The potential for material circularity and independence in the U.S. steel sector

Daniel R. Cooper1 | Nicole A. Ryan1,2 | Kyle Syndergaard1 | Yongxian Zhu1

1George G. Brown Laboratory, Mechanical Engineering Department, University of Michigan, Ann Arbor, Michigan
2School for Environment and Sustainability, University of Michigan, Ann Arbor, Michigan

Correspondence
Daniel R. Cooper, George G. Brown Laboratory, Mechanical Engineering Department, University of Michigan, 2350 Hayward Street, Ann Arbor, MI 48109.
Email: drcooper@umich.edu

Funding information:
Kyle Syndergaard was partially funded by a gift from Ford Motor Company and Yongxian Zhu was partially funded by the Michigan-Cambridge Research Initiative.

Editor Managing Review: Ester van der Voet

Abstract
Achieving a U.S. circular economy would reduce environmental impacts and increase material independence. This article calculates maximum recycled contents (RCs) and recycling rates (RRs) in an independent U.S. steel sector, and estimates the potential to displace current imports with recycled scrap that is currently destined for landfill, hibernating stocks, or export (LHSE). A U.S. dynamic material flow analysis (1880–2100) is conducted to estimate annual steel consumption and scrap generation. The results are coupled with a linear optimization model that minimizes primary steel demand while satisfying the volumetric and compositional demands of new consumption. The compositional analysis examines only copper content because it is of greatest concern to recyclers.

The best estimate is that the maximum independent RR is already constrained by copper contamination. Without interventions, this maximum RR will gradually decline throughout the century. The annual consumption to scrap availability ratio (C2SR) will decrease from around 1.4 today. Concurrently, the maximum RC rises but then plateaus below 75% as the RR falls. This highlights a conflict in the conditions for a circular economy: a C2SR approaching unity is a necessary condition for a high RC but leads to fewer opportunities for scrap contaminant dilution, which decreases the RR. Improved product design for recycling and deployment of scrap refining technologies will be needed to reach higher RCs. In 2017, the mass of U.S. scrap destined for LHSE exceeded direct steel imports. Domestic recycling of scrap exports alone could have displaced 36% of direct steel imports, reducing the U.S. deficit by $5.5 billion.

KEYWORDS
circular economy, copper contamination, dynamic material flow analysis, industrial ecology, recycling, tariffs

1 | INTRODUCTION

A circular economy (CE) requires high end-of-life recycling rates (RR) and plateauing material demand (Material Economics, 2018). Under these conditions, the material required to make new products is supplied from discarded end-of-life (EOL) products with minimal need for new material production from virgin resources. Globally, a CE is not currently possible because socioeconomic stocks are still growing at around 17 Gt per annum (Haas, Krausmann, Wiedenhofer, & Heinz, 2015). Around two-thirds of current global metal production is used to add to existing stocks (Haas et al., 2015) and, for steel specifically, Allwood, Cullen, and Milford (2010) report that 60% of global production was used to increase stocks in 2006. Unlike the world at large, per-capita ownership of metal in the United States has been largely unchanged since the late 1970s (Müller, Wang, & Duval, 2011). This plateauing of per capita stocks suggests that the average U.S. consumer needs very little new metal made from naturally occurring ore if existing steel resources are carefully managed.

Estimates of the current U.S. steel RR (50–90%: Section S1-1 in the Supporting Information) show that the United States is far from utilizing all its discarded metal. Environmental concerns alone may not lead to significant industry and government support for increased recycling (Motel, 2014). However, the Wang, Müller, and Graedel (2007) vision of a CE in which a nation achieves the benefits of independence from foreign resources, as
well as the environmental benefits of reduced primary production, may receive wider support and is a pertinent consideration given that in 2018 the United States imposed 25% steel tariffs to cut imports (BBC, 2018; U.S. Department of Commerce, 2018).

### 1.1 Barriers to increased steel recycling

The literature reveals three technical barriers to increased EOL steel scrap recycling. First, the availability of steel scrap limits the quantity that can be recycled (Oda, Akimoto, & Tomoda, 2013). Only small amounts of scrap are available for recycling in countries that are only now acquiring or have recently acquired significant steel stocks; for example, China. Globally, the annual steel consumption to scrap availability ratio (C2SR) is greater than two. This limits the global recycled content (RC) of new steel to less than 50%.

Second, recycling is limited by the ability to collect scrap (McMillan, Skerlos, & Keoleian, 2012; Material Economics, 2018). Some EOL scrap is economically inaccessible in submarine and underground pipes or as reinforcement in foundations (Cooper & Allwood, 2012; Cooper, Skelton, Moynihan, & Allwood, 2014). Other steel is left uncollected in abandoned buildings and factories. Despite these barriers, it is widely predicted that metal scrap collection rates will reach 90% by 2050 (Allwood et al., 2010; Ayres, 2006).

The third barrier is contamination of the steel scrap. Some contaminants (e.g., silicon, manganese, and aluminum) can be easily removed from molten scrap because they oxidize and dissolve in the slag. Zinc from galvanized scrap is also not a concern because its high vapor pressure causes it to evaporate during the steelmaking process (Janke, Savov, Weddige, & Schulz, 2000). However, tramp elements including copper, tin, antimony, and lead have a low affinity for oxygen (Janke et al., 2000) and cannot be easily removed from the melt using common pyrometallurgical processes (Nakajima, Takeda, Miki, Matsubae, & Nagasaki, 2011). Copper is the greatest concern (Daehn, Cabrera Serrenho, & Allwood, 2017; Material Economics, 2018) to recyclers because, as described by Daehn et al. (2017), copper is widespread in EOL steel scrap and no economic route currently exists for removing it.

Copper in steel scrap comes from electrical wiring in discarded products and from copper-containing steels such as reinforcing bars and stainless steels. Copper can be a useful alloying element but also causes hot shortness: surface cracking during hot steel processing above 1050°C (Savov, Volkova, & Janke, 2003) due to liquid copper penetrating the grain boundaries (Rod, Becker, & Nylén, 2006). Thus, copper contamination can reduce corrosion and fatigue resistance, often making recycled EOL steel unsuitable for use in flat products such as sheets for the car industry. Nakamura et al. (2012) showed using an input–output analysis that most EOL steel scrap is instead recycled for use in construction. This is because long structural steel sections undergo less hot rolling than flat products and because surface defects in reinforcing bars are not a concern as they are embedded in concrete (Rod et al., 2006).

A recent study by Daehn et al. (2017) suggests that copper contamination is unlikely to constrain global steel recycling rates as long as free trade in scrap allows contaminated scrap to be recycled where there is a high demand for construction products. Consistently, Söderholm and Ejdemo (2008) and Gesing (2004) all argue that unrestricted international trade is needed to ensure that scrap flows to appropriate uses. A recent European Union funded report on the prospect of a European CE identified copper contamination of steel scrap as a major barrier to being able to recycle European EOL scrap for use within Europe’s borders (Material Economics, 2018). No such study has been conducted for the United States but is needed because the relative size of the U.S. construction sector (acting as a copper contaminant sink) is much smaller than the global construction sector. Zhu, Syndergaard, and Cooper (2019) recently showed that only around 38% of U.S. steel consumption is destined for construction compared with a 55% share globally (Cullen, Allwood, & Bambach, 2012). In addition, the U.S. preference for steel framed buildings means that just 21% of construction demand is for rebar (the most impurity-tolerant steel product), compared to 28% globally.

### 1.2 U.S. steel production and trade

The United States made 82 Mt of steel in 2017 (USGS, 2018a), of which 68% was made using electric arc furnaces (EAFs) and predominantly from scrap metal. While this may sound like an impressive amount of recycling, over two-thirds of U.S. steel consumption can be attributed to imports (Cooper, 2018). The United States produces an abundance of steel scrap, much of which is destined either for landfill, hibernating stocks, or export (LHSE). In recent years, the United States has exported more steel scrap than any other nation, with Turkey and India the primary destinations (USGS, 2016). Cooper (2018) suggested that U.S. steel imports could be displaced by recycled steel if U.S. scrap currently destined for LHSE were diverted toward domestic recycling; however, there have not been any rigorous studies examining the feasibility of this approach.

### 1.3 Scope of this article

This article addresses the potential for material circularity and independence in the U.S. steel sector by answering the following questions:

- What are the maximum recycled contents and recycling rates in an independent U.S. steel sector?
- Could current U.S. steel imports be displaced by recycled steel originating from scrap that the United States currently fails to recycle (LHSE scrap)?
This work focuses on the technical barriers to realizing a steel CE. This is needed to evaluate the scope of potential change but future work must also consider the economic and behavioral barriers at the aggregated and firm level. These include dumping of cheap primary steel made overseas (Greenwood & Hudson, 2017), the lack of an established supply chain for steel that is difficult to retrieve from service (Cooper & Allwood, 2012), and consumer recycling habits (Park & Ha, 2014). Individual recycling firms are motivated to reduce costs not the use of primary steel. These objectives are reasonably equivalent when the purchasing and processing of steel scrap is cheaper than buying primary steel. However, scrap steel processing may become considerably more expensive in the future if refining technologies (e.g., sulfide slagging) are used extensively to remove tramp elements.

2 | METHODOLOGY

Sections 2.1–2.3 present the methodology used to estimate the maximum EOL recycled content (RC) and the maximum EOL recycling rate (RR) in an independent U.S. steel sector. Perfect circularity is when \( RC = RR = 1 \). The RC is constrained by the steel consumption to scrap availability ratio, \( C_{2SR} = RC \leq 1/C_{2SR} \), and the necessity to dilute scrap contaminants through additions of primary iron in the form of pig iron or direct reduced iron (DRI). The concentration of copper in the steel scrap determines how much primary iron must be added to the scrap in the EAF. This study concentrates on copper contamination because, as discussed in Section 1, copper contamination has been found by previous researchers to be the greatest compositional concern when recycling steel scrap. The primary additions dilute the copper to a concentration equal to or less than the copper tolerance for the intended application. The copper concentration in both pig iron and DRI is very low at approximately 0.01 wt.% (DJJ, 2018).

In order to calculate maximum RCs for future years, it is necessary to first estimate the quantity and composition of scrap available for recycling and the quantity and compositional tolerance of new steel demand in those future years. Section 2.1 describes the dynamic material flow analysis used to estimate future annual U.S. steel consumption and scrap discards to 2100, and Section 2.2 describes the copper contamination and tolerances assigned to the future scrap and new steel flows, respectively. In Section 2.3, a minimum primary metal consumption optimization model is formulated as a linear blending problem repeated for each year from 2020 to 2100 using the data generated in Sections 2.1 and 2.2. The calculated minimum primary metal consumption is used to then calculate the annual maximum RC and RR.

Sections 2.4–2.6 present the methodology used to assess the technical potential to displace current U.S. steel imports with increased domestic recycling. Section 2.4 estimates the quantity and compositional tolerance of imports in 2017, and Section 2.5 estimates the quantity and composition of U.S. steel scrap destined for LHSE in 2017. Section 2.6 presents a linear optimization model used to estimate the current import substitution potential. Data for 2017 is used as it is the last full year before the implementation of U.S. steel import tariffs.

2.1 | Future U.S. steel consumption and scrap discards

A flow-driven dynamic material flow analysis (DMFA) is used to calculate current and historical steel stocks and scrap discards, while a stock-driven DMFA model is used to calculate future consumption and scrap discards. These quantities are calculated for varying product lifespan and population growth scenarios. Product lifespans in each of the sectors are modeled as normal distributions (Table S1-2 in the Supporting Information). Historical U.S. stocks are calculated across the four end-use sectors (construction, transport, industrial equipment, and metal goods) commonly used in steel DMFA studies (Gerst & Graedel, 2008; Muller, Wang, Duval, & Graedel, 2006; Pauliuk, Milford, Müller, & Allwood, 2013; Pauliuk, Wang, & Müller, 2013). Calculated per capita stocks are extrapolated to 2100 for use in the stock-driven DMFA.

2.1.1 | Flow-driven DMFA (1880–2017)

The steel stock in each end-use sector in year \( t \) is calculated using Equation (1), where Embedded consumption\(_{t_0} \) is the quantity of steel entering use in each end-use sector in a previous year (\( t_0 \)), and \( F(t) \) is the complementary cumulative normal distribution function of the corresponding product lifespan. Embedded consumption is referred to as consumption elsewhere in this article but here must be distinguished from other definitions of consumption used by the steel industry.

\[
\text{Stock}_t = \sum_{t_0 = 1880}^{t} \left( \text{Embedded consumption}_{t_0} \times F_{t-t_0} \right)
\]

The quantity of steel entering U.S. societal stocks annually (Embedded consumption\(_{t_0} \)) is not reported by trade organizations or presented in government statistics. Instead, the steel industry commonly uses two other national consumption metrics: apparent consumption and true consumption. Apparent consumption equals the quantity of U.S. steel deliveries (material leaving domestic producers’ facilities) plus net direct imports of steel intermediate products (e.g., steel plates) but does not consider trade of finished goods (e.g., refrigerators). True consumption is defined as the apparent consumption plus the steel used to make (not embedded within) net indirect (finished good) imports (e.g., automobiles and refrigerators). True consumption therefore accounts for international trade but is a measure of the steel used in manufacturing not the steel embedded in the final...
product. True consumption includes steel destined for manufacturing scrap and is an aggregated value not revealing the breakdown of consumption among the different end-use sectors.

In this article, reported and estimated true consumption values are converted into embedded consumption values for each end-use sector. First, true consumption is split into sectoral true consumption according to the breakdown recorded by the American Iron and Steel Institute (Table S1-3 in the Supporting Information; AISI, 2019). Second, sectoral true consumption is converted to sectoral embedded consumption in final goods by multiplying by 84%, which is the global average material yield for the conversion of intermediate steel products (e.g., steel sheet) to final steel goods (e.g., automobiles) according to the seminal work on global steel flows by Cullen et al. (2012). For a given year, U.S. true consumption is either taken directly from World Steel Association reports (if available) or otherwise calculated from data on annual apparent consumption, indirect trade, domestic steel shipments, direct imports, direct exports, steel production, and process yields. Table S1-4 in the Supporting Information summarizes the true consumption calculation performed for each year from 1880 to 2017.

The historical stock of steel in each end-use sector (calculated using Equation (1)) is divided by the historical U.S. population (U.S. Census Bureau, 2019) to calculate the stock per capita \( \text{spc} \) in each end-use sector in each year. The aggregated \( \text{spc} \) values calculated for 2017 (8.5–13 t/cap, Figure S1-1 in the Supporting Information) align well with existing estimates from the literature (9.1–14.3 t/cap). A full comparison between these results and previous U.S. steel stock studies is presented in Section S1-2 in the Supporting Information.

2.1.2 Stock-driven DMFA (2017–2100)

Historical \( \text{spc} \) for each sector are extrapolated to 2100 and used to calculate annual steel consumption and steel discards (Figure S1-4 in the Supporting Information). In order to extrapolate the \( \text{spc} \) curves, logistic models (as shown in Equation (2)) are fitted to the historical \( \text{spc} \) curves, where \( t \) is the year, \( c_1 \) is the saturated \( \text{spc} \) value, and \( c_2 \) and \( c_3 \) are parameters used to define the logistic curve shape. Logistic curves are commonly used to model stock saturation (Müller, Hilty, Widmer, Schluep, & Faulstich, 2014) with Equation (2) also used by Tio and Sato (1998) in their modeling of steel \( \text{spc} \) in Japan. The values of \( c_1, c_2 \) and \( c_3 \) were determined using Matlab’s nonlinear optimization solver (fmincon) to minimize the sum of the square residuals between the fitted curve and the historical estimates with the constraint that the fitted and historical \( \text{spc} \) are equal in 2017 (see Figures S1-13 to S1-24 in the Supporting Information).

\[
\text{Stocks per cap}_t = \frac{c_1}{1 + e^{c_2 + c_3 t}} 
\]

Future absolute stocks are calculated for three U.S. population growth scenarios: a no growth scenario, a baseline expected population growth scenario, and a high population growth scenario. The expected population growth scenario (590 million people by 2100), and high population growth scenario (1.2 billion people by 2100) correspond to the U.S. Census Bureau’s (2018a) medium and high population projections, respectively. Steel consumption and scrap discards in future years are calculated using Equations (3) and (4), respectively. The DMFA results are summarized in Figure 1; data for this figure is provided in Supporting Information S2.

\[
\text{Embedded consumption}_t = \text{Stock}_t - \sum_{t_0=1880}^{t} \left( \text{Embedded consumption}_{t_0} \times F_{t-t_0} \right) - \sum_{t_0=1880}^{t-1} \left( \text{Embedded consumption}_{t_0} \times F_{t-t_0} \right) 
\]
where $F(t)$ is the cumulative normal distribution function of the corresponding product lifespan.

\[
\text{Scrap}_t = \text{Embedded consumption}_t - \left( \text{Stock}_t - \text{Stock}_{t-1} \right)
\] (4)

### 2.2 Composition of new steel and collected scrap

The DMFA provides estimates of the steel annually consumed within the four end-use sectors (2018–2100). As in Daehn et al.’s analysis, the steel consumed in the U.S. end-use sectors is disaggregated into consumption of 20 intermediate steel products according to the fractional breakdown of intermediate products entering end-use products shown in the U.S. steel flow analysis by Zhu et al. (2019) (Table S1-6 in the Supporting Information). The copper tolerance of the intermediate products (Table S1-11 in the Supporting Information) was assigned according to the limits described by Daehn et al. (2017) for 18 of the steel products and by Alro (2015) for steel castings and tool steel. For each product, an expected, loose, and strict copper tolerance is defined. The strictest copper tolerance is 0.04 wt.% for tinned cold rolled coil and the loosest copper tolerance is 0.75 wt.% for castings.

The DMFA results provide estimates of the annual quantity of EOL scrap discards from each of the four end-use sectors. The copper concentration in each of these scrap categories was recently estimated by Daehn et al. (2017). They consider three scenarios corresponding to expected, low, and high copper concentration levels to account for uncertainty. Daehn et al.’s global estimates were reconciled against U.S. specific scrap contamination data (see Section S1-4 in the Supporting Information) and used in this article’s analysis (Table S1-8 in the Supporting Information).

### 2.3 Analyzing circularity (2020–2100)

In the recycling industry, different scrap metal sources are mixed together with primary metal additions (pig iron and DRI) to satisfy the volumetric and compositional demand for new metal. The model implemented in this work allows mixing of the scrap and primary metal sources to satisfy demand while simultaneously minimizing the demand for primary metals (the sum of pig iron and DRI). The maximum RC and RR is calculated from the results for each year of the analysis using Equations (5) and (6), respectively.

\[
\text{RC} = \frac{\text{U.S. steel consumption (Mt)} - \text{Primary metal consumption (Mt)}}{\text{U.S. steel consumption (Mt)}}
\] (5)

\[
\text{RR} = \frac{\text{U.S. steel consumption (Mt)} - \text{Primary metal consumption (Mt)}}{\text{Domestic scrap discards (Mt)}}
\] (6)

In the optimization model, for each year there are 20 types of new steel demand, corresponding to each of the intermediate products (Table S1-6 in the Supporting Information), and six metal sources (pig iron, DRI, and the four EOL scrap categories from each end-use sector). Elements in the $20 \times 6$ matrix $\theta_{d,s}$ represent the quantity (in mass units) of metal source $s$ used to produce new steel product $d$. The decision parameters are physically independent; however, it is recognized that weak coupling may exist through socioeconomic effects (e.g., scrap price elasticity of the collection rate). Linear programming is used to reduce computation times and to avoid multiple local minima. The model assumes that there is an unlimited availability of primary metal and that steel scrap EAF melt losses equal 7% of the charge (American Iron and Steel Institute, Energetics, Inc., 2003). Manufacturing scrap is neither added to the steel consumption data nor included in the scrap arising estimates because it is an industry loop that acts neither as a net contaminant source nor sink.

The objective function for each year is shown in Equation (7), subject to the inequality constraints shown in Equations (8)–(11). Note that $s = 1$ represents pig iron, $s = 2$ represents DRI, and $s = 3:6$ represents the scrap from the four EOL scrap categories (construction, transport, machinery, and products).

Minimize:

\[
\sum_{d=1}^{20} \sum_{s=1}^{6} \theta_{d,s}
\] (7)

Subject to:

\[
\sum_{s=1}^{6} \theta_{d,s} \geq \text{Product\_demand}_d
\] (8)

\[
\sum_{d=1}^{20} \theta_{d,s} - 3:6 \leq \text{Metal\_source}_s - 3:6
\] (9)

\[
\frac{\sum_{s=1}^{6} (\theta_{d,s} \times \text{copper\_source}_s)}{\sum_{s=1}^{6} \theta_{d,s}} \leq \text{copper\_limit}_d
\] (10)
Absolute quantity of copper in product, d, from scrap sources:

\[ \sum_{s=3}^{6} (\theta_{d,s} \times \text{copper}_s) \tag{11a} \]

Minimum quantity of product, d, that must be produced in order to dilute copper from scrap (assumes primary sources contain no copper):

\[ \frac{\sum_{s=3}^{6} (\theta_{d,s} \times \text{copper}_s)}{\text{copper}_{\text{limit},d}} \leq \sum_{s=3}^{6} \theta_{d,s} \tag{11b} \]

A minimum fraction, \( \alpha \), of the minimum demand for product, d, must be satisfied by scrap sources:

\[ \alpha \times \left( \frac{\sum_{s=3}^{6} (\theta_{d,s} \times \text{copper}_s)}{\text{copper}_{\text{limit},d}} \right) \leq \sum_{s=3}^{6} \theta_{d,s} \tag{11c} \]

There are four groups of inequality constraints: new steel demand constraints; scrap supply constraints; new steel chemistry constraints; and economic furnace constraints. The 20 \( (d = 1:20) \) new steel demand constraints (Equation (8)) ensure that the simulated production of each new steel product, \( d \), is greater than or equal to the demand for that product \( (\text{Product}_{\text{demand}}) \). The four \( (s = 3:6) \) scrap supply constraints (Equation (9)) ensure that the quantity of a used scrap source does not exceed the amount available \( (\text{Metal}_{\text{source}}) \). The 20 \( (d = 1:20) \) steel product chemistry constraints (Equation (10)) ensure that the simulated copper concentrations of the new steel products are lower than or equal to the copper tolerance, \( \text{copper}_{\text{limit},d} \), of that intermediate product as described in Table S1-11 in the Supporting Information. The 20 \( (d = 1:20) \) economic furnace constraints (Equation (11c)) ensure that the scrap content in a recycling furnace is not less than the required weight fraction \( \alpha \). This parameter was determined through interviews with industry experts (Pretorius, 2018), suggesting \( \alpha = 1/3 \) reflects a sensible EAF operation limit.

In addition, recycling emissions may rise to be higher than from primary production if the scrap content in the recycling furnace reduces below one-third; the energy and emissions associated with primary steelmaking is only three times higher than for remelting steel scrap (Ashby, 2012).

The optimizations were run using MATLAB’s linprog solver (Zhang, 1998) and an Intel(R) CoreTM i7-6600U CPU, 2.81 GHz, with 16 GB of RAM. Convergence times were less than 5 s for each year analyzed.

2.4 | U.S. steel imports in 2017 (see Section S1-5 in the Supporting Information)

Imports are classified as either direct ( semifinished and steel mill goods containing only steel) or indirect ( finished goods). Direct imports are extracted from the U.S. Census Bureau (2018b), which splits these imports into 20 subcategories. Imported billets, blooms, and slabs (BBS) are formed into intermediate products within the U.S. steel industry and in this work is allocated to the various intermediate product import categories in the same ratio as domestically produced steel, as reported for steel mill products in table 3 of the USGS (2014) Iron and steel Minerals Yearbook (USGS, 2014) and as reported for steel and iron castings in the USGS mineral commodity summary report (USGS, 2015).

Finished goods are either subassemblies ( e.g., automobile engines) that are imported by U.S. manufacturers to help assemble a final product or are completed products ( e.g., automobiles) that will undergo no more than superficial physical changes between import and sale. The mass of finished product imports in 2017 was estimated using trade data on the 29 goods with the largest mass of embedded steel traded—as defined by Wang et al. (2007)—recorded in the United Nations Commodity Trade (Comtrade) Database ( U.N., 2018). The traded product mass is converted to the traded steel mass by multiplying with steel content coefficients defined for each good by Wang et al. (2007). For 2017, the Comtrade database records the economic value of indirect trade in all 29 product categories of interest; however, the mass of the imports is only recorded for 15 of the categories. In order to estimate the traded mass in the remaining 14 categories, an empirical relationship is derived for the dependence of the product steel intensity (kilogram steel per dollar of imports) on product characteristics such as the steel content and fabrication complexity (Section S1-5 in the Supporting Information). Thus, indirect steel imports for 2017 are estimated at 58 Mt (Table S1-15 in the Supporting Information), which is further broken down according to the intermediate products entering global end-use products (Table S1-7 in the Supporting Information). The copper tolerance was assigned using the values in Table S1-11 in the Supporting Information.

2.5 | U.S. steel scrap destined for LHSE in 2017

The steel scrap the United States currently fails to recycle domestically is either exported or is destined for landfill and hibernating stocks (LHS). Only aggregated scrap flow data is currently available for 2017 from USGS (USGS, 2018b); therefore, in this study a detailed analysis is conducted on 2014 data and the results then scaled according the ratios of the exported and collected steel scrap flows in the 2 years.
It is estimated that the United States exported 14 Mt of steel scrap in 2017 (USGS, 2018b), as summarized in Table S1-16 in the Supporting Information. The quantity of scrap sent to LHS is not recorded directly and must be estimated from the quantity of scrap collected and the RR, as shown in Equation (12).

\[
\text{Scrap}_{\text{landfill or hibernating stocks}} = \text{Scrap}_{\text{collected}} \times \left( \frac{1}{RR} - 1 \right)
\]

The quantity of scrap collected includes scrap destined for both domestic recycling and export, and was estimated for 2014 by examining the USGS steel scrap statistics (Table S1-17 in the Supporting Information). In 2014, U.S. consumer receipts for steel scrap totaled 40.3 Mt (including 3.3 Mt of imported scrap), and a further 15.1 Mt of scrap was exported; thus, 52.1 Mt of U.S. EOL scrap was collected. Coupled with the DMFA result that 71.1 Mt of steel scrap was available for recycling in 2014 (baseline scenario, Figure 1), this scrap collection figure (52.1 Mt) implies an overall RR of 73%, which is between previous estimates of the U.S. recycling rate (Table S1-1 in the Supporting Information).

Sectoral recycling rates were derived by estimating the sectoral composition of the different industry scrap categories recorded by USGS—such as #1 Heavy melting steel—through additional literature reviews and interviews with local scrap dealers (Tables S1-17 and S1-18 in the Supporting Information). For example, the largest single steel scrap category—standard shredded scrap of which 19.28 Mt was collected in 2014—was assumed to consist of 62.5% transport scrap and 37.5% metal product scrap (including 30% appliance scrap) as found by the Center for Automotive Research (Brahmst, 2006). Equation (12) was then used to estimate the quantity of scrap destined for LHS belonging to each industry scrap category (Table S1-17 in the Supporting Information).

The concentration of copper in the different industry scrap categories was determined in collaboration with The David J. Joseph company (DJJ, 2018), a scrap dealer owned by Nucor who are the world’s largest steel recycling company. Copper concentration values from DJJ (2018) are supplemented with additional data from Leroy (1995) for copper concentration in steel can scrap, and Kostetsky, Troyansky, and Samborsky (2000) for alloy steel scrap and all other carbon steel scrap. It is assumed in the main analysis that scrap destined for LHS has the same copper concentration as currently collected scrap.

### 2.6 Analyzing import substitution

The United States currently fails to domestically recycle U.S. scrap destined for LHSE. If this material were made available for domestic recycling, what percentage could be recycled to directly displace imports? The answer depends on the quantity and compositional tolerance of the imports (Figure 2 (top)) and the quantity and composition of the scrap (Figure 2 (bottom)). The import substitution potential (ISP) is defined in Equation (13), and the scrap utilization potential (SUP) is defined in Equation (14). The numerator in the equations is calculated by adapting the linear optimization objective function introduced in Section 2.3 to calculate the maximum quantity of LHSE scrap that can be recycled to meet import demand (Equation (15)). In the adapted optimization model, there are again 20 types of new steel demand, corresponding to each of the imported intermediate products (Table S1-11 in the Supporting Information and Figure 2 (top)) and now 18 metal sources (pig iron, DRI, and the 16 EOL industry scrap categories, Figure 2 (bottom)). Elements in the \(20 \times 18\) matrix \(\theta_{d,s}\) represent the quantity of metal source \(s\) used to produce currently imported steel product \(d\). All constraints and assumptions from Section 2.3 apply.

\[
\text{ISP} = \frac{\text{Maximum LHSE scrap domestically recycled to meet import demand (Mt)}}{\text{Current U.S. new steel imports (Mt)}}
\]

\[
\text{SUP} = \frac{\text{Maximum LHSE scrap domestically recycled to meet import demand (Mt)}}{\text{Current LHSE scrap (Mt)}}
\]

Maximize:

\[
\sum_{d=1}^{20} \sum_{s=3}^{18} \theta_{d,s}
\]

### 3 RESULTS

The potential for material circularity and independence in the U.S. steel sector is revealed by first examining the results on the maximum EOL scrap RRs and RCs; and, second, by examining the import substitution potential of recycling U.S. scrap currently destined for LHSE.

#### 3.1 Maximum recycling rates and recycled contents in an independent U.S. steel sector

The maximum RRs and RCs between 2020 and 2100 are shown in Figure 3. The best estimate (under the baseline scenario: expected copper contents, population growth, and product lifespans) is that the copper content of the steel scrap already constrains the maximum
FIGURE 2  U.S. steel imports (top) and LHSE scrap (bottom) in 2017. Top: Direct imports = 34 Mt; indirect imports = 58 Mt; total = 92 Mt. Bottom: Exports = 14.0 Mt; landfill & hibernating stocks (LHS) = 20.3 Mt; total = 34.3 Mt. Underlying data used to create this figure can be found in Supporting Information S2.
FIGURE 3  Maximum recycling rates and recycled contents in an independent U.S. steel sector. (a) Maximum RRs and RCs under the expected copper content scenario. (b) RRs and RCs under the baseline (expected) population growth and product lifespan scenario. Varying scrap copper contamination & product copper tolerance. Underlying data used to create this figure can be found in Supporting Information S2.
FIGURE 4  Potential import substitution from domestic recycling of LHSE scrap (2017). Error bars are from product copper tolerance scenarios. Underlying data used to create this figure can be found in Supporting Information S2.

A: All LHSE scrap diverted to displace direct imports
B: Scrap exports diverted to displace direct imports
C: All LHSE scrap diverted to displace all imports
D: LHS scrap diverted to displace direct imports

The results shown in Figure 4 were generated under the assumption that scrap destined for LHS has the same copper concentration as currently collected scrap. Further work is needed to understand the correlation between scrap purity and collection rates; however, as part of a sensitivity study, the analysis on displacement of direct imports with LHS scrap (Figure 4: Scenario D) was repeated assuming a homogenous LHS scrap composition equal to the highest copper content level of all collected scrap types in 2017 (Table S1-17 in the Supporting Information: 0.46 wt.%). The corresponding ISP was reduced from 48% to 21%; thus, highlighting the importance of collecting accurate LHS copper content data in future, more refined, analyses.
This study has quantified the potential for U.S. steel circularity and the potential to displace current steel imports with increased domestic recycling. A new U.S. steel DMFA has been used to predict steel demand and scrap arising to 2100. The DMFA results have been used in a linear optimization model to estimate maximum RRs and RCs in an independent U.S. steel sector, accounting for both the C2SR and the compositional mismatch between scrap and new steel. Multiple product lifespan, population growth, scrap copper contamination, and new product copper tolerance scenarios have been modeled to reflect the inherent uncertainty in the projections. Despite the uncertainties, several transferable lessons can be learned from the results, as discussed below.

4.1 Overcoming the limits to the recycling rate and recycled content

The RC is limited by both the balance between steel consumption and scrap availability (C2SR) and, particularly at higher RCs in the future, the compositional mismatch between copper-rich steel scrap and copper-intolerant new steel products. C2SRs approaching unity are needed to enable high RCs but this results in fewer opportunities for scrap down-cycling and contaminant dilution, decreasing the RR. The RC is expected to be less than 60% even under a near balanced steel flow scenario (C2SR ≈ 1), when RRs start to fall due to contamination. The conflict is present in other material systems too such as aluminum; a recent study by Zhu and Cooper (2019) found that the compositional mismatch between aluminum scrap and new aluminum products has already helped to limit the U.S. domestic aluminum RR.

The source and destination of optimal steel flows are examined in order to target CE strategies. Figure 5 shows that EOL scrap is largely recycled into products for the construction industry. It shows that transport scrap and metal product scrap (old appliances, packaging, etc.) are the most difficult to recycle because of high levels of copper contamination from wiring (Table S1-8 in the Supporting Information) and because of the tight copper tolerances of the new metal used in these sectors (e.g., cold rolled galvanized sheet has an expected copper tolerance of just 0.06 wt.%). Reuse and recycling strategies should focus on these sectors and are discussed below.

EOL vehicles and appliances are typically shredded before remelting. Shredding embeds copper strands into the steel and makes magnetic separation ineffective (Daehn, Serrenho, & Allwood, 2019). Advanced design for recycling (e.g., product modularity that allows easy separation of copper containing electronics from the product’s steel structure) could be motivated by extended producer responsibility policies (Ayres, Ferrer, & Leynseele, 1997; Nasr & Thurston, 2006). For vehicles, there is the potential to design copper wiring harnesses that are easily detachable at EOL (Toyota, 2018) or even to replace copper wiring with aluminum wiring that could also reduce the weight of the harness by around 30% (Oba, 2013). Aluminum is easy to remove from molten steel scrap if not separated beforehand. Elsewhere, Cooper and Allwood (2012) highlight the potential
to reuse steel packaging that contains copper-intolerant tin coated sheet. They find limited potential to directly reuse steel sheet from domestic appliances because the profiled paneling and interior connections are unique to the brand and product generation; however, they report that there may be potential to reshape and reuse some of the steel sheet from cars, trucks, and domestic appliances, as demonstrated by Tekkaya, Franzen, and Trompeter (2010).

If product designs do not change and if shredding at EOL persists; then, new affordable scrap separation and refining technologies must be developed. Higher-density shredding causes greater separation and sulfide slagging and vacuum induction melting (Savov & Janke, 2000; Savov et al., 2003) are two current technologies capable of removing copper from steel scrap but are expensive and have seen negligible commercial uptake. Daehn et al. (2019) provide an up-to-date and comprehensive review of the refining options. Government could support the development of EOL strategies through funding for research into economical scrap separation and refining methods and by creating programs, for example, tax breaks or interest-free loans, for recyclers who invest in advanced recycling technologies.

4.2 Displacing imports with diverted scrap exports

The U.S. trade surplus in scrap metal has led some commentators to argue that steel scrap exports are good for the U.S. trade balance (Fitzgerald, 2004; Paben, 2018). Figure 6 shows the wider context: the United States sells steel scrap at an average of $350/t and pays an average of $810/t for direct new steel imports. The exchange of cheap steel scrap for expensive new metal widens the U.S. deficit and represents a loss of base metal and valuable alloying elements such as magnesium and manganese, which are now listed as critical by the U.S. government (U.S. Department of the Interior, 2018). If the United States domestically recycled currently exported scrap, displacing 36% of current direct (industry product) imports (see Figure 4: Scenario B), then the U.S. trade deficit could be reduced by $5.5 billion.

This analysis provides an initial assessment of the potential to displace imports with domestically recycled scrap metal. However, the U.S. Census Bureau import categories used in this analysis mask the many specific steel grades imported for automotive and other applications, some of which may not be manufactured in the U.S. The testimonies submitted to the U.S. Department of Commerce’s (2018) report on the national security implications of steel imports reveal importers’ concerns regarding domestic supplies of, for example, high-quality wire rod for making safety critical automotive fasteners, some advanced and ultra-high strength steel sheets for the car industry, and plates for large diameter line pipe. Any policy designed to transition the United States to an independent steel sector needs to address such concerns.
4.3 Achieving a circular economy with a growing population

In this study’s model, future consumption, maximum RRs, and maximum RCs are all sensitive to population growth (see Figures 1 and 3). A low-resolution map of the U.S. 2017 steel flow was derived to help explain the ongoing significance of population growth (Figure 7). The map was constructed using the DMFA results (baseline analysis) and publically available USGS statistics (See Section S1-8 in the Supporting Information for more details).

Figure 7 shows that the U.S. consumption in 2017 was 30 Mt greater than the U.S. scrap discards (EOL scrap). This addition to stock is not due to an increasing per person demand but an increasing number of people. The increase in the U.S. population in 2017 (2.31 million people, 0.7% per annum: U.S. Census Bureau, 2019) multiplied by 13 t/capita, which is within the saturated spc range shown in Figure S1-1 of the Supporting Information, results in the 30 Mt additional stock. Thus, we cautiously hypothesize that even low population growth in a mature economy can limit RCs to around 70% even if RRs approach 100%. A CE with a growing population therefore requires reductions in spc through material efficiency strategies such as product light weighting (Carruth, Allwood, & Moynihan, 2011; Serrenho, Norman, & Allwood, 2017) and more intensive use of products (Liao & Cooper, 2019).

4.4 Closing remarks

In this U.S. focused analysis, the best estimate is that copper contamination already restricts the maximum independent RR to less than 88%. The best estimate of the actual U.S. collection rate is 76% (though estimates range from 50 to 90%: Section S1-1 in the Supporting Information). A quarter of the collected scrap is exported. The exported scrap is estimated to be 14% more contaminated with copper than the U.S. scrap that is recycled domestically (0.35 vs. 0.31 wt.%).

Over the coming decades, the maximum independent U.S. RR will decrease and—without interventions—continuing scrap exports to the construction intensive developing world will be needed to keep the compositional constraint inactive. Design, reuse, and recycling strategies that promote an independent U.S. steel CE have been introduced and should focus on the transport and metal products sector.

ACKNOWLEDGMENTS

We thank all the recycling companies who assisted in this study.

CONFLICT OF INTEREST

The authors declare no conflict of interest.

REFERENCES


Additional supporting information may be found online in the Supporting Information section at the end of the article.

How to cite this article: Cooper DR, Ryan NA, Syndergaard K, Zhu Y. The potential for material circularity and independence in the U.S. steel sector. J Ind Ecol. 2020;24:748–762. https://doi.org/10.1111/jiec.12971