$

$$\mathrm{SF(t)} = 1 - \mathrm{F(t)} \quad (3.3)$$

where t is the time step, μ is the mean lifetime of the respective product and σ is the standard deviation of the lifetime distribution. The SF distributions of copper products are shown in Figure S3.2.

The survival function presents the fraction of copper products consumed in year (t') that are still in use in year t. We then calculated the copper content in surviving copper products (CSR) from the copper consumption in all products CC_i of previous years (t') as follows:

$$CSR_i(t) = \sum_{t'}^{t} CC_i(t') \times SF_i(t - t') \quad (3.4)$$

$$CC_i(t') = PC_i(t') \times \theta_i(t') \quad (3.5)$$

$$CC_i(t) = CS_i(t) - CSR_i(t) \quad (3.6)$$

$$CD_i(t) = CC_i(t) - (CS_i(t) - CS_i(t-1)) \qquad (3.7)$$

where i is the type of copper product, $CS_i(t)$ is the in use stock of copper in each product in year t. $CD_i(t)$ is the discarded copper in each product in year t. $PC_i(t')$ is the consumption of copper products in year (t'). $\theta_i(t')$ is the copper content of each product in year t.

3.2.3 Modelling future primary and secondary copper supply

Copper demand is met by primary and secondary copper. To estimate the future primary and secondary copper supply, we first estimated the future supply of secondary copper (n, Figure 3.2) from EoL scrap (j), new scrap generated in the fabrication process (m) and scrap coming from international trade (M_w, X_w). Next, we estimated the demand for finished (f) and semi-finished (c) products. Finally, the amount of primary copper (b) was estimated through backwards calculation from this final demand via manufacturing of semi-finished products and trade. Part of this primary copper comes from domestic mining (a). The remainder was assumed to be imported (M_{cr}, X_{cr}). The simple stock-driven model for estimating future copper supply is shown in Figure 3.2. The detailed calculation steps can be found in the Supporting files.

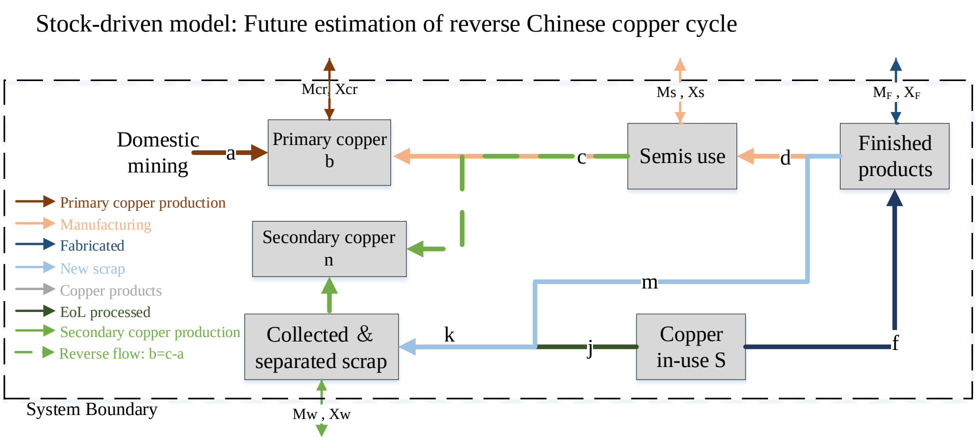


Figure 3.2 Stock-driven model: Future estimation of reverse Chinese copper cycle

3.2.3.1 Secondary copper production

Domestic secondary production-EoL scrap (old scrap). Copper scrap from

EoL products (j), also called old copper scrap, refers to copper scrap from products that have been used by consumers or other end-users. It is a huge potential resource. The generation of copper scrap in China is forecasted to increase massively in the future owing to the historical rise in copper consumption and stocks (Dong et al., 2019; Soulier et al., 2018b; Wang et al., 2017). Recovery of old copper scrap generally goes through two stages: pretreatment and recycling. The pretreatment stage refers to the collection and separation of mixed copper scrap and removal of other waste on the surface of scrap products to obtain a single and relatively pure variety of copper scrap. There is a certain amount of dissipation or abandoned copper in place in the pretreatment stage; for example, some copper scrap is not collected, such as underground cables that are just left in the ground. In addition, some copper use is at least partly dissipative in nature (e.g. copper used in fungicides; material loss through abrasion or corrosion or copper and copper alloy parts in architecture and machinery, ammunition), or present in too small quantities in products to be practically collected for recycling (e.g. brass rivets and zippers in clothing). Dissipative losses (L3) occur mainly during the use phase of copper applications due to corrosion and abrasion and are accounted for in their entirety on a market-by-market basis at the time products leave the use phase. Some of the dissipated copper will become part of sewage sludge and might be reused as a fertilizer. However, as copper is not recycled from this flow, sewage sludge and its copper content are not included in waste management.

Copper scrap is usually divided into six types: construction and demolition waste, end of-life vehicles, municipal solid waste, waste of electrical and electronic equipment, industrial electrical waste, and industrial non-electrical waste (Ruhrberg, 2006; Schlesinger et al., 2011; Soulier et al., 2018b). To estimate the amount of EoL copper recycled, the generation of EoL scrap (g), the collection rate of EoL scrap (CR, for flows i), the dissipation rate (DR, for flows h, L3) and loss rate of separation (PR, for flow i, L5) for each type were used based on Soulier et al. (2018b). We matched the rate for each of the copper sub-categories to the six types in their study.

Domestic secondary production-New scrap. New copper scrap is generated during the fabrication of copper products, in amounts that depend on the fabrication efficiency. As a result of the upgrading of equipment and continuous technological innovation, copper product fabrication efficiency increased from 80% in 1990 to 89% in 2016. In this study, we assumed it remains unchanged in the future. To estimate new scrap from fabrication, we assumed a collection efficiency of 95%, leading to a loss rate (SR, for flow L2) of 5%.

Imported copper scrap. The customs codes of the General Administration of Customs of the People's Republic of China (GACC) for imported solid wastes are divided into ten categories, corresponding to different types of solid waste, including waste paper, waste plastics and metal scrap. Among them, "category 6" and "category 7" contain copper. We therefore likewise divided imported copper scrap into two categories: "category 6" and "category 7" (Table 3.2). High-grade Category 6 scrap and brass scrap with a clear classification and little impurities (copper content above 95%) can be directly processed and used. Part of the lower-grade red copper scrap and brass scrap (copper content ranging from 70% to 95%) is also Category 6 scrap and can be imported, but needs to be smelted again. Lower copper content scrap belongs to Category 7.

Under the "Green Fence" policy, category 7 scrap will no longer be allowed to be imported from 2019 onwards, while imports of category 6 scrap will be restricted as of July 2019 and will be gradually phased out as well. To estimate future imports and exports of category 6 scrap, the historical relationship between GDP and the volume of imports and exports of copper scrap was used.

Total secondary copper supply. The secondary copper utilized in Chinese production derives from three sources: new copper scrap, old copper scrap and imported copper scrap. Based on the calculation of these three elements, secondary copper supply is estimated using equation (8).

$$F_{n,t} = F_{m,t} + F_{j,t} + M_{w,t} - X_{w,t} \times (1 - SRR) \qquad (3.8)$$

where $F_{n,t}$ is future secondary copper supply, $M_{w,t}$ and $X_{w,t}$ are future import and export of copper scrap and SRR is the loss rate during scrap refining, which is related to flow $L6$.

Table 3.2 Overview of the "Green Fence" policy: categories 6 and 7 of imported copper scrap

Customs code	**Scrap category**	**Scrap covered**	**Copper content**	**Import timeline**	**Sources**
7404000010	7	Precipitated copper	40%	Automatic import from 2013.01.01 Restriction from 2014.01.01 Ban from 2015.01.01	Automatic import license of China (2014); Automatic import license of China (2015)
7404000010	7	Low-grade waste cables, waste motors, waste transformers and other hardware	20%-70%	Automatic import from 2013.01.01 Restriction from 2015.01.01 Ban from 2018.12.31	Automatic import license of China (2014); Catalogue for Administration of Import of Solid Wastes (2018, 06); Catalogue for Administration of Import of Solid Wastes (2018, 68)
7404000090	6	Bright copper, No.1, No.2 and clean or new yellow, red and semi-red brass	70-99.9%	Restriction from 2019.07.01	Catalogue for Administration of Import of Solid Wastes (2018, 68)

3.2.3.2 Manufacturing and fabrication

To satisfy final domestic demand, we analyzed the production of semi-finished (c) and finished (d) products. The international trade simulation of semi-finished and finished products was based on the historical relationship between GDP and the volume of imports and exports of semi-finished products and finished products, respectively (Figure S3.4). Using these numbers on trade together with a loss rate (FR) of 1% during semi-finished goods production and a fabrication efficiency (F_{Ex}) of 89% during finished goods production, we determined the copper input for production of semi-finished goods.

3.2.3.3 Primary copper production

Primary production-Domestic copper mining. The starting point for calculating primary copper is modelling of domestic copper mining (a). As it is unknown how domestic production of copper concentrates will develop in the future, we referred to historical data on copper mining, primary copper trade and primary copper supply to project the share of domestic mining in primary copper supply (as shown in Figure S3.3). To estimate future domestic mining, we extrapolated the share of domestic primary production in total supply based on the data from 2006 to 2015 (a rate of change of -1.5% per year).

Total primary copper supply. Having determined the copper input required for production of semi-finished products, we set primary supply to cover the difference between recycled supply from secondary sources and the required production input:

$$F_{b,t} = F_{c,t} - F_{n,t} \quad (3.9)$$

$$MX_{cr,t} = F_{b,t} - F_{a,t} \quad (3.10)$$

where $F_{b,t}$ is future primary copper supply in year t, $MX_{cr,t}$ is net trade of primary copper in year t, $F_{a,t}$ is future domestic copper mining in year t and $F_{c,t}$ is future domestic production of semi-finished products in year t.

3.2.4 Design of copper demand and supply scenarios

Two copper demand scenarios are explored: the Continuity Policy (CP) scenario and the Circular Economy (CE) scenario. They are summarized in Table 3.3. Both scenarios were constructed based on the same stock as estimated above. In the CE scenario, we modeled reduced demand by assuming that re-use, refurbishing and waste prevention will lead to extended lifetimes of copper products. Lifetimes have a direct impact on outflow, which means that extending product lifetime can reduce copper scrap generation in a determined period of time. Therefore, to model the waste generation before

2016, the lifetimes of end-use copper applications in the CP and CE scenarios were assumed to be the same. For the period from 2017 to 2100, we kept the lifetimes of end-use copper applications constant in the CP scenario, while in the CE scenario we extended the lifetimes of 29 sub-categories of end-use copper applications with a logistic regression based on the current level. The average lifetimes of copper products under sub-categories (e.g. cables in buildings) were assumed to be the same with the average lifetimes of sub-categories. Detailed lifetimes of each sub-categories can be found in Table S3.1 of appendix 1.

To explore how the country should meet its future copper demand, copper supply scenarios (CP and CE) were designed corresponding to the CP and CE demand scenarios (Table 3.3). Compared with the CP supply scenario, higher recycling rates were applied in the CE supply scenario. Recycling rates determine how much copper waste is recyclable from EoL copper products. An increase in the EoL recycling rate therefore means more copper remains in the cycle as a valuable secondary resource, while permanent losses to landfill, dissipation and other material cycles are reduced. Under the CP scenario, the recycling rate, which is determined by the collection rate (83%) and processing rate (20%~90%, category-specific), was assumed to remain constant from 2016 onwards (Table S3.3). Under the CE scenario, the collection rate and processing rate were assumed to increase from the 2016 value under the CP scenario to 90% and 95% for each end-use sub-category in 2100, with a linear regression.

Regarding the importance of secondary copper in the copper supply chain and the benefits of moving toward a more circular economy, we propose two hypotheses based on the CP (Hypothesis 1, H1) and CE (Hypothesis 2, H2) scenarios. In these hypotheses, all Chinese copper demand (from the CP and CE scenarios) is met by secondary copper.

Table 3.3 Summary of copper demand and supply scenarios

Scenario	**Scenario description**	**Common parameters**	**Parameter changes**	
			Lifetime	Recycling rate
Demand perspective				
Continuity Policy scenario	A scenario for future development that assumes there will be certain changes under activities of population, economic development and government policies, so that the copper industry can be expected to change and develop correspondingly.	GDP, Population, Urbanization rate, Copper intensity and Chinese government policy	Regular lifetimes of copper products	-
Circular Economy scenario	A scenario for future development based not only on certain changes in demographics, economic welfare and government policies, but also on a reduction of copper demand, e.g. due to prolonged product lifetimes.		Extended lifetimes of copper products	-
Supply perspective				
Continuity Policy scenario	A supply scenario to meet the Continuity Policy scenario for copper demand.	"Green Fence" policy on imported copper scrap	-	Constant recycling rates of EoL products
Circular Economy scenario	A supply scenario to meet the Circular Economy scenario for copper demand, and encourage the reduction, recycling and reuse of copper.		-	Higher recycling rates of EoL products

3.2.5 Data source

The data used for estimating copper demand were derived mainly from Dong et al. (2019) and other publications and reports, as specified above. The historical data on fabrication efficiency, loss rates during copper production (semis production, new scrap collection, scrap refining, separation, dissipation) were taken mainly from Soulier et al. (2018a), Soulier et al. (2018b) and Pfaff et al. (2018). Data on historical trade of copper products were taken from the United Nations Comtrade Database and China Nonferrous Metals Industry Association, as shown in Figures S3.3 and S3.4. To examine the robustness of the results according to the data used, a sensitivity analysis was conducted. Detailed definitions of and data on the items used for the Chinese copper cycle are provided in Table S3.3. The Supporting files can be found online at https://doi.org/10.1016/j.resconrec.2020.104943.

3.3 Results and discussion

Here, we present results and discussions for the historical copper supply of China (Section 3.3.1), the resulting of future copper demand (Section 3.3.2), the resulting of future copper supply (Section 3.3.3), the discussions on sustainability performance of future Chinese copper cycle (Section 3.3.4) and uncertainty and limitations of this study (Section 3.3.5).

3.3.1 The historical copper supply of China: pre- and post-implementation of the “Green Fence” policy

Figure 3.3a shows copper supply in China between 2005 and 2016 from our model, compared to statistical data. To meet the country’s growing demand, the supply of both primary and secondary copper have increased in our model. Primary copper has served as the main source, with imported primary copper accounting for more than 70%.

Secondary copper production has increased in recent years, but its share in total copper supply has been declining. Before the implementation of the “Green Fence” policy, secondary copper was satisfied mainly by imported copper scrap. In 2005, the imported scrap, domestic new scrap and domestic old scrap accounted for 51%, 34% and 15% of total scrap supply, respectively. After the implementation of the policy, secondary copper has been supplied mainly by domestic scrap (new and old scrap). Generation of new copper scrap is determined mainly by the processing efficiency of fabrication and the loss rate of new scrap collection, which means that the increased processing efficiency has contributed to slower growth of new scrap generation.

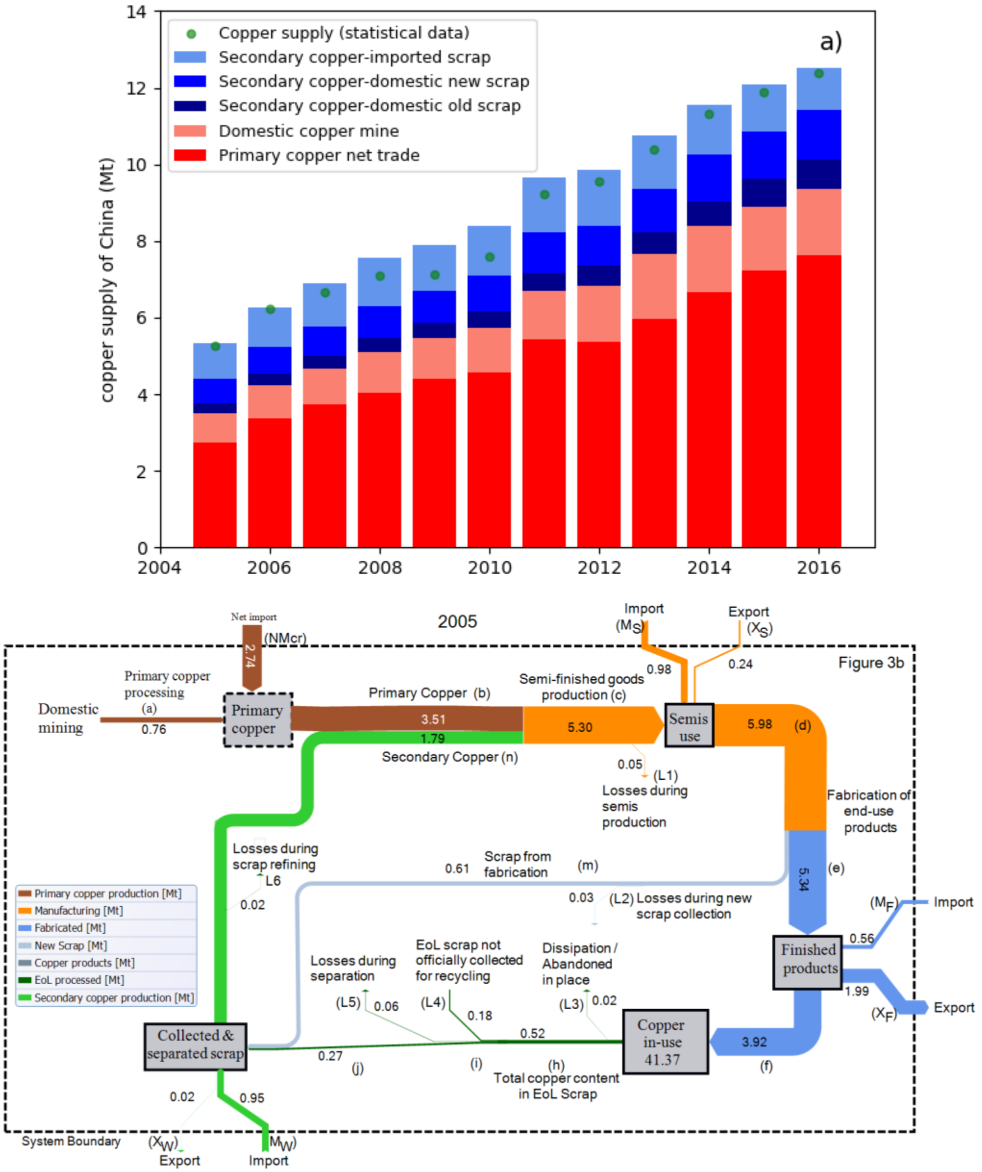


Figure 3.3 a) Copper supply of China from 2005 to 2016; b) copper cycle of China in 2005

At the same time, although the EoL recycling rate has increased, most of China's copper products had not entered its retirement period prior to 2016, given the 30-year overall average lifetime of copper products, resulting in very slow growth of EoL copper recycled in China. Limited domestic copper mining implied that primary copper production within China was insufficient to meet historical copper demand. At the same time, the "Green Fence" policy has affected the copper supply structure of China, resulting in inadequate copper scrap for producing secondary copper and consequently in the need to import more primary copper.

3.3.2 Copper demand projections for China

Figure 3.4 shows the developments of the copper stock and copper demand in the CP and CE scenarios up to 2100. The copper stock in China (identical under the CP and CE scenarios) is increasing rapidly owing to the enduring growth of copper use in power generation and transmission, buildings and transportation. Infrastructure is expected to hold the greatest copper stock, within which electricity applications account for the largest fraction. The stock of buildings will likely stabilize in the latter half of the century and still occupies the second largest copper stock.

Copper demand is expected to increase in near future, with different growth rates for different copper categories in the CP scenario, reaching a level in 2100 around 2.9 times higher than in 2016. Unsurprisingly, copper demand is lower in the CE scenario, since extended lifetimes of copper products mean a slower replacement rate and hence reduced demand. As a result, copper demand peaks around 2050 and then gradually declines in the CE scenario. Previous studies show different trends of future Chinese copper demand due to different methods, considered copper products and assumptions. For a short-term estimation, Yang et al. (2017) and Wang, J. et al. (2019) estimate the demand in 2030 to be around 15.4 and 12 Mt respectively. The outcomes in our model are comparable and in the range of existing studies. Adopting a long-term perspective, Zhang, L. et al. (2015a) estimated that demand will peak in 2042 with around 8.3 Mt and decline until 2080 to around 7 Mt. The values are very low in comparison with our results. Possible reasons for this large difference might be categorization of copper products that had been included and assumptions in model. In this study we included new energy applications that will demand major amounts of copper. In particular, our simulation is conducted according to the government development plans rather than assuming that the development level in 2050 is based on the regression of the previous year's data, such as the electricity generation and demand plan, which makes a vast difference. Subsequently, OECD (2019) displayed that the demand will increase to around 27 Mt in 2060 based on a central baseline scenario, which fits better with our results.

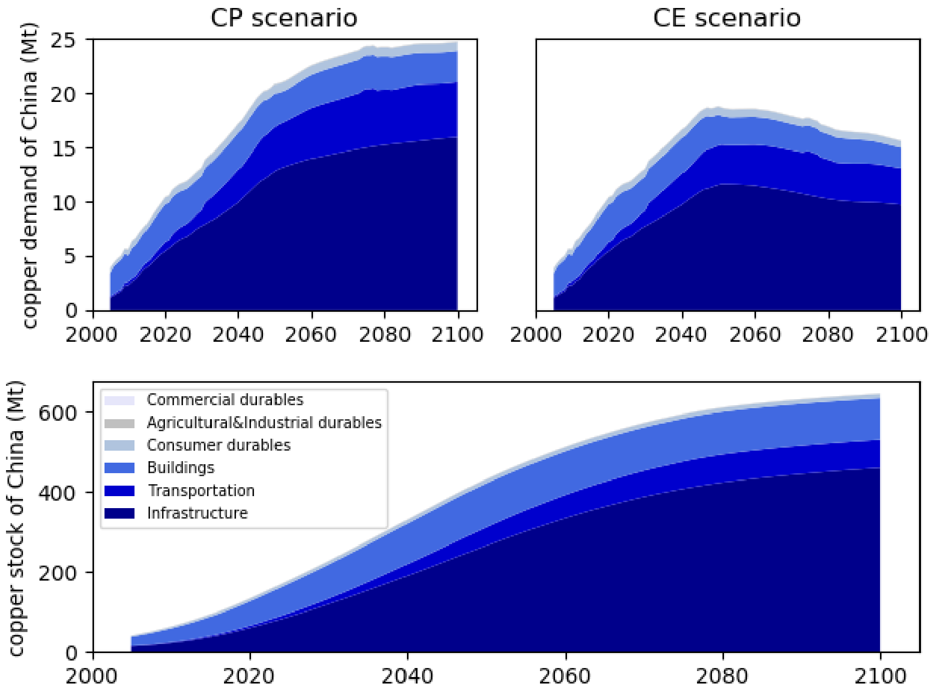


Figure 3.4 CP and CE scenarios of copper stock and final domestic copper demand in China up to 2100

In both scenarios, demand for copper is generated predominantly by the infrastructure sector, followed by transportation, buildings, consumer durables, agricultural & industrial durables and commercial durables. Note that the CP and CE scenarios are both based on the same projection of in-use copper stock, with the differences in copper demand emerging through different replacement rates under different lifetime assumptions.

3.3.3 Copper supply projections for China

Figure 3.5 shows the copper supply scenarios for China up to 2100. Primary copper supply reaches its maximum around the year 2060 and then declines in the CP scenario, whereas this point is already reached before mid-century in the CE scenario. Moreover, the proportion of primary copper to total copper supply decreases after 2020 in both scenarios. By then, production of secondary copper has gradually increased and become a major source of copper supply. In 2100, secondary copper will supply 55% of copper demand in the CP scenario, while this figure will be 60% in the CE scenario.

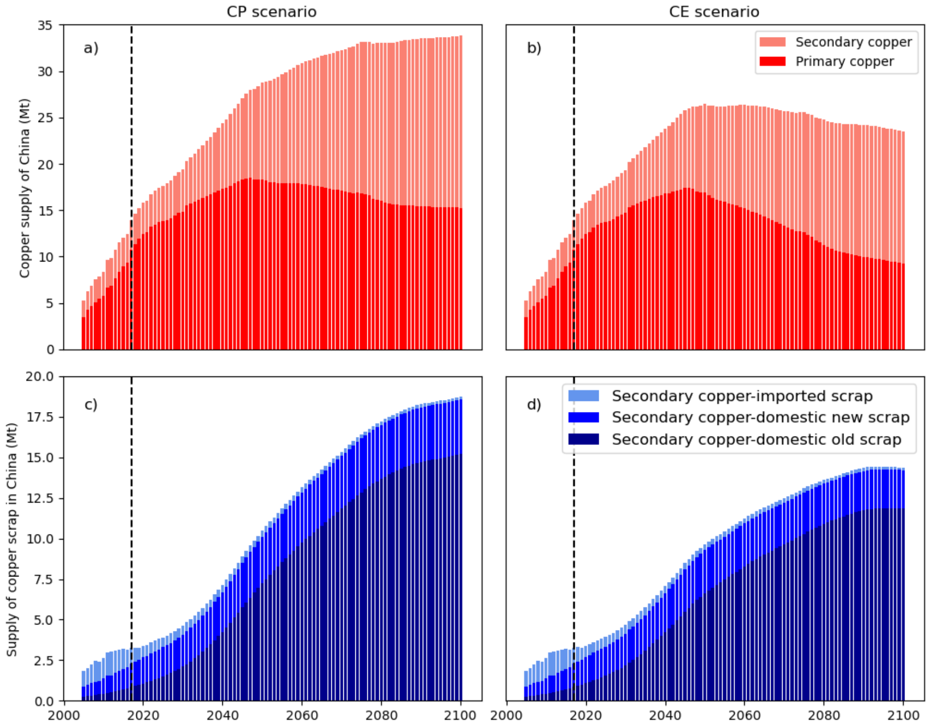


Figure 3.5 Chinese copper supply scenarios from 2005 to 2100. The vertical dashed line marks the boundary between historical data and future scenarios. Copper supply in the CP (a) and CE (b) scenario. Different scrap sources for secondary copper production in CP (c) and CE (d) scenario.

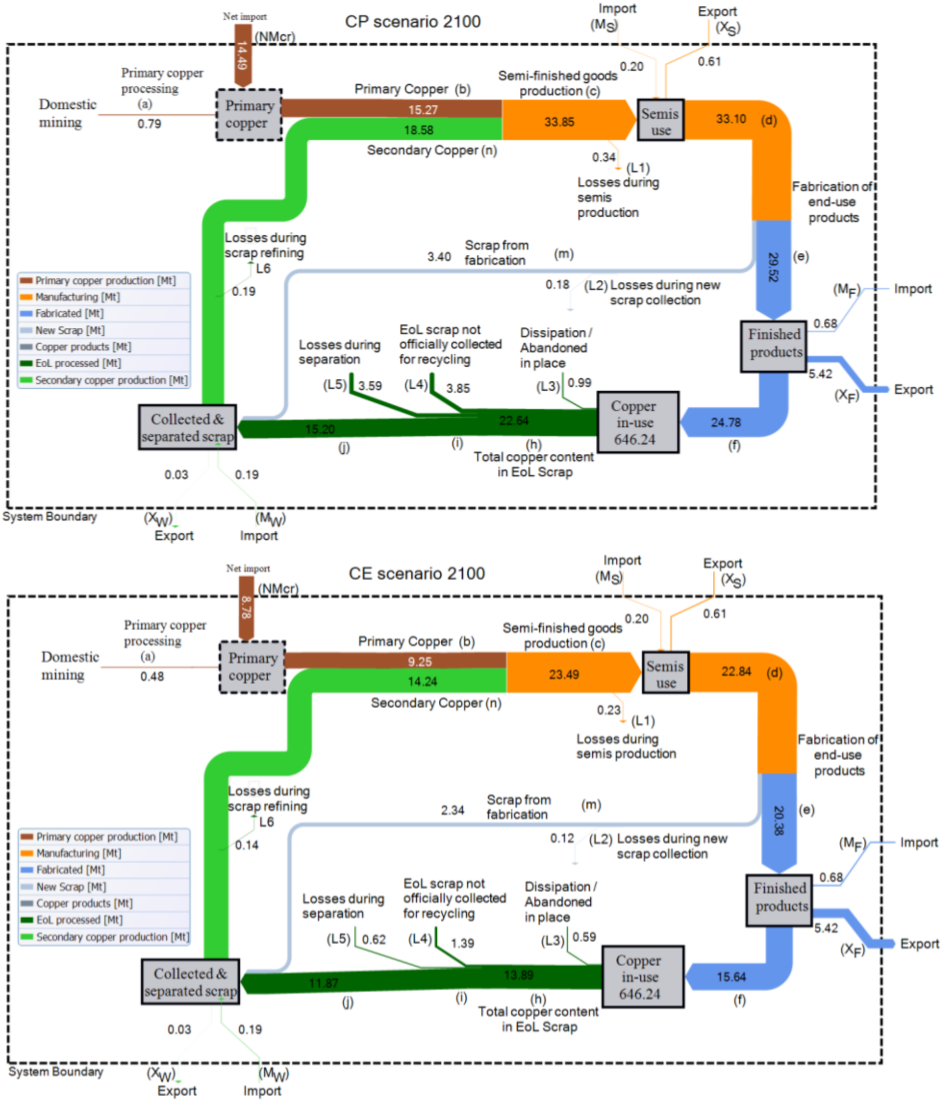


Figure 3.6 Future copper cycle of China in 2100 under the CP and CE scenarios

As mentioned above, before the implementation of the "Green Fence" policy, imported copper scrap played an important role in Chinese secondary copper production. However, with the increase in the generation of old copper scrap in China, both scenarios show that domestic old copper scrap will become the main source of secondary production. As shown in Figure 3.6, in the CP scenario, the proportion of old scrap to total copper scrap increases from 19%

in 2013 to approximately 81% in 2100. In the CE scenario, the proportion is even higher, with 83% in 2100. Comparing the two scenarios, we find there is 9% less domestic scrap in the CE scenario than in the CP scenario in 2050, with the figure rising to 30% in 2100. The reason for this is the time delay between copper use in production and scrap generation, especially for products with long lifetimes. Longer product lifetimes will reduce the amount of scrap generated, while increased copper recycling rates will boost secondary copper production. Future copper demand in China will therefore be satisfied substantially through increased copper recycling from domestic old scrap.

3.3.4 Discussion on sustainability performance of future Chinese copper cycle

As part of the transition to a circular economy in China's industrialization and urbanization process, the copper industry is also actively exploring moving from a linear development pattern to a circular economy pattern. Given this process of transformation along with implementation of the "Green Fence" policy, our scenarios point to a number of opportunities as well as challenges for the future development of the Chinese copper industry.

3.3.4.1 Prospective development of circular economy for copper demand

Copper is widely used in buildings, transportation, home appliances and industrial machinery, which implies that Chinese copper demand is expected to grow continuously as these sectors develop further. In order to reduce the environmental burden caused by the copper production related to this expanded copper demand, many scholars have proposed that the prime option should be to simply reduce copper demand. At the same time, though, they also recognize that copper demand is not readily reduced by cutting back on the use of copper products (Dong et al., 2019; Elshkaki et al., 2018; Van der Voet et al., 2018), which will not be easy to realize in China since it is still in the midst of development. Additionally, China is currently shifting from a fossil-based energy system to a renewables-based energy system. In particular, electrification of the energy system, a key strategy in the energy transition, requires massive amounts of copper.

Extending the lifetimes of copper products is an obvious option to achieve slower replacement rates and thus reduce future copper demand. Furthermore, many studies have shown that extending product lifetimes is not only beneficial to consumers, but can also alleviate the environmental burden caused by waste treatment by reducing the amount of waste that has to be dealt with, and thus achieve the goal of a more circular economy (Ardente and Mathieux, 2014; Wang et al., 2017). However, a product's lifetime is determined during the design stage, when establishing such matters as product structure and material selection. Another option is to repair and reuse products that would otherwise be scrapped. This option is also related to product design, since easy replacement and easy maintenance of product parts will facilitate product reuse and repair, thus extending lifetimes. Although product design is complex, it is certainly an effective way to lengthen copper product lifetimes, maintaining the copper's value for as long as possible by keeping it in the economic system.

3.3.4.2 Challenges in primary copper production

Worthy of discussion are the copper supply projections in this paper, since they are built on the assumption that no limitations exist on copper reserves in China. However, such limitations do exist. A report issued by the China Geological Survey states that copper reserves and identified resources of China in 2016 are 28 and 101.1 Mt, respectively (CGS, 2016). It can be seen that China's domestic copper production has been increasing in recent years, but it is still unable to meet Chinese copper demand. Based on our results, domestic primary copper production is not and will not be adequate to meet the required amount of primary copper, and China will have to import substantially more copper ore or refined copper (as shown in Figure 3.7). Consequently, the share of imported primary copper in total primary copper supply is likely to increase significantly, which is undoubtedly a supply security issue worth considering for the Chinese copper industry. This is in agreement with Wang, J. et al. (2019) who show a large gap between Chinese copper demand and supply in the future. Furthermore, on the global level, Mohr (2010) and Northey et al. (2014) have provided projections of future potential copper production and concluded that global copper production will peak around 2032 and rapidly decline thereafter (Figure 3.7b). Following these projections, China would not have enough primary copper sources to import from in the future.

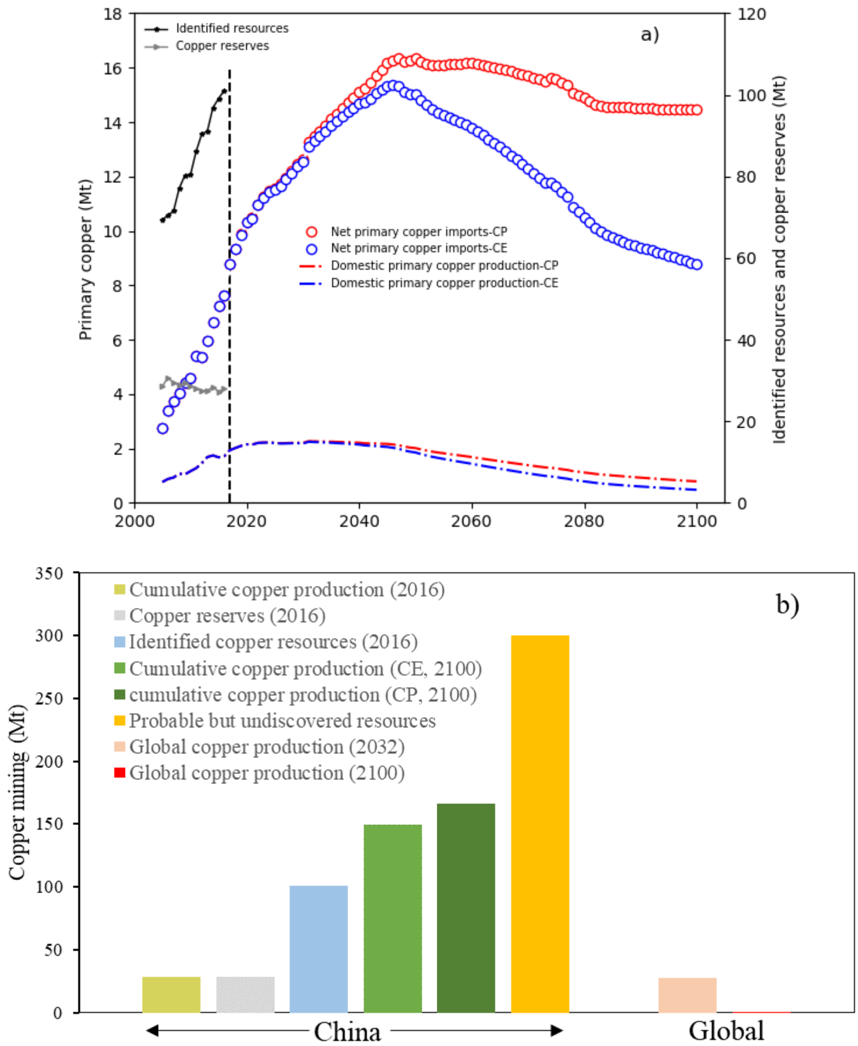


Figure 3.7 a) Right y-axis: Domestic primary copper production, copper reserves, identified copper resources in China (data: National Bureau of Statistics of China and Ministry of Natural Resources of China); left y-axis: domestic primary copper production and primary copper that needs to be imported in CP and CE scenarios. The vertical dashed line marks the boundary between historical data and future scenarios; b) Copper mining in China and the world (global data source: Mohr (2010) and Northey et al. (2014)).

Despite these projections, future Chinese primary copper production is unlikely to be physically limited. On the one hand, to ensure long-term supply security in line with the National Mineral Resource Planning (MNR, 2016), the Government will likely provide strong support to domestic exploration as well as overseas acquisition. Copper reserves are likely to increase in the future as additional deposits are discovered and/or new technology or economic variables improve their economic viability (e.g. probable but undiscovered resources in Figure 3.7b). Under these assumptions, the potential cumulative copper production up to 2100 might be satisfied by domestic resources. On the other hand, improved copper refining and copper recycling technologies and possibly the use of copper substitutes will also reduce demand for primary copper or total copper. No matter what kind of developments occur in the future, however, the issue of security of supply of primary copper remains relevant for China owing to its high import dependence.

3.3.4.3 Opportunities and risks for China related to the import of copper scrap

One possible way to improve security of supply is to increase reasonable proportion of secondary production. As the amount of copper scrap generated domestically in China increases, the implementation of this measure becomes more feasible. It has the additional advantage that the environmental impacts of producing 1 kg of secondary copper are much less than that for 1 kg of primary copper (Kuipers et al., 2018; Norgate, 2001; Van der Voet et al., 2018). It has been pointed out that the energy requirements and associated GHG emissions of secondary copper production may surpass those of primary copper production with EoL recycling rates approaching 100% (Norgate, 2004; Schäfer and Schmidt, 2020). This implies that primary production and secondary production need to be balanced to minimize energy consumption and environmental impacts in a circular economy scenario. However, considering the advantage of conservation of natural resources and utilization of waste, increasing secondary production is still an important long-term option to achieve a circular economy.

Relying entirely on secondary production is also problematic for additional reasons. If China were to rely solely on secondary production, the amounts of copper scrap that would have to be imported would be quite substantial in both scenarios, especially in the CP scenario (Figure 3.8). According to the data presented by Elshkaki et al. (2016) and Schipper et al. (2018), global copper scrap availability is expected to reach around 25 Mt in 2040 and 50 to 220 Mt in 2100, depending on the scenario. In fact, China would use around 24 Mt or 23 Mt of copper scrap, of which around 17 Mt are from imports, under the hypotheses (H1 and H2) where secondary production dominates in 2040. This is a major share of all the EoL scrap expected to be generated globally in 2040, which clearly is not feasible. Furthermore, some countries or regions proposed to transition to a more circular economy and will increase recycling. What this indicates, then, is that other countries will also need large amounts of scrap and there would be a major shortage of copper scrap if China decides to move to a completely closed-loop economy before mid-century. By 2100, however, China would use half or less of global copper scrap. In the CP scenario, therefore, China will probably face an even tighter scarcity of copper scrap.

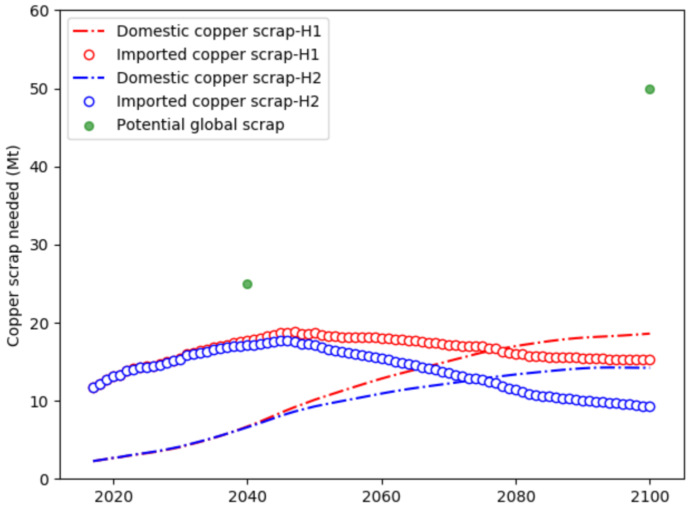


Figure 3.8 H1/H2: Domestic copper scrap supply and required imported copper scrap under the CP and CE scenarios, assuming a phase-out of primary production. Calculations based on the assumption that Chinese copper demand under the CP (H1)/CE (H2) scenario is met solely by secondary copper produced under the corresponding CP (H1)/CE (H2) supply scenario. Data of potential global scrap is derived from Elshkaki et al. (2016) and Schipper et al. (2018) that ranges from 50 to 220 Mt in 2100. Here we used the least value (50 Mt) in 2100.

Therefore, the implementation of "closing the loop" on the national level is challenging because of domestic resources availability, environmental benefits and trade-offs on the way towards a circular economy on the global level. It may mean that creating a circular economy at the global level will take some time, not just because it is hard to organize, but because it can only be established after stock saturation has occurred on a global scale, so copper demand will no longer be growing, allowing scrap generation to catch up.

Considering that the production of domestic copper scrap and primary copper are still unable to meet current copper demand in China, the most reasonable way forward, by and large, is probably somewhere between a major dependence on primary copper import and a major dependence on scrap imports. Given that secondary copper production has (up to a certain point, see above) less environmental impact than primary copper production, China should still consider importing copper scrap to partially meet its copper demand. An increase of high-quality copper scrap imports instead of importing copper ore or refined copper is a feasible and environmentally beneficial option for further development of China's copper industry. This option may indeed force more industries, in China as well as elsewhere, to establish facilities for copper dismantling and recycling.

3.3.5 Uncertainty and limitations

Exploring copper demand and supply throughout the 21st century involves significant uncertainty. A simulation exercise like the present one does not aim for precise numbers, but for patterns and trends reproducible under many different simulation parameters and assumptions. The long-term projections of GDP, population, copper content, trade and many other parameters are based on historical data and policy-oriented inference. Although our simulation is based on reliable historical data and policies, there is no mechanism for unambiguously indicating the future development trajectory, especially for the trend post-2050 and trade in copper-containing products. However, it is highly likely that the output of Chinese copper scrap will increase significantly under any assumption, which will accelerate the development of secondary copper supply.

As pointed out above, the lifetimes of copper products are absolutely crucial for the modelling of copper demand and supply. Glöser et al. (2013) found that the effect of changing the shape (functional form) of product lifetime distributions is small compared with the effect of changes in average lifetimes. To investigate the effect of changes in average lifetimes, an initial sensitivity analysis was therefore carried out in which it was assumed that the average lifetime of each sub-category in the CE scenario deviates in the same way (either all shorter or all longer) and in the same proportion. A +10% or -10% change in the average lifetime of each sub-category was assumed. The results indicate that, as could be expected, copper demand is lower in the scenario with longer copper product lifetimes (Figure 3.9). Over the next 20 years, the difference between the effects of +10% lifetimes and -10% lifetimes on secondary copper supply is limited, but in the long term it is considerable.

In addition to uncertainties in average lifetimes, there is considerable uncertainty concerning the recycling rate of copper scrap. In the CE scenario, the recycling rate is determined by the loss rate during separation and the collection rate. The separation rate was modeled to be around 95% in 2100, which is already a very high recovery efficiency. To examine the effect of variable recycling rates, therefore, only the collection rate was considered and assumed to change by ±5%. Higher collection rates could increase the amount of recycled copper, thus improving the secondary copper supply, as shown in Figure 3.9. Several studies have also found that the recycling rate is the most important indicator for analyzing the efficiency of EoL copper products treatment and have recommended that it be improved (Graedel et al., 2011; Ruhrberg, 2006). This is not easy to implement, however, since it involves technical updates to a range of processes, including product design, disassembly and material separation (Glöser et al., 2013; Ruhrberg, 2006; Spatari et al., 2002).

Copper content is another key factor in determining copper stocks, but it is difficult to make any reliable forecast for future changes. In some applications, such as power plants (e.g. using copper to substitute Ag in Photovoltaic power plant), new energy technologies and new energy vehicles, the copper content is rising (Månberger and Stenqvist, 2018). In other cases, copper content could be decreasing due to substitution by other materials. Aluminum is considered to have the greatest potential for replacing copper in energy infrastructure, but primary aluminum has higher energy requirements and corresponding CO_2 emissions per kg production than copper (Van der Voet et al., 2018). Given the required properties, other products may also be eligible for partially replacing copper, for example through stainless steel, zinc and plastics, but this could come with the drawback of more difficult recycling (García-Olivares, 2015). Furthermore, it is by no means straightforward to find suitable alternatives that have the same functionality and can be produced in equally large quantities (Commission, 2017). Therefore, more analysis is still needed to assess the long-term use and impacts of these substitutes.

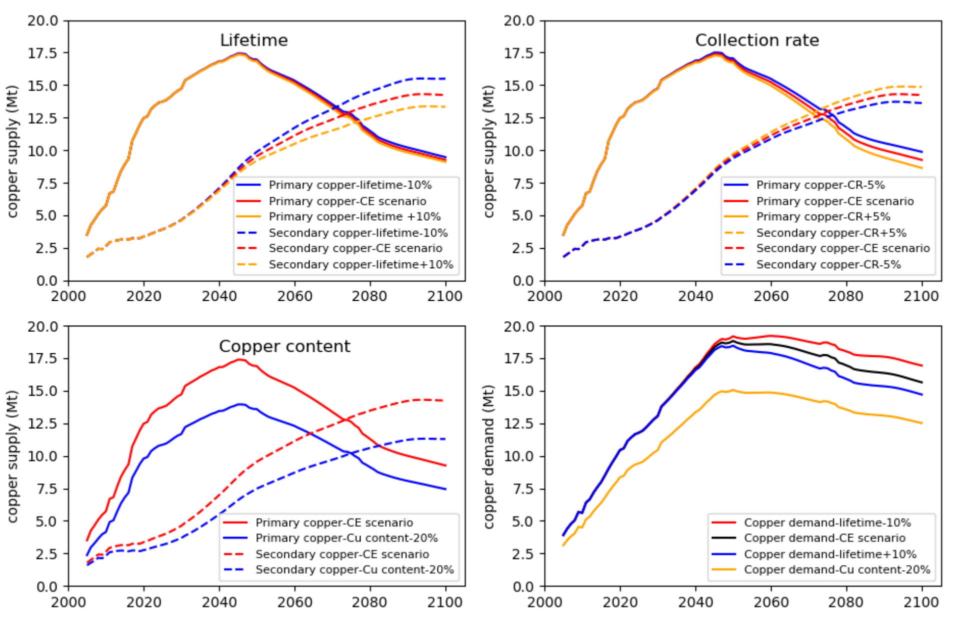


Figure 3.9 Sensitivity analysis of copper product lifetimes, recycling rate and copper content based on CE scenario

Even though copper content data are quite uncertain, a sensitivity analysis was conducted to examine the effect of it to copper demand and supply. The copper content of each sub-category was assumed to be 20% less than current level based on CE scenario. The results show that reducing copper content would lead to rather straightforward changes: reducing the copper content by 20% would decrease the demand by the same percentage (Figure 3.9).

3.4 Conclusions and implications

In this study, we used a stock-driven method to estimate the implications of a continuity policy scenario ("Green Fence" policy: ban of cat. 7 scrap and restrictions on cat. 6) and a circular economy scenario ("Green Fence" policy: ban of cat. 7 scrap and restrictions on cat. 6; increasing product lifetimes and recycling rates) for copper demand and supply in China up to 2100. We explored the scenarios with respect to the question of how China's copper demand can be met, and what impact Chinese government proposals to ban the import of copper scrap will have on the Chinese copper supply. On the basis of our results, we suggest that China could benefit from an increase in secondary copper supply by importing more copper scrap. Although certain limitations may affect the accuracy of the results, current trends of primary and secondary copper supply are unlikely to change significantly. There are therefore still good grounds for drawing the following conclusions.

- Before implementing the "Green Fence" policy, primary copper was the main source of China's copper supply. Imported copper scrap was the major source of secondary copper production, mainly because domestic old scrap production was not yet significant.
- Effective measures to increase product lifetimes could lead to significantly reduced copper demand. Under the assumptions made in this study, decreasing copper demand in the second half of the century appears possible.
- The contribution of secondary copper to total supply is likely to steadily increase in the coming decades as a result of increased availability of domestic scrap. When combined with decreasing copper demand (CE scenario), secondary copper could provide the bulk of China's copper supply towards the end of the century.
- However, there will be a substantial gap between Chinese copper demand and the amount of scrap available domestically. In the future, this gap needs to be closed by means of domestic mining or through imports. Given the limited scope for domestic mining, China will still have to depend largely on imports of primary material in the form of concentrates and refined copper or, otherwise, put major emphasis on its recycling industry and continue to import scrap. In this manner, secondary sources would be able to meet a large part of China's growing copper demand. In combination with the establishment of a state-of-the-art, efficient and environmentally friendly recycling industry, this could be an opportunity for China to transition to a more circular economy with regard to copper.

Chapter 4

Assessing the future environmental impacts of copper production in China: implications of the energy transition

Abstract

Copper demand in China is expected to grow considerably over the coming decades, driving energy use and environmental impacts related to copper production. To explore the environmental impacts of copper production in China, we used a variant of Life Cycle Sustainability Analysis that combined the Life Cycle Assessment methodology with the Chinese copper demand projections from 2010 to 2050. The results indicate that the environmental impacts of pyrometallurgical copper production are expected to increase more than twofold during this period and remain the largest contributor to the environmental footprint. Secondary copper production emits the least pollutions. Increasing the share of secondary copper production is the most environmental friendly option for copper production. To this end, China may focus on improving the classification of waste copper products and recycling infrastructure for end-of-life management. Hard coal use and production are crucial contributors to climate change in the context of copper production. Cleaning up copper production processes and improving energy efficiency would also help reduce environmental impacts. Energy transition can significantly reduce the environmental impacts of copper production, but it also can increase copper requirement. It does not visibly contribute to reduce human toxicity as well.

Published as: Dong, D., van Oers, L., Tukker, A., & van der Voet, E. (2020). Assessing the future environmental impacts of copper production in China: Implications of the energy transition. Journal of Cleaner Production, 274, 122825. https://doi.org/10.1016/j.jclepro.2020.122825.

4.1 Introduction

Copper production is a basic raw material industry that provides one of the key non-ferrous metals for infrastructure and buildings. It is also energy intensive as energy is used in the whole life cycle of copper production, including mining, beneficiation, smelting and refining, not only in the directly processes but also through the indirectly production of inputs, e.g. electricity generation. Globally, copper production requires around 600 million Gj of energy annually and contributes 0.21% of total greenhouse gases (John, 2012). On the other hand, copper demand has been increasing in the past decade and will continuously expand due to growing population, developed infrastructure and the application of copper-intensive technologies. A further consequence of this is the occurrence of serious environmental pollution and ecological damage, including biodiversity and water-quality losses, around mining sites.

Life Cycle Assessment is a methodology that is widely used to assess the environmental impacts of products and materials (Norgate, 2001; Norgate et al., 2007; Van Genderen et al., 2016). The environmental impacts per unit (kilogram) metal production (steel, aluminum, zinc and lead, among others) have been analyzed by various authors based on the inputs and outputs of production processes (Davidson et al., 2016; Zhang et al., 2016). Tan and Khoo (2005), Nunez and Jones (2016) and Farjana et al. (2019a) studied environmental impacts of primary aluminum production. Ferreira and Leite (2015) and Gan and Griffin (2018) analyzed environmental impacts of iron ore mining and processing . However, Farjana et al. (2019b) pointed out that many LCA studies have an incomplete coverage of production processes due to data limitations. Several scholars investigated the environmental impacts of mining processes with LCA, and identified key contributors to one or more impact categories (coal, Burchart-Korol et al. (2016); gold, Haque and Norgate (2014); nickel, Khoo et al. (2017); rare earth elements, Weng et al. (2016); uranium, Parker et al. (2016)).

LCA studies of copper production have been published as well. Some scholars have distinguished different copper production routes and have compared the environmental impacts of primary (pyrometallurgical and hydrometallurgical) and secondary production (Hong et al., 2018; Kuipers et al., 2018; Wang et al., 2015). More specifically, Song et al. (2014) and Kulczycka et al. (2016)

have quantified the environmental impacts of copper production processes based on specific technologies and explored potential options for reducing the energy use and environmental pressure associated with these processes. These studies have generally focused on assessing the environmental impacts of product manufacture using LCA and discussed the impacts per kilogram production.

However, analyzing the environmental impacts of copper production requires not only calculating the environmental impacts of producing one kilogram of copper. To assess the impacts of copper production, the total amount of produced copper has to be considered. Life Cycle Sustainability Analysis (LCSA) takes a life cycle approach but has a wider perspective (Guinée, 2016), which has been applied to assess the prospective environmental impacts and large scale system. On the one hand, it can include different types of impacts in addition to environmental ones, especially economic and social impacts. On the other hand, it can widen the spatial and temporal scope to include larger systems and future developments, such as the total use of materials and products in an economy and future developments. Plenty of publications have applied the LCSA method to analyze one or more aspects of environmental, social and economic impacts of production and use of products (Atilgan and Azapagic, 2016; Finkbeiner et al., 2010; Guinee et al., 2011; Keller et al., 2015; Nzila et al., 2012; Onat et al., 2016). Some studies have broadened the scope at global level by multiplying impacts per kg by global supply (in kg), in an attempt to assess past and future developments scenarios (Ayres et al., 2003; Norgate and Jahanshahi, 2011; Van der Voet et al., 2018). At lower scale levels it is more complicated: environmental impacts of mining are different per location, and copper production does not match copper consumption due to imports and exports, which is difficult to trace in many cases. Therefore, this study provides a comprehensive framework to assess environmental impacts of copper production at a national scale level, in this case, China. China is one of the world's largest economies. Presently, it accounts for 35% of global refined copper production. China has become the world's largest copper consumer and now uses 46% of global copper supply. Moreover, the Chinese copper demand is expected to rise considerably in the future (OECD, 2019; Zhang, L. et al., 2015a). Considering the intensive energy consumption, the total amount of energy consumed in the mining and beneficiation processes of the non-ferrous metals industry in China was around 3.5 billion Standard Coal units in 2015, with copper mining and beneficiation accounting for one-quarter of this figure (CNMIA, 2016). This number is likely to rise, given the increasing copper use in infrastructures, buildings and transportation, especially the increase use of renewable energy to power infrastructure and vehicles. Therefore, it is important to understand the copper production and consumption system to formulate recommendations for a more sustainable copper metabolism.

In this paper, we assessed the environmental impacts of copper production implied by projected future copper demand and supply of China. To this end, the objectives are:

- To assess the environmental impacts of 1 kg of copper produced by pyrometallurgical, hydrometallurgical and secondary production using LCA.
- To assess the environmental impacts of copper production combined with copper supply scenarios for meeting demand scenarios.
- To identify potential options for improving the environmental performance of Chinese copper production and consumption, by means of contribution analysis and other analysis.

In our explorations of the future, we include climate and energy policies to assess the impact of renewable energy use on copper demand, which may go up considerably as a result of electrification. On the other hand, copper production related to impacts per kg may go down as a result of a cleaner production of energy, which is included in our assessment as well. Furthermore, the imported copper concentrates and imported copper scrap and its differences in impacts compared to domestic copper production in China were explored in this study. The data used for domestic production represents China's average situation instead of a typical enterprise.

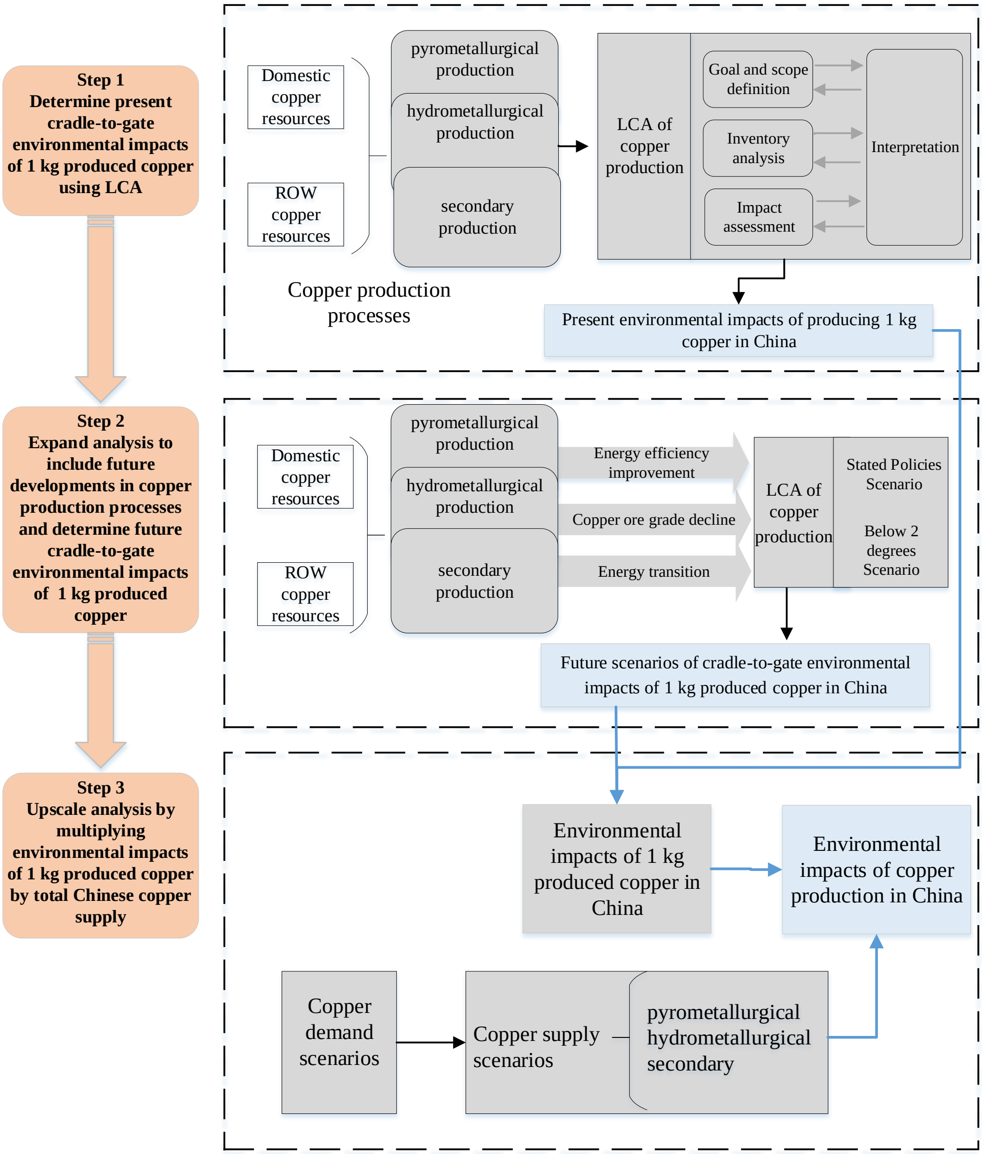


Figure 4.1 Framework of assessing future environmental impacts of copper production; “ROW” = Rest of the world

4.2 Methodology and data

To assess the environmental impacts of copper production combined with Chinese copper demand scenarios, in essence we used a variant of Life Cycle Sustainability Analysis (LCSA) as presented by Guinée (2016), but without including economic social or economic impacts (note further the abbreviation

LCSA is not meant to indicate a social assessment). In our case study on China, we focus the upscaling and forecasting. Or, alternatively formulated, we applied the LCA methodology not on a typical small-scale functional unit, but on the total current and future Chinese copper demand. The methodology comprises the following steps (Figure 4.1).

4.2.1 Methodology

4.2.1.1 Step 1 Determine present cradle-to-gate environmental impacts of 1 kg produced copper using LCA

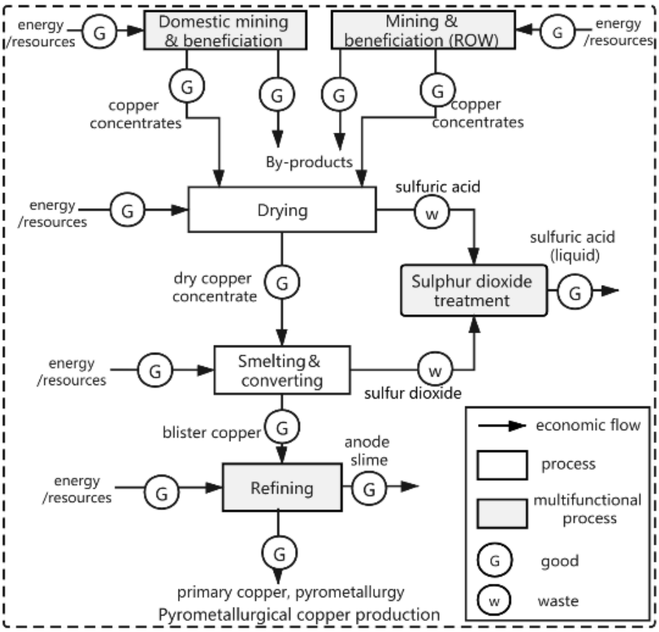
1) Copper production system

The "copper production system" refers to the technological routes adopted by the industries to produce copper. At present in China, there are three basic routes: pyrometallurgical (primary production), hydrometallurgical (primary production) and secondary production. Pyrometallurgical production is currently the dominant route in China, in general use for sulfide copper concentrates (Wang et al., 2015). In this study we distinguish the following main processes of this technology: mining & beneficiation, drying, smelting & converting, and refining (Figure 4.2).

Because of the continuous decline in quality of copper ore and the increase in complex minerals that are difficult to handle, China is shifting towards hydrometallurgical production (Wang et al., 2012). In this study this production route comprises the following main processes: mining, beneficiation, leaching & extraction, and electrowinning (Figure 4.2). In many cases this route still needs to be combined with pyrometallurgical production and depends on the grade and type of copper concentrates involved.

Secondary copper production can use the pyrometallurgical as well as the hydrometallurgical production route (Kuipers et al., 2018). The more modern hydrometallurgical technology has been used for recycling some EoL products, especially waste circuit boards and lithium-ion batteries (Liu et al., 2019; Perez et al., 2019). However, this method has not been applied on a large scale in China based on the investigation of the main copper recycling plants. Also in view of data issues, the secondary production presented in this study is assumed to follow the pyrometallurgical production route. The relevant processes are generally divided into two elements: pretreatment and recycling. Pretreatment includes collection and classification of the mixed waste copper and removal of other wastes on its surface to obtain a single stream of relatively pure waste copper. Copper recycling consists of smelting and converting followed by refining (Figure 4.2).

Besides domestic production, a significant amount of copper is imported from abroad as well. Imported copper concentrates have become an important source of copper production and are expected to remain so for decades to come. We therefore also included the impacts of imported copper concentrates, accounting for the variations in mining and beneficiation processes in other countries by taking a global average for imported copper concentrates. For secondary production, we included both domestic and imported scrap.



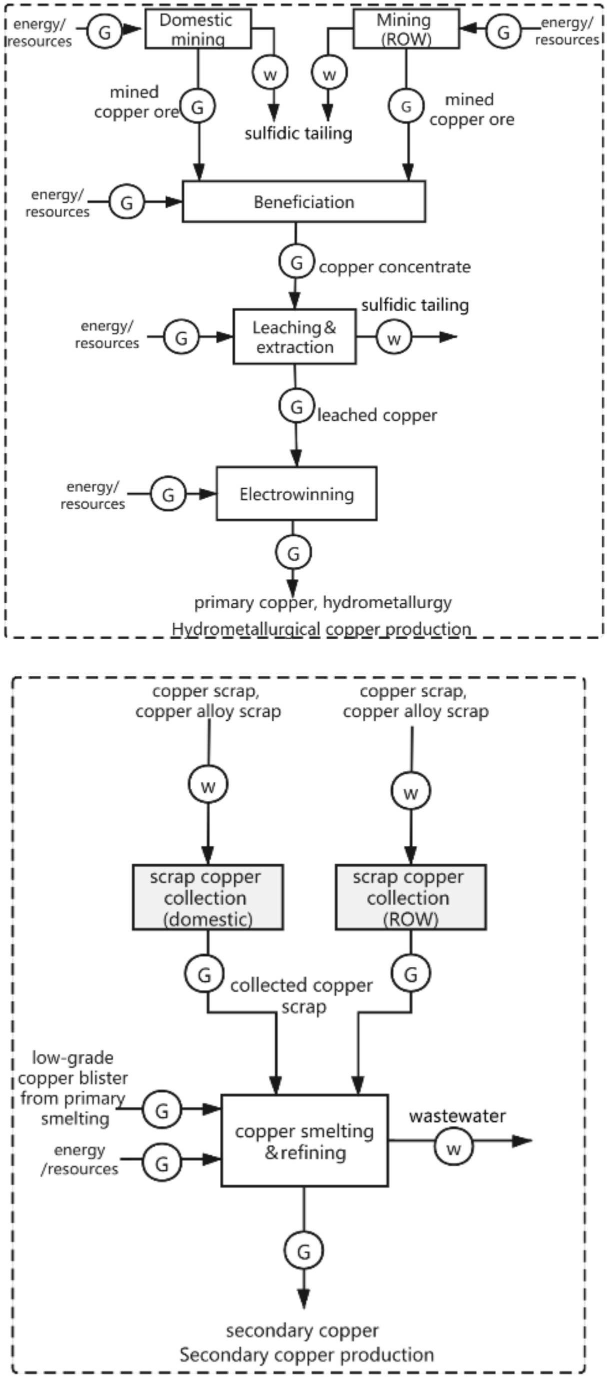


Figure 4.2 Chinese copper production system: pyrometallurgical, hydrometallurgical and secondary copper production

2) LCA of the present copper production system

The LCA methodology is widely used to assess the environmental impacts of metal production system (Liu et al., 2011; McMillan and Keoleian, 2009). It comprises four main phases: 1) goal and scope definition, 2) life cycle inventory, 3) life cycle impact assessment, and 4) interpretation (ISO, 2006a). For our assessment of the present environmental impacts of producing 1 kg copper, we took the following specifications:

- The goal is to assess the environmental impacts of copper production in China. The scope is cradle-to-gate production of 1 kg refined copper in China by pyrometallurgical, hydrometallurgical and secondary production.
- Life cycle inventory of the present copper production system. For the foreground processes related to copper production, separate data were collected. The mining and beneficiation processes associated with pyrometallurgical and hydrometallurgical production include domestic production in China and the rest of the world. The copper scrap refined in secondary production similarly derives from both domestic collection and imports. Allocation is part of the LCI stage (see the blue processes in Figure 4.2). Environmental impacts need to be partly allocated to by-products of copper mining, such as Molybdeen (Mo) and Silver (Ag). To this end, the partition method of economic allocation was adopted on the basis of revenue (market price multiplied by amount produced). The inputs and outputs of producing 1 kg refined copper with three technologies in 2015 is shown in Table 4.1. In addition, we used the Ecoinvent 3.4 database for the background processes of copper production system (Moreno Ruiz et al., 2017).
- The life cycle environmental impacts were conducted using the CMLCA 6.0 software and the CML2001 impact categories (CML, 2016; Guinée, 2002), which include eight commonly used indicators: acidification potential, climate change, freshwater aquatic ecotoxicity, human toxicity, photochemical oxidation (summer smog), abiotic depletion of resources-fossil fuels, abiotic depletion of resources-elements and cumulative energy demand.
- In the interpretation stage, a contribution analysis to identify the impacts in

different production processes was conducted. Sensitivity analysis on energy efficiency and copper ore grade were performed to examine the variability of environmental impacts due to data uncertainty.

Table 4.1 Inputs and outputs of the production process, scaled to the production of 1 kg refined copper with three technologies in 2015

	Material	**Pyrometallurgy**	**Hydrometallurgy**	**Secondary**	**Unit**
Economic inflows	Electricity	1.65E+00	3.97E+00	4.11E-01	kWh
	Diesel	1.23E+00	1.86E+00	8.80E-03	MJ
	Limestone	1.11E+00	2.77E-01	7.39E-02	kg
	Sulfuric acid	1.11E-02	2.45E-01	6.80E-03	kg
	Hard coal	3.21E-01	-	1.86E-01	kg
	Coke	2.70E-01	-	1.12E-01	kg
	Natural gas	4.44E-02	-	4.92E-01	m^3
	Oxygen	2.63E-01	-	9.14E-02	kg
	Xanthate	1.67E-02	-	-	kg
	Butylamine	1.99E-02	-	-	kg
	Heavy fuel oil	2.05E-02	-	-	MJ
	Copper extractant	-	6.19E-03	-	kg
Economic outflows	Refined copper	1.00E+00	1.00E+00	1.00E+00	kg
	Molybdenum concentrate	4.11E-03	-	-	kg
	Sulfidic tailing	9.70E+01	5.54E+02	-	kg
	Sulfur dioxide	4.45E-01	-	3.00E-03	kg
	Sulfuric acid	1.20E+00	-	-	kg
	Waste water	5.68E-03	1.41E-01	1.00E-03	m^3
Environmental resources	Copper ore	1.10E+02	3.07E+02	-	kg
	Imported-copper concentrate	2.41E+00	-	-	kg
	Domestic copper scrap	-	-	1.31E+00	kg
	Imported-copper scrap	-	-	5.73E-01	kg
	Molybdenum ore	2.64E+00	-	-	kg
	Water	1.17E+00	2.20E-02	1.17E+00	m^3
Environmental emissions	Carbon dioxide	5.88E+00	7.37E+00	1.59E+00	kg
	Antimony	4.49E-09	-	3.00E-06	kg
	Carbon monoxide	2.94E-08	-	2.00E-03	kg
	Dioxins	9.80E-15	-	-	kg
	Arsenic	3.36E-08	-	2.00E-06	kg
	Mercury	1.12E-09	-	-	kg

	Material	Pyrometa llurgy	Hydrome tallurgy	Secondary	Unit
	Nickel	1.79E-06	-	1.00E-06	kg
	NMVOC	1.47E-08	-	-	kg
	Particulates	1.14E-02	-	2.82E-04	kg
	Sulfur dioxide	2.28E-01	6.80E-02	3.00E-03	kg
	Water	6.52E-02	1.66E-01	1.76E-04	m^3
	Zinc	4.26E-06	-	3.75E-04	kg
	Leaching residues	-	2.76E+02	-	kg

Table 4.2 Main assumptions and data sources in assessing environmental impacts of producing 1 kg copper

Variables	Stated Policies Scenario (SP)	Below 2°C Scenario (B2D)	Data source
Ore grade decline	Chinese copper ore grade from 2016 to 2050 modeled using historic time series. The ore grades related to imported copper concentrates are based on global trends.		China Nonferrous Industry Statistical Yearbook, Kuipers et al. (2018), Northey et al. (2014)
Energy efficiency, foreground processes	Primary pyrometallurgical production: • Domestic production based on historical trends of Chinese mine production • Global level production: global historical trend data were used. • Primary hydrometallurgical production: • conservative estimate of 1% annual improvement of all processes, both for China and the world. • Secondary production: • scrap collection changes in efficiency assumed for China or the world. • refining: the same energy efficiency improvement rate of pyrometallurgical production.		China Nonferrous Industry Statistical Yearbook, Jiang et al. (2006); Kuipers et al. (2018); Ruan et al. (2010).
Energy transition, background processes	Domestic: the SP Scenario. Global: the New Policies scenario from IEA (IEA, 2017).	Domestic: the B2D Scenario. Global: the 450 scenario from IEA (2017), consistent with the goal of limiting global temperature rise to 2°C.	CNREC (2017), (IEA, 2017), ecoinvent v3.4 database (Moreno Ruiz et al., 2017)

4.2.1.2 Step 2 Expand analysis to include future developments in copper production processes

This step models future changes in the foreground and background systems and determines the resultant changes in environmental impacts of producing per kg copper. These impacts are affected by numerous variables associated with the various links in the copper production chain, including copper ore grade decline, energy efficiency improvements, the energy mix used for electricity production and transport (which the energy transition may alter substantially) and changes in material transport requirements (Norgate et al., 2007; Van der Voet et al., 2018; Vieira et al., 2012). Our analysis took into account ore grade decline, energy efficiency improvements and changes in electricity mix. Main assumptions are summarized in Table 4.2.

1) Copper ore grade

Ore grade decline is already occurring and is expected to continue in the future (Northey et al., 2014). Several studies have analyzed the energy inputs related to mining and beneficiation and established that energy use rises as ore grade declines (Alvarado et al., 2002; Ballantyne and Powell, 2014; Calvo et al., 2016; Norgate and Jahanshahi, 2010). The likely further decline of ore grade will therefore drive up the energy requirements of copper production (Mudd et al., 2013; Northey et al., 2014).

Using the historical data on copper ore grade in the China Nonferrous Industry Statistical Yearbook (CNMIA, 2016), we applied the power regression method reported by Crowson (2012) to simulate the change in copper ore grade up to 2050.

$$G = \mu * y^{-\varepsilon} \quad (4.1)$$

Where G is the copper ore grade in year y. The parameters μ and ε are regression parameters, These parameters are calculated to match the past trend as closely as possible. The past and projected future trend of copper ore grade in China is shown in Figure S4.1.

To estimate the energy requirements of copper production we applied the power regression model, as presented by Northey et al. (2014). This model defines the relationship between copper ore grade and energy inputs for mining and beneficiation processes for different type of copper extraction sites as follows:

$$E = \alpha \times G^{-\beta} \qquad (4.2)$$

where G represents the ore grade, E is the energy use for obtaining copper concentrate in MJ per kg and the parameters α and β are regression parameters calculated by Northey et al. (2014) . This equation can be used to describe the accelerating growth of energy use due to declining ore grade. The trends of copper ore grade and energy requirements are reported in the Supporting Information.

For the ore grade of imported copper concentrates, we adopted the assumptions of Kuipers et al. (2018) for the global level, indicating that energy consumption will increase on average by 0.66% per year between 2010 and 2050, as shown in Figure S4.2.

2) Energy efficiency

The International Energy Agency (IEA) pointed out that energy efficiency is one of the keys to advancing the transformation of the global energy system and addressing environmental issues associated with energy consumption (IEA, 2017). Energy efficiency improvements depend on numerous factors, including production technology, energy structure and pricing, of which technology is the most fundamental (Wei et al., 2016).

Since 1990 China has gradually developed its copper production technology, resulting in an improvement in the energy efficiency of production processes. The future energy efficiency of pyrometallurgical mining and beneficiation was estimated using historic data from 1995 to 2015 in China Nonferrous Industry Statistical Yearbook. The energy efficiency of the smelting and refinery processes improved considerably from 1995 to 2007 and slightly from 2007 to 2015 (Figure S4.3). Based on the past trends, the energy efficiency improvements of mining & beneficiation, smelting & converting and refining of pyrometallurgical copper production were assumed of 0.95%, 0.71% and 0.7% per year, respectively. Consideration of imported copper concentrate, the average energy consumption in mining and beneficiation process in other countries, we used the global level data presented by Kuipers et al. (2018), as shown in Table S4.1.

Data on the average energy requirements for Chinese hydrometallurgical production from 2006 to 2009 are also available (Hong et al., 2018; Jiang et al., 2006; Ruan et al., 2010; Song et al., 2014). Given this short period, no accurate trend in energy efficiency improvement could be established for hydrometallurgical production. It has previously been observed from other research, though, that technological innovation can increase it by 1-4% per year (Marsden, 2008; Wiechmann et al., 2010), as reported in Table S4.2. We used a conservative estimate of 1% annual improvement in energy efficiency for mining, beneficiation, leaching and electrowinning process in China. For the mining and beneficiation process of hydrometallurgical production in other countries, we worked with the same assumptions as for domestic production.

For secondary copper production we found no data on efficiency improvement. As this is allied to pyrometallurgical production, we assumed no changes in the energy efficiency in the copper scrap collection process either in China or elsewhere and the same improvement rate in the energy efficiency of the refining process in China.

3) Energy supply mix

China Renewable Energy Center (CNREC) has developed two energy supply scenarios in China Renewable Energy Outlook 2017 based on current development status (CNREC, 2017). This report focuses on specifying a feasible path for China's low-carbon transition until 2050 and the measures required to address barriers to renewable energy development in the near term. The "Stated Policies Scenario (SP)" examines the impact of current strategic energy transformation policies, while the "Below 2°C scenario (B2D)" explores the measures China needs to implement to fulfill its obligations under the Paris Agreement.

These scenarios define two roadmaps for Chinese electricity production from 2016 to 2050, as shown in Figure 4.3 and Table S4.3. The share of fossil fuels goes down, not just in a relative sense but also absolutely. Under the SP Scenario, fossil fuel use will peak in 2025, while under the B2D scenario this will already be in 2020. There is accelerated adoption of renewable energy technologies in both scenarios, but in the B2D scenario take-up is faster. In both scenarios nuclear power and wind energy come to provide the bulk of electrical power. In the B2D scenario, solar power generation increases to become the second largest source.

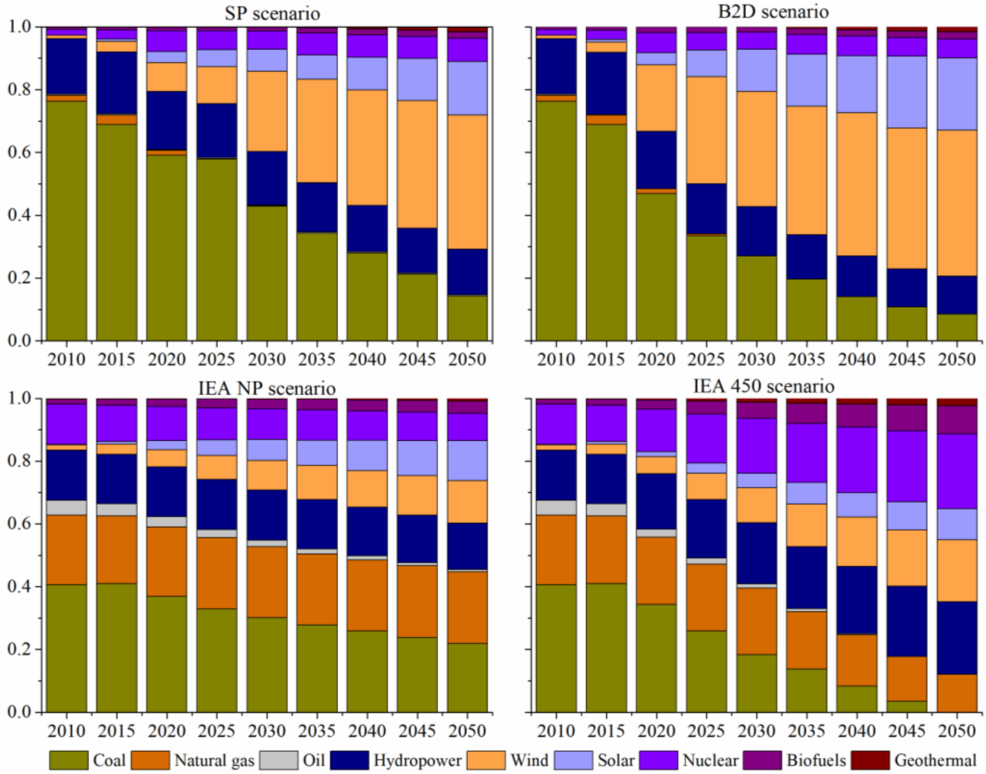


Figure 4.3 Electricity supply mix scenarios for background system of China: Stated Policies scenario (SP scenario) and Below 2 Degree scenario (B2D scenario), the world: IEA New policies scenario (IEA NP scenario) and IEA 450 scenario.

For the global electricity supply mix, we took the "New Policies (NP)" scenario and "450" scenario from IEA (2012) as corresponding to the Chinese SP and B2D scenarios, the latter being consistent with the policy goal of limiting global temperature rise to 2°C. The trends embodied in the NP and 450 scenarios are shown in Figure 4.3.

4.2.1.3 Step 3 Upscaling: Upscale analysis by multiplying environmental impacts of producing 1 kg copper by total Chinese copper supply

The aim of this step is to specify the shares of the three production routes in meeting Chinese copper demand. For future copper demand scenarios, we designed the Stated Policies scenario and Below 2°C scenario, which basically adopted the results from Dong et al. (2019). The copper demand scenarios were modeled with a dynamic material flow analysis, information on this step can be found in the Supporting information. Then the next step is to translate copper demand scenarios into copper supply scenarios. Main assumptions and data sources for copper demand and supply scenarios are summarized in Table 4.3.

Table 4.3 Main assumptions and data sources for copper demand and supply scenarios

Variables	**Stated Policies Scenario (SP)**	**Below 2°C Scenario (B2D)**	**Data source**
Ratio domestic/imported copper concentrates	The share of domestic production was 42% and 30% in 2010 and 2015, respectively. The latter figure was assumed for both pyro- and hydrometallurgical production up to 2050, under both scenarios.		China Nonferrous Industry Statistical Yearbook, UN Comtrade
Ratio domestic/imported copper scrap	The proportion of domestic copper scrap in the total copper scrap used to produce secondary copper increased from 49% to 96% as a result of import restrictions between 2010 and 2050.		Dong et al. (2019), UN Comtrade
Copper demand	SP scenario	B2D scenario	Dong et al. (2019)
Secondary copper production	In both scenarios the secondary copper will increase based on the domestic copper scrap that generated from use phases.		Supporting information
Ratio pyro/hydro primary production	Modeled based on historical data, as shown in appendix 2 in Supporting information.		the United States Geological Survey (USGS)

1) Translate copper demand scenarios into copper supply scenarios

The primary and secondary copper production was measured based on assumptions of the in-use stock, semi and finished copper production. To establish future trends in the primary production routes, we applied regression analysis to historical data and projected the results into the future. With the decline of copper ore grade and the requirement to reduce the cost of copper production, hydrometallurgy is more widely used in copper production. We modeled the shares of pyrometallurgical and hydrometallurgical routes in Chinese primary copper production to 2050 based on the historical data from

2004 to 2015 (Figure S4.4). As stated above, the Chinese copper supply derives from both domestic production and imports, broken down further into pyrometallurgical, hydrometallurgical and secondary production from 2010 to 2050. Additional details are provided in the Appendix 2 of Supporting Information.

2) Assess environmental impacts of copper supply scenarios

The SP and B2D supply scenarios were correspondingly translated from Stated Policies scenario and Below 2°C scenario of copper demand, which were then assessed as to their environmental impacts. Each scenario has two (domestic and foreign) times three (pyrometallurgical, hydrometallurgical and secondary) different production routes that correspond to a time series impact assessment per kg produced copper. In each scenario, the amounts of copper produced via these six routes were multiplied by the corresponding impacts per kg. The impacts for these six routes were then summed to yield a total.

4.2.2 Data

Data sources were summarized in Tables 4.2 and 4.3. The data on electricity production and other processes related to copper production in the background system were derived from the ecoinvent v3.4 database, which has updated a lot of data for the region ‘China’. Compared with previous research on China (Jiang et al., 2006; Ruan et al., 2010; Song et al., 2014; Wang et al., 2015), the present study included the data of imported copper and statistical data that represents China’s average situation instead of a typical enterprise for each constituent process of the three copper production routes, especially with respect to energy and resource use in foreground processes and electricity production in background processes. Detailed information can be found online at https://doi.org/10.1016/j.jclepro.2020.122825.

4.3 Results and discussion

4.3.1 Present environmental impacts per kg produced copper

Table 4.4 presents the environmental impacts of per kg copper produced in China by the three production routes in 2015. Pyrometallurgical and hydrometallurgical production are both highly energy-intensive.

Table 4.4 Environmental impacts of producing 1 kg copper by pyrometallurgical, hydrometallurgical and secondary production in 2015

Category	Pyro-	Hydro-	Secondary	Unit
CED	81.72	95.393	24.073	MJ
AD-e	0.8	0.633	0.0578	kg antimony-eq.
AD-f	85.8	98.7	27	MJ
PO	0.0131	0.00295	0.000651	kg ethylene-eq.
HT	193	634	16.3	kg 1,4-DCB-eq.
FE	132	443	9.88	kg 1,4-DCB-eq.
CC	5.88	7.37	1.59	kg CO2-eq.
AC	0.354	0.134	0.0163	kg SO2-eq.

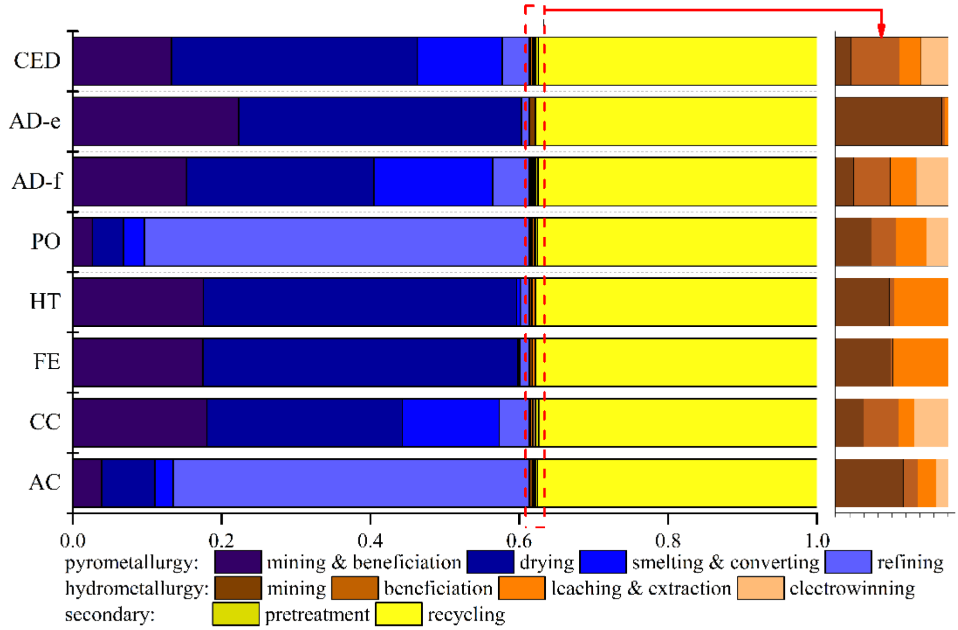


Figure 4.4 Contribution of processes from three different production routes (mix of production routes in year 2015: pyrometallurgical 61.35%, hydrometallurgical 0.80 %, secondary 37.85%) to the cradle-to-gate environmental impacts of 1 kg copper production in China. Contribution is given per impact category: acidification (AC), climate change (CC), freshwater aquatic ecotoxicity (FE), human toxicity (HT), photochemical oxidation (PO), abiotic depletion of resources-fossil fuels (AD-f), abiotic depletion of resources-elements (AD-e) and cumulative energy demand (CED); detailed data refers to the appendix 4 in SI.

Pyrometallurgical production contributes the most to acidification, photochemical oxidation (summer smog) and depletion of abiotic resources-elements, with these impacts due mainly to the mining, beneficiation and drying processes (Figure 4.4). Hydrometallurgical production contributes more to cumulative energy demand and toxicity, owing mainly to the mining, leaching and extraction processes. Both production processes primarily produce sulfuric acid, sulfur dioxide and large amounts of toxic chemicals. As one would expect, secondary production always has the lowest environmental impact for producing 1 kg copper, for the reason that it does not involve the early stages of mining, beneficiation and drying.

4.3.2 Future environmental impacts per kg produced copper in different scenarios

Developments of the per-kg environmental impacts until 2050 in the two scenarios are shown in Tables S4.8-S4.13 in the Supporting Information. We restrict ourselves to analysis of three key impact categories: climate change, human toxicity and cumulative energy demand in the main discussion. The developments are shown graphically in Figure 4.5.

The overall trends with respect to climate change, human toxicity and cumulative energy demand are similar in the two scenarios. Climate change exhibits a clear downward trend, while human toxicity shows a gradual rise in three copper production routes. The assumptions regarding to the energy transition mean the cumulative energy demand decreases in pyrometallurgical and secondary production, and a slight increase in hydrometallurgical production. In absolute terms, secondary production still scores much lower than primary production on all environmental impacts, which confirms that secondary copper production always has considerably lower impacts than primary copper production in per kg (Norgate, 2001; Song et al., 2014). An interesting finding is that the climate change impact of hydrometallurgical production declines, while energy demand increases. The assumed rate at which the energy efficiency of hydrometallurgical production improves is insufficient to offset the increase in energy use due to declining copper ore grade. The declining quality of copper ore appears to be a key factor on increasing environmental performance, with the assumed decline driving up energy use by more than the gains from the assumed improvement in energy efficiency. On the other hand, the higher energy requirements are met more by renewable sources, leading to a decline in energy-related greenhouse gas emissions. Since the extent to which these assumed developments will actually occur is fairly uncertain, we conducted a sensitivity analysis on energy efficiency improvement and declining copper ore grade.

All the environmental impacts of 1 kg copper production are lower in the B2D scenario than in the SP scenario. This result indicates that the energy transition could reduce the environmental impacts of copper production, and thus could offset the increase caused by the decline in copper ore grade.

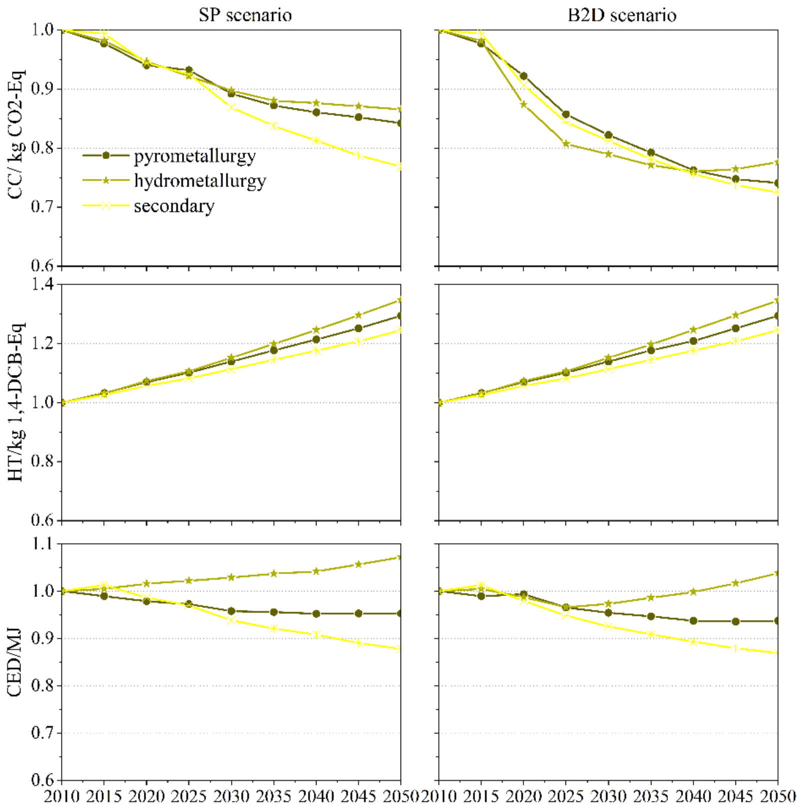


Figure 4.5 Developments in cradle-to-gate environmental impacts of 1 kg copper production in China from 2010-2050 for three copper production technologies (2010 = 1), climate change (CC), human toxicity (HT), cumulative energy demand (CED).

4.3.3 Environmental impacts of Chinese copper supply scenarios

Figure 4.6 provides an overview of Chinese copper demand and supply scenarios. Both scenarios of copper demand are expected to increase considerably, especially for Below 2°C scenario. This is due to the fact that the energy transition assumed under the Below 2°C scenario requires a larger amount of copper. For the copper supply, pyrometallurgical production is still the leading production technology under both scenarios. Secondary copper production is expected to increase as a result of increased domestic copper scrap generation and higher collection and recycling rate in China. There is no difference in domestic copper scrap production between the two scenarios, though, because copper scrap generation is a result of past developments rather than future policies. However, combined with China's restrictions on import of copper scrap as an environmental protection measure, the volume of imported foreign scrap is expected to fall. In a growing market, this will need to be replaced by either domestic scrap or primary production.

The reduction of environmental impacts per-kg appears to be counteracted by the growth in copper production for all three shown impacts categories. Overall, the impacts of copper production via all three production routes are projected to more than double from 2010 to 2050. The impacts of pyrometallurgical production are still expected to contribute most to the aggregate impacts of copper production, since it is still assumed to be the dominant mode of production.

The most striking finding is that the impacts of both scenarios are fairly similar. While the per-kg impacts are lower in the B2D scenario, copper demand is in fact lower in the Stated Policies scenario (SP supply scenario). The levelling off of the trend in both scenarios can be attributed to the growth of the share of secondary production.

Multiplying the environmental impacts of production of 1 kg copper by the copper supply scenarios, we obtained insight into the potential environmental impacts of copper supply scenarios from 2010 to 2050. Figure 4.7 presents the results for climate change, cumulative energy demand and human toxicity of the two copper supply scenarios for China. The complete results for all the impacts in each supply scenario are reported in Tables S4.14-S4.19.

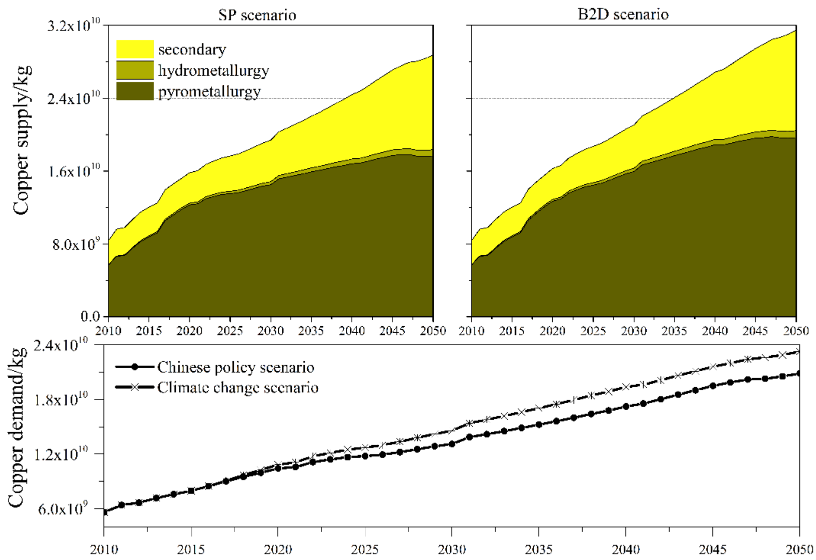


Figure 4.6 Copper demand and supply scenarios of China from 2010 to 2050

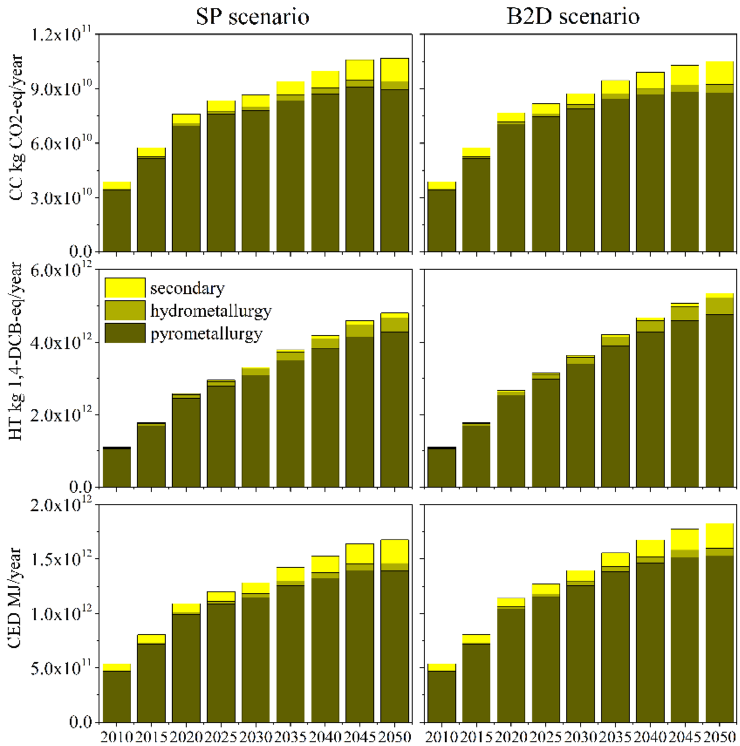


Figure 4.7 Environmental impacts of copper supply scenario of China per year.

4.3.4 Uncertainties analysis

In this study the LCSA method was used to estimate the environmental impacts of Chinese copper production, encompassing domestic copper production and imported copper, and taking into account three variables: energy efficiency improvement, energy transition and copper grade decline. The results were analyzed from aspects on copper production technology and scenarios analysis. There are several uncertainties in the results, however, owing to limited data or the assumptions made in the scenarios.

First, improved production process data need to be obtained, in particular on the energy efficiency of hydrometallurgical and secondary production processes. The results of LCA are often affected by changes in inventory data inputs, making these a major source of uncertainty in LCA. To examine the effect of variables on the environmental impacts of copper production, sensitivity analysis was performed using varying assumptions on energy efficiency improvement and ore grade decline. As Figure S4.5 shows, if the energy efficiency of the three copper production routes remains constant from 2015 onwards, the environmental impact is about 1-6% higher than the improved energy efficiency model in both scenarios, though the rate of change is lower in the B2D scenario. Copper ore grade also has an important influence on the results, as shown in Figure S4.6 a constant grade for pyrometallurgical and hydrometallurgical production will reduce impacts by around 1-15% in both scenarios. Because of the energy transition, the impacts are far lower in the B2D scenario, furthermore.

Second, we made several assumptions about the potential development of copper demand and copper supply. There is no doubt that the results stemming from these assumptions have a high degree of uncertainty. It should be borne in mind, though, that scenario analysis is not concerned with predicting the future, but with exploring the implications of the continuation of present developments and of the present policies from the Chinese government. In the scenario analysis conducted in this study, this means to examine alternative future Chinese copper supply scenarios.

Table 4.5 Studies on environmental impacts of copper production at different scale levels (Functional unit: 1 kg copper)

Country or region	**Production process**	**Impact category**	**Value**	**Source**
Global	Metal production and refining	CC	4.3~8.9	Norgate (2001)
		AC	0.03	
		CED	51	
	Copper ore processing	CC	4.5	Rötzer and Schmidt (2020)
	Primary production	CC	2.8	Nuss and Eckelman (2014)
		CC	6.44	Van der Voet et al. (2018)
		CED	106	
		AC	0.08	Kuipers et al. (2018)
		HT	184	
	Secondary production	CC	1.58	Van der Voet et al. (2018)
		CED	22.4	
		AC	0.02	Kuipers et al. (2018)
		HT	6.77	
Australia	Mining and smelting	CC	2.5~8.5	Memary et al. (2012)
		AC	0.05~0.5	
	In-situ leaching mining	CC	4.78	Haque and Norgate (2014)
Canada	Primary production	CC	2.3	Northey et al. (2013)
Chile	Primary production	CC	1.1~3.9	
South Africa	Primary production	CC	8.5	
USA	Primary production	CC	4.44	
Japan	Primary production	CC	2.5	Adachi and Mogi (2007)
India	Secondary production	CC	1.5	Chaturvedi et al. (2012)
Poland	Primary production	CC	5.44~7.65	Kulczycka et al. (2016)
		AC	0.03~0.05	
China	Primary production, specific copper production site	CC	1.91	Hong et al. (2018)
		HT	0.09	
		AC	0.01	
	Secondary production	CC	0.69	
		HT	0.22	
		AC	0.002	
	Primary production	CC	3.42	Chen, J. et al. (2019)
		AC	0.02	

Country or region	Production process	Impact category	Value	Source
		HT	1.79	
	Secondary production	CC	0.32	
		AC	0.001	
		HT	0.22	
	Primary, pyrometallurgical production, 2015	CC	5.88	This study
		AC	0.35	
		CED	81.72	
	Primary, hydrometallurgical production, 2015	CC	7.37	
		AC	0.13	
		CED	95.39	
	Secondary production	CC	1.59	
		AC	0.016	
		CED	24.07	

4.3.5 Critical analysis of environmental impacts of copper production: comparison with other countries or regions

To put our results into a broader perspective, we compared them with results of studies on the environmental impacts of copper production at global and national level: main copper producing countries including Chile, the United States, Australia, Canada, South Africa, India and Poland (Table 4.5). In view of the scope of the selection of studies, a comparison is possible on the impact categories of climate change, acidification potential and energy consumption of copper production. The ranges of impact value are quite large – a factor 2 for energy demand, a factor 8 for climate change and orders of magnitude for acidification, both for primary and secondary production. Results for China are not exceptional, they fall within the range of the studies included in the comparison. The reasons for these differences are not apparent. They may be due to methodological choices, calculation procedures or data uncertainty. They may also be real-world reasons related to the use of different production technologies, different ore qualities or local energy mixes.

4.3.6 Discussion and identifying options to improve the environmental performance of copper production

While these uncertainties have some impact on the accuracy of the results, they are unlikely to change the overall trend of increased environmental impacts of aggregate copper supply as copper demand continues to grow. This implies a need to further reduce the environmental impacts of copper production, to which end several options can be identified.

Reducing copper demand is the first and most obvious option. Several studies indicate that copper demand is closely related to GDP (Schipper et al., 2018; Soulier et al., 2018b). Once a certain GDP threshold is reached, copper demand may become saturated. However, this will not be easy to realize in 2050 in a country like China, which is still in the midst of development. During this period, China will be shifting from a fossil-based to a renewables-based energy system. In particular, electrification – all the more important in view of the energy transition – requires massive amounts of copper. It is also evident from our research (Below 2°C scenario of copper demand) that the use of new energy will lead to increased copper demand. As a result, Chinese copper demand is expected to continue to grow, a trend that cannot readily be halted before 2050.

Substitution of copper by other materials is a second option to consider (Batker and Schmidt, 2015). Substitution of copper by another material is an option to reduce demand, but poses challenges as well. It is by no means straightforward to find suitable alternatives that have the same functionality and can be produced in equally large quantities. Replacing copper by other materials is therefore not always feasible, especially for electricity infrastructure (Graedel et al., 2015). Primary aluminum is considered to have the greatest potential for replacing copper in energy infrastructure. Unfortunately aluminum has even higher climate change and energy demand impacts per kg than copper (Li and Guan, 2009; Van der Voet et al., 2018). In view of lower environmental impacts, secondary aluminum may be a more effective substitute for copper. Depending on required performance and application, other products may also be eligible for partially replacing copper, such as stainless steel, zinc and plastics. However, further study is still needed to assess the long-term use and impacts of these substitutes, to develop a broader understanding of more environmentally compatible copper alternatives.

The energy transition deserves greater attention. Hard coal use and production are the principal sources of GHG emissions in pyrometallurgical production in 2015 and 2050 of SP scenario, as shown in Figure 4.8. With the energy transition, the climate change impact for all processes is expected to decline, especially for hard coal. Extending its policy of encouraging use of renewables, China could initiate a drive to use renewable energy instead of hard coal for copper production, as implemented at the Zaldivar copper mine in Chile (Antofagasta et al., 2018). This is the first copper mine in the world to use 100% renewables, such as wind and solar. It is expected to lead to a reduction in greenhouse gas emissions of about 350,000 tonnes per year. A further bolstering of the energy transition in the foreground and background systems is therefore a promising option for reducing the overall environmental impacts of copper production.

Cleaning up copper production processes and improving energy efficiency are further options for reducing impacts**.** To comprehensively reduce or eliminate environmental pollution, clean production systems should be applied across the board to the entire copper production chain. An extremely important first step is therefore to identify the origins of impacts occurring in the various links in the chain. This option has close connection to the energy transition, for example when traditional coal is replaced by low-sulfur clean coal or other clean fuels (Figure 4.8). Another approach is by way of technological innovation and improved plant and equipment. While technological innovation is crucial for improving energy efficiency, additionally presented in sensitivity analysis, it is a challenging issue. Previous studies have reported a boom in technological innovation in copper production in the mid-20th century. Prior to that, the technology had remained unchanged for 65 years (Council, 2002; Radetzki, 2009). Truly substantial efficiency improvements are not to be anticipated unless a completely new production process is developed, a perspective that is unlikely to have any impact before 2050.

While energy efficiency improvements and energy transition have an important role to play in reducing most of the impacts of copper production, these make no obvious contribution to reduce human toxicity. The main impact of declining ore grade is on the direct environment of copper mine (Table S4.7). There will inevitably be increased production of tailings, which can leach out into the runoff after mixing with rainwater and affect human health. To reduce this impact, further comprehensive recycling of tailings is required.

Increasing the share of secondary copper production is probably the most important option of all. As this study has confirmed, secondary copper production has lower per-kg environmental impacts. This option hinges on the amount of available copper scrap in China. However, given that domestically generated scrap will only be able to cover just over 50% of demand in 2050, the scope for this option appears to be fairly limited in China. Currently, the copper recycling rates in China are somewhat lower than in Europe (Soulier et al., 2018a; Soulier et al., 2018b). Although several studies argue for promoting increased copper recovery (Brahmst, 2006; Giurco and Petrie, 2007; McMillan et al., 2012), this is difficult to implement since it involves technical improvements to a range of processes, including product design, disassembly and end-of-life collection. An appropriate recycling infrastructure for end-of-life management of complex products is presently lacking as well. Therefore, to protect the environment of recycling facilities and achieve high copper recovery rates, the waste management sector needs to reorganize. In addition, China has taken steps to restrict the import of copper scrap, limiting the potential for recycling even further. Against this background, improving the efficiency of copper scrap collection and processing as well as reconsideration of import restrictions stand out as potentially very effective options.

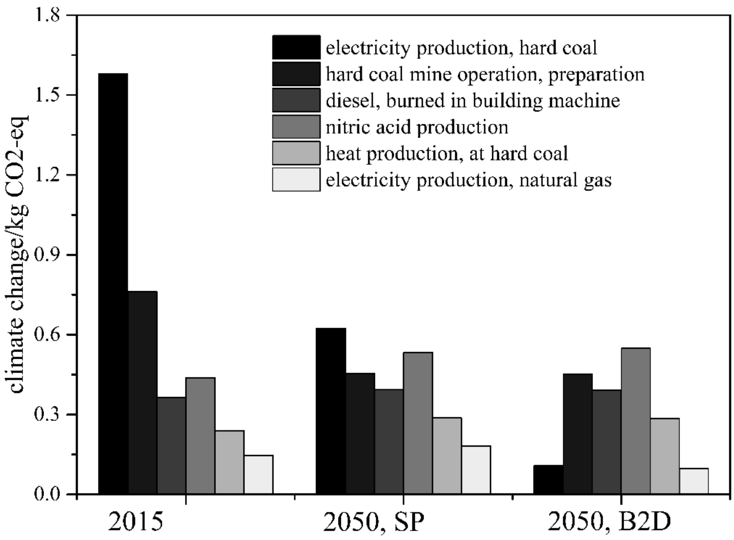


Figure 4.8 Main sources (original processes) of GHG emissions in pyrometallurgical production in SP and B2D scenarios in 2015 and 2050

4.4 Conclusions

The environmental impacts of 1 kg of copper and total supply of copper produced by pyrometallurgical, hydrometallurgical and secondary production from 2010 to 2050 were assessed using LCSA. Moreover, we identified the potential options for improving the environmental performance of Chinese copper production and consumption. This study provides strong support for establishing how to reduce the environmental impacts of copper production, permitting a better examination of the challenges confronted by China's copper industry and thus enabling better identification of solutions to these challenges.

The results indicate that the environmental impacts of pyrometallurgical copper production are expected to increase more than twofold during this period, which means this will remain the largest contributor to the environmental footprint. Secondary copper production is the most environmentally friendly production route, increasing the share of secondary copper production is the most relevant option for reducing the environmental impacts of copper production. To this end, China may focus on improving the classification of waste copper products and recycling infrastructure for end-of-life management.

Hard coal production and use are crucial contributors to climate change in the context of copper production. Cleaning up copper production processes and improving energy efficiency would also help reduce environmental impacts. The energy transition has the potential to significantly reduce the environmental impacts of copper production, but it also will increase copper demand considerably.

Energy efficiency improvements and energy transition do not visibly contribute to reduce human toxicity. With the declining ore grade, further comprehensive production of copper mine and recycling of tailings is required.

Further research into the possibilities for a circular economy, by exploring options to reduce demand via repair, reuse, refurbishing, remanufacturing and recycling, is recommended. It will be important information to support the transition towards a sustainable resource use, in China and in the world.

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.

Dong, D., Tukker, A., Steubing, B., van Oers, L., Rechberger, H., Aguilar, Hernandez G., Li, H., van der Voet, E., Towards "Zero waste" management of copper in China: dematerialization and environmental impact minimization. Environmental Science & Technology, under review.

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).

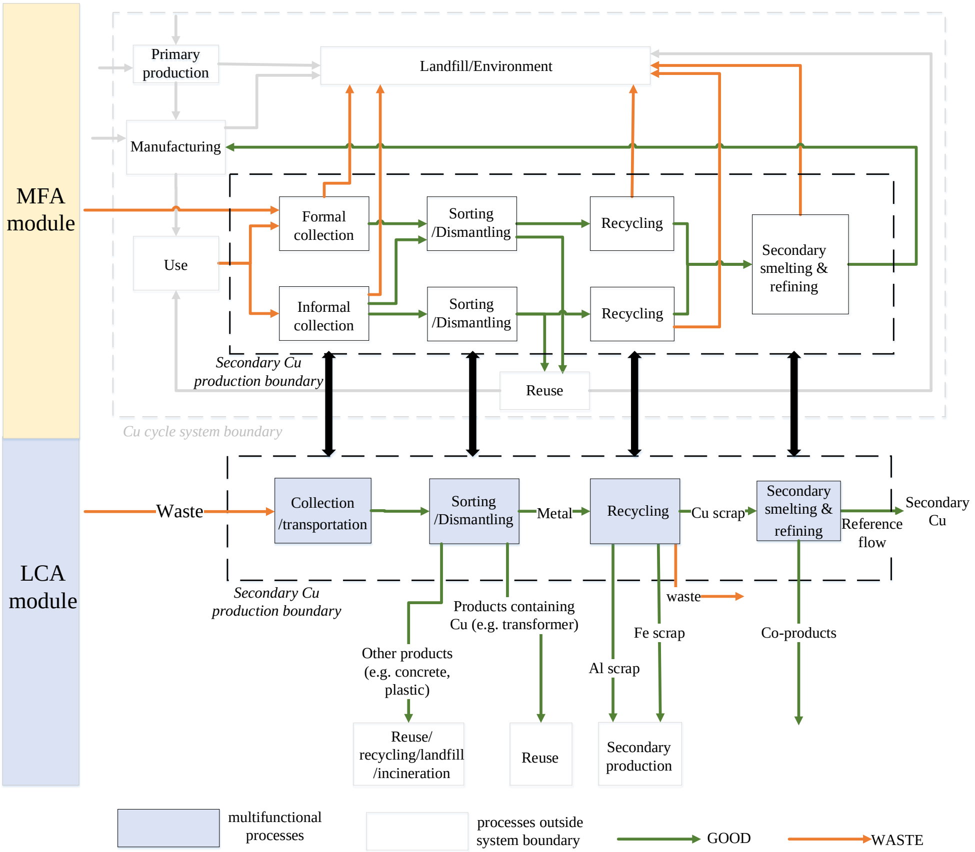


Figure 5.1 Copper cycle system boundary and definition for China: schematic representation of the MFA and LCA combination on the secondary copper production. The MFA processes (collection, sorting & dismantling, recycling, secondary smelting & refining) are represented by the LCA processes. Note: The 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.

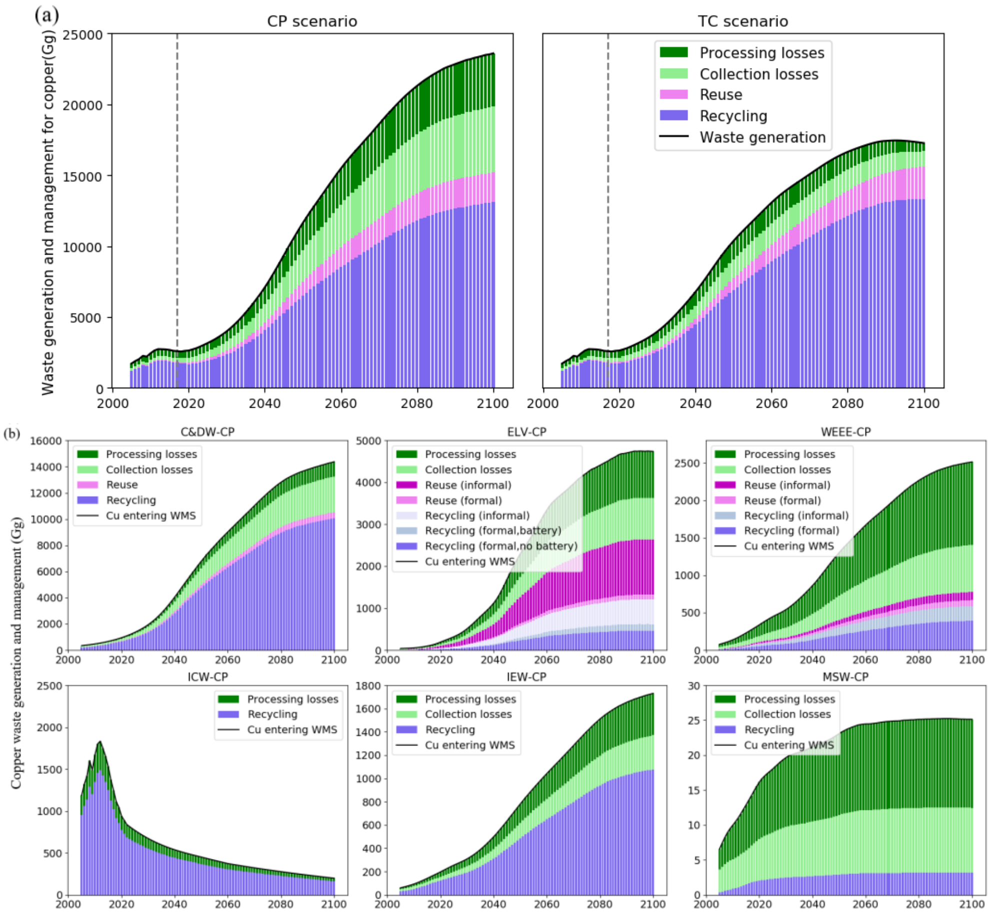
A detailed data and assumptions are available in supporting files (https://doi.org/10.6084/m9.figshare.14593737.v1).

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.



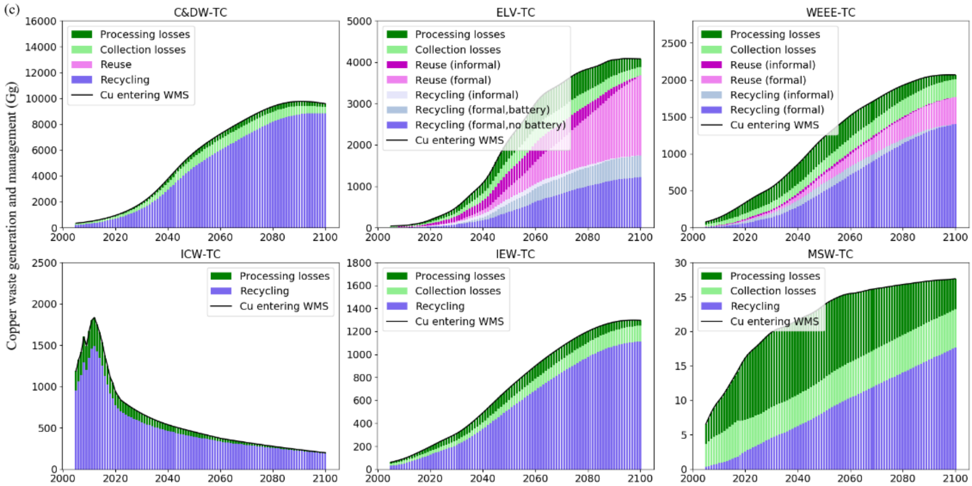
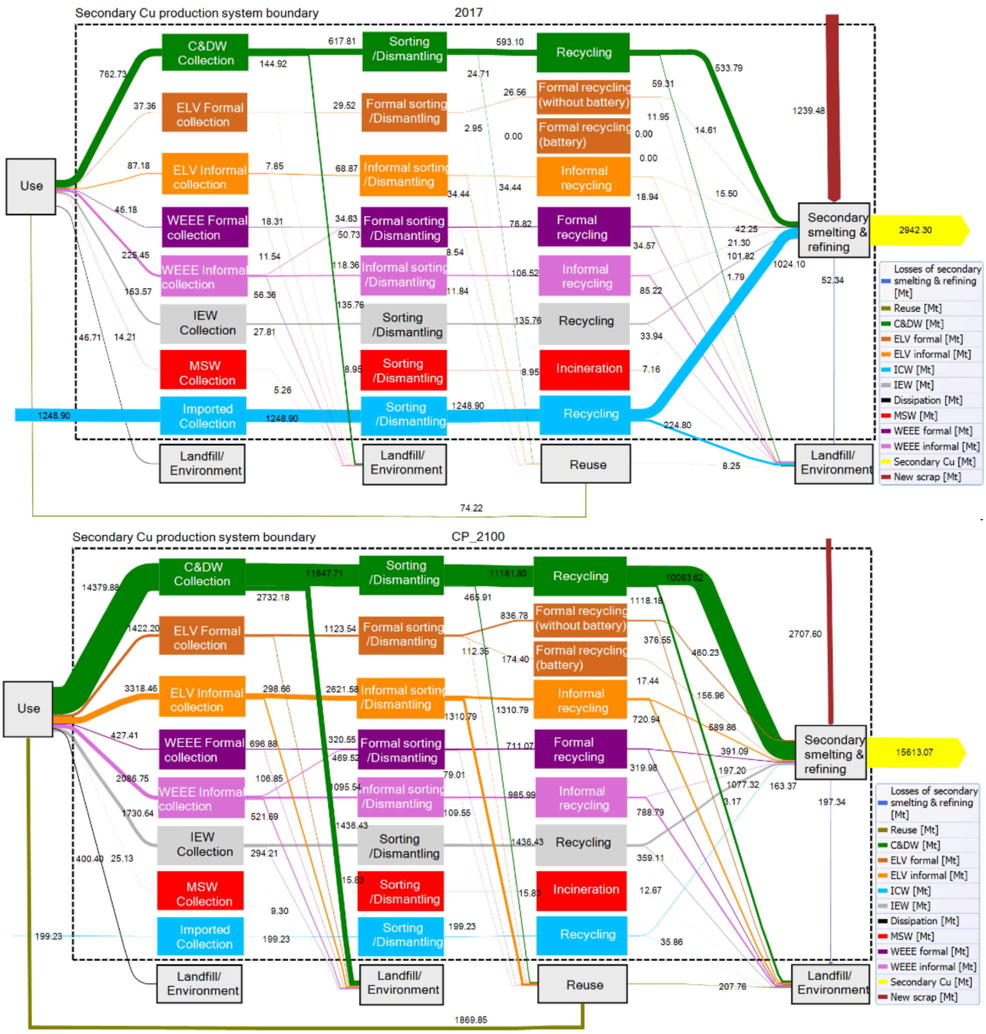


Figure 5.2 Copper waste generation and management from 2005 to 2100 in the CP and TC scenarios in China: (a) total copper waste generation and its destination, the vertical black dashed line marks the boundary between historical data and future scenarios, (b) by waste source in the CP scenario, (c) by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



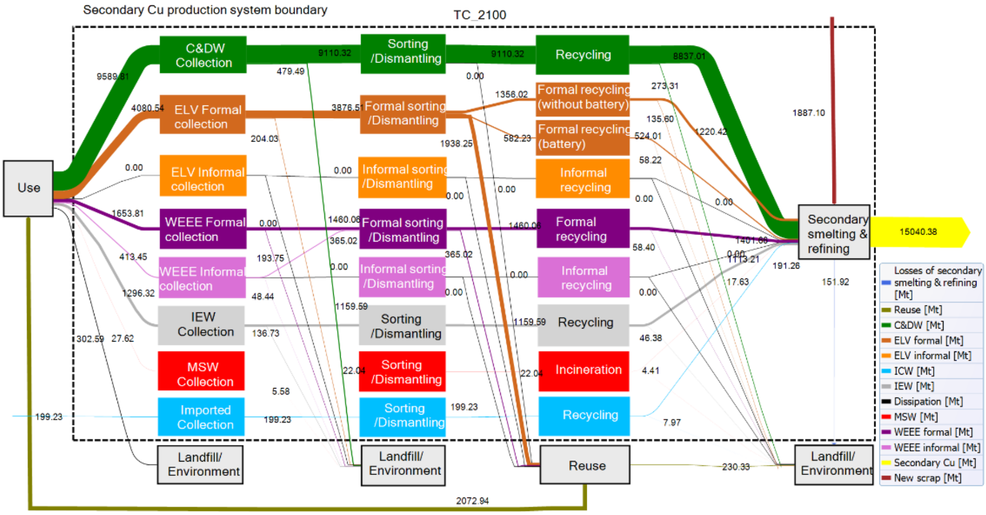


Figure 5.3 Sankey diagram of copper waste management and secondary copper production in China in 2017 and 2100 in the CP and TC scenarios

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).

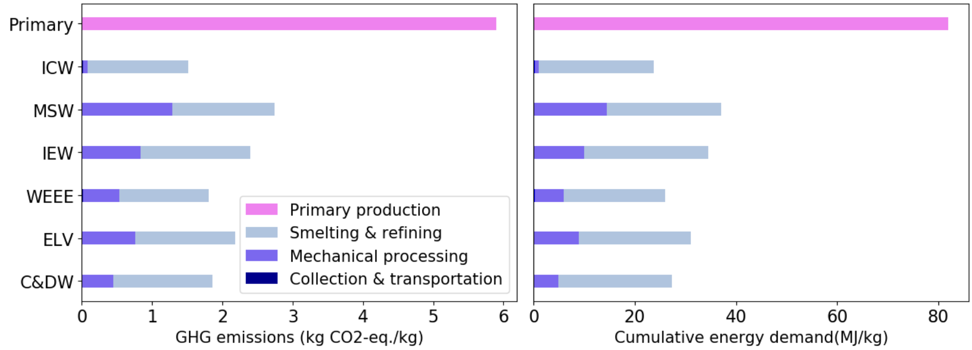


Figure 5.4 GHG emissions and cumulative energy demand of 1 kg secondary copper produced from different waste types in China in 2017, broken down into constituent processes, and comparison with production of 1 kg primary copper in China in 2017. The primary copper data is derived from Dong et al.(2020b)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

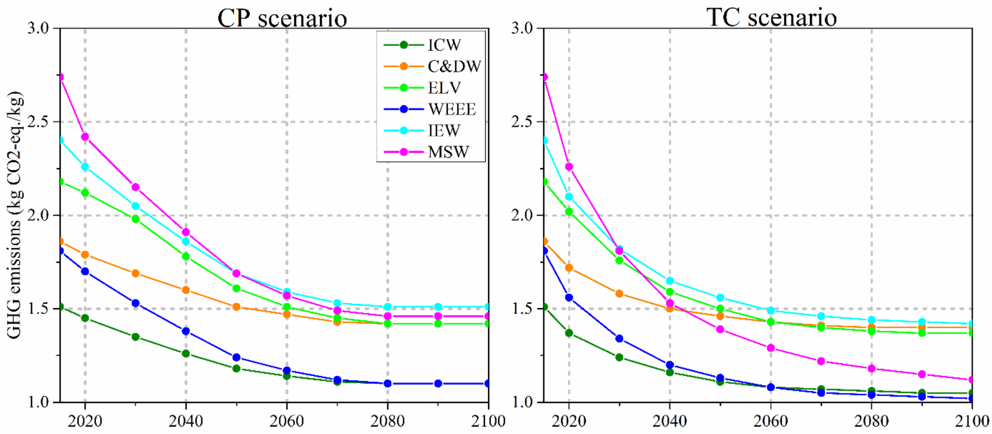


Figure 5.5 GHG emissions of production of 1 kg secondary copper from different waste types in the CP and TC scenarios (Figure S5.5 shows the results for CED).

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.

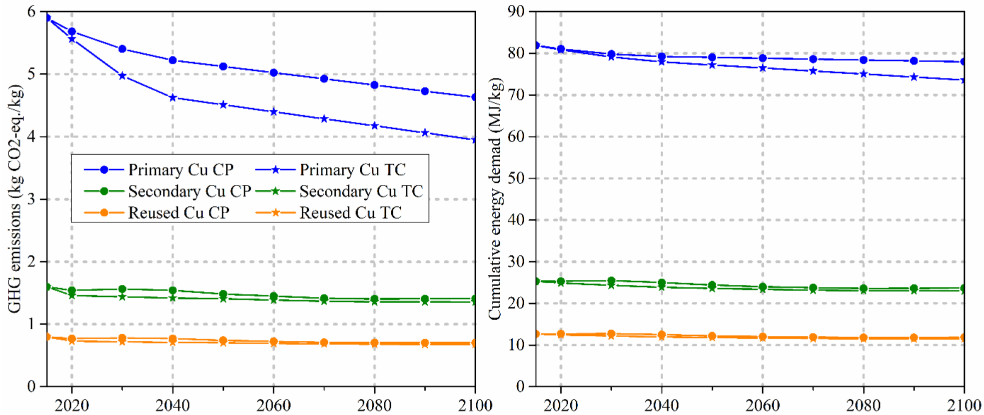


Figure 5.6 GHG emissions and cumulative energy demand of 1 kg primary and secondary copper production and comparison with copper reuse in CP and TC scenarios. The primary copper is derived from Dong et al. (2020b) and represents an average value based on pyrometallurgical and hydrometallurgical production 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.

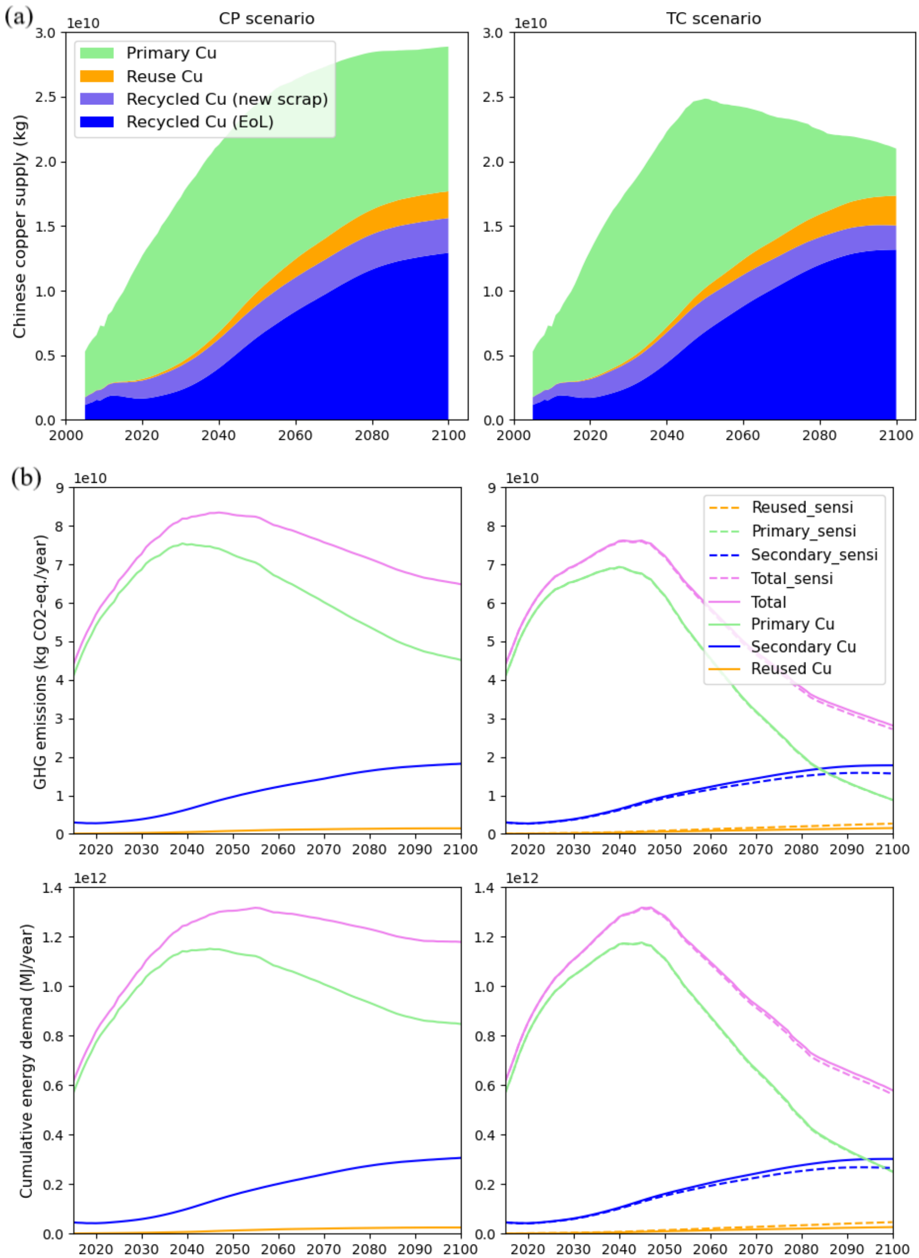


Figure 5.7 (a) Sources of Chinese copper supply in CP and TC scenarios, (b) GHG emissions and CED of primary, secondary and reused copper production. The dash lines in Figure (b) indicate the sensitivity analysis on reused copper that modeled based on the assumptions of increasing 60% of reuse copper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

Chapter 6

Conclusions and General Discussion

The aim of this thesis was to explore how the copper cycle in China could develop into a sustainable and circular system. The main focus was on quantifying in-use stocks, the demand for and supply of copper, and associated environmental impacts in several scenarios for the future. The results of this thesis can be used to identify possible measures and policy options in response to future environmental and supply challenges.

This thesis starts in Chapter 2 by investigating a business-as-usual scenario (Chinese Policy or CP scenario) of the in-use stocks, and the demand for and waste generation of copper in China using dynamic Material Flow Analysis. To explore how to transition to a more sustainable development of copper, as well as the effect of the "Green Fence" policy on copper supply, a circular economy scenario (CE scenario) has been designed to compare with the CP scenario in Chapter 3. This CE scenario includes several strategies such as extending lifetimes of copper products and increasing the recycling rate of EoL products. In Chapter 4, a Prospective Life Cycle Assessment was used to analyze the environmental performance of future copper production, where the effects of the energy transition on both copper use and the environmental impacts of copper production were considered. From these combined efforts, potential options for improving the environmental performance of Chinese copper production were identified. As a final step in Chapter 5, a combination of MFA and LCA was used to identify an optimized copper waste management system, where the utilization of copper waste is maximized with minimum environmental impacts. Finally, the influence of such an optimized system on the copper cycle in China was assessed. This concluding chapter will first answer the research questions proposed in the introduction, and then discussion and make recommendations for future research.

6.1 Answer to research questions

6.1.1 Question 1- How are copper demand, in-use stocks and waste generation expected to develop under the current Chinese policies related to general economic development, the energy transition and ambitions with regard to the circular economy?

Chapter 2 reports a dynamic MFA involving a bottom-up approach to quantify in-use stocks, as well as the demand for and waste generation of copper under the current Chinese policies. The results show that in-use stocks for copper in China are expected to increase significantly and reach about 400 million tons in 2050, however, with different growth rates for different copper applications. The copper stocks in infrastructure and transportation will not yet have reached saturation by 2050, while the copper stocks in buildings will likely be stabilized. The copper stock in buildings was the largest at present. In 2050, that place is expected to be taken by infrastructure, mainly the electricity grid. Buildings will then be second in size. By sub-categories, the copper stocks in the electricity infrastructure, new energy vehicles, and home appliances (including air conditioners and refrigerators) are likely to increase considerably up to 2050. As a result, the copper stock per capita is expected to increase about 10 times in the 2005 - 2050 period.

The total copper demand as well as the demand per capita also increase significantly during this period, especially in infrastructure, transportation and buildings. In 2050, under the scenario specifications made, demand will have increased by a factor of 3 compared to 2015. The main end-use sector is infrastructure, accounting for around 50% of the total copper demand by 2050, followed by transportation, buildings, and different types of durable goods.

With the increasing consumption of diverse copper-containing products, in-use stocks for copper have become a large reservoir that can be considered as an urban mine, a source of materials for the future. Domestic waste generation has been increasing in recent years, however, and most of China's copper products have not yet entered the waste stage at present. Given the 30-year overall average lifetime of copper products, the amount of copper that is now entering the waste stage is still very small, and the growth of EoL copper recycling therefore very slow. The findings of this thesis suggest that only a small part of the copper demand in 2050 under the current Chinese policies can be met through the supply of secondary copper.

6.1.2 Question 2- How could China meet its future copper demand in the context of moving towards a circular economy, and how may this be affected by the import restrictions of copper scrap?

In order to understand how to develop the copper cycle sustainably in China, a circular economy scenario was defined that assumed longer lifetimes of copper-containing products and a higher EoL copper recycling rate on top of the business-as-usual assumptions. This scenario has been used to explore how China's copper demand can be met up to 2100. It has also been used to assess the impact of the Chinese government proposals to restrict the import of copper scrap on the future Chinese copper supply. The results are presented in Chapter 3.

The findings made clear that the Chinese copper cycle could reduce its primary input significantly under the circular economy scenario. Extending the lifetimes of copper-containing products may lead to a decrease in the overall copper demand in the second half of the century, while maintaining the functionality of the in-use stocks. Regarding copper supply, the imported copper scrap was the principal source for secondary copper production in China before implementing the "Green Fence" policy. The scrap import restriction or ban will reduce secondary copper production significantly. Domestic copper scrap will become the main source of secondary production in the future; however, even with an increasing availability of domestic scrap and an assumed high copper recycling rate, the share of secondary copper supply can only increase to 60% in 2100.

There will be a substantial gap between Chinese copper demand and the amount of scrap available domestically throughout the 21st century. In the future, this gap needs to be closed either by domestic mining of primary copper, or through imports of concentrates and refined copper. Meanwhile, China will still have to put major emphasis on its recycling industry to reach the high recycling rates assumed in the CE scenario. If China wants to further reduce primary production, an important suggestion is that they should continue to import copper scrap. Another reason for this proposal is that importing high-quality copper scrap instead of copper ore or refined copper is an environmentally beneficial option. In combination with the establishment of a state-of-the-art, efficient and environmentally friendly recycling industry, this could be an opportunity for China to transition to a more circular economy with regard to copper.

6.1.3 Question 3- What are the environmental benefits and drawbacks related to present and future copper production in China, and how could the environmental performance be improved in the future?

Copper production consumes a lot of energy and releases harmful emissions along its life cycle. To understand the environmental benefits and drawbacks related to future copper production in China, a prospective LCA approach was used and up-scaled with several prospective changes related to copper production.

In Chapter 4, the future environmental impacts of pyrometallurgical, hydrometallurgical and secondary copper production in China were assessed with the prospective changes of declining copper ore grade, energy efficiency improvement of production processes and the transition of the electricity supply towards a renewable system (named as the Below 2 degree, B2D scenario), and compared with the business-as-usual scenario. The main finding was that the environmental impacts of the production of 1 kg copper in the B2D scenario are considerably lower than that of the business-as-usual scenario. However, the energy transition by shifting from fossil fuels to renewable energy in the B2D scenario will increase copper demand by more than 10% compared to the business-as-usual scenario. As a result, the total environmental impacts of copper production in the B2D scenario are no lower than that of the business-as-usual scenario.

The results also support the idea that the environmental impacts of production of 1 kg secondary copper are much lower than those of primary copper. Pyrometallurgical and hydrometallurgical production are both highly energy-intensive. Pyrometallurgical copper production is the largest contributor to the environmental impacts of copper production, and this is not expected to change until 2050.

To further explore options to reduce the environmental impacts of copper production, China's copper waste management system was assessed in some detail. The aim of Chapter 5 was to optimize this waste management system by applying more circular economy and “Zero waste” strategies. Optimization in this context means: to produce as much secondary copper as possible, with as little environmental impacts as possible. The waste streams were divided into six types, including C&DW, ELV, WEEE, MSW, IEW and ICW. The circular economy strategies consist of waste prevention, reuse (including repair, remanufacturing or refurbishment), and the recycling of copper-containing products, as well as the transition from informal recycling to formal recycling.

Combining with the environmental impacts of primary copper production in Chapter 4, the main conclusion was that under the present Chinese policies, the reuse and recycling of copper-containing products will lead to somewhat lower GHG emissions and lower energy demand for total copper production. Maximizing such circular economy strategies may lead to a further reduction, but can also be counter-productive if applied too stringently. GHG emissions related to secondary copper production may become greater than those of primary copper production despite lower per kg GHG emissions of secondary production, depending on the copper content of those waste streams and the need for transport. The environmental impacts of secondary copper produced from the different types of waste (C&DW, ELV, WEEE, IEW, MSW, ICW) are different, attributable mostly due to the differences in impacts by mechanical processing and collection & transportation.

The findings and discussion on the options to improve the environmental performance of copper production in Chapters 4 and 5 suggest that increasing the share of secondary copper production is the most environmentally friendly option. To this end, the copper waste management system needs be improved, which requires actions across the full product lifecycle, including waste prevention and circular economy strategies to the EoL products as well as early-stage product design. An optimized copper waste management system is expected to reduce pressures associated not only with copper ore extraction and refinery, but also with copper waste disposal. Such an expansion from waste management to life cycle management may be the best way to reduce the environmental impacts of copper production in China.

6.1.4 Question 4- What is the potential to close the copper cycle in China?

Since increasing the supply of secondary copper is a crucial way to alleviate both resource supply constraints and environmental pressure, it is important to understand the potential to close the copper cycle in China. There are many aspects that have to be considered in this context. The chapters in this thesis all provide pieces of that puzzle.

The overall conclusion is that neither at present nor in the future will copper scrap be a sufficient source of copper. Secondary copper production will not be able to meet copper demand under the current Chinese policies in China, even assuming high recycling rates. A large amount of primary copper supply will be still needed. However, with several more circular economy strategies including extending the lifetimes of copper-containing products, the increasing copper recycling rate, increasing copper reuse and the transition from informal to formal recycling, the share of secondary copper supply could increase to 80% in 2100. Accordingly, this could mitigate some of the pressures associated with copper ore extracting and processing, and achieve the dematerialization of the copper cycle in China.

However, to completely close the Chinese copper cycle is challenging. Several barriers, such as the establishment of adequate recycling technologies as well as recycling infrastructure, need to be overcome. Moreover, there are constraints in the availability of secondary resources, and environmental trade-offs may be expected on the path towards a circular economy at global level. Finally yet importantly, stock dynamics imply that secondary supply can catch up only if demand levels off. This could occur with more circular measures in the CE scenario in 2100 since the in-use stocks for copper will likely stabilize at that time; meanwhile, the copper demand is expected to decline.

6.1.5 Overall research question: how can the copper cycle in China be transformed into a circular and sustainable economy?

By and large, this thesis revealed four major prerequisites that need to be fulfilled to establish a circular and sustainable economy for copper cycle in China.

(1) circular design of copper-containing products,

(2) increased reuse or recycling, particularly formal reuse or recycling,

(3) a transition of the energy system towards renewable sources,

(4) a saturation of the in-use stocks.

Early-stage design is key to the principles of circular economy, aiming at keeping the products in use as long as possible. By designing products that use less raw materials, and products that can be easily shared, repaired, remanufactured, refurbished or recycled, the lifetimes of the copper-containing products can be lengthened, resulting in a lower copper demand.

Reuse can reduce the volumes of copper waste requiring treatment and often is more environmentally friendly since no new materials have to be produced. From that point of view, policies to support the organization of reuse (second-hand markets, remanufacturing plant) centers and networks, are an essential part of a successful implementation of the circular economy. Recycling however remains an essential part of a circular economy as all products at some point will end up as waste. To increasing the copper recycling rate, enhancing the collection rate is probably the most important strategy. Therefore, a waste management system needs to be constructed as a socio-technical system, including people's decision-making on how they handle waste, which probably requires a transformation of our consumer culture.

Renewable energy systems play an essential role in shifting towards a sustainable society in general and copper cycle in particular. The emissions related to copper production are for a large part related to the energy use of those processes. A fossil free energy system therefore reduces emissions of copper production considerably.

The analysis of copper stocks and flows in this thesis shows that the Chinese copper cycle is still far from closed. Even when copper reuse and recycling are assumed to improve significantly, closing copper loops cannot be achieved. This is mostly due to the expectation that copper stocks still grow by 2050. As a result, the demand will still be larger than the supply of secondary copper. An essential requirement for closing cycles is therefore a saturation of the in-use stocks. Secondary supply could then catch up with demand. Primary copper extraction could then be reduced, leading to lower environmental impacts of copper production and a more sustainable copper cycle in China.

6.2 Discussion and recommendations

In this thesis, the in-use stocks, demand and supply of copper in China and the associated environmental impacts are explored using dynamic MFA and prospective LCA. This provides insights that are relevant for identifying options to move toward a sustainable and circular economy of the Chinese copper cycle. This systems analysis is valuable, but has a rather limited scope. Many relevant aspects related to economy, policy and society have not been taken into account. Below, a critical step back is taken to identify limitations and potential bottlenecks that need to be the focus of further research in this field.

Uncertainty is a common challenge to any long-term scenario research. The forward-looking study of copper cycle in terms of resource use and environmental performance in this thesis does not aim for predicting precise future, but for explorations of potential patterns and trends that assumed based on multiple parameters. Such an analysis may cause a degree of uncertainty due to data availability and quality as well as the methods used for research, however, it could provide guidance for long-term critical assessment.

High-quality data from reliable sources are essential for convincing results. For the Chinese copper cycle, the main uncertainty of data for copper production concerns the statistic accuracy of data that the copper companies and regional government provide. The data regarding to primary copper production and energy use is well-established on the whole, while the data about copper recycling and reuse still needs to be greatly improved. In other words, the copper cycle is an illustration of the need to improve and develop more reliable and robust data on waste streams in China. In addition, a bottom-up dynamic MFA stock model was used in this thesis. In this approach, in-use stocks of products are the starting point. Each of these products has a life span, and for each of these products different driving forces determine its behavior over time. Inflows (demand) and outflows (discarded products) are calculated from stock dynamics. These stocks and flows of products are translated into stocks and flows of copper using information of the copper content of each product. This is a data intensive exercise, since these copper-containing products may have different growth trends, requiring specific stocks models. Data collection needs to be done for each separate product.

This is not only an elaborate process, but also fraught with difficulty: missing data, uncertain data, indirectly estimated data and suchlike. The fact that in MFA mass balances have to be closed can help to fill in data gaps and force to assess a best fit if data sets are used that lead to an unbalanced total. The result of this process is often highly uncertain, for instance due to estimate of in-use lifetimes of products, that may deviate in practice significantly from estimates made now. An alternative would be a top-down approach of MFA that projects material flows based on some drivers such as population and GDP (Schiller et al., 2017; Schipper et al., 2018). However, this approach does not capture essential system characteristics related to stock dynamics and especially stock saturation that net additions to stocks stop growing. Material demand estimated based on the two approaches can be expected to be similar in a short period of growth, such as the comparison results by Schipper et al. (2018). On the longer run, we expect stock saturation, which is an essential step in the realization of a circular economy. This process is completely missed by a top-down approach. Despite difficulties and uncertainties, a preference for the bottom-up approach can be indicated.

The copper content of products is an important factor in determining copper stocks. Copper contents of products and appliances have changed in the past and will change in the future. However, in our scenarios, we have kept material contents constant except for a rough estimation of its changes in buildings. A sensitivity analysis assessing the influence of changes in copper content shows the effect of its change on total copper demand. It merely seems to lead to rather straightforward relatively modest changes. In some cases, copper is substituted altogether by different materials, leading to a reduced demand. Around 0.24 million metric tons of copper have been substituted with different other materials globally in 2016, in power cables, winding wires, copper tubes, and telecom cables (ICA, 2016). This indicates that such substitution does indeed take place, with consequences for the amount of present copper stocks and future available copper waste. Aluminum is considered to have the greatest potential for replacing copper in energy infrastructure, other materials such as stainless steel, zinc and PVC (e.g. building water supply system) may be used as substitutes in other products (Månberger and Stenqvist, 2018; Xiong et al., 2020). On the other hand, copper is substituting other materials: for example, silver is substituted with copper in photovoltaic power plants (García-Olivares, 2015). Another relevant effect of copper substitution is the environmental impacts related to the production of the substitutes. Several studies have investigated the environmental impacts of aluminum, steel and zinc production, which shows that primary aluminum has higher energy requirements and corresponding CO_2 emissions per kg production than copper while the impacts of primary steel and zinc are lower than that of copper (Van der Voet et al., 2018; Van Genderen et al., 2016). Regarding secondary production, an analysis of 48 metals and their application in various products by Schäfer and Schmidt (2020) shows that in many cases the metal concentration in EoL products is lower than that in natural ores. Recycling of this kind of metal is impeded by such low concentrations. These substitution and technological developments have not yet been included in the scenarios in this thesis. For future research, more analysis is still needed to assess the long-term use and impacts of substitution.

More efficient resource use by moving from a linear to a circular economy may cause rebound effects. In other studies, authors found that increasing efficiency of e.g. energy carriers or resource use can even lead to a 'backfire' effect - energy use or material use has become so efficient and cheap, that more use takes place instead of less, even beyond initial levels (Font Vivanco et al., 2016). A large body of literature has discussed the interplay between circular economy and rebound effects, mainly focusing on lower energy cost and efficiency improvements (Binswanger, 2001; Figge and Thorpe, 2019; Zink and Geyer, 2017). For example, improving efficiency leads to less energy cost, thus the users could spend the resulting savings on other products and services. The improving efficiency also could lower energy price, which in turn may cause more energy consumption. Another aspect of rebound is opportunity costs. Several circular economy polices target at enhancing material recycling, which means that the option of reuse cannot be chosen as a priority for the same source of secondary material. We refer to the discussion in Chapter 5 on copper reuse and recycling. Many policies related to resource efficiency regarding to copper have been implemented in China (see Table 1.1), however, further research on the rebound effects and related social as well as economic consequences induced from those policies is needed. In addition to the rebound effects, the transition to a circular economy of copper cycle may have other undesirable side effects. The analysis in this thesis is only focus on copper stocks and flows, the assumptions and changes of these flows may have implications for other resources. This is specifically true for copper reuse and recycling from various waste streams, since multiple co-products (e.g. Fe, Al) are processed simultaneously as well.

One direction of model development concerns expansion to include other aspects besides technology. This thesis focused on physical models, and the emphasis was on scenarios that describe the overall development of the copper cycle and its associated environmental impacts. Improving the environmental performance by optimizing the technical parameters (e.g. recycling rate, energy transition) is important, but there are other important considerations as well. Physical models therefore should be used in a wider context. For example, improving the collection rate and formalizing the recycling system involves economic and behavioral changes to become effective, however, strong emphasis on recycling may cause some rebound effects such as more disposable products consumption. Connecting physical models (e.g. MFA of waste management system) to models describing behavioral changes may give insights in these potential rebound effects. LCSA could be used to assess social and/or economic impacts of copper production (Guinée, 2016). Adding social and or economic information in the assessment could help for a better optimization of measures (e.g. lower production costs) in the copper cycle. Moreover, integrating the MFA with other economic models (e.g. Computable General Equilibrium models (CGE)) can give additional insights into economy-wide implications of policy measures focused on specific materials or sectors. This combination of MFA with CGE has been used for the Chinese building sector by Cao et al. (2019) to ensure that both mass and monetary balances would hold, and to assess economy-wide rebound effects of a saturation of building stock availability per capita via prospective modeling. In addition, the MFA can also be linked to spatially explicit models, such as the GIS-based models, which could give spatially explicit information on the environmental stocks and flows and hence where the (future) urban mine is concentrated (Verhagen et al., 2021; Yang et al., 2020).

6.3 Final remarks

The transition to a circular economy is an important ambition for China and other countries in the world. Copper is critical to the transition to a circular economy. This thesis explores the future copper demand and supply as well as associated environmental impacts, and addresses how the copper cycle in China can be transformed in a circular way. China currently consumes around 50% of the copper globally. With copper being very relevant for infrastructure development in other fast developing countries in the world, a transition towards a circular economy in China is likely to provide an important reduction of the pressure on copper demand for those countries. Since copper is a global market commodity, international partnerships are greatly needed to transfer the experiences of how to realize such transitions, but also to see how the circularity of copper flows across nations best can be organized. From a methodological perspective, this thesis provides a perspective on the integration of MFA and LCA. Such an approach could be very supportive to future research on other materials for which a transition to circularity is required, in order to move towards a sustainable development in the world.

References

Adachi, T., Mogi, G., 2007. Life cycle inventory for base metal ingots production in Japan including mining and mineral processing processes by cost estimating system database. Transactions of Nonferrous Metals Society of China 17(s1A), s131-s135.

Allegrini, E., Vadenbo, C., Boldrin, A., Astrup, T.F., 2015. Life cycle assessment of resource recovery from municipal solid waste incineration bottom ash. Journal of Environmental Management 151, 132-143.

Alvarado, S., Maldonado, P., Barrios, A., Jaques, I., 2002. Long term energy-related environmental issues of copper production. Energy 27(2), 183-196.

Antofagasta, Barrick, Colbun, 2018. Zaldívar copper mine moves to 100% renewable power, Mining Journal, https://www.mining-journal.com/copper-news/news/1342892/zald%C3%ADvar-copper-mine-moves-to-100-renewable-power.

Ardente, F., Mathieux, F., 2014. Environmental assessment of the durability of energy-using products: method and application. Journal of cleaner production 74, 62-73.

Ardente, F., Talens Peiró, L., Mathieux, F., Polverini, D., 2018. Accounting for the environmental benefits of remanufactured products: Method and application. Journal of Cleaner Production 198, 1545-1558.

Arvidsson, R., Tillman, A.-M., Sandén, B.A., Janssen, M., Nordelöf, A., Kushnir, D., Molander, S., 2018. Environmental Assessment of Emerging Technologies: Recommendations for Prospective LCA. Journal of Industrial Ecology 22(6), 1286-1294.

Asiedu, E., 2006. Foreign Direct Investment in Africa: The Role of Natural Resources, Market Size, Government Policy, Institutions and Political Instability. The World Economy 29(1), 63-77.

Assefa, G., Björklund, A., Eriksson, O., Frostell, B., 2005. ORWARE: an aid to environmental technology chain assessment. Journal of Cleaner Production 13(3), 265-274.

Atilgan, B., Azapagic, A., 2016. An integrated life cycle sustainability assessment of electricity generation in Turkey. Energy Policy 93, 168-186.

Ayres, R.U., Ayres, L.W., Råde, I., 2003. The Life Cycle of Copper, Its Co-Products and Byproducts.

References

Babaei, M.J., Molaei, M.A., Dehghani, A., 2015. Estimating the function of copper consumption in Iran between 1991- 2011 using Johansen model. Journal of Mining and Environment 6(2), 183-189.

Baccini, P., Bader, H.-P., 1996. Regionaler stoffhaushalt: erfassung, bewertung und steuerung. Spektrum Akademischer Verlag Heidelberg.

Bader, H.-P., Scheidegger, R., Wittmer, D., Lichtensteiger, T., 2011. Copper flows in buildings, infrastructure and mobiles: a dynamic model and its application to Switzerland. Clean Technologies and Environmental Policy 13(1), 87-101.

Ballantyne, G., Powell, M., 2014. Benchmarking comminution energy consumption for the processing of copper and gold ores. Minerals Engineering 65, 109-114.

Batker, D., Schmidt, R., 2015. Environmental and Social Benchmarking Analysis of Nautilus Minerals Inc: Solwara 1 Project. Earth Economics.

Bertram, M., Graedel, T.E., Fuse, K., Gordon, R., Lifset, R., Rechberger, H., Spatari, S., 2003. The copper cycles of European countries. Regional Environmental Change 3(4), 119-127.

Binswanger, M., 2001. Technological progress and sustainable development: what about the rebound effect? Ecological Economics 36(1), 119-132.

Brahmst, E., 2006. Copper in end-of-life vehicle recycling, The Center for Automotive Research, Ann Arbor, MI. Center for Automotive Research.

Brunner, P.H., Rechberger, H., 2003. Practical handbook of material flow analysis. CRC press.

Buchner, H., Laner, D., Rechberger, H., Fellner, J., 2015. Dynamic material flow modeling: An effort to calibrate and validate aluminum stocks and flows in Austria. Environmental science & technology 49(9), 5546-5554.

Burchart-Korol, D., Fugiel, A., Czaplicka-Kolarz, K., Turek, M., 2016. Model of environmental life cycle assessment for coal mining operations. Science of The Total Environment 562, 61-72.

Calvo, G., Mudd, G., Valero, A., Valero, A., 2016. Decreasing ore grades in global metallic mining: A theoretical issue or a global reality? Resources 5(4), 36.

Cao, Z., Liu, G., Zhong, S., Dai, H., Pauliuk, S., 2019. Integrating Dynamic Material Flow Analysis and Computable General Equilibrium Models for Both Mass and Monetary Balances in Prospective Modeling: A Case for the

Chinese Building Sector. Environmental Science & Technology 53(1), 224-233.

CELVE, 2019. Remanufacturing of components in ELVs: from "ELV waste, component waste" to "ELV waste, component reuse". China's End-of-life-vehicles Recycling Reprocess Economy.

CGS, 2016. National Mineral Resources Planning (2016-2020). Geological Survey of China.

Chang, M.M.L., Ong, S.K., Nee, A.Y.C., 2017. Approaches and Challenges in Product Disassembly Planning for Sustainability. Procedia CIRP 60, 506-511.

CHARI, 2018. Green Supply Chain Management of Electrical and Electronic Products. CHARI of the People's Republic of China.

Charles, R.G., Douglas, P., Hallin, I.L., Matthews, I., Liversage, G., 2017. An investigation of trends in precious metal and copper content of RAM modules in WEEE: Implications for long term recycling potential. Waste Management 60, 505-520.

Chaturvedi, A., Strasser, C., Eisinger, F., Raghupathy, L., Henzler, M.P., Arora, R., 2012. The carbon footprint of e-waste recycling - Indian scenarios, 2012 Electronics Goes Green 2012+. pp. 1-6.

Chen, J., Wang, Z., Wu, Y., Li, L., Li, B., Pan, D., Zuo, T., 2019. Environmental benefits of secondary copper from primary copper based on life cycle assessment in China. Resources, Conservation and Recycling 146, 35-44.

Chen, W.-Q., Graedel, T.E., 2012. Anthropogenic Cycles of the Elements: A Critical Review. Environmental Science & Technology 46(16), 8574-8586.

Chen, W., Shi, L., Qian, Y., 2010. Substance flow analysis of aluminium in mainland China for 2001, 2004 and 2007: Exploring its initial sources, eventual sinks and the pathways linking them. Resources Conservation & Recycling 54(9), 557-570.

Chen, W., Wang, M., Li, X., 2016. Analysis of copper flows in the United States: 1975–2012. Resources, Conservation and Recycling 111, 67-76.

Chen, Y., Ding, Z., Liu, J., Ma, J., 2019. Life cycle assessment of end-of-life vehicle recycling in China: a comparative study of environmental burden and benefit. International Journal of Environmental Studies 76(6), 1019-1040.

Chen, Y., Yang, Y., Hu, S., Xie, L., Yang, Y., Huang, W., Chen, Z., 2018.

References

Analysis of the current situation and policy suggestions of the recycling and utilization of ELVs in China Engineering 01, 113-119.

Chi, X., Streicher-Porte, M., Wang, M.Y.L., Reuter, M.A., 2011. Informal electronic waste recycling: A sector review with special focus on China. Waste Management 31(4), 731-742.

Chi, X., Wang, M.Y.L., Reuter, M.A., 2014. E-waste collection channels and household recycling behaviors in Taizhou of China. Journal of Cleaner Production 80, 87-95.

Choi, S.G., Kim, C.S., Ko, E.M., Kim, S.Y., Jo, H.Y., 2008. Mineral Economic Index and Comprehensive Demand Prediction for Strategic Minerals: Copper, Zinc, Lead, and Nickel. Economic and Environmental Geology 41(3), 345-357.

Ciacci, L., Fishman, T., Elshkaki, A., Graedel, T.E., Vassura, I., Passarini, F., 2020. Exploring future copper demand, recycling and associated greenhouse gas emissions in the EU-28. Global Environmental Change 63, 102093.

CML, 2016. CML-IA Characterisation Factors., Department of Industrial Ecology, Leiden University. https://www.universiteitleiden.nl/en/research/research-output/science/cml-ia-characterisation-factors.

CNMIA, 2016. China Nonferrous Industry Statistical Yearbook. China Nonferrous Metals Industry Association.

CNMIA, 2019. China Nonferrous Industry Statistical Yearbook. China Nonferrous Metals Industry Association.

CNREC, 2017. China Renewable Energy Outlook 2017. Renewable Energy Center of China.

Commission, E., 2017. Study on the review of the list of critical raw materials. European Commission Brussels.

Council, N.R., 2002. Evolutionary and revolutionary technologies for mining. National Academies Press.

Crane, W., Krausmann, F., Eisenmenger, N., Giljum, S., Hennicke, P., Kemp, R., Lankao, P.R., Manalang, B.S., Sewerin, S., 2011. Decoupling Natural Resource Use and Environmental Impacts from Economic Growth. International Resource Panel.

Crowson, P., 2012. Some observations on copper yields and ore grades. Resources Policy 37(1), 59-72.

Daigo, I., Hashimoto, S., Matsuno, Y., Adachi, Y., 2007. Dynamic analysis on material balance of copper and copper-alloy scraps in Japan. Journal of the Japan Institute of Metals 71(7), 563-569.

Daigo, I., Hashimoto, S., Matsuno, Y., Adachi, Y., 2009. Material stocks and flows accounting for copper and copper-based alloys in Japan. Resources Conservation and Recycling 53(4), 208-217.

Das, S., Lee, S.H., Kumar, P., Kim, K., Lee, S.S., Bhattacharya, S.S., 2019. Solid waste management: Scope and the challenge of sustainability. Journal of Cleaner Production 228, 658-678.

Davidson, A.J., Binks, S.P., Gediga, J., 2016. Lead industry life cycle studies: environmental impact and life cycle assessment of lead battery and architectural sheet production. The International Journal of Life Cycle Assessment 21(11), 1624-1636.

Davis, J., Geyer, R., Ley, J., He, J., Clift, R., Kwan, A., Sansom, M., Jackson, T., 2007. Time-dependent material flow analysis of iron and steel in the UK: Part 2. Scrap generation and recycling. Resources Conservation & Recycling 51(1), 118-140.

De Meester, S., Nachtergaele, P., Debaveye, S., Vos, P., Dewulf, J., 2019. Using material flow analysis and life cycle assessment in decision support: A case study on WEEE valorization in Belgium. Resources, Conservation and Recycling 142, 1-9.

Deetman, S., Pauliuk, S., van Vuuren, D.P., van der Voet, E., Tukker, A., 2018. Scenarios for Demand Growth of Metals in Electricity Generation Technologies, Cars, and Electronic Appliances. Environmental Science & Technology 52(8), 4950-4959.

Dong, D., An, H., Huang, S., 2017. The transfer of embodied carbon in copper international trade: An industry chain perspective. Resources Policy 52, 173-180.

Dong, D., Gao, X., Sun, X., Liu, X., 2018. Factors affecting the formation of copper international trade community: Based on resource dependence and network theory. Resources Policy 57, 167-185.

Dong, D., Tercero Espinoza, L.A., Loibl, A., Pfaff, M., Tukker, A., Van der Voet, E., 2020a. Scenarios for anthropogenic copper demand and supply in China: implications of a scrap import ban and a circular economy transition. Resources, Conservation and Recycling 161, 104943.

Dong, D., Tukker, A., Van der Voet, E., 2019. Modeling copper demand in

China up to 2050: A business-as-usual scenario based on dynamic stock and flow analysis. Journal of Industrial Ecology.

Dong, D., van Oers, L., Tukker, A., van der Voet, E., 2020b. Assessing the future environmental impacts of copper production in China: Implications of the energy transition. Journal of Cleaner Production 274, 122825.

Duan, H., Hou, K., Li, J., Zhu, X., 2011. Examining the technology acceptance for dismantling of waste printed circuit boards in light of recycling and environmental concerns. Journal of environmental management 92(3), 392-399.

Eheliyagoda, D., Wei, F., Shan, G., Albalghiti, E., Zeng, X., Li, J., 2019. Examining the Temporal Demand and Sustainability of Copper in China. Environmental Science & Technology 53(23), 13812-13821.

Ellen MacArthur Foundation, 2013. Towards the circular economy. Journal of Industrial Ecology 2, 23-44.

Elshkaki, A., Graedel, T., 2013. Dynamic analysis of the global metals flows and stocks in electricity generation technologies. Journal of Cleaner Production 59, 260-273.

Elshkaki, A., Graedel, T.E., Ciacci, L., Reck, B.K., 2016. Copper demand, supply, and associated energy use to 2050. Global Environmental Change-Human and Policy Dimensions 39, 305-315.

Elshkaki, A., Graedel, T.E., Ciacci, L., Reck, B.K., 2018. Resource Demand Scenarios for the Major Metals. Environmental Science & Technology 52(5), 2491-2497.

Elshkaki, A., Van der Voet, E., Timmermans, V., Holderbeke, M.V., 2005. Dynamic stock modelling: A method for the identification and estimation of future waste streams and emissions based on past production and product stock characteristics. Energy 30(8), 1353-1363.

Eriksen, M.K., Pivnenko, K., Faraca, G., Boldrin, A., Astrup, T.F., 2020. Dynamic Material Flow Analysis of PET, PE, and PP Flows in Europe: Evaluation of the Potential for Circular Economy. Environmental Science & Technology 54(24), 16166-16175.

European Commission, 2003. Communication from the commission to the council.

Fan, Y., Fang, C., 2020. Circular economy development in China-current situation, evaluation and policy implications. Environmental Impact Assessment Review 84, 106441.

Farjana, S.H., Huda, N., Mahmud, M.P., 2019a. Impacts of aluminum production: A cradle to gate investigation using life-cycle assessment. Science of the Total Environment 663, 958-970.

Farjana, S.H., Huda, N., Parvez Mahmud, M.A., Saidur, R., 2019b. A review on the impact of mining and mineral processing industries through life cycle assessment. Journal of Cleaner Production 231, 1200-1217.

Farjana, S.H., Li, W., 2021. Integrated LCA-MFA Framework for Gold Production from Primary and Secondary Sources. Procedia CIRP 98, 511-516.

Ferreira, H., Leite, M.G.P., 2015. A Life Cycle Assessment study of iron ore mining. Journal of cleaner production 108, 1081-1091.

Figge, F., Thorpe, A.S., 2019. The symbiotic rebound effect in the circular economy. Ecological Economics 163, 61-69.

Finkbeiner, M., Schau, E.M., Lehmann, A., Traverso, M., 2010. Towards life cycle sustainability assessment. Sustainability 2(10), 3309-3322.

Fiore, S., Ibanescu, D., Teodosiu, C., Ronco, A., 2019. Improving waste electric and electronic equipment management at full-scale by using material flow analysis and life cycle assessment. Science of The Total Environment 659, 928-939.

Foelster, A.-S., Andrew, S., Kroeger, L., Bohr, P., Dettmer, T., Boehme, S., Herrmann, C., 2016. Electronics recycling as an energy efficiency measure – a Life Cycle Assessment (LCA) study on refrigerator recycling in Brazil. Journal of Cleaner Production 129, 30-42.

Font Vivanco, D., Kemp, R., van der Voet, E., 2016. How to deal with the rebound effect? A policy-oriented approach. Energy Policy 94, 114-125.

Forti, V., Balde, C.P., Kuehr, R., Bel, G., 2020. The Global E-waste Monitor 2020: Quantities, flows and the circular economy potential. United Nations.

Fu, X., Ueland, S.M., Olivetti, E., 2017. Econometric modeling of recycled copper supply. Resources, Conservation and Recycling 122, 219-226.

GACC, 2018. Catalogue for Administration of Import of Solid Wastes. General Administration of Customs of the People's Republic of China

Gan, Y., Griffin, W.M., 2018. Analysis of life-cycle GHG emissions for iron ore mining and processing in China—Uncertainty and trends. Resources Policy 58, 90-96.

García-Olivares, A., 2015. Substituting silver in solar photovoltaics is feasible and allows for decentralization in smart regional grids. Environmental

Innovation and Societal Transitions 17, 15-21.

Geng, J., Hao, H., Sun, X., Xun, D., Liu, Z., Zhao, F., 2021. Static material flow analysis of neodymium in China. Journal of Industrial Ecology 25(1), 114-124.

Geng, Y., Doberstein, B., 2008. Developing the circular economy in China: Challenges and opportunities for achieving 'leapfrog development'. International Journal of Sustainable Development & World Ecology 15(3), 231-239.

Geng, Y., Sarkis, J., Bleischwitz, R., 2019. How to globalize the circular economy. Nature Publishing Group.

Gerst, M.D., 2009. Linking material flow analysis and resource policy via future scenarios of in-use stock: an example for copper. Environmental Science & Technology 43(16), 6320.

Gharfalkar, M., Court, R., Campbell, C., Ali, Z., Hillier, G., 2015. Analysis of waste hierarchy in the European waste directive 2008/98/EC. Waste Management 39, 305-313.

Ghisellini, P., Cialani, C., Ulgiati, S., 2016. A review on circular economy: the expected transition to a balanced interplay of environmental and economic systems. Journal of Cleaner Production 114, 11-32.

Giurco, D., Petrie, J., 2007. Strategies for reducing the carbon footprint of copper: New technologies, more recycling or demand management? Minerals Engineering 20(9), 842-853.

Giusti, L., 2009. A review of waste management practices and their impact on human health. Waste Management 29(8), 2227-2239.

Glöser, S., Soulier, M., Tercero Espinoza, L.A., 2013. Dynamic analysis of global copper flows. Global stocks, postconsumer material flows, recycling indicators, and uncertainty evaluation. Environmental Science & Technology 47(12), 6564-6572.

Gradin, K.T., Luttropp, C., Björklund, A., 2013. Investigating improved vehicle dismantling and fragmentation technology. Journal of cleaner production 54, 23-29.

Graedel, T.E., Allwood, J., Birat, J.P., Buchert, M., Hagelüken, C., Reck, B.K., Sibley, S.F., Sonnemann, G., 2011. What do we know about metal recycling rates? Journal of Industrial Ecology 15(3), 355-366.

Graedel, T.E., Bertram, M., Kapur, A., Reck, B., Spatari, S., 2004a.

Exploratory data analysis of the multilevel anthropogenic copper cycle. Environmental Science & Technology 38(4), 1253-1261.

Graedel, T.E., Harper, E.M., Nassar, N.T., Reck, B.K., 2015. On the materials basis of modern society. Proceedings of the National Academy of Sciences 112(20), 6295-6300.

Graedel, T.E., van Beers, D., Bertram, M., Fuse, K., Gordon, R.B., Gritsinin, A., Kapur, A., Klee, R.J., Lifset, R.J., Memon, L., Rechberger, H., Spatari, S., Vexler, D., 2004b. Multilevel Cycle of Anthropogenic Copper. Environmental Science & Technology 38(4), 1242-1252.

Guan, D., Meng, J., Reiner, D.M., Zhang, N., Shan, Y., Mi, Z., Shao, S., Liu, Z., Zhang, Q., Davis, S.J., 2018. Structural decline in China's CO2 emissions through transitions in industry and energy systems. Nature Geoscience 11(8), 551-555.

Guinée, J., 2016. Life cycle sustainability assessment: What is it and what are its challenges?, Taking stock of industrial ecology. Springer, Cham, pp. 45-68.

Guinée, J.B., 2002. Handbook on life cycle assessment operational guide to the ISO standards. The international journal of life cycle assessment 7(5), 311-313.

Guinee, J.B., Heijungs, R., Huppes, G., Zamagni, A., Masoni, P., Buonamici, R., Ekvall, T., Rydberg, T., 2011. Life cycle assessment: past, present, and future. ACS Publications.

Guo, X.Y., Song, Y., 2008. Substance flow analysis of copper in China. Resources Conservation and Recycling 52(6), 874-882.

Haberl, H., Wiedenhofer, D., Virág, D., Kalt, G., Plank, B., Brockway, P., Fishman, T., Hausknost, D., Krausmann, F., Leon-Gruchalski, B., Mayer, A., Pichler, M., Schaffartzik, A., Sousa, T., Streeck, J., Creutzig, F., 2020. A systematic review of the evidence on decoupling of GDP, resource use and GHG emissions, part II: synthesizing the insights. Environmental Research Letters 15(6), 065003.

Han, H., Wang, H.W., Ouyang, M.G., Cheng, F., 2011. Vehicle survival patterns in China. Science China Technological Sciences 54(3), 625-629.

Haque, N., Norgate, T., 2014. The greenhouse gas footprint of in-situ leaching of uranium, gold and copper in Australia. Journal of cleaner production 84, 382-390.

Haupt, M., Zschokke, M., 2017. How can LCA support the circular economy?—63rd discussion forum on life cycle assessment, Zurich, Switzerland, November 30, 2016. The International Journal of Life Cycle Assessment 22(5), 832-837.

Hedbrant, J., 2001. Stockhome: A Spreadsheet Model of Urban Heavy Metal Metabolism. Water Air & Soil Pollution Focus 1(3-4), 55-66.

Holm, O., Wollik, E., Johanna Bley, T., 2018. Recovery of copper from small grain size fractions of municipal solid waste incineration bottom ash by means of density separation. International Journal of Sustainable Engineering 11(4), 250-260.

Hong, J., Chen, Y., Liu, J., Ma, X., Qi, C., Ye, L., 2018. Life cycle assessment of copper production: a case study in China. The International Journal of Life Cycle Assessment 23(9), 1814-1824.

Hong, J., Li, X., Zhaojie, C., 2010. Life cycle assessment of four municipal solid waste management scenarios in China. Waste Management 30(11), 2362-2369.

Hong, J., Shi, W., Wang, Y., Chen, W., Li, X., 2015. Life cycle assessment of electronic waste treatment. Waste Management 38, 357-365.

Hu, M., Bergsdal, H., van der Voet, E., Huppes, G., Müller, D.B., 2010a. Dynamics of urban and rural housing stocks in China. Building Research & Information 38(3), 301-317.

Hu, M., Pauliuk, S., Wang, T., Huppes, G., van der Voet, E., Müller, D.B., 2010b. Iron and steel in Chinese residential buildings: A dynamic analysis. Resources Conservation & Recycling 54(9), 591-600.

Huang, T., Shi, F., Tanikawa, H., Fei, J., Han, J., 2013. Materials demand and environmental impact of buildings construction and demolition in China based on dynamic material flow analysis. Resources, Conservation and Recycling 72, 91-101.

Hunt, A.J., 2013. Element recovery and sustainability. Royal Society of Chemistry.

ICA, 2013. Copper Recycling.

ICA, 2016. Global Copper Substitution and Regulatory Trends, International Copper Association. https://copperalliance.org/wp-content/uploads/2017/05/2017.04-Substitution-and-Regulation-Factsheet.pdf.

ICA, 2017. Global Copper Substitution and Regulatory Trends. International

Copper Association.

ICA, 2021. Stocks and flows. https://copperalliance.org/about-copper/stocks-and-flows/. (Accessed 23 March 2021).

ICGS, 2021. Latest Copper Market Forecast, in: Group, I.C.S. (Ed.).

ICSG, 2020. World Copper Factbook 2020. International Copper Study Group.

IEA, 2017. Global Energy Efficiency Report 2017, International Energy Agency.

IFs, 2017. International Futures (IFs) modeling system, Version x.xx. Frederick S. Pardee Center for International Futures, Josef Korbel School of International Studies, University of Denver, Denver, CO.

Igarashi, Y., Kakiuchi, E., Daigo, I., Matsuno, Y., Adachi, Y., 2008. Estimation of Steel Consumption and Obsolete Scrap Generation in Japan and Asian Countries in the Future. Isij International 93(12), 782-791.

Ioannidou, D., Sonnemann, G., Suh, S., 2020. Do we have enough natural sand for low-carbon infrastructure? Journal of Industrial Ecology 24(5), 1004-1015.

IRP, 2019. Global Resources Outlook 2019: Natural Resources for the Future We Want., in: Panel, I.R. (Ed.). United Nations Environment Programme.

ISO, 2006a. Environmental Management—Life Cycle Assessment—Principles and Framework. International Organization for Standardization 14040: 2006 (E) Series.

ISO, 2006b. Environmental management: Life cycle assessment; requirements and guidelines. International Organization for Standardization.

Jiang, J., Dai, J., Feng, W., Xu, J., 2006. Life cycle assessment of the processes of producing copper by pyrometallurgy and hydrometallurgy. Journal of Lanzhou University of Technology 32(1), 19-21.

Jiang, M., Behrens, P., Wang, T., Tang, Z., Yu, Y., Chen, D., Liu, L., Ren, Z., Zhou, W., Zhu, S., He, C., Tukker, A., Zhu, B., 2019. Provincial and sector-level material footprints in China. Proceedings of the National Academy of Sciences 116(52), 26484.

John, R., 2012. Energy Use in Metal Production High Temperature Processing Symposium 2012 CSIRO, Process Science and Engineering, Australia.

Kapur, A., 2006. The future of the red metal - A developing country perspective from India. Resources Conservation and Recycling 47(2), 160-182.

Kaufman, S.M., Krishnan, N., Themelis, N.J., 2010. A Screening Life Cycle Metric to Benchmark the Environmental Sustainability of Waste Management Systems. Environmental Science & Technology 44(15), 5949-5955.

Keller, H., Rettenmaier, N., Reinhardt, G.A., 2015. Integrated life cycle sustainability assessment–A practical approach applied to biorefineries. Applied Energy 154, 1072-1081.

Khoo, J.Z., Haque, N., Woodbridge, G., McDonald, R., Bhattacharya, S., 2017. A life cycle assessment of a new laterite processing technology. Journal of cleaner production 142, 1765-1777.

Kiddee, P., Naidu, R., Wong, M.H., 2013. Electronic waste management approaches: An overview. Waste Management 33(5), 1237-1250.

Kleijn, R., 2012. Materials and energy: a story of linkages Date: 2012-09-05.

Krausmann, F., Wiedenhofer, D., Lauk, C., Haas, W., Tanikawa, H., Fishman, T., Miatto, A., Schandl, H., Haberl, H., 2017. Global socioeconomic material stocks rise 23-fold over the 20th century and require half of annual resource use. Proceedings of the National Academy of Sciences 114(8), 1880-1885.

Kuipers, K.J.J., van Oers, L., Verboon, M., van der Voet, E., 2018. Assessing environmental implications associated with global copper demand and supply scenarios from 2010 to 2050. Global Environmental Change-Human and Policy Dimensions 49, 106-115.

Kulczycka, J., Lelek, Ł., Lewandowska, A., Wirth, H., Bergesen, J.D., 2016. Environmental Impacts of Energy-Efficient Pyrometallurgical Copper Smelting Technologies: The Consequences of Technological Changes from 2010 to 2050. Journal of Industrial Ecology 20(2), 304-316.

Lassesson, H., Fedje, K.K., Steenari, B.-M., 2014. Leaching for recovery of copper from municipal solid waste incineration fly ash: Influence of ash properties and metal speciation. Waste management & research 32(8), 755-762.

Li, H., Qi, Y., 2011. Comparison of China's carbon emission scenarios in 2050. Advances in Climate Change Research 2(4), 193-202.

Li, J., Liang, J., Zuo, J., Guo, H., 2020. Environmental impact assessment of mobile recycling of demolition waste in Shenzhen, China. Journal of Cleaner Production 263, 121371.

Li, Y., Guan, J., 2009. Life cycle assessment of recycling copper process from copper-slag, 2009 International Conference on Energy and Environment Technology. IEEE, pp. 198-201.

Ling, Z., Zengwei, Y., Jun, B., 2012. Estimation of Copper In-use Stocks in Nanjing, China. Journal of Industrial Ecology 16(2), 191-202.

Linzner, R., Salhofer, S., 2014. Municipal solid waste recycling and the significance of informal sector in urban China. Waste management & research 32(9), 896-907.

Liu, C., Lin, J., Cao, H., Zhang, Y., Sun, Z., 2019. Recycling of spent lithium-ion batteries in view of lithium recovery: A critical review. Journal of Cleaner Production 228, 801-813.

Liu, G., Bangs, C.E., Müller, D.B., 2013. Stock dynamics and emission pathways of the global aluminium cycle. Nature Climate Change 3(4), 338.

Liu, G., Bangs, C.E., Müller, D.B., 2011. Unearthing potentials for decarbonizing the US aluminum cycle. Environmental science & technology 45(22), 9515-9522.

Liu, M., Chen, X., Zhang, M., Lv, X., Wang, H., Chen, Z., Huang, X., Zhang, X., Zhang, S., 2020. End-of-life passenger vehicles recycling decision system in China based on dynamic material flow analysis and life cycle assessment. Waste Management 117, 81-92.

Liu, X., Tanaka, M., Matsui, Y., 2006. Electrical and electronic waste management in China: progress and the barriers to overcome. Waste Management & Research 24(1), 92-101.

Macquarie, R., 2015. Copper In China-A Bottom-up Approach To Long-term Demand, Macquarie Research

Magalini, F., Wang, F., Huisman, J., Kuehr, R., Baldé, K., van Straalen, V., Hestin, M., Lecerf, L., Sayman, U., Akpulat, O., 2014. Study on collection rates of waste electrical and electronic equipment (WEEE). EU Commission.

Månberger, A., Stenqvist, B., 2018. Global metal flows in the renewable energy transition: Exploring the effects of substitutes, technological mix and development. Energy Policy 119, 226-241.

Marsden, J.O., 2008. Energy efficiency and copper hydrometallurgy, Hydrometallurgy. pp. 29-42.

Maung, K.N., Hashimoto, S., Mizukami, M., Morozumi, M., Lwin, C.M., 2017. Assessment of the Secondary Copper Reserves of Nations.

Environmental Science & Technology 51(7), 3824-3832.

McMillan, C.A., Keoleian, G.A., 2009. Not all primary aluminum is created equal: life cycle greenhouse gas emissions from 1990 to 2005. Environmental science & technology 43(5), 1571-1577.

McMillan, C.A., Skerlos, S.J., Keoleian, G.A., 2012. Evaluation of the metals industry's position on recycling and its implications for environmental emissions. Journal of Industrial Ecology 16(3), 324-333.

MEEC, 2006. Discarded household appliances and electronic products pollution control technology policy. Ministry of Ecology and Environment of the People's Republic of China

MEEC, 2018. Work Plan for the Pilot Program of 'Zero-Waste City' Construction. Ministry of Ecology and Environment of the People's Republic of China

Melo, M.T., 1999. Statistical analysis of metal scrap generation: the case of aluminium in Germany. Resources, Conservation and Recycling 26(2), 91-113.

Memary, R., Giurco, D., Mudd, G., Mason, L., 2012. Life cycle assessment: a time-series analysis of copper. Journal of Cleaner Production 33, 97-108.

Mendoza, J.M.F., Sharmina, M., Gallego-Schmid, A., Heyes, G., Azapagic, A., 2017. Integrating Backcasting and Eco-Design for the Circular Economy: The BECE Framework. Journal of Industrial Ecology 21(3), 526-544.

Meng, L., Zhong, Y., Guo, L., Wang, Z., Chen, K., Guo, Z., 2018. High-temperature centrifugal separation of Cu from waste printed circuit boards. Journal of Cleaner Production 199, 831-839.

Mercante, I.T., Bovea, M.D., Ibáñez-Forés, V., Arena, A.P., 2011. Life cycle assessment of construction and demolition waste management systems: a Spanish case study. The International Journal of Life Cycle Assessment 17(2), 232-241.

MIIT, 2020. Requirements of the Industry Standards for the Comprehensive Utilization of Waste Power Storage Batteries of New Energy Vehicles. Ministry of Industry and Information Technology

MNR, 2016. the National Mineral Resource Planning (2016-2020). Ministry of Natural Resources.

MNR, 2018. China mineral resources, Geological Publishing House. Ministry of Natural Resources.

www.mnr.gov.cn/sj/sjfw/kc_19263/zgkczybg/201811/t20181116_2366032.html.

MNR, 2020. China mineral resources, Geological Publishing House. Ministry of Natural Resources. http://www.mnr.gov.cn/dt/ywbb/202010/t20201023_2573150.html.

Mohr, S., 2010. Projection of world fossil fuel production with supply and demand interactions. University of Newcastle.

MOHURD, 2005. Regulations on Urban Construction Waste Management. Ministry of Housing and Urban-Rural Development of the People's Republic of China

MOHURD, 2019. Technical Standards for Construction Waste Disposal. Ministry of Housing and Urban-Rural Development of the People's Republic of China

Molteni, D., 2017. Plant and process for the recovery of wires from car fluff. Google Patents.

Moreno Ruiz, E., Valsasina, L., Fitzgerald, D., Brunner, F., Vadenbo, C., Bauer, C., Bourgault, G., Symeonidis, A., Wernet, G., 2017. Documentation of changes implemented in the ecoinvent database v3. 4. ecoinvent. Zür Switz.

Muchová, L., Rem, P., 2006. Metal content and recovery of MSWI bottom ash in Amsterdam. WIT Transactions on Ecology and the Environment 92.

Mudd, G.M., Weng, Z., Jowitt, S.M., 2013. A detailed assessment of global Cu resource trends and endowments. Economic Geology 108(5), 1163-1183.

Müller, D.B., 2006. Stock dynamics for forecasting material flows—Case study for housing in The Netherlands. Ecological Economics 59(1), 142-156.

Müller, E., Hilty, L.M., Widmer, R., Schluep, M., Faulstich, M., 2014. Modeling metal stocks and flows: A review of dynamic material flow analysis methods. Environmental science & technology 48(4), 2102-2113.

Nakem, S., Pipatanatornkul, J., Papong, S., Rodcharoen, T., Nithitanakul, M., Malakul, P., 2016. Material Flow Analysis (MFA) and Life Cycle Assessment (LCA) Study for Sustainable Management of PVC Wastes in Thailand, in: Kravanja, Z., Bogataj, M. (Eds.), Computer Aided Chemical Engineering. Elsevier, pp. 1689-1694.

NDRC, 2008. Administrative Measures for Pilot Remanufacturing of Automobile Parts & Accessories. National Development and Reform Commission

NDRC, 2017. Initiative to Guide the Shift Toward Circular Development. National Development and Reform Commission

Norgate, T., 2001. A comparative Life Cycle Assessment of copper production processes. Clayton South: CSIRO Minerals.

Norgate, T., 2004. Metal recycling: an assessment using life cycle energy consumption as a sustainability indicator. CRISO Minerals Report DMR-2616.

Norgate, T., Jahanshahi, S., 2010. Low grade ores–smelt, leach or concentrate? Minerals Engineering 23(2), 65-73.

Norgate, T., Jahanshahi, S., 2011. Reducing the greenhouse gas footprint of primary metal production: Where should the focus be? Minerals Engineering 24(14), 1563-1570.

Norgate, T., Jahanshahi, S., Rankin, W., 2007. Assessing the environmental impact of metal production processes. Journal of Cleaner Production 15(8-9), 838-848.

Northey, S., Haque, N., Mudd, G., 2013. Using sustainability reporting to assess the environmental footprint of copper mining. Journal of Cleaner Production 40, 118-128.

Northey, S., Mohr, S., Mudd, G.M., Weng, Z., Giurco, D., 2014. Modelling future copper ore grade decline based on a detailed assessment of copper resources and mining. Resources Conservation and Recycling 83, 190-201.

Nunez, P., Jones, S., 2016. Cradle to gate: life cycle impact of primary aluminium production. The International Journal of Life Cycle Assessment 21(11), 1594-1604.

Nuss, P., Eckelman, M.J., 2014. Life cycle assessment of metals: a scientific synthesis. PLoS One 9(7).

Nzila, C., Dewulf, J., Spanjers, H., Tuigong, D., Kiriamiti, H., Van Langenhove, H., 2012. Multi criteria sustainability assessment of biogas production in Kenya. Applied Energy 93, 496-506.

OECD, 2018. GDP long-term forecast (indicator).

OECD, 2019. Global Material Resources Outlook to 2060.

Onat, N.C., Kucukvar, M., Tatari, O., Egilmez, G., 2016. Integration of system dynamics approach toward deepening and broadening the life cycle sustainability assessment framework: a case for electric vehicles. The International Journal of Life Cycle Assessment 21(7), 1009-1034.

Padeyanda, Y., Jang, Y.-C., Ko, Y., Yi, S., 2016. Evaluation of environmental impacts of food waste management by material flow analysis (MFA) and life cycle assessment (LCA). Journal of Material Cycles and Waste Management 18(3), 493-508.

Park, J., Sarkis, J., Wu, Z., 2010. Creating integrated business and environmental value within the context of China's circular economy and ecological modernization. Journal of Cleaner Production 18(15), 1494-1501.

Parker, D.J., McNaughton, C.S., Sparks, G.A., 2016. Life cycle greenhouse gas emissions from uranium mining and milling in Canada. Environmental science & technology 50(17), 9746-9753.

Pauliuk, S., 2014. Python Dynamic Stock Model; Python Software: Trondheim, Norway.

Pauliuk, S., Wang, T., Müller, D.B., 2012. Moving Toward the Circular Economy: The Role of Stocks in the Chinese Steel Cycle. Environmental Science & Technology 46(1), 148-154.

Perez, J.P.H., Folens, K., Leus, K., Vanhaecke, F., Van Der Voort, P., Du Laing, G., 2019. Progress in hydrometallurgical technologies to recover critical raw materials and precious metals from low-concentrated streams. Resources, Conservation and Recycling 142, 177-188.

Pfaff, M., Gloser-Chahoud, S., Chrubasik, L., Walz, R., 2018. Resource efficiency in the German copper cycle: Analysis of stock and flow dynamics resulting from different efficiency measures. Resources Conservation and Recycling 139, 205-218.

Piatkowski, M.M., Coste, A., Shi, L., Du, Y., Cai, Z., 2019. Enhancing China's Regulatory Framework for Eco-Industrial Parks: Comparative Analysis of Chinese and International Green Standards. The World Bank.

Pita, F., Castilho, A., 2018. Separation of copper from electric cable waste based on mineral processing methods: A case study. Minerals 8(11), 517.

Qiang, Y., WANG, H.-m., LU, Z.-w., 2012. Quantitative estimation of social stock for metals Al and Cu in China. Transactions of Nonferrous Metals Society of China 22(7), 1744-1752.

Radetzki, M., 2009. Seven thousand years in the service of humanity—the history of copper, the red metal. Resources Policy 34(4), 176-184.

Rechberger, H., Graedel, T.E., 2002. The contemporary European copper cycle: statistical entropy analysis. Ecological Economics 42(1), 59-72.

Reijnders, L., 2003. Recovery of dissipated copper and the future of copper supply. Resources Conservation and Recycling 38(1), 59-66.

Rincón, L., Castell, A., Pérez, G., Solé, C., Boer, D., Cabeza, L.F., 2013. Evaluation of the environmental impact of experimental buildings with different constructive systems using Material Flow Analysis and Life Cycle Assessment. Applied Energy 109, 544-552.

Rochat, D., Binder, C.R., Diaz, J., Jolliet, O., 2013. Combining Material Flow Analysis, Life Cycle Assessment, and Multiattribute Utility Theory. Journal of Industrial Ecology 17(5), 642-655.

Rötzer, N., Schmidt, M., 2020. Historical, Current, and Future Energy Demand from Global Copper Production and Its Impact on Climate Change. Resources 9(4), 44.

Ruan, R., Zhong, S., Wang, D., 2010. Life cycle assessment of two copper metallurgical processes: Bio-heap leaching and Flotation-flash smelting. Multipurpose Utilization of Mineral Resources.

Rubin, R.S., de Castro, M.A.S., Brandão, D., Schalch, V., Ometto, A.R., 2014. Utilization of Life Cycle Assessment methodology to compare two strategies for recovery of copper from printed circuit board scrap. Journal of Cleaner Production 64, 297-305.

Ruhrberg, M., 2006. Assessing the recycling efficiency of copper from end-of-life products in Western Europe. Resources, Conservation and Recycling 48(2), 141-165.

Salhofer, S., Steuer, B., Ramusch, R., Beigl, P., 2016. WEEE management in Europe and China – A comparison. Waste Management 57, 27-35.

Santini, A., Morselli, L., Passarini, F., Vassura, I., Di Carlo, S., Bonino, F., 2011. End-of-Life Vehicles management: Italian material and energy recovery efficiency. Waste Management 31(3), 489-494.

Sato, F.E.K., Furubayashi, T., Nakata, T., 2019. Application of energy and CO2 reduction assessments for end-of-life vehicles recycling in Japan. Applied Energy 237, 779-794.

Schäfer, P., Schmidt, M., 2020. Discrete-Point Analysis of the Energy Demand of Primary versus Secondary Metal Production. Environmental Science & Technology 54(1), 507-516.

Schandl, H., Fischer-Kowalski, M., West, J., Giljum, S., Dittrich, M., Eisenmenger, N., Geschke, A., Lieber, M., Wieland, H., Schaffartzik, A., Krausmann, F., Gierlinger, S., Hosking, K., Lenzen, M., Tanikawa, H., Miatto, A., Fishman, T., 2018. Global Material Flows and Resource Productivity: Forty Years of Evidence. Journal of Industrial Ecology 22(4), 827-838.

Schandl, H., Hatfield-Dodds, S., Wiedmann, T., Geschke, A., Cai, Y., West, J., Newth, D., Baynes, T., Lenzen, M., Owen, A., 2016. Decoupling global environmental pressure and economic growth: scenarios for energy use, materials use and carbon emissions. Journal of Cleaner Production 132, 45-56.

Scheel, C., Aguiñaga, E., Bello, B., 2020. Decoupling Economic Development from the Consumption of Finite Resources Using Circular Economy. A Model for Developing Countries. Sustainability 12(4), 1291.

Schiller, G., Müller, F., Ortlepp, R., 2017. Mapping the anthropogenic stock in Germany: Metabolic evidence for a circular economy. Resources, Conservation and Recycling 123, 93-107.

Schipper, B.W., Lin, H.-C., Meloni, M.A., Wansleeben, K., Heijungs, R., van der Voet, E., 2018. Estimating global copper demand until 2100 with regression and stock dynamics. Resources, Conservation and Recycling 132, 28-36.

Schlesinger, M.E., King, M.J., Sole, K.C., Davenport, W.G., 2011. Extractive metallurgy of copper. Elsevier.

Schneider, L., Berger, M., Schüler-Hainsch, E., Knöfel, S., Ruhland, K., Mosig, J., Bach, V., Finkbeiner, M., 2014. The economic resource scarcity potential (ESP) for evaluating resource use based on life cycle assessment. The International Journal of Life Cycle Assessment 19(3), 601-610.

Seliger, G., Kernbaum, S., Zettl, M., 2006. Remanufacturing approaches contributing to sustainable engineering. Gestão & Produção 13, 367-384.

Seniunaite, J., Vasarevicius, S., 2017. Leaching of Copper, Lead and Zinc from Municipal Solid Waste Incineration Bottom Ash. Energy Procedia 113, 442-449.

Sevigné-Itoiz, E., Gasol, C.M., Rieradevall, J., Gabarrell, X., 2014. Environmental consequences of recycling aluminum old scrap in a global market. Resources, Conservation and Recycling 89, 94-103.

Singer, D.A., 2017. Future copper resources. Ore Geology Reviews 86, 271-279.

Song, C.H., 2005. Whole life and highgrade quality-stick to the implement housing performance certification. Housing Science 290(8), 287-302.

Song, Q., Li, J., Zeng, X., 2015. Minimizing the increasing solid waste through zero waste strategy. Journal of Cleaner Production 104, 199-210.

Song, Q., Wang, Z., Li, J., Zeng, X., 2013. The life cycle assessment of an e-waste treatment enterprise in China. Journal of Material Cycles and Waste Management 15(4), 469-475.

Song, X., Yang, J., Lu, B., Li, B., Zeng, G., 2014. Identification and assessment of environmental burdens of Chinese copper production from a life cycle perspective. Frontiers of Environmental Science & Engineering 8(4), 580-588.

Soulier, M., Gloser-Chahoud, S., Goldmann, D., Tercero Espinoza, L.A., 2018a. Dynamic analysis of European copper flows. Resources Conservation and Recycling 129, 143-152.

Soulier, M., Pfaff, M., Goldmann, D., Walz, R., Geng, Y., Zhang, L., Tercero Espinoza, L.A., 2018b. The Chinese copper cycle: Tracing copper through the economy with dynamic substance flow and input-output analysis. Journal of Cleaner Production 195, 435-447.

Spatari, S., Bertram, M., Fuse, K., Graedel, T.E., Rechberger, H., 2002. The contemporary European copper cycle: 1 year stocks and flows. Ecological Economics 42(1-2), 27-42.

Spatari, S., Bertram, M., Gordon, R.B., Henderson, K., Graedel, T.E., 2005. Twentieth century copper stocks and flows in North America: A dynamic analysis. Ecological Economics 54(1), 37-51.

State Council, 2001. Administrative Measures for the Recycling of Scrapping Automobiles. State Council of the People's Republic of China

State Council, 2013. General Office of the State Council of the People's Republic of China, ban trash imports and reform the solid waste import management system. State Council of the People's Republic of China

State Council, 2018. Work Plan for the Pilot Program of "Zero-Waste City" Building. State Council of the People's Republic of China

State Council, 2019. Measures for the Management of End-of-Life Vehicle Recycling. State Council of the People's Republic of China.

Steuer, B., Ramusch, R., Salhofer, S., 2018. Is There a Future for the Informal Recycling Sector in Urban China? Detritus(4), 189.

Sun, X., Hao, H., Zhao, F., Liu, Z., 2017. Tracing global lithium flow: A trade-linked material flow analysis. Resources, Conservation and Recycling 124, 50-61.

Šyc, M., Simon, F.G., Hykš, J., Braga, R., Biganzoli, L., Costa, G., Funari, V., Grosso, M., 2020. Metal recovery from incineration bottom ash: State-of-the-art and recent developments. Journal of Hazardous Materials 393, 122433.

Tan, R.B., Khoo, H.H., 2005. An LCA study of a primary aluminum supply chain. Journal of Cleaner Production 13(6), 607-618.

Tanimoto, A.H., Durany, X.G., Villalba, G., Pires, A.C., 2010. Material flow accounting of the copper cycle in Brazil. Resources Conservation and Recycling 55(1), 20-28.

Tatsumi, K., Daigo, I., Matsuno, Y., Adachi, Y., 2008. Analysis on Recycling Potential of Copper in Japan. Journal of the Japan Institute of Metals 72(8), 617-624.

Terakado, R., Ichino, T.K., Daigo, I., Matsuno, Y., Adachi, Y., 2009a. Estimation of In-Use Stock of Copper in China, Korea and Taiwan. Journal of the Japan Institute of Metals 73(11), 833-838.

Terakado, R., Takahashi, K.I., Daigo, I., Matsuno, Y., Adachi, Y., 2009b. In-use stock of copper in Japan estimated by bottom-up approach. Journal of the Japan Institute of Metals 73(9), 713-719.

Tercero Espinoza, L.A., Soulier, M., 2016. An examination of copper contained in international trade flows. Mineral Economics 29(2), 47-56.

Troschinetz, A.M., Mihelcic, J.R., 2009. Sustainable recycling of municipal solid waste in developing countries. Waste Management 29(2), 915-923.

Tukker, A., 2000. Life cycle assessment as a tool in environmental impact assessment. Environmental Impact Assessment Review 20(4), 435-456.

United Nations, D.o.E.a.S.A., Population Division, 2017. World Population Prospects: The 2017 Revision. DVD Edition.

United Nations, D.o.E.a.S.A., Population Division, 2018. World Urbanization Prospects: The 2018 Revision. Online Edition.

USGS, 2021. Copper Statistics and Information. https://www.usgs.gov/centers/nmic/copper-statistics-and-information. (Accessed 23 March 2021).

Van der Voet, E., Van Oers, L., Moll, S., Schütz, H., Bringezu, S., De Bruyn, S., Sevenster, M., Warringa, G., 2005. Policy Review on Decoupling: Development of indicators to assess decoupling of economic development and environmental pressure in the EU-25 and AC-3 countries. EU

Commission, DG Environment, Brussels.

Van der Voet, E., Van Oers, L., Verboon, M., Kuipers, K., 2018. Environmental implications of future demand scenarios for metals: methodology and application to the case of seven major metals. Journal of Industrial Ecology.

Van Genderen, E., Wildnauer, M., Santero, N., Sidi, N., 2016. A global life cycle assessment for primary zinc production. The International Journal of Life Cycle Assessment 21(11), 1580-1593.

Vanegas, P., Peeters, J.R., Cattrysse, D., Tecchio, P., Ardente, F., Mathieux, F., Dewulf, W., Duflou, J.R., 2018. Ease of disassembly of products to support circular economy strategies. Resources, Conservation and Recycling 135, 323-334.

Venkatesh, G., Hammervold, J., Brattebø, H., 2009. Combined MFA-LCA for Analysis of Wastewater Pipeline Networks. Journal of Industrial Ecology 13(4), 532-550.

Vergara, S.E., Damgaard, A., Gomez, D., 2016. The Efficiency of Informality: Quantifying Greenhouse Gas Reductions from Informal Recycling in Bogotá, Colombia. Journal of Industrial Ecology 20(1), 107-119.

Verhagen, T.J., van der Voet, E., Sprecher, B., 2021. Alternatives for natural-gas-based heating systems: A quantitative GIS-based analysis of climate impacts and financial feasibility. Journal of Industrial Ecology 25(1), 219-232.

Vieira, M.D., Goedkoop, M.J., Storm, P., Huijbregts, M.A., 2012. Ore grade decrease as life cycle impact indicator for metal scarcity: the case of copper. Environmental science & technology 46(23), 12772-12778.

Villares, M., Işıldar, A., van der Giesen, C., Guinée, J., 2017. Does ex ante application enhance the usefulness of LCA? A case study on an emerging technology for metal recovery from e-waste. The International Journal of Life Cycle Assessment 22(10), 1618-1633.

Virtanen, M., Manskinen, K., Uusitalo, V., Syvänne, J., Cura, K., 2019. Regional material flow tools to promote circular economy. Journal of Cleaner Production 235, 1020-1025.

Wäger, P.A., Hischier, R., Eugster, M., 2011. Environmental impacts of the Swiss collection and recovery systems for Waste Electrical and Electronic Equipment (WEEE): A follow-up. Science of The Total Environment 409(10), 1746-1756.

Wang, b., 2012. Research on Shenzhen Construction Waste Treatment Mode

Based on Life Cycle Assessment, Huazhong University of Science and Technology.

Wang, H., Schandl, H., Wang, G., Ma, L., Wang, Y., 2019. Regional material flow accounts for China: Examining China's natural resource use at the provincial and national level. Journal of Industrial Ecology 23(6), 1425-1438.

Wang, H.T., Liu, Y., Gong, X.Z., Wang, Z.H., Gao, F., Nie, Z.R., 2015. Life cycle assessment of metallic copper produced by the pyrometallurgical technology of China, Materials Science Forum. Trans Tech Publ, pp. 559-563.

Wang, J., Ju, Y., Wang, M., Li, X., 2019. Scenario analysis of the recycled copper supply in China considering the recycling efficiency rate and waste import regulations. Resources, Conservation and Recycling 146, 580-589.

Wang, M., Chen, W., Zhou, Y., Li, X., 2017. Assessment of potential copper scrap in China and policy recommendation. Resources Policy 52, 235-244.

Wang, T., Zhang, Y., Yu, H., Wang, F., 2012. Advanced manufacturing technology in China: a roadmap to 2050. Springer.

Watari, T., Nansai, K., Giurco, D., Nakajima, K., McLellan, B., Helbig, C., 2020. Global Metal Use Targets in Line with Climate Goals. Environmental Science & Technology 54(19), 12476-12483.

Weber, R., Kaplan, S., Sokol, H., 2009. Market Analysis of Construction and Demolition Material Reuse in the Chicago Region. Chicago: University of Illinois (commissioned by the Delta Institute). Market Analysis of Used Building Materials in Metropolitan Vancouver 67.

Wei, W., Chen, D., Hu, D., 2016. Study on the evolvement of technology development and energy efficiency—A case study of the past 30 years of development in Shanghai. Sustainability 8(5), 457.

Weng, Z., Haque, N., Mudd, G.M., Jowitt, S.M., 2016. Assessing the energy requirements and global warming potential of the production of rare earth elements. Journal of cleaner production 139, 1282-1297.

Wiechmann, E.P., Morales, A.S., Aqueveque, P., 2010. Improving productivity and energy efficiency in copper electrowinning plants. IEEE Transactions on Industry Applications 46(4), 1264-1270.

Wiedenhofer, D., Steinberger, J.K., Eisenmenger, N., Haas, W., 2015. Maintenance and expansion: modeling material stocks and flows for residential buildings and transportation networks in the EU25. Journal of industrial ecology 19(4), 538-551.

World Bank, 2019. Global GDP, in: World Bank national accounts data, a.O.N.A.d.f. (Ed.). https://data.worldbank.org/indicator/NY.GDP.MKTP.CD.

Worldometer, 2020. World Population

Wormser, F.E., 1921. The Importance of Foreign Trade in Copper and Other Metals. The ANNALS of the American Academy of Political and Social Science 94(1), 65-76.

Wu, Z., Yuan, W., Li, J., Wang, X., Liu, L., Wang, J., 2017. A critical review on the recycling of copper and precious metals from waste printed circuit boards using hydrometallurgy. Frontiers of Environmental Science & Engineering 11(5), 8.

Xiong, J., Zhu, J., He, Y., Ren, S., Huang, W., Lu, F., 2020. The application of life cycle assessment for the optimization of pipe materials of building water supply and drainage system. Sustainable Cities and Society 60, 102267.

Xu, B.-S., 2013. Progress of remanufacturing engineering and future technology expectation. Advances in Manufacturing 1(1), 8-12.

Yan, L., Wang, A., Chen, Q., Li, J., 2013. Dynamic material flow analysis of zinc resources in China. Resources Conservation & Recycling 75(2), 23-31.

Yang, J., Li, X., Liu, Q., 2017. China's Copper Demand Forecasting Based on System Dynamics Model: 2016-2030.

Yang, X., Hu, M., Heeren, N., Zhang, C., Verhagen, T., Tukker, A., Steubing, B., 2020. A combined GIS-archetype approach to model residential space heating energy: A case study for the Netherlands including validation. Applied Energy 280, 115953.

Yoshimura, A., Matsuno, Y., 2018. Dynamic Material Flow Analysis and Forecast of Copper in Global-Scale: Considering the Difference of Recovery Potential between Copper and Copper Alloy. Materials Transactions 59(6), 989-998.

Yu, X., Lu, B., Wang, R., 2018. Analysis of low carbon pilot industrial parks in China: Classification and case study. Journal of Cleaner Production 187, 763-769.

Yue, Q., Lu, Z.W., Zhi, S.K., 2009. Copper cycle in China and its entropy analysis. Resources, Conservation and Recycling 53(12), 680-687.

Zalmon, I., Carvalho, G., Ferreira, C.A., 1998. Regional population projections for China, Higher Education in Europe. pp. 351-356.

Zamagni, A., Pesonen, H.-L., Swarr, T., 2013. From LCA to Life Cycle

Sustainability Assessment: concept, practice and future directions. The international journal of life cycle assessment 18(9), 1637-1641.

Zeltner, C., Bader, H.P., Scheidegger, R., Baccini, P., 1999. Sustainable metal management exemplified by copper in the USA. Regional Environmental Change 1(1), 31-46.

Zhang, L., Cai, Z., Yang, J., Yuan, Z., Chen, Y., 2015a. The future of copper in China—A perspective based on analysis of copper flows and stocks. Science of the Total Environment 536, 142-149.

Zhang, L., Chen, T.M., Yang, J.M., Cai, Z.J., Sheng, H., Yuan, Z.W., Wu, H.J., 2017. Characterizing copper flows in international trade of China, 1975-2015. Science of the Total Environment 601, 1238-1246.

Zhang, L., Yang, J., Cai, Z., Yuan, Z., 2014. Analysis of copper flows in China from 1975 to 2010. Science of The Total Environment 478, 80-89.

Zhang, L., Yang, J., Cai, Z., Yuan, Z., 2015b. Understanding the spatial and temporal patterns of copper in-use stocks in China. Environmental science & technology 49(11), 6430-6437.

Zhang, Q., Wu, Y., Wang, W., Zhang, Y., Cheng, H., Zuo, T., 2011. Method for preparing high-purity copper oxide superfine powder from waste printed circuit boards, in: Technology, B.U.o. (Ed.). China.

Zhang, S., Ding, Y., Liu, B., Pan, D.a., Chang, C.c., Volinsky, A.A., 2015. Challenges in legislation, recycling system and technical system of waste electrical and electronic equipment in China. Waste Management 45, 361-373.

Zhang, T., He, Y., Wang, F., Ge, L., Zhu, X., Li, H., 2014. Chemical and process mineralogical characterizations of spent lithium-ion batteries: an approach by multi-analytical techniques. Waste management 34(6), 1051-1058.

Zhang, X., Zhang, M., Zhang, H., Jiang, Z., Liu, C., Cai, W., 2020. A review on energy, environment and economic assessment in remanufacturing based on life cycle assessment method. Journal of Cleaner Production 255, 120160.

Zhang, Y., Sun, M., Hong, J., Han, X., He, J., Shi, W., Li, X., 2016. Environmental footprint of aluminum production in China. Journal of Cleaner Production 133, 1242-1251.

Zhao, W., Rotter, S., 2008. The current situation of construction & demolition waste management in China, 2008 2nd International Conference on Bioinformatics and Biomedical Engineering. IEEE, pp. 4747-4750.

Zhu, J., Fan, C., Shi, H., Shi, L., 2019. Efforts for a Circular Economy in China: A Comprehensive Review of Policies. Journal of Industrial Ecology 23(1), 110-118.

Zink, T., Geyer, R., 2017. Circular Economy Rebound. Journal of Industrial Ecology 21(3), 593-602.

Zongguo, W., Xiaoli, J., 2013. Copper resource trends and use reduction measures in China. Journal of Tsinghua University, 09.

Summary

In-use stocks of products can be considered as intermediaries between human needs and the physical world. These stocks are regarded as an important indicator for society's metabolism. During use, they fulfil important functions, but they can also be seen as a source of materials for the future: when the products are discarded, the materials embedded in them become available to be used again: the production of secondary materials. Stocks of materials are therefore increasingly the subject of investigation.

This idea of an urban mine can be applied to all kinds of materials and resources, and applies at all scale levels, from the local to the global level. In this dissertation, the urban mine of copper in China is the subject. Consumption of diverse copper-containing products in China is increasing rapidly. The in-use copper stocks have become a large reservoir for urban mining. The aim of this research is therefore to explore how the stocks and flows related to the Chinese copper cycle can be transformed into a sustainable and circular economy.

The research was performed with four complementary and successive research questions:

1. How are copper demand, in-use stocks and waste generation expected to develop under the current Chinese policies related to development, energy transition and the circular economy? (Chapter 2)
2. How could China meet its future copper demand in the context of moving towards a circular economy, and how may this be affected by the import restrictions on copper scrap? (Chapter 3)
3. What are the environmental benefits and drawbacks related to future copper production in China, and how could the country's environmental performance be improved in the future? (Chapters 4 and 5)
4. What is the potential to close the copper cycle in China? (Chapters 2, 3 and 5)

The starting point of this thesis is the material resources use and the concurrent challenges for resource availability, waste generation and management and environmental impacts of material production. These challenges make it clear that there is a need for resource efficiency and transition to a circular economy for the copper cycle in China. A logical point of departure from this view was therefore to quantify the in-use stocks, as well as demand and waste generation for copper under the current Chinese policies (question 1). The findings indicate that copper demand in 2050 under the current Chinese policies related to development, energy transition and the circular economy cannot be met through the supply of secondary copper. More diverse circular economy strategies need to be employed to develop the copper cycle sustainably in China, especially given the expected restriction on the import of copper scrap (question 2). In the move towards sustainable development, the environmental performance of copper cycle is a crucial dimension. An LCA approach was used and up-scaled with several prospective changes related to copper production to understand what the environmental impacts are related to future copper production in China (question 3). Finally, combining the different scenarios assessed in this thesis, an overview was given of the potential to close the copper cycle in China (research question 4).

Chapter 2 presents the in-use stocks, as well as the demand for and waste generation of copper under the current Chinese policies. The main conclusion is that the in-use stocks and demand for copper in China are expected to increase significantly, albeit with different growth rates for different copper categories. The copper stocks for some copper-containing products like buildings are likely to have stabilized by 2050. For infrastructure and transportation, saturation will likely not have occurred yet by that time. Specifically, the copper stocks and demands of electricity infrastructure and new energy vehicles are expected to increase considerably up to 2050 as a result of the ongoing energy transition. Domestic waste generation has been increasing in recent years; however, most of China's copper products have not entered their retirement period at present, resulting in a growth of EoL copper recycled in China that is much slower than that of copper demand. For that reason, it is unlikely that copper demand in 2050 under the current Chinese policies could be met through the supply of secondary copper.

Chapter 3 answers question 2 and explores the potential of a number of more stringent circular economy strategies to close the Chinese copper cycle. A special point of attention is the effects of China's import restrictions on copper scrap. The main findings are that the Chinese copper cycle could benefit

considerably from these circular economy strategies, but that there still may be a substantial gap between the Chinese demand for copper and the amount of scrap available domestically. Extending the lifetimes of copper-containing products could lead to a reduction in the overall copper demand. The restriction or ban on imported copper waste will reduce the availability of scrap to be used for secondary copper production. Even with a growing amount of domestic scrap and assuming a high copper recycling rate, the share of secondary copper supply may not be more than 60% even in the more distant future. The substantial gap between Chinese copper demand and the amount of scrap available domestically needs to be closed, either by means of primary copper in the form of domestic mining, or through imports of concentrates and refined copper in the future. An alternative scenario could be to consider lifting the ban on import of copper scrap. In combination with the establishment of a state-of-the-art, efficient and environmentally friendly recycling industry, this could be an opportunity for China to transition to a more circular economy with regard to copper.

Chapter 4 assesses the future environmental impacts of pyrometallurgical, hydrometallurgical and secondary copper production in China, under influence of declining copper ore grade, energy efficiency improvement of production processes and energy transition on electricity supply. Potential options to improve the environmental performance of copper production are also identified in this chapter. Environmental impacts related to the production of 1 kg of copper could go down considerably. Nevertheless, the total environmental impact of copper production with these prospective changes will not be lower than that of the business-as-usual scenario. The energy transition by shifting from fossil fuels to more renewable energy will reduce CO2 emissions, but at the same time will increase copper demand compared to the business-as-usual scenario. The results also support the idea that the environmental impact of the production of 1 kg secondary copper is and will remain much lower than that of primary copper, which suggests that increasing the share of secondary copper production is the most environmental friendly option.

Chapter 5 takes a closer look at copper waste management. Circular economy and "Zero waste" strategies for the copper waste management system are investigated, and the potential to close the copper cycle based on an optimized waste management system is explored in this chapter. In combination with the environmental impact of primary copper production in Chapter 4, the main conclusion is that under the present Chinese policies, the reuse and recycling of copper-containing products will lead to somewhat lower GHG emissions and reduced energy demand for total copper production. Maximizing such circular economy strategies may lead to a further reduction, but if applied too stringently can also be counter-productive. 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 findings indicate that the copper waste management system should be improved to keep the utility and value of copper at the highest level. This requires actions across the full product lifecycle, including waste prevention and circular economy strategies for the EoL products as well as early-stage product design. In this way, the dematerialization and environmental sustainability of the copper cycle could be realized simultaneously in China.

Chapter 6 concludes that this thesis has contributed to the exploration on how to move toward a sustainable and circular economy of copper cycle in China, with respond to the challenges for the close cycle and environmental dimension. Several recommendations are provided to improve the existing analysis of copper cycle. The first one is to improve uncertainties with regard to data and model, such as obtaining more information on the changes of copper content over time. Second, the rebound effects and side effects should be considered in future research. The final one is the need to consider other models to comprehensively estimate the copper stocks and flows. Further step in this direction is to integrate the MFA and LCA with other models to include environmental, economic and social dimensions, such as economic models and spatial models. This, by the way, could realize the dynamic modeling of copper cycle and provide more reliable and comprehensive policy suggestions to move toward a sustainable development of copper cycle in the future.

Samenvatting

Voorraden van producten die in gebruik zijn kunnen worden opgevat als een schakel tussen menselijke behoeften enerzijds, en de fysieke wereld van grondstoffen en materialen anderzijds. Ze worden beschouwd als een belangrijke indicator in onderzoek naar het metabolisme van de maatschappij. Deze voorraden vervullen een functie tijdens gebruik, maar kunnen ook worden beschouwd als een toekomstige bron van materialen: wanneer de producten afgedankt worden, komen de materialen die erin verwerkt zijn weer beschikbaar voor hergebruik, de productie van secondair materiaal. Zo kunnen de voorraden van producten in gebruik worden beschouwd als een *urban mine*, een stedelijke mijn. Deze voorraden zijn dan ook steeds vaker onderwerp van onderzoek.

Het idee van de stedelijke mijn kan op allerlei verschillende materalen en grondstoffen worden toegepast, en ook op allerlei ruimtelijke schaalniveaus, van lokaal tot wereldwijd. In deze dissertatie is de stedelijke mijn van koper in China het onderwerp. Het gebruik van diverse koperhoudende producten in China neemt snel toe. De voorraden van koper in gebruik groeien daarom in hoog tempo, en vormen een groot reservoir waaruit geput kan worden. Met dit onderzoek wordt daarom beoogd na te gaan hoe de koperkringloop in China kan worden omgevormd tot een duurzame en circulaire economie.

Het onderzoek is gebaseerd op vier elkaar aanvullende en opeenvolgende onderzoeksvragen:

1. Hoe zullen de vraag naar koper, voorraden in gebruik en afvalproductie zich naar verwachting ontwikkelen onder het huidige Chinese beleid met betrekking tot de algemene economische ontwikkeling, de energietransitie en de ambities ten aanzien van de circulaire economie? (Hoofdstuk 2)
2. Hoe kan China voldoen aan de toekomstige vraag naar koper tegen de achtergrond van de overgang naar een circulaire economie, en welke invloed kunnen de invoerbeperkingen voor koperschroot daarop hebben? (Hoofdstuk 3)
3. Welke milieu-effecten gaan gepaard met de huidige en toekomstige koperproductie in China, en hoe kunnen de milieuprestaties in de

toekomst worden verbeterd? (Hoofdstukken 4 en 5)

4. In hoeverre kan de koperkringloop in China worden gesloten? (Hoofdstukken 2, 3 en 5)

Het proefschrift begint met het in kaart brengen van het gebruik van koper in China en de problemen die dit met zich meebrengt, waaronder verminderde beschikbaarheid van grondstoffen, productie van afval, en milieueffecten van materiaalproductie. Om deze problemen beheersbaar te houden, is een efficiënter gebruik van grondstoffen en met name een overgang naar een circulaire economie gewenst. Vanuit deze optiek is het dan ook een logisch vertrekpunt om zowel de voorraden in gebruik als de vraag naar en afvalproductie van koper te kwantificeren, en een inschatting te maken van hoe deze zich ontwikkelen onder het huidige Chinese beleid (onderzoeksvraag 1). Het lijkt erop dat het bij het huidige Chinese beleid in 2050 niet mogelijk is de koperkringloop te sluiten. Dit is het gevolg van de sterk toenemende vraag naar koper, die het gevolg is van economische ontwikkeling en van de transitie naar een duurzaam energiesysteem. Voor een duurzame ontwikkeling van de Chinese koperkringloop moeten daarom meer circulaire economie strategieën worden toegepast. Een complicerende factor is de te verwachten beperking van de invoer van koperschroot (onderzoeksvraag 2) in China: deze is ingesteld om dumping van afval te voorkomen, maar zorgt er tegelijkertijd voor dat een belangrijke bron voor secondaire productie niet benut kan worden.

De milieuprestaties van de kopercyclus zijn een cruciale factor op weg naar een duurzame ontwikkeling. Met behulp van levenscyclusanalyse (LCA), opgeschaald met verschillende verwachte veranderingen in de koperproductieprocessen, zijn de milieu-effecten van de toekomstige koperproductie in China in kaart gebracht (onderzoeksvraag 3). Ten slotte zijn de verschillende scenario's die in dit proefschrift zijn ontwikkeld gecombineerd voor de beantwoording van onderzoeksvraag 4, de algemene vraag naar de mogelijkheid om de koperkringloop in China te sluiten om te vormen tot een circulaire en duurzame economie.

Hoofdstuk 2 beschrijft de voorraden in gebruik alsmede de vraag naar en afvalproductie van koper onder het huidige Chinese beleid. De belangrijkste conclusie is dat de voorraden in gebruik en de vraag naar koper in China naar verwachting aanzienlijk zullen toenemen, zij het met verschillende groeicijfers voor verschillende kopercategorieën. De kopervoorraden in sommige koperhoudende producten, zoals gebouwen, zullen tegen 2050 waarschijnlijk gestabiliseerd zijn, terwijl het verzadigingspunt voor infrastructuur en vervoer nog niet bereikt zal zijn. Met name de kopervoorraden en de vraag naar koper bij de elektriciteitsinfrastructuur en elektrische voertuigen zullen naar verwachting tot 2050 aanzienlijk toenemen. Hoewel de binnenlandse afvalproductie de afgelopen jaren is toegenomen, zijn de meeste koperproducten in China nog niet aan het einde van hun levensduur. Er komt dus nog maar weinig koper in het afval, waardoor ook de end-of-life-koper recycling in China maar langzaam toeneemt. Al met al ziet het er naar uit dat bij het huidige Chinese beleid, waarin economische ontwikkeling, energietransitie en circulaire economie alle een plaats hebben, er in 2050 onvoldoende secondair koper geproduceerd kan worden om in de vraag naar koper te voorzien.

Hoofdstuk 3 behandelt onderzoeksvraag 2. In dit hoofdstuk worden de mogelijkheden verkend om met strategieën van een meer circulaire economie toe te werken naar een circulaire economie van de koperkringloop, met name onder invloed van de invoerbeperkingen op koperschroot. De belangrijkste bevindingen zijn dat de Chinese kopercyclus zeer gebaat zou zijn met deze strategieën van een circulaire economie. Tegelijkertijd blijft het een uitdaging om de aanzienlijke kloof tussen de Chinese vraag naar koper en de in eigen land beschikbare hoeveelheid schroot te overbruggen. Verlenging van de levensduur van koperhoudende producten kan leiden tot een afname van de vraag naar koper. De beperking van of het verbod op de invoer van koperafval zal de beschikbaarheid van grondstoffen voor de productie van secondair koper verminderen. Hierdoor zal het aandeel van het secondaire koperaanbod toenemen tot slechts 60% in 2100, zelfs als in het circulaire-economiescenario de beschikbaarheid van binnenlands schroot toeneemt en er veel koper gerecycled wordt. De aanzienlijke kloof tussen de Chinese vraag naar koper en de in eigen land beschikbare hoeveelheid schroot moet in de toekomst worden overbrugd door primair koper te winnen in binnenlandse mijnen of door concentraten en geraffineerd koper te importeren. Een alternatief kan zijn om het importverbod op schroot op termijn op te heffen, en koperschroot te blijven importeren. In combinatie met het opzetten van een ultramoderne, efficiënte en milieuvriendelijke recyclingindustrie zou dit voor China een kans kunnen zijn om over te schakelen op een meer circulaire economie, waarin een groter deel van de vraag kan worden voldaan met secondair koper.

Hoofdstuk 4 gaat over de toekomstige milieueffecten van koperproductie. Binnen de drie productieroutes - pyrometallurgische, hydrometallurgische en secondaire koperproductie - worden veranderingen verwacht als gevolg van van afnemende ertsgraden (het kopergehalte van kopererts), de verbetering van de energie-efficiëntie van de productieprocessen en de energietransitie in de elektriciteitsvoorziening in China. Ook andere opties om de milieuprestaties van de koperproductie te verbeteren worden besproken in Hoofdstuk 4. Naar verwachting zullen de milieueffecten van de productie van 1 kg koper als gevolg van deze veranderingen veel kleiner zijn dan in het *business-as-usual*-scenario. Dat geldt echter niet voor de totale milieueffecten van de koperproductie Dit komt doordat als gevolg van de energietransitie de vraag naar koper zal toenemen in vergelijking met het *business-as-usual*-scenario. Een duurzaam energiesysteem bevat veel meer koper dan een op fossiele brandstoffen gebaseerd systeem. De resultaten bevestigen ook dat de milieueffecten van de productie van 1 kg secondair koper veel geringer zijn dan die van primair koper, zodat een hoger aandeel van de secondaire koperproductie de meest milieuvriendelijke optie lijkt.

In hoofdstuk 5 wordt verder ingegaan op milieuvriendelijke koperproductie in een meer circulaire economie en met *zero waste*-strategieën in het koper afvalbeheer. Ook bevat dit hoofdstuk een verkenning van de mogelijkheden om de koperkringloop te sluiten vanuit een geoptimaliseerd afvalbeheersysteem. De belangrijkste conclusie is dat bij het huidige Chinese beleid hergebruik en recycling van koperhoudende producten in combinatie met de milieueffecten van de primaire koperproductie zoals besproken in hoofdstuk 4 zullen leiden tot een kleine reductie in zowel de uitstoot van broeikasgassen als de energiebehoefte van de totale koperproductie. Maximale toepassing van dergelijke strategieën van een circulaire economie kan tot een verdere reductie leiden, maar een te strikte toepassing kan ook contraproductief zijn. De broeikasgasemissies die samenhangen met de secondaire koperproductie kunnen groter worden dan die van de primaire koperproductie, ondanks de lagere broeikasgasemissies per kg van de secondaire productie. De bevindingen wijzen erop dat het koper afvalbeheersysteem moet worden verbeterd om koper zijn hoogste nut en waarde te laten behouden. Dit vergt maatregelen gedurende de gehele levenscyclus van het product, waaronder zowel afvalpreventie en strategieën voor een circulaire economie bij end-of-life-producten, als productontwerp in de beginfase. Op die manier kunnen de dematerialisatie en de ecologische duurzaamheid van de kopercyclus in China tegelijkertijd worden gerealiseerd.

In hoofdstuk 6 wordt geconcludeerd dat dit proefschrift heeft bijgedragen aan de verkenning van de weg naar een duurzame en circulaire economie van de koperkringloop in China vanuit oogpunt van het sluiten van de kringloop en het verminderen van milieu-effecten. Er worden verschillende aanbevelingen gedaan voor nader onderzoek naar de koperkringloop. Een eerste aanbeveling betreft het reduceren van onzekerheden in data en model, bijvoorbeeld door meer informatie over de veranderingen van het kopergehalte van koperhoudende producten door de tijd heen te verzamelen. Ten tweede zou onderzoek naar neveneffecten en rebound-effecten een waardevolle aanvulling vormen. Ten slotte zou het zinvol zijn andere modellen toe te passen om een bredere analyse te maken van –het kopersysteem. Een volgende stap in deze richting is het integreren van materiaalstroomanalyse (MFA) en levenscyclusanalyse (LCA) met andere modellen, zoals economische en ruimtelijke modellen, om milieu-, economische en sociale dimensies op te nemen. Met deze aanbevelingen zou een completere dynamische modellering van de kopercyclus kunnen worden gerealiseerd. Dit zou betrouwbaardere en volledigere beleidsaanbevelingen kunnen opleveren om te komen tot een duurzame ontwikkeling van de kopercyclus.

Acknowledgements

Throughout the writing of this thesis I have received a great deal of support and assistance.

First and foremost, I wish to express my deep gratitude and appreciation to my promotores Dr. Ester van der Voet and Prof. dr. Arnold Tukker for their professional and valuable guidance, support and encouragement of my PhD study and personal development. Dr. Ester van der Voet drew my attention to material flow analysis and life cycle assessment and broadened my perspective on the different aspects of sustainable development, nationally and globally. Thank you for all the opportunities I was given to further my research. My other promoter, Prof. dr. Arnold Tukker, has always been a cheerful motivator, who taught and supported me to manage projects. In daily life, he also helped me learning about Dutch culture. I still remember the first traditional Dutch food he recommended-Boerenkool.

Many thanks to China Scholarship Council (CSC) and EIT Raw materials for the financial support of my PhD study. Special thanks to the coordinator of the IDS-FunMat-Inno project, Prof. Thierry Toupance from Université de Bordeaux, and all the project members for their inspiration and help throughout the project.

I would like to acknowledge to the collaborators Luis Tercero Espinoza, Antonia Loibl and Matthias Pfaff, Helmut Rechberger and Huajiao Li for their important contributions to our research and the supportive arrangement of my visit. I also want to thank the colleagues Lauran van Oers, Bernhard Steubing and Glenn A. Aguilar Hernandez from CML, for their support and many insightful discussions about the research, which not only guided me through the completion of my PhD research but also motivated me to think deeper about the meaning of Industrial Ecology.

I would like to express my sincere gratitude to my thesis reading committee: Prof. dr. ir. P.M. van Bodegom, Prof. dr. ing. Martina G. Vijver, Prof. Dr. H. Rechberger, Dr. Ing. L. Tercero Espinoza, Dr. E.G.M. Kleijn and Dr. T. Fishman for evaluating this thesis and providing valuable comments.

My gratitude extends to the CML Support and Scientific Staff for organizing different kinds of meetings and workshops, they are always helpful and considerate. Thanks to my officemates Franco Donati, Sebastiaan Deetman, Pim Wassenaar and Olivier Burggraaff in office A320 for nice talks and joyful company throughout the years. I would like to extend sincere thanks to my Chinese colleagues for their help and company, I really enjoyed the parties and the hanging out together. Special thanks to my LADIES Feibo Duan, Weilin Huang, Chen Tang, Juan Wu, Qi Yu, Jianhong Zhou and some great friends in China Zhi Qin, Zhiqiang Yu and Lina Zhu for cherished time spent together, the long evening chats about life, and the opportunity to vent my frustration. I would like to thank all members in my Study Group and Sports Group for the great moments we have shared during the special situation (COVID-19) and happy distractions to distract my mind from my research.

I dedicate this work to my family, who always encourage me to explore new pathways in life and who support me with unconditional love. I could not have finished this PhD without their support. 我爱你们！ ❤

Curriculum Vitae

Di Dong was born on June 30th, 1991 in Jining, Shandong Province, China. She graduated from No.2 High School in 2010, and started her Bachelor study in Engineering Management in Zhejiang University of Finance and Economics from 2010 to 2014. She was one of the "Outstanding Graduates" in 2014. She continued her Master study in Management Science and Engineering in China University of Geosciences (Beijing) from 2014 to 2017. During this period, she was supervised by Prof. Haizhong An and Prof. Wei Fang. Her Master thesis was focused on "Embodied carbon in international copper trade from an industry chain perspective" and was awarded one of the "Outstanding Master Thesis".

After her Master study, she was granted a scholarship by the Chinese Scholarship Council for a PhD project, and started her PhD study in the Institute of Environmental Sciences in Leiden University in 2017 under the supervision of Dr. Ester van der Voet and Prof. dr. Arnold Tukker. Her PhD research was focused on the sustainable and circular metals economy, specifically on the case of copper in China. The copper model designed in her research has been used by the International Copper Association (ICA) to do further research.

She participated in the EIT-labelled International Doctoral School Programme- the IDS-FunMat-Inno project during PhD study. She visited the Fraunhofer Institute for Systems and Innovation Research ISI (Karlsruhe, Germany) in 2018 and the Vienna University of Technology (Vienna, Austria) in 2019, where she focused on modelling of copper waste management and supply.

In addition, she presented her research findings in the International Society for Industrial Ecology (ISIE) 6th Asia-Pacific (AP) conference, the ISIE conference in 2019 and the 5th international conference on Final Sinks. During the PhD study, she also assisted the course of "Resilient Cities-Minor Sustainable Development".

List of Publications

1. Dong, D., Tercero Espinoza, Luis A. Loibl, A., Pfaff, M., Tukker, A., & van der Voet, E. (2020). Scenarios for anthropogenic copper demand and supply in China: implications of a scrap import ban and a circular economy transition. Resources, Conservation and Recycling, 161, 104943.
2. Dong, D., van Oers, L., Tukker, A., & van der Voet, E. (2020). Assessing the future environmental impacts of copper production in China: Implications of the energy transition. Journal of Cleaner Production, 274, 122825.
3. Dong, D., Tukker, A., & van der Voet, E. (2019). Modeling copper demand in China up to 2050: A business-as-usual scenario based on dynamic stock and flow analysis. Journal of Industrial Ecology, 23(6), 1363-1380.
4. Dong, D., Tukker, A., Steubing, B., van Oers, L., Rechberger, H., Aguilar, Hernandez G., Li, H., van der Voet, E., Towards "Zero waste" management of copper in China: dematerialization and environmental impact minimization. Environmental Science & Technology, under review.

辛丑仲夏董迪著