SOCIAL ECOLOGY WORKING PAPER 182

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ISSN 1726-3816

January 2020

Quirin Dammerer (2020):

Material stocks and sustainable resource use in the United States of America from 1870 to 2017

Social Ecology Working Paper 182
Vienna, January 2020

ISSN 1726-3816

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Material stocks and sustainable resource use in the United States of America from 1870 to 2017*

von

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** Masterarbeit verfasst am Institut für Soziale Ökologie, Studium der Sozial- und Humanökologie. Diese Arbeit wurde von Univ.-Prof. Dr. Fridolin Krausmann und Mag. Dr. Dominik Wiedenhofer betreut. Die vorliegende Fassung ist eine geringfügig überarbeitete Version der Masterarbeit.*

Abstract

More than half of all extracted materials worldwide are used to build and maintain in-use material stocks of manufactured capital in the form of infrastructures, buildings and durable goods. These material stocks provide services to societies but also drive material and energy flows and contribute to socio-ecological challenges like climate change. Understanding stock dynamics is thus of paramount importance for reconciling societal wellbeing and ecological sustainability. Here, we employ a dynamic inflow-driven stock-flow modelling approach to estimate total in-use stocks, waste and recycling for the United States of America from 1870 to 2017. We find that in-use stocks increased 160-fold, from 0.6 Gt (16 t/capita) in 1870 to 96 Gt (295 t/capita) in 2017. The existing stock of the USA provides a broad range of services to society, but the production and maintenance of stocks require an annual input of 1.9 Gt of materials and their operation 69 EJ of energy, producing 3.6 Gt of CO_2-eq. This is incompatible with the goal of keeping global warming below 1.5°C. Stocks need to be reduced, transformed and redistributed to ensure both a sufficient level of services and ecological sustainability. This would require a transformation towards new patterns of production and service-provisioning.

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List of Abbreviations

BTS	Bureau of Transportation Statistics
Comtrade	International Trade Statistics Database of the United Nations
DMC	Domestic material consumption
EIA	Energy Information Administration
EoL	End-of-life
EPA	Environmental Protection Agency
Ew-MFA	Economy-wide material flow accounting
FAOSTAT	Database of the Food and Agriculture Organization of the United Nations
GDP	Gross domestic product
GHG	Greenhouse gas
Gt	Gigatonne (10^9 metric tonnes)
HFP	High flexible pavement
IPCC	Intergovernmental Panel on Climate Change
MISO	Material Inputs, Stocks and Outputs
Mt	Megatonne (10^6 metric tonnes)
NAS	Net additions to stock
SI	Supplementary Information
T	Metric tonne
UNEP	United Nations Environment Programme
USA	United States of America
USDT	US Department of Transportation
USGS	United States Geological Survey
WWII	World War II
Yr	Year

1 Introduction

In-use material stocks of manufactured capital in infrastructures, buildings and durable goods play a crucial role for sustainability. On the one hand, material stocks provide services (e.g. communication, shelter and mobility), which contribute to human well-being and are a requirement for human activities and societal development (Haberl et al. 2017, 2019). On the other hand, they require a continuous inflow of material and energy as well as physical space to provide these services and thereby contribute to socio-ecological challenges like climate change (IPCC 2014, 2018) and biodiversity loss (Maxwell et al., 2016). Understanding material stock dynamics is thus of utmost importance for developing strategies to reconcile societal wellbeing and ecological sustainability.

On a global level, more than half of all extracted materials are currently used to build and maintain material stocks (Wiedenhofer et al., 2019). It has been estimated that using existing stocks in infrastructures (energy production, industry, transport and buildings) until the end of their lifetime would result in 496 Gigatonnes (Gt) of CO_2 emissions from 2010 to 2060 (Davis et al., 2010). Bringing the worlds stocks of cement, steel and aluminum up to the level of industrialised nations would result in 350 Gt of CO_2 emissions (Müller et al., 2013). Together, this amounts to more than the remaining CO_2 budget from 2018 (580 Gt) for limiting global warming to 1.5°C with a 50 % chance (IPCC, 2018).

Global stocks are distributed unequally (Cao et al. 2017, Liu and Müller 2013, Müller et al. 2011), with two thirds of all stocks residing in industrialised countries in 2010 (Krausmann et al., 2017). Since a certain amount of stocks is crucial for any society's wellbeing, a space for manoeuvre must be created for the rest of the world to build up, maintain and use stocks, while already industrialised nations have to decrease their stock-related emissions (Krausmann et al., 2017). From a climate justice perspective, the USA bears substantial responsibility for reducing its emissions (Adams and Luchsinger, 2009). The USA is the second largest emitter of global GHG emissions in 2017 (UNEP, 2018). Its per capita CO_2 emissions (16.2 tonnes/year) rank among the highest worldwide and its cumulative emissions amount to 25 % of all anthropogenic CO_2 emissions ever produced (Ritchie and Roser, 2017). At the same time, many infrastructures in the USA are in a poor condition, requiring material and energy for maintenance and repair (BTS, 2019). The population of the USA is expected to continue to grow, requiring more services from stocks in the future (Bureau of the Census, 2018). Solutions need to be found which simultaneously decrease emissions from stocks and guarantee a sufficient level of services for society.

Decreasing stock-related emissions and resource use, while ensuring a high level of societal wellbeing, requires a robust understanding of the (historic) interplay between material inputs, stocks, waste and recycling flows, energy consumption and greenhouse gas emissions (Mayer et al., 2017). Most studies on the USA have so far confined their analyses to specific stock types. Kapur et al. (2008) studied the evolution of cement in-use stocks in the USA during the 20^{th} century. Miatto et al. (2017) quantified the in-use material stocks (rammed earth, sand and gravel, cement, bitumen) in the road network of the USA from 1905 to 2015. Other studies have estimated and analysed stocks of the USA for various metals such as iron (e.g. Müller et al. 2006, Sullivan 2005), aluminum (e.g. Chen and Graedel 2012, McMillan et al. 2010), copper (e.g. Rauch, 2009) as well as gold, lead, tungsten, zinc, chromium and nickel (Gerst and Graedel, 2008). Using long term material flow data from Gierlinger and Krausmann (2012), Fishman et al. (2014) produced the first estimates of the total stock of the USA for the period of 1930 to 2005. However, Fishman et al. (2014) limited their analysis to aggregated material groups, omitted fossil fuel stocks and did not account for re- and downcycling in their stock-flow model.

While previous studies have produced valuable knowledge about the development of material stocks, a more systemic analysis of the total stock of manufactured capital is needed to better understand the role of stocks for sustainability transformations. In this study, we investigate the long term (1870-2017) dynamics of total material stocks, waste and recycling in the USA, using a dynamic inflow-driven stock-flow model (Wiedenhofer et al. 2019, Krausmann et al. 2017). This study therefore goes beyond the approach of Fishman et al (2014) by explicitly taking issues of circularity into account, while also performing a more in-depth assessment of the evolution of various material stock types over time. We also, for the first time, explicitly connect the entire framework of economy-wide material flow accounting (ew-MFA) with a dynamic stock-modelling approach for the USA and relate total in-use material stocks to energy consumption and CO_2 emissions. In doing so we aim at answering the following questions: what share of all consumed materials is used for building and maintaining stocks? How did stocks develop over time, what is their size and composition? What are the corresponding waste and re- and downcycling flows? How large are stocks of the USA in comparison to the world and other countries? How have energy consumption and CO_2 emissions evolved in relation to stocks? And, finally, what strategies seem promising to shift the USA towards a path of higher sustainability?

The remainder of this paper is structured as follows. Section 2 presents the methodological approach and data sources. Section 3 shows results for inputs to stock, the size and development of stocks, recycling and waste and a comparison of stocks with socio-economic indicators. In section 4 we compare our estimates with the results of other studies and relate the evolution of energy use and CO_2 emissions to stock development. Based on these results, we discuss the role of material stocks in reconciling societal wellbeing and ecological sustainability for the USA. Section 5 recaps and concludes.

2 Methodology and Data

2.1 Model Description

Studies that quantify stocks have used a variety of different approaches; the selection of an appropriate approach is often constrained by data availability and research objectives. Applied methods and models can generally be characterised as top down or bottom up, dynamic or static and inflow-or stock-driven, while many studies also use hybrid approaches (Wiedenhofer et al. 2019, Augiseau and Barles 2017, Müller et al. 2014). This study employs the MISO-model (**M**aterial **I**nputs, **S**tocks and **O**utputs), which is a dynamic inflow-driven modelling approach (Wiedenhofer et al. 2019, Krausmann et al., 2017). Dynamic inflow-driven models use exogenous data on material flows and service lifetimes to endogenously calculate in-use material stocks, End-of-life outflows, recycling and downcycling and waste flows over time within predefined spatial and temporal system boundaries.

Material flows in the MISO-model are derived from ew-MFA. Ew-MFA is commonly used in science and policy and provides information on the annual extraction and trade of all materials (excl. water and air) flowing into socio-economic systems (Mayer et al. 2017, Fischer-Kowalski et al. 2011). Full consistency with the ew-MFA framework and principles is thus “an important feature of the MISO-model, which has been built to complement and expand material flow accounting tools and to provide information consistent with MFA headline indicators” (Krausmann et al. 2017: Si-3). We describe the key mechanisms of the MISO-model here. For a more detailed explication see the Supplementary Information (SI) of this study and the papers by Wiedenhofer et al. (2019) and Krausmann et al. (2017).

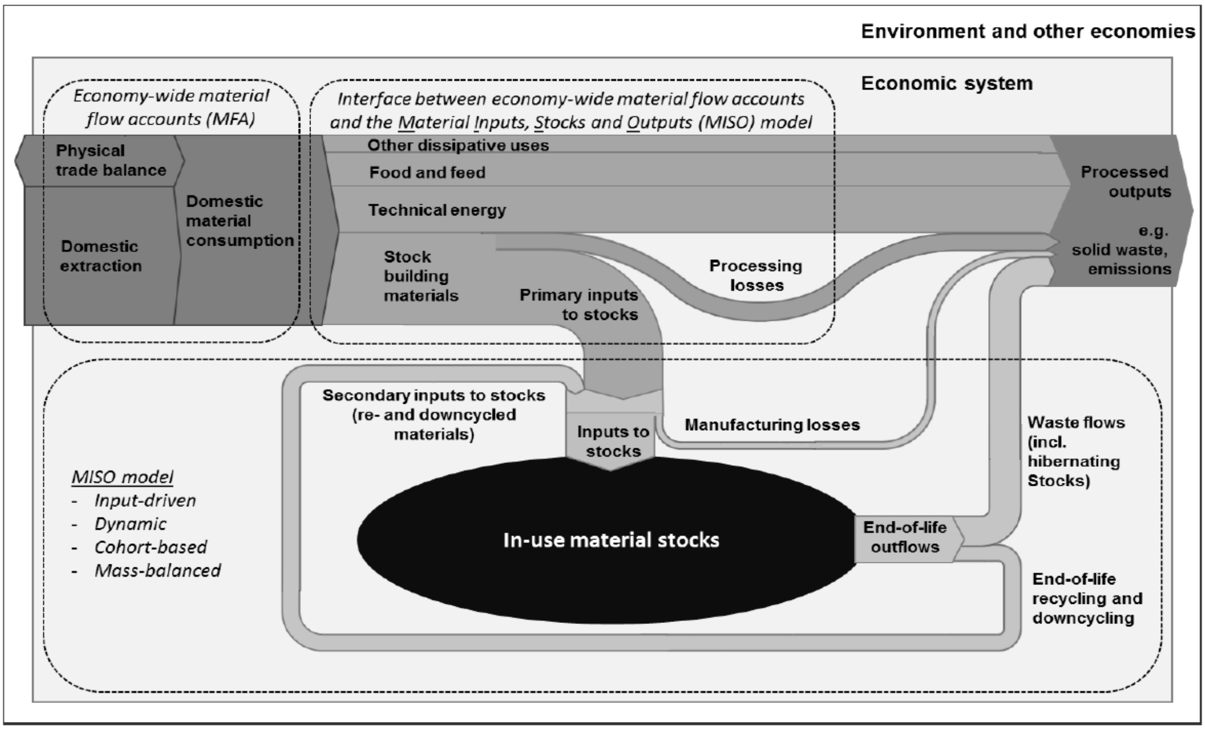

Figure 1: The MISO-model: System boundaries and material stocks and flows. Exogenous parameters are: stock-building materials, processing and manufacturing losses, lifetimes and re- and downcycling rates or flows. The MISO model uses these exogenous parameters to endogenously calculate in-use stocks, End-of-life outflows, re- and downcycling flows or rates and waste flows. Dark grey colours show inputs from the environment and other socio-economic systems and outputs to the environment (standardized ew-MFA indicators). Medium grey colours show the interface module, which connects the MISO-model to the ew-MFA framework; here materials are classified according to their uses and processing losses are deducted from stock-building materials. Light grey colours show flows within the MISO-model and black colours depict stocks. Source: Krausmann et al. (2017)

The MISO-model (Figure 1) requires data for the following *exogenous parameters*: the annual inflow of stock-building materials, processing and manufacturing losses, lifetimes and re- and downcycling rates. *Stock-building materials* include all materials in ew-MFA which are not used for either energy provision (e.g. food, feed, fossil energy carriers for thermal conversion) or other dissipative uses (e.g. fertilizers, salt). The main conceptual distinction in the classification of these materials is the residence time in the socio-economic system before they are discarded. Materials for energy and other dissipative uses are consumed within one year and leave the system immediately as emissions or solid waste. Stock-building materials, on average, stay in the socio-economic system for longer than one year. *Processing losses* accrue during the first processing step of the raw materials (e.g. CO_2 from limestone calcination, tailings from ore processing) and are deducted from stock-building materials as reported in material flow accounts to obtain primary material inputs to stock (e.g. cement, metal).

Manufacturing losses (e.g. wastage of cement during the construction of buildings, losses during metal manufacturing) are subtracted from primary inputs to stock to obtain actual inputs to stock. These actual inputs become in-use stocks and stay in the socio-economic system until the end of their service lifetime. *Lifetimes* in this regard are understood as average lifetimes of different products with different lifetimes. To obtain average lifetimes end-use shares for materials (e.g. % clay in bricks for building construction or % clay in tiles) are multiplied by their respective lifetimes (e.g. 75/25 years for bricks/tiles). Some short-lived products, like plastics packaging or paper in newsprint and magazines, are included in inputs to stock, despite being in use for less than one year, because plastics and paper as stock types exhibit an average lifetime of more than one year (e.g. 10 years for plastics in 2017).

At the end of their lifetime, stocks become *End-of-life (EoL) outflows*. *EoL outflows* are either *re- or downcycled* to become secondary material inputs to stock or they become *waste flows* returned to the environment. The model either estimates *recycling rates* (if exogenous information on flows of secondary materials are available) or *recycling flows* (if rates are available). Note that recycling does not include the thermal conversion of materials for energy recovery and only EoL recycling is considered; recycling of scrap within the manufacturing process is considered an internal flow and not explicitly dealt with in the MISO-model. We refer to *waste flows* as *final waste flows*, to separate waste from EoL outflows from other forms of waste returned to the environment (e.g. processing losses and manufacturing losses in the form of solid waste or emissions). *Final waste flows* comprise waste going to controlled and uncontrolled landfills, other treatment facilities (e.g. incineration), export or simply remains in place as hibernating stocks (e.g. abandoned rail tracks or buildings that are not demolished). The different pathways of *final waste flows* currently cannot be fully separated due to a lack of data.

The MISO-model estimates all material stocks of manufactured capital, that is materials accumulated in all human-made artefacts including buildings, infrastructures and all durable goods (machinery, furniture, electronic devices, etc.) along with their EoL outflows, final waste and re- and downcycling flows or rates. The total material stock of a socio-economic system also includes the human population and livestock (Haberl et al., 2019), which are not estimated in this study. The human population and livestock, however, are comparatively small, only making up 0.1 % of the global total material stock in 2015 (Krausmann et al., 2018a). In this study, we estimate stocks, outflows, waste and re-/downcycling for the four main material groups distinguished in ew-MFA: biomass, metals, non-metallic minerals and fossil energy carriers, and we distinguish thirteen stock types: concrete, asphalt, aggregates in sub-base and base-course layers, bricks (incl. stones and tiles), solidwood, paper and paperboard, plastics, iron/steel, aluminum, copper, all other metals, container and flat glass. These types comprise more than 98 % of the mass of the total material stock of manufactured capital (Krausmann et al., 2017).

The spatial boundary of this study is the geographical border of the USA and the temporal boundaries are the years 1870 and 2017. In addition, we use a spin-up period of 70 years from 1800-1870, which is necessary to derive robust initial values for stocks, outflows, re-/downcycling and final waste flows in the starting year of the period. The length of the spin-up period is based on the material stock type with the longest lifetime in the model (here: aggregates in sub-base and base-course layers, 80 years).

2.2 Dataset

2.2.1 Material Flows

Data descriptions, sources, assumptions and calculation procedures used for quantifying all exogenous parameters are described in detail in the Supplementary Information (SI). An overview of sources for all exogenous parameters is shown in Tables SI.1, SI.2 and SI.3. In general, data on stock-building materials or primary inputs to stock, processing and manufacturing losses were sourced from databases, MFA accounting guides (e.g. Krausmann et al., 2018b) and academic papers (e.g. Glöser et al. 2013, Cochran and Townsend 2010, Ruth and Dell´Anno 1997). Primary sources for the quantification of stock-building materials and primary inputs to stock were the online database of the United States Geological Survey (USGS; Kelly and Matos, 2014) and its Mineral Commodity Summaries (USGS, 2019a), which offer detailed, high-quality material flow data from 1900 to 2017.[1] In some instances, time series were extended using secondary sources such as the online databases of UN Comtrade (2019), FAOSTAT (2019) and the International Energy Administration (IEA, 2019). Historical documents from the Bureau of the Census (1949, 1975) and ew-MFA data presented in Gierlinger and Krausmann (2012) were used to extend the time series back to the 19th century. For some materials, data were available as far back as 1800 (e.g. lumber for solidwood primary inputs to stock) or the year of their introduction (e.g. aluminum was used from 1886 onwards). In cases were data were not available back to 1800, per capita material consumption was held constant at the earliest available year (e.g. 1869 for iron/steel) and multiplied with population data from the Maddison Project (2018).

In contrast to other inputs to stock (e.g. metals), material flow data for aggregates in concrete, asphalt and sub-base and base-course layers is often of poor quality in national and international sources (Miatto et al., 2016). Aggregates consumption for concrete and asphalt production were therefore estimated based on bitumen and cement consumption while demand for aggregates in sub-base and base-course layers was estimated based on construction standards for buildings and roads (Krausmann et al. 2017, 2018b). Aggregates demand for sub-base layers of buildings is assumed constant: for every tonne of concrete and bricks consumed annually, aggregates demand was assumed to be 70 kg and 45 kg, respectively. Assumptions on the demand for aggregates in sub-base and base-course layers of roads depends on asphalt use and take into account that with the expansion of the road network of the USA, an increasing share of asphalt is used for road maintenance (Miatto et al., 2017). Demand for aggregates in roads therefore decreases over time: from 2.6 tonnes (t) per t of asphalt in 1870 to 0.8 t in 2017 (see Figure SI.34). For further information on how aggregates demand is estimated see the SI.

2.2.2 Lifetimes

To calculate the average residence time of material inputs to stock in the socio-economic system of the USA, data on end-use shares for inputs to stock (e.g. concrete in buildings, roads/bridges, other infrastructures) were multiplied by their corresponding lifetimes (75, 32

[1] We do not consider trade of final products. Although this possibly causes considerable underestimations for certain stock types (e.g. metals in machinery or vehicles), the impact on overall stocks is likely small (Gierlinger and Krausmann, 2012).

and 35 years, respectively) to obtain weighted lifetimes (e.g. 46/52 years for concrete inputs to stock in 1902/2002). End-use data, however, is often fragmentary and not available for long periods of time. The USGS online database and its Mineral Commodity Summaries for example offer end-use statistics for different materials from 1975 to 2017. In some cases, information on historical end-uses was available in the literature. Cochran and Townsend (2010) for example show end-use shares for concrete for single years for 1902-2002, Chen and Graedel (2012) show end-use shares for aluminum for 1960-2009 and Spatari et al. (2005) for copper for 1845-1999. Whenever end-use shares were not available, we linearly interpolated between available data points or kept shares constant before the earliest available year. Lifetimes for different end-uses were taken entirely from the literature (e.g. from Geyer et al. 2017, IPCC 2003).

2.2.3 Recycling and Downcycling

Time series data for End-of-Life recycling flows (also referred to as secondary inputs to stock) for aluminum, copper, iron/steel, other metals and paper and paperboard was taken from the USGS database (USGS 2019a, Kelly and Matos, 2014). Data on asphalt recycling was available for the years 1993 (USDT, 1993), 1996 (Wilburn and Goonan, 1998) and 2009 to 2017 (NAPA, 2019). We assume that asphalt recycling started in 1970 (Federal Highway Administration, 2016) and linearly interpolated asphalt recycling flows between these data points. Recycling rates for glass and plastics were taken from the Environmental Protection Agency (EPA 2015, 2018a) and linearly interpolated between available data points from 1960 and 1980 to 2015. Information on recycling of concrete, solidwood, bricks, stones and tiles and aggregates in sub-base and base-course layers is scarce in the literature. We assume that concrete recycling started in 1970 (similar to asphalt) and increased to 5 % in 2017 (Krausmann et al. 2017, Kelly 1998, Wilburn and Goonan 1998). We further assume that solidwood recycling is insignificant and recycling of bricks, stones and tiles stopped in 1960. Recycling of aggregates in sub-base and base-course layers is assumed constant at 60 % (Krausmann et al., 2017).

High-quality data for downcycling of concrete, asphalt and bricks, stones and tiles to aggregates in sub-base and base-course layers is similarly scarce. For bricks, stones and tiles, we assume a constant downcycling rate of 35 % for 1800-1920, which decreases to 10 % in 1970 and then grows again to a level of 47 % in 2017 (Krausmann et al., 2017). For concrete, Wilburn and Goonan (1998) report that, 50 % of concrete debris was landfilled in 1996. The remains were either used for cement concrete (3 %), asphaltic concrete (4 %) or road base (43 %; Kelly, 1998). According to Sandler (2003), studies have placed re- and downcycling rates for concrete between 50 and 57 %. For asphalt, Wilburn and Goonan (1998) report that 8 % of asphalt pavement debris was downcycled in 1996. According to NAPA (2019), 4-12 % of reclaimed asphalt pavement from roads and parking lots was downcycled annually from 2009 to 2017. Based on these studies we assume that concrete downcycling linearly increases from 10 % in 1970 to 55 % in 2017 and asphalt downcycling linearly decreases from 10 % in 1970 to 5 % in 2017. Prior to 1970, asphalt and concrete downcycling is assumed constant at 10 % (Krausmann et al., 2017). Downcycling of materials other than asphalt, concrete and bricks, stones and tiles is not considered.

2.3 Limitations of the Approach

Uncertainties for results arise out of the chosen modelling approach and the data quality of the exogenous parameters. This study applies an inflow-driven model using material flows and average lifetimes to estimate the development of stocks and outflows over time. Since this approach operates with age cohorts of inputs which are assigned average lifetime distributions, sudden changes in the lifetimes of in-use stocks cannot be properly accounted for (Fishman et al., 2014). This includes events which abruptly decrease (e.g. wars or natural disasters such as Hurricane Katrina in 2005) or increase the lifetime of stocks (e.g. economic crises, where the in-use phase of infrastructures and durable goods might be extended) for a certain period of time. Krausmann et al. (2017) have argued in their analysis of global stocks that while massive destruction of stocks during World War II in Europe and Japan had a short-term impact, they had little long-term effects on the development and size of material stocks. The effects of economic crises and recessions (e.g. the oil price shocks in the 1970s and the financial crisis and great recession in 2007-09) on the other hand have not been fully understood yet. We will revisit this issue when discussing our results in the next section of this study.

Concerning data quality, we generally regard the quality of material flow data (stock-building materials, processing and manufacturing losses, primary inputs to stock) as good. Material flows were cross-checked between primary and secondary sources and overall showed a high level of similarity, which is in line with previous research on this matter (Fischer-Kowalski et al., 2011). Uncertainties are highest for sand and gravel in sub-base and base-course layers, because estimated demand for these aggregates depends on exact information of construction standards for roads and buildings. While data on the length of different road types in the USA is quite good, exact information on their material intensities is not always available. The US Department of Transport e.g. characterizes the high flexible pavement (HFP) road type as "having a combined surface and base thickness *of 7" (177.8 mm) or more*" (Miatto et al. 2017: SI.8; emphasis added), i.e. it is unclear how much asphalt and aggregates are used (on average) for the construction of HFP roads. HFP roads were, at the same time, the fastest growing road type for 1960-2015 (see Figure SI.29) and HFP is currently the most common road type in the USA (31 % of road km in 2015). It is therefore possible that aggregates demand for road construction was significantly different from 0.8 to 2.6 tonnes per tonne of asphalt between 1870 and 2017 (see Figure SI.34), which would result in higher or lower stocks of aggregates in sub-base and base-course layers.

In contrast to material flow data, end-use shares for the calculation of average lifetimes are often only available for the second half of the 20th century or even shorter periods of time. Information on the variability of lifetimes for different end-uses over time is similarly scarce. Studies have shown that variations in average lifetimes can have considerable implications for individual stocks (e.g. steel, aluminum), EoL outflows and recycling rates (Pauliuk et al. 2013, Chen and Graedel 2012). However, when systematically changing lifetimes (±30-50 %) for total global stocks, Wiedenhofer et al. (2019) and Krausmann et al. (2017) found a small overall uncertainty range of stock development of 5-15 %.

Data quality for recycling flows is quite good, especially for certain metals (e.g. aluminum, copper) as well as paper and paperboard. Information on downcycling of construction materials is of low quality (e.g. for concrete, asphalt). Downcycling uncertainties have no immediate implications for stocks in the MISO-model, because aggregates demand which is not fulfilled by downcycling is fulfilled by virgin or recycled aggregate flows (see the SI).

However, downcycling flows have greater significance for the estimations of EoL re- and downcycling rates. We will discuss this issue in the next section of this study.

While a full-scale sensitivity analysis goes beyond the scope of this study, we have applied several cross-checks comparing the estimates of this study with results from other studies. A summary of these comparisons is presented in section 4, detailed comparisons can be found in the SI. In general, we found either a high degree of consistency or reasonable explanations for observed differences. Despite the stated limitations, we are convinced that our results are robust and the most detailed and comprehensive historical account of the total in-use material stock of the USA currently available.

3 Results

3.1 Material Inputs to Stock

From 1870 to 2017 domestic material consumption (DMC; equals the domestic extraction of materials plus imports minus exports; Gierlinger and Krausmann, 2012) grew 14-fold from 0.4 Gigatonnes/year (1 Gt = 10^9 metric tonnes) in 1870 to 5.4 Gt/yr in 2017 (Figure 2). The share of stock-building materials in DMC grew from 15 % in 1870 to 49 % in 1973 and then declined to 35 % in 2017. The composition of stock-building materials changed substantially over time. In 1870, stock-building materials were predominantly biomass (industrial roundwood). In 2017 non-metallic minerals made up 68 % of stock-building materials. Deducting processing losses from stock-building materials we arrive at primary inputs, which grew 47-fold from 0.03 Gt/yr in 1870 to 1.41 Gt/yr in 2017 (see also Figures SI.15, SI.16).

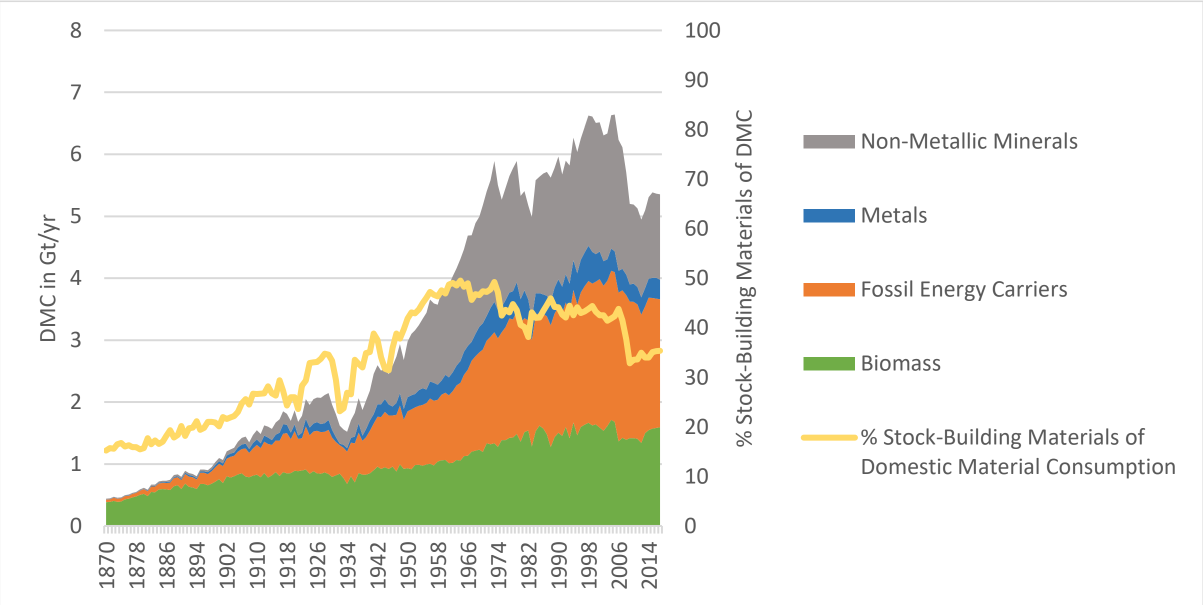

Figure 2: Development of domestic material consumption (DMC) and the share of stock-building materials in DMC in the USA from 1870 to 2017. Stock-building materials include all materials which remain in the socio-economic system on average for longer than one year, thus excluding energy and other dissipative uses. Sources: data on biomass and fossil energy carriers for 1870-2005 is from Gierlinger and Krausmann (2012) and for 2005-2017 from UNEP (2019), metals, non-metallic minerals and stock-building materials are from own calculations.

From 1870 to 2017, cumulative material consumption of the economy of the USA was 468 Gt of materials, of which 184 Gt were stock-building materials (39 %). The largest share of stock-building materials were non-metallic minerals (125 Gt, 68 %) followed by metals and ores (31 Gt, 17 %), biomass (25 Gt, 14 %) and fossil energy carriers (3 Gt, 2 %). Cumulative primary inputs to stock from 1870 to 2017 were 139 Gt while 45 Gt were lost during processing of raw materials (as solid waste or emissions, e.g. waste rock from ore processing, CO_2 emissions from cement production or loss of moisture content in bricks production). Cumulative manufacturing losses from 1870 to 2017 were only 2 Gt.

3.2 In-Use Stocks

Over the whole period, total stocks grew 160-fold, from 0.6 Gt in 1870 to 96 Gt in 2017 (Figure 3). During the same period, per capita stocks increased from 16 metric tonnes (t) in 1870 to 295 t in 2017. Approximately 70 % of all inputs which entered the stock of the USA from 1870 to 2017 were still in-use in 2017, while 30 % were discarded and either landfilled, shipped to other treatment facilities (e.g. incineration), exported or remain in place as hibernating stocks. The biggest stocks in 2017 were sand and gravel in sub-base and base course layers (44.3 Gt, 136 t/cap), concrete (27.8 Gt, 85 t/cap) and asphalt (13.6 Gt, 42 t/cap), followed by solidwood (4.4 Gt, 13 t/cap), iron/steel (3.5 Gt, 11 t/cap), bricks, stones and tiles (1.1 Gt, 3 t/cap), plastics (0.6 Gt, 2 t/cap), paper (0.24 Gt, 0.7 t/cap), glass (0.16 Gt, 0.5 t/cap), aluminum (0.15 Gt, 0.5 t/cap), other metals (0.14 Gt, 0.4 t/cap) and copper (0.08 Gt, 0.3 t/cap).

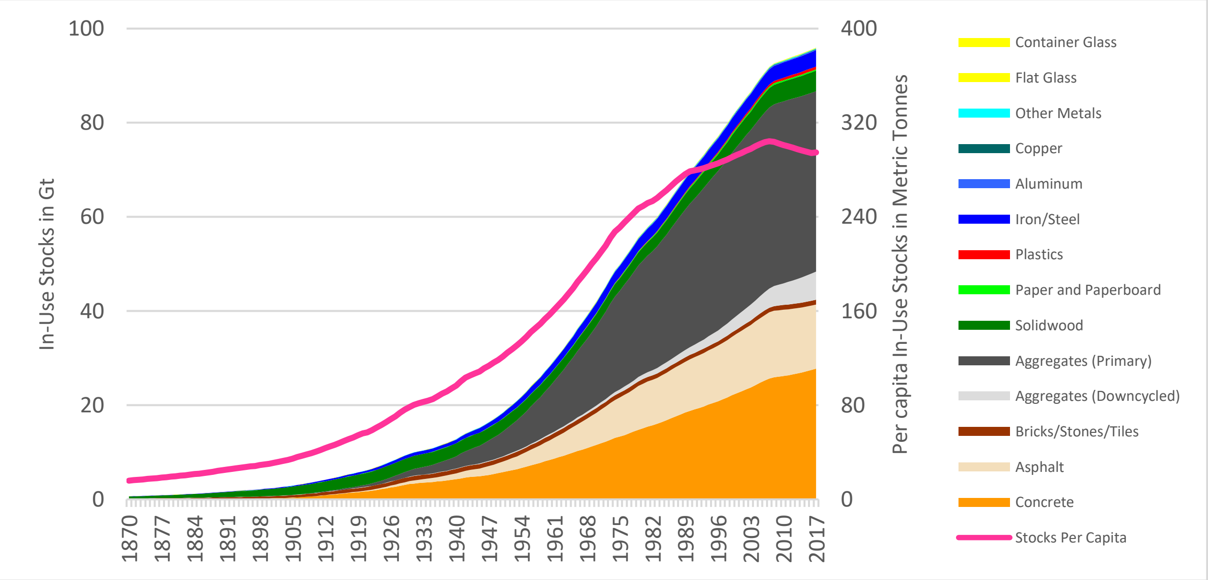

Figure 3: Development of total and per-capita stocks of manufactured capital in the USA from 1870 to 2017 by 14 stock types. Total stocks are shown on the left axis per capita stocks on the right axis. Sources: total stocks are from own calculations, population data from the Maddison Project (2018) and Bureau of the Census (2019a).

3.3 Net Additions to Stock

The development of the material stock of the USA reflects one and a half centuries of urbanization, industrialization and economic growth (Krausmann et al., 2017). Net additions to stock (NAS, equal to primary and secondary inputs to stock minus manufacturing losses minus End-of-Life outflows) initially were 0.03 Gt/yr in 1870 and steadily increased to 0.4 Gt/yr in 1929 (see Figure 4). Stock growth during this period was especially strong during the so-called roaring twenties, a decade of economic prosperity in the USA. The stock market crash of 1929 and subsequent great depression marked the end of this period, resulting in a strong decline of NAS. However, NAS started to rise again already from 1933, supported by various policy measures of the New Deal (Gierlinger and Krausmann, 2012) and continued to increase until 1941 when the USA entered WWII. After WWII, NAS growth accelerated until the first oil price shock, reaching an all-time high of 1.85 Gt/yr in 1973. After three decades of accelerating stock growth, the USA then entered a period of more linear stock growth from 1973 to 2007. NAS severely fluctuated during this period and frequently decreased in times of economic crises such as the oil price shocks in 1973 and 1979, the recession in 1990 and the dot-com crash in 2000. Nevertheless, NAS fluctuated around a high average of 1.3 Gt/yr between 1973 and 2007. The rapid stock expansion during these two periods (1945-1973 and 1973-2007) went along with large infrastructure projects (e.g. buildings and road construction), the emergence of mass production and consumption, economic growth and rising material living standards (Gierlinger and Krausmann, 2012).

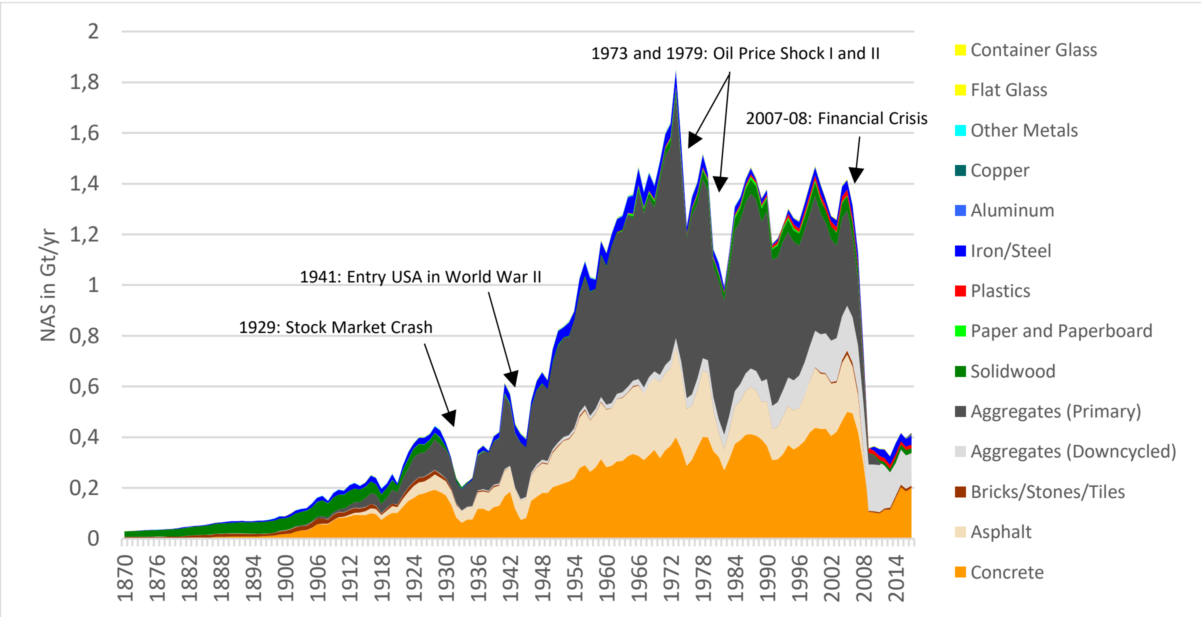

Figure 4: Development of annual net additions to stock (NAS) in the USA from 1870 to 2017 by 14 stock types. NAS are equal to primary inputs to stock plus secondary inputs to stock (recycled materials) minus manufacturing losses minus End-of-Life outflows. NAS are given in Gigatonnes/year. Source: own calculations

Similar to the oil crisis in 1973 (end of accelerated stock growth), the financial crisis and the great recession from 2008 to 2009 mark another turning point for stock growth in the USA. From 2007 to 2009, NAS plummeted to the level of 1945 (0.4 Gt/yr) and stayed low until 2017. The massive decline in NAS resulted in a slowdown of stock growth below the level of population growth; as a consequence, per capita stocks even declined (Figure 3). The impact of NAS decline on stock growth may, however, be overestimated if stocks remained longer in use before replacement during and after the economic crisis than assumed in the average life times in our model. It is also possible that lifetimes of stocks are higher than assumed, e.g. for concrete, asphalt and sand and gravel in sub-base and base-course layers which formed 86 %

of EoL outflows in 2017 and accounted for 87 % of the decline in NAS from 2007 to 2009. Longer lifetimes would ceteris paribus result in less outflows and consequently higher NAS. Rising outflows on the other hand could also be a sign of deteriorating infrastructures: stocks could still be in use but in a deficient condition. These issues should be further explored by future studies.

3.4 Recycling, Downcycling and Final Waste

At the end of their lifetime, discarded stocks become End-of-Life (EoL) outflows. EoL outflows grew roughly 570-fold from 0.003 Gt/yr in 1870 to 1.7 Gt/yr in 2017 and were either re- or downcycled to secondary inputs to stock or they were released to the environment as final waste. We distinguish two recycling rates here, which offer complementary views on the circularity of the economy of the USA: the EoL recycling rate, which is defined as the share of recycled flows in EoL outflows and the recycling input rate which is defined as the share of recycled materials in primary and secondary inputs to stock (Krausmann et al., 2017). Exported secondary materials are not considered in these recycling rates.

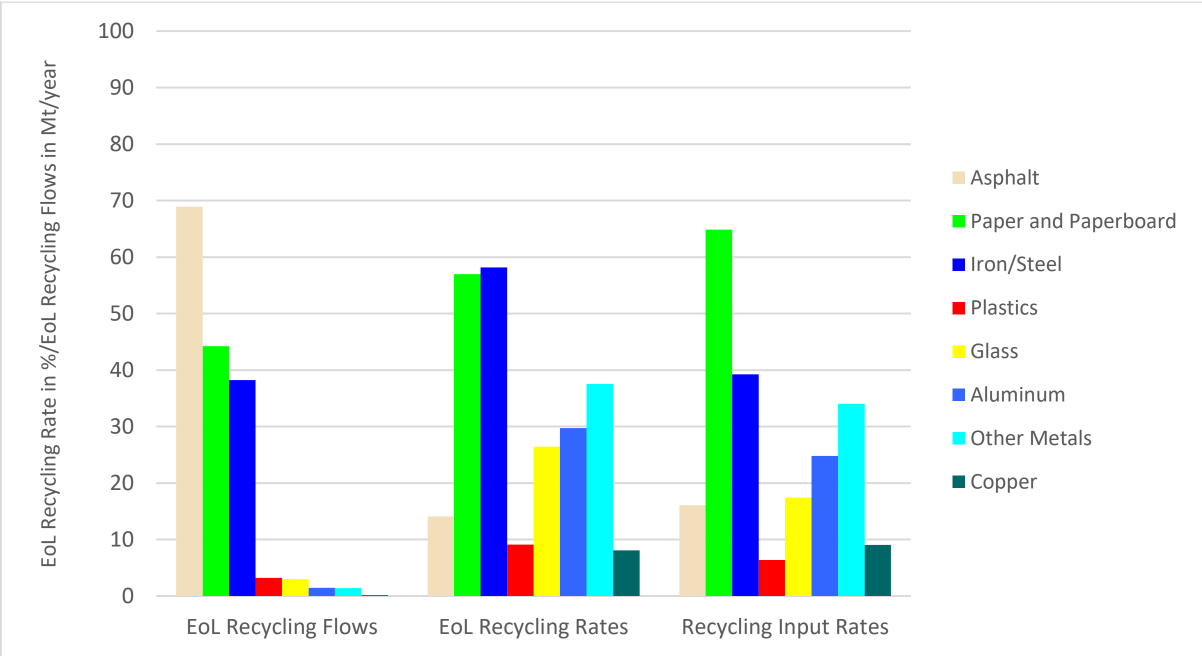

Figure 5: End-of-Life (EoL) recycling flows, EoL recycling rates and recycling input rates in the USA in 2015. The EoL recycling rate is defined as the share of recycled materials in EoL outflows of the same material. The recycling input rate is defined as the share of recycled materials in primary and secondary inputs to stock of the same material. The unit for the EoL recycling rates and the recycling input rates is %, recycling flows are given in Megatonnes/year. Both units are depicted on the left axis. Sources: recycled EoL asphalt flows are from NAPA (2019), paper and paperboard and all secondary metal flows are from Kelly and Matos (2014) and USGS (2019a), glass and plastics EoL recycling rates are from EPA (2018a), all other values shown are from own calculations.

By mass (Figure 5), asphalt was the most recycled material in 2015 (69 Megatonnes/year = 10^6 metric tonnes) and copper was the least recycled material (0.2 Mt/yr). EoL recycling flows per year increased substantially from 1960 to 2015 (Figure 6). EoL glass recycling flows, for example, grew 18-fold from 1960 to 2015 and plastics recycling flows grew 195-fold from 1980 to 2015. Recycled copper flows on the other hand declined by 57 % from 1960 to 2015. The USA have been a net exporter of copper scrap from 1982 onwards (USGS 2019b, Nathan Associates 2004). For 2004, Goonan (2009) reports that net exports of EoL copper scrap were 0.63 Mt/yr, which was more than triple the amount of recycled copper (0.19 Mt/yr). The large exports of copper scrap result in low domestic EoL recycling of copper.

Comparing recycling rates of different material stock types shows a different picture (Figure 5). EoL recycling rates were highest for iron/steel (58 %), paper and paperboard (57 %) and other metals (38 %) in 2015.[2] EoL recycling rates were below 30 % for the other materials and even below 15 % for asphalt, plastics and copper. Interestingly the input recycling rate was higher than the EoL recycling rate for paper and paperboard, asphalt and copper, which reflects a decline of stocks in the year 2015.[3]

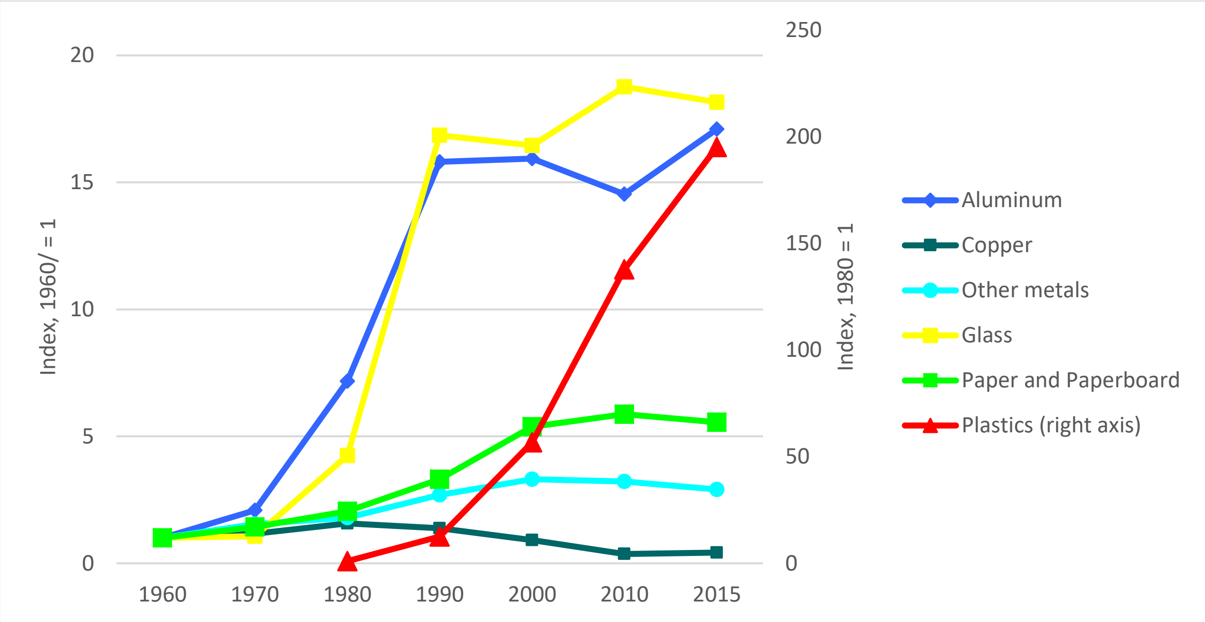

Figure 6: Development of End-of-Life recycling flows for selected material stock types from 1960 and 1980 to 2015. Values normalized to 1960 = 1 for aluminum, copper, other metals, glass and paper and paperboard (left axis), values normalized to 1980 = 1 for plastics (right axis). Sources: Kelly and Matos (2014), own calculations

Total EoL recycling flows were 0.52 Gt and total downcycling flows were 0.23 Gt in 2015. The total EoL re- and downcycling rate (total re- and downcycled flows for all stocks as a share of total EoL outflows) was 44 % in 2015 and the total re- and downcycling input rate (total re- and downcycling flows for all stocks divided by total primary and secondary inputs to stock) was 35 %. However, recycling flows for aggregates and downcycling flows are considerable (0.58 Gt) but at the same time exhibit high uncertainties. If re- and downcycled aggregates are excluded in total re- and downcycling flows, outflows and inputs to stock, the total EoL recycling rate was 16 % and the total recycling input rate was 13 % in 2015. Both recycling rates indicate a low level of loop closing in the economy of the USA.

[2] Within the other metal's category, the EoL recycling rate was highest for lead (98.7 % in 2015 according to EPA, 2018b).

[3] The input recycling rate can only be higher than the EoL recycling rate when EoL outflows are higher than total inputs to stock (EoL outflows > primary inputs + secondary inputs), which is equivalent to a decline of stocks.

Deducting recycling and downcycling from EoL outflows, we obtain the resulting final waste flows. Final waste from end of life stocks grew from 0.003 Gt/yr in 1870 to 1 Gt/yr in 2017. Waste composition changed considerably over time. The largest waste type in 1870 was solidwood (50 %), in 2017 it was asphalt (41 %). In total, 41.5 Gt of EoL outflows became final waste from 1870 to 2017. This includes 30.3 Gt of non-metallic minerals, 7.1 Gt of biomass, 2.6 Gt of metals and 1.4 Gt of fossil energy carriers.

3.5 Socio-Economic Indicators

In this section we show how different socio-economic indicators – residential housing units, commercial buildings, road length, Gross Domestic Product (GDP) and population – have developed in comparison to material stocks. We present data for the period of 1940 to 2010 because comparable figures were available for the considered indicators. Figure 7 shows the growth of the chosen indicators over time, indexed at 1 in 1940. Note that the period for commercial buildings is from 1946 to 2012.

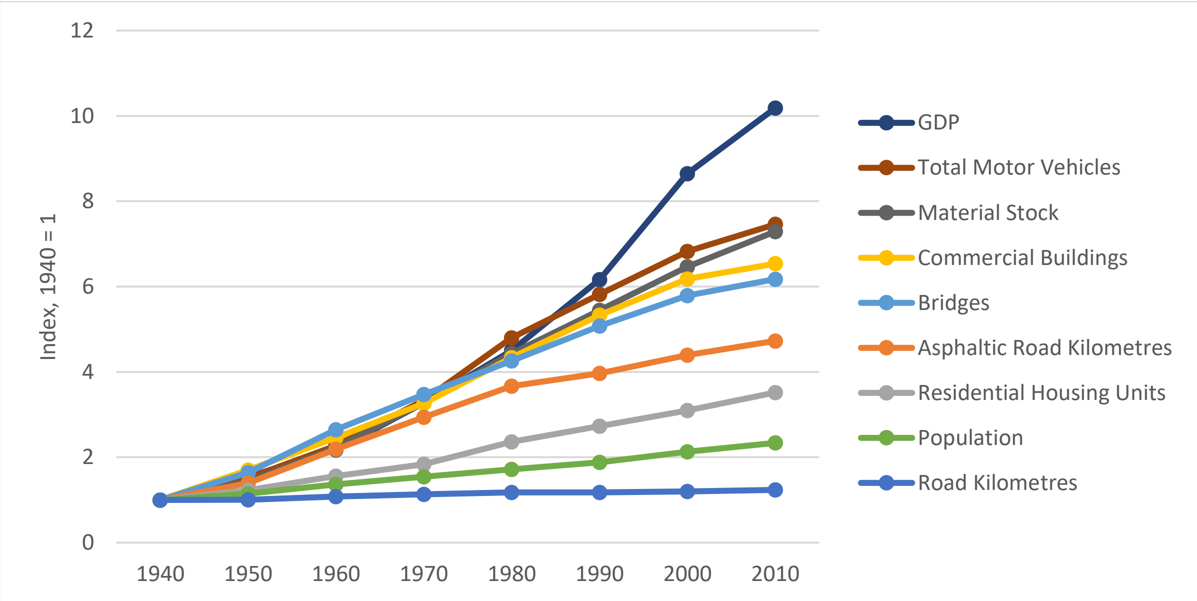

Figure 7: Stocks and socio-economic development: Development of material stocks, residential housing units, commercial buildings, road length, total motor vehicles, bridges, GDP, population and GDP per capita in the USA from 1940 to 2010. Values are normalized to 1940 = 1. GDP and GDP per capita are in constant international dollars of 2011. Note that the period for commercial buildings is from 1946 to 2012 (1946=1940, 2010=2012). Sources: residential housing units are from Bureau of the Census (2012), commercial buildings from EIA (2012), road length from Miatto et al. (2017), bridges and total motor vehicles from the Federal Highway Administration (2018), GDP and population data are from the Maddison Project (2018) and stocks are from own calculations.

Stocks have grown much faster than population from 1940 to 2010, as a result per capita stocks have also strongly increased (Figure 3). GDP has grown roughly in line with stocks until 1980, relative decoupling of stocks from GDP can only be observed from 1980 onwards. The growth of stocks is also reflected in the quantitative growth of infrastructures (e.g. buildings, roads) and durable goods (e.g. vehicles) in the USA. Infrastructures and buildings, however, not only changed in quantitative, but also in qualitative terms. While total road length increased only 1.2-fold, asphaltic road length increased 4.7-fold, due to a rapid improvement of roads to cope with the rising speed and loads of vehicles (Miatto et al., 2017). The size of residential housing units and commercial buildings also increased. The average size of a newly constructed single-family house was 154 m² in 1973 and grew to 222 m² in 2010 (Bureau of the Census, 2019b). The average size of commercial buildings was 1094 m² for buildings constructed from 1946 to 1959 and increased to 1755 m² for buildings constructed from 2008 to 2012 (EIA, 2012). Stocks in infrastructures, buildings and durable goods deliver services, which are crucial for the societal wellbeing in the USA but also need energy to deliver these services. We discuss the connection between stocks, services and energy consumption in the next section of this study.

4 Discussion

4.1 Comparison with the results from other studies

The study by Fishman et al. (2014) so far is the only other study which has estimated total stocks for the USA for 1930-2005. Figure 8 shows a comparison between their estimates and the results of this study. From 1930 until 1970, the development and size of total stocks is almost identical. The gap between Fishman et al. and MISO results then increases from 2.5 Gt in 1970 to 18.1 Gt in 2005, largely because of an increasing gap in the estimates for non-metallic minerals. Fishman et al. (2014) use ew-MFA data by Gierlinger and Krausmann (2012) to estimate stocks. Gierlinger and Krausmann (2012) assume that for every tonne of asphalt consumed annually, demand for aggregates in sub-base and base-course layers in roads is 0.5 tonnes. This should actually lead to a lower stock for Fishman et al. (2014), because demand for aggregates in roads, although decreasing over time, is always higher than 0.5 tonnes per tonne of asphalt in this study. Therefore, we tried to replicate the figures of Gierlinger and Krausmann (2012) for aggregates consumption and find that reported and estimated figures differ significantly from 1945 onwards (Figure SI.63). Although we cannot explain these differences, material inputs to stock for non-metallic minerals for 1970-2005 are therefore significantly higher in Fishman et al. (2014) which largely explains the increasing gap for total stocks from 1970 (see also Figure SI.64).

Miatto et al. (2017) quantified the in-use material stocks (rammed earth, sand and gravel, cement, bitumen) in the road network of the USA from 1905 to 2015. They employ a bottom-up methodology and used information on road length and construction standards for different road types to estimate total stocks in roads. When comparing our estimates for asphalt and sand and gravel in sub-base and base-course layers (Figure SI.44 and Figure SI.45) to their estimates, we find a similar development for these stocks over time but also that our results are 4 times (aggregates) and 9 times (asphalt) higher in 2017. As construction standards for several road types in the USA are unknown and these are likely the roads with the highest material intensities and longest road length, we assume that the bottom-up methodology of Miatto et al (2017) has resulted in an underestimation of total stocks in roads.

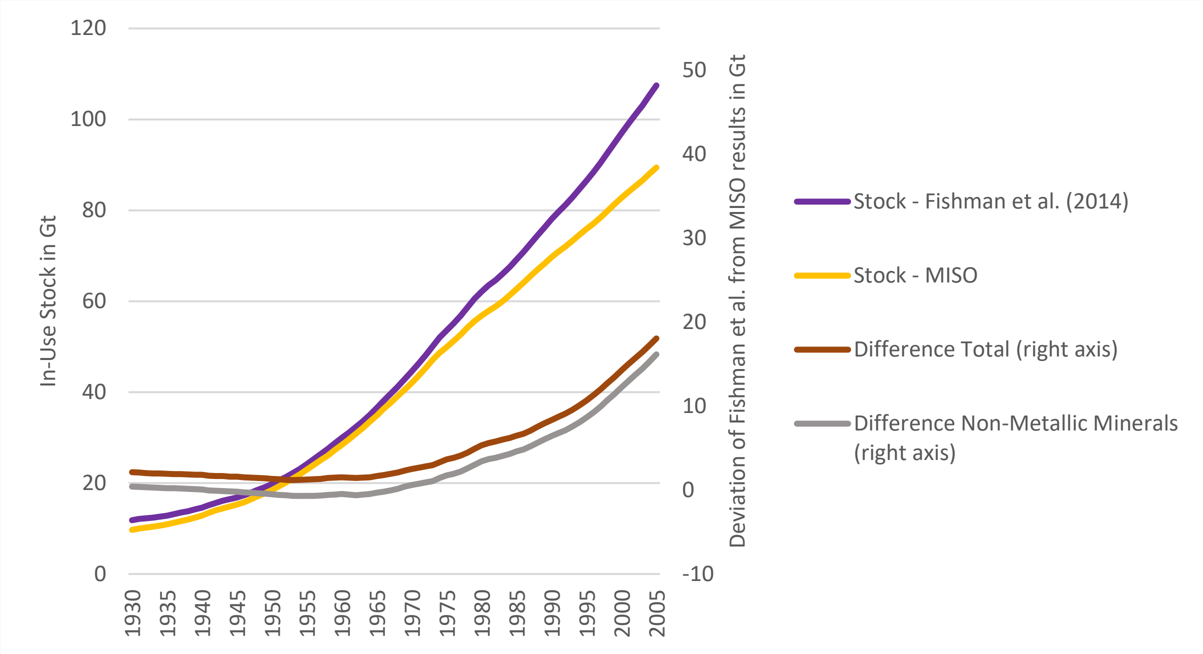

Figure 8: Comparison of material stocks from 1930 to 2005 from this study (MISO) and Fishman et al. (2014). The differences between total material stocks and non-metallic minerals stocks are calculated as stocks of Fishman et al. minus MISO-stocks (right axis). In MISO estimates, non-metallic minerals stocks include sand and gravel in asphalt, concrete, aggregates, bricks, stones and tiles and glass. Comparisons for different material groups (biomass, metals, non-metallic minerals) can be found in the SI. Sources: Fishman et al. (2014), own calculations

Estimates for individual stocks, EoL outflows, recycling and final waste were also compared against results available from other studies. The comparison in Figure 9 shows that results for metals and concrete stocks as well as EoL outflows and recycling flows for other materials (bricks, glass, solidwood, plastics) generally agree well with other sources. Most differences are within a range of ±30 % (Liu and Müller, 2013). The 57 % lower aluminum stock in 2007 by McMillan et al. (2010) may be explained due to shorter lifetimes used in their study (Chen and Graedel, 2012). The methodological details used by the studies which estimated copper stocks for 1948, 1957 and 2003 and iron stocks for 1950 could not be identified. Therefore, we cannot explain the larger differences. Values for glass and plastics recycling flows and EoL outflows differ by greater margins before 1990 but are robust for recent years. Further comparisons between MISO estimates and the results of other studies can be found in the SI.

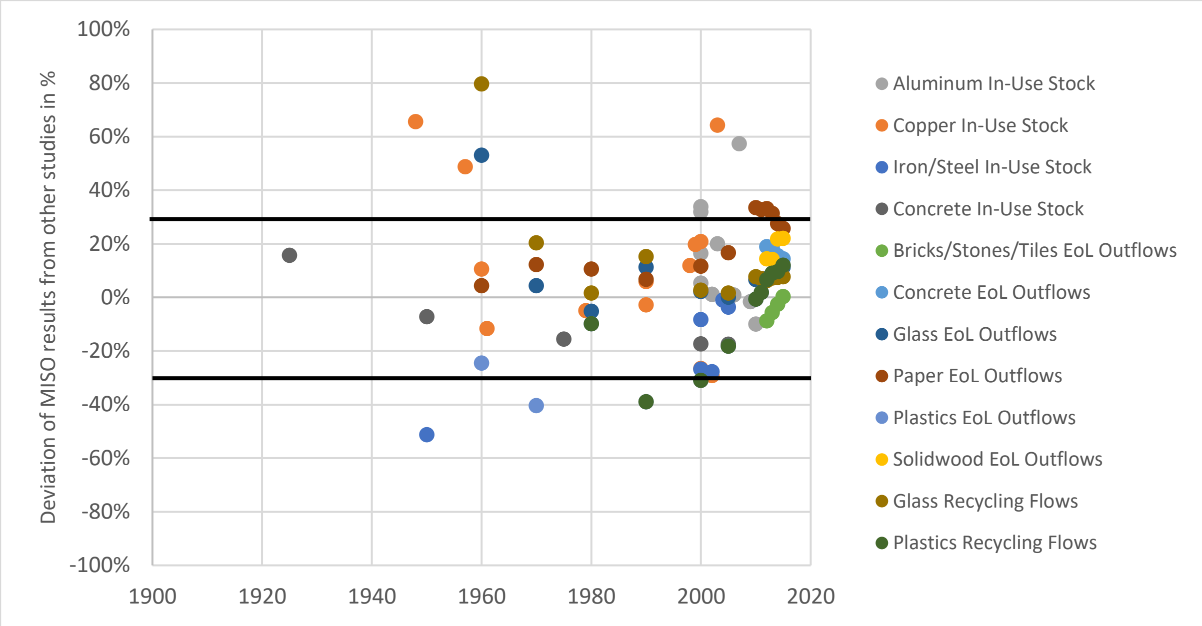

Figure 9: Deviation of MISO-results from other studies. Positive and negative values indicate that MISO results are higher and lower by x % than the results of other studies, respectively. The black lines indicate a deviation of ±30 % and show that most of the compared results are within this range. Sources: figure is based on Tables SI.5-7 and Figures SI.42, 47-48, 50-53 and 59-60. Comparisons for time series data can be found in the SI.

4.2 How big are stocks, GDP, CO_2-emissions and population in comparison to the world?

The stock of the USA constituted 10 % of the global stock in 2014. On a per capita basis, our stock estimate for 2010 (301 t/cap) matches well with the estimate for industrialised countries (335 t/cap) in Krausmann et al. (2017). Per capita stocks in 2017 for iron/steel (11 t), aluminium (0.5 t) and concrete (85 t) also represent typical estimates for industrialised countries (Cao et al. 2017, Liu and Müller 2013, Pauliuk et al. 2013). The per capita stock of the USA in 2010 was 2.2 times higher than the Chinese per capita stock, 2.6 times higher than the world average per capita stock and 7.9 times bigger than the average per capita stock of the rest of the world (excl. China, ind. countries; Krausmann et al., 2017).

The development of the US share of global stocks followed an inverse u-shaped trend from 1900 to 2014 (Figure 10). The share of US stocks in global stocks increased from 7 % in 1900 to a peak in 1959 at 24 % and then decreased to 10 % in 2014. Compared to the share of the USA in global population (4-6 %) during the whole period, the share in global stocks is over proportionally high, which resembles the share of the USA in global GDP. The USA were a major economic player throughout the 20th and 21st century. The US share of world GDP was 13 % in 1900, increased to 23 % in 1950 and then declined to 15 % in 2014. CO_2 emissions produced in the USA followed a similar trend as GDP but at a much higher level. In 1900, the US share of CO_2 emissions from fossil fuels combustion and cement production was 36 %, increased to a peak of 55 % in 1945 and then declined to a share of 15 % in 2014.

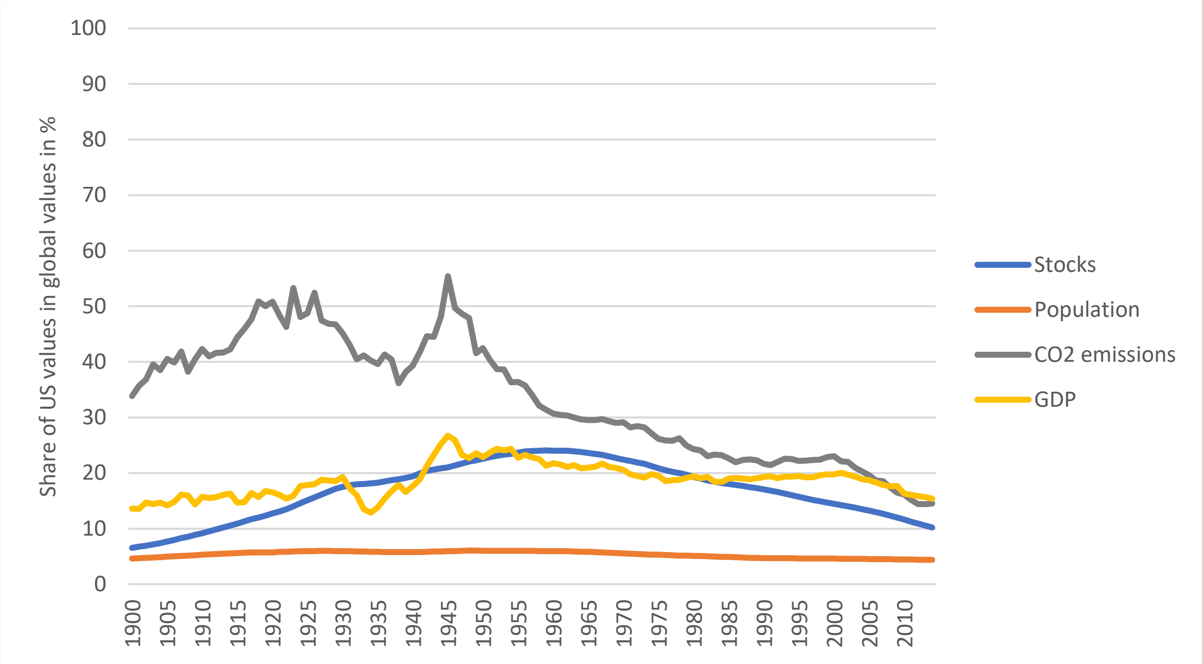

Figure 10: Share of the USA in global material stocks, population, GDP in constant international dollars of 2011 and CO_2 emissions from fossil fuel combustion and cement production from 1900 to 2014. World GDP was linearly interpolated between available data points for 1900-1913, 1914-1939 and 1941-1950. Sources: global stocks are from Wiedenhofer et al. (2019), USA stocks are from own calculations, global population data is from Roser et al. (2013), USA population data is from the Maddison Project (2018), global GDP is from Roser (2018), USA GDP is from the Maddison Project (2018), global and USA CO_2-emissions are from the CDIAC (2017).

These figures underline the dominant position of the USA in the global socio-economic system in the first half of the twentieth century. When European countries and Japan began catching up after WWII due to large-scale investments for reconstruction, economic growth and urbanisation, the dominance of the USA in global GDP and emissions and with some delay also in stocks declined. From 1990 onwards, growth of stocks, GDP and emissions in China accelerated, driving in a further decline of the shares of the USA (Krausmann et al. 2017, Fishman et al. 2014). In absolute terms, however, the USA still outranks most other countries. The USA is the second largest emitter of CO_2 and GHG emissions (UNEP 2018, CDIAC 2017) and has the largest GDP and the third largest population (Maddison Project 2018, Roser 2013). The USA produced 25 % of all anthropogenic CO_2 emissions from 1751 (the largest national share) and consumed 436 Gt of materials from 1900 to 2015, which equals 13 % of global extraction of materials (3400 Gt) during this period (Krausmann et al. 2018a, Ritchie and Roser 2017).

4.3 How have stocks developed in relation to energy demand and CO_2 emissions?

Material stocks in specific combinations with resource flows are necessary for socio-economic systems, as they enable “human activities and societal development by providing services for societal wellbeing” (Haberl et al. 2017: 2). To provide these services, stocks need to be produced, maintained and eventually replaced. This requires a continuous inflow of stock-building materials. However, stocks not only drive the demand for stock-building materials and the corresponding recycling and waste flows. Practically all technical energy produced is, in one way or another, related to stocks (Krausmann et al., 2017). Energy is needed to produce, maintain and dispose stocks and other short-lived products, and to provide services from stocks, e.g. through lighting and heating residential buildings (shelter), operating railways and cars (mobility) or guaranteeing access to the internet (communication). Energy consumption in the USA is, at the same time, mainly provided by thermal conversion of fossil fuels (77.6 % of primary energy production in 2017; EIA, 2018a). Fossil fuel combustion produces CO_2 emissions, which are the main cause of anthropogenic climate change (82 % of GHG emissions in the USA were CO_2 emissions in 2017, the global share was 76 % in 2010[4]; EPA 2018c, IPCC 2014). Energy use thus directly connects stocks to climate change. The manufacturing of stocks produces additional emissions (e.g. CO_2 emissions during cement production or from incineration of waste, EPA 2018c).

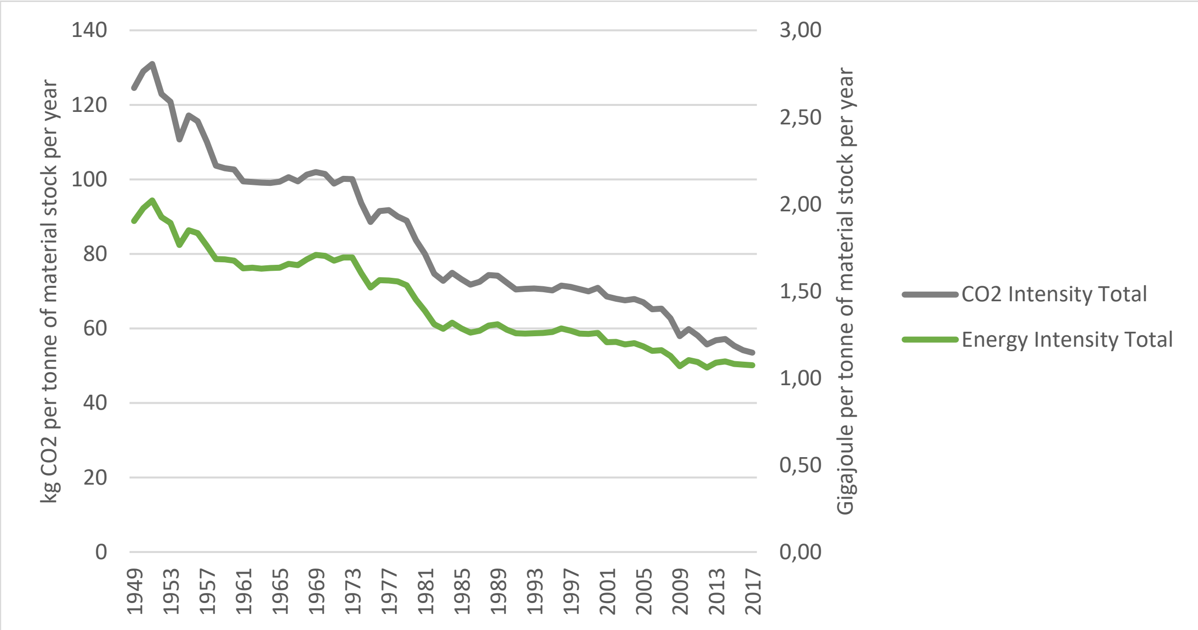

Figure 11: Development of energy intensity (total primary energy consumption/total stocks) and carbon intensity (CO_2 emissions from total energy consumption/total stocks), 1949-2017. Sources: EIA (2019a), own calculations

The Energy Information Administration (EIA, 2019a) provides a breakdown of primary energy consumption by 4 main sectors: industry, transport, residential and commercial.[5] The industry sector comprises all energy "used for producing, processing, or assembling goods" (EIA, 2019b). The transport sector includes all energy used in "vehicles whose primary purpose is transporting people and/or goods from one physical location to another" (EIA, 2019c).[6] Energy

[4] About 65 % of global GHG emissions were CO_2 emissions from fossil fuel combustion and industrial processes in 2010; in the USA 76 % of GHG emissions were CO_2 emissions from fossil fuel combustion in 2017 (EPA 2019, IPCC, 2014).

[5] The EIA also breaks down primary energy consumption of the electricity sector, which is used to produce electricity and useful thermal output for the other sectors (EIA, 2019a).

[6] In this sector all "automobiles; trucks; buses; motorcycles; trains, subways, and other rail vehicles; aircraft; and ships, barges, and other waterborne vehicles" (EIA, 2019b) are included. Vehicles on the other hand "whose

in the residential and commercial sectors is used for buildings (e.g. space and water heating, lighting, running appliances; EIA 2019d). Energy use of the industrial sector can be allocated to the extraction and processing of stock-building materials (e.g. limestone), the manufacturing of stocks (e.g. construction, manufacturing of machinery and durable goods) and the production of other goods (e.g. oil, food) while the energy consumption of the transport, commercial and residential sectors can be allocated to the use-phase of stocks (i.e. service provisioning like mobility, shelter). Over the last 70 years, the share of energy consumption for services has strongly increased. In 1949, 54 % of energy (18 Exajoule/year, EJ/yr) was used for services, by 2017 it was already 67 % (69 EJ/yr; see also Figures SI.61-62). Energy for service provisioning was responsible for 55 % of GHG emissions in 2017 (3.6 Gt CO_2-equivalents, CO_2-eq.), industrial energy accounted for 20 % (1.3 Gt CO_2-eq.) and other emissions[7] for 25 % of GHG emissions (1.6 Gt CO_2-eq.; EPA, 2019).[8]

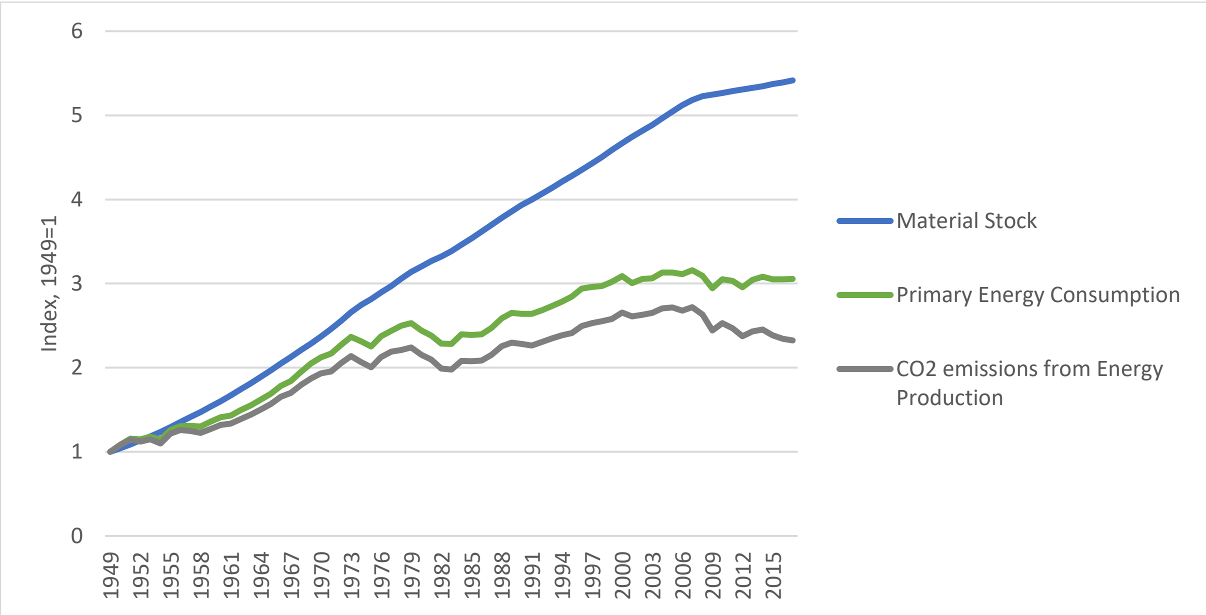

Figure 12: Development of stocks in relation to total primary energy consumption and CO_2-emissions from energy consumption in the USA, 1949-2017. Values normalized to 1949 = 1. Sources: EIA (2019a), own calculations

Because almost all technical energy is related to stocks, we can compare stock size, energy consumption and CO_2 emissions to determine how energy and carbon intensities (energy consumption and emissions per unit of stock) have developed over time. Primary energy consumption per unit of stock has strongly declined, from 1.9 Gigajoule/tonne of stocks/year (GJ/t/yr) in 1949 to 1.1 GJ/t/yr in 2017 or a reduction of 42 % (Figure 11).[9] Total CO_2 emissions

primary purpose is not transportation (e.g., construction cranes and bulldozers, farming vehicles, and warehouse tractors and forklifts) are classified in the sector of their primary use" (ibid.).

[7] Other emissions include e.g. CO_2 emissions from cement production and CH_4 emissions from landfills or from rice cultivation.

[8] Primary energy consumption for services in the USA in 2017 is equivalent to 11.6 % of global primary energy consumption in 2016 (EIA, 2018b). The share of GHG emissions of the USA in global total GHG emissions was 13.1% in 2017 (UNEP, 2018).

[9] For comparisons: the energy intensity of global stocks (total primary energy supply/stocks) was 0.7 Gj/t/yr in 2010 (Krausmann et al., 2017). This lower figure resides well with the fact that US houses are larger, household appliances more inefficient and driven distances by car higher than in other countries around the globe (Gierlinger and Krausmann, 2012).

per tonne of stock were 125 kg CO_2 in 1949 and decreased to 53 kg CO_2 in 2017 (-58 %). Emission intensities decreased more strongly than energy intensities because the USA switched to energy sources with lower emission factors per GJ produced (e.g. from coal to oil, natural gas and renewables, EIA 2011). Decreasing intensities have resulted in a long-term relative decoupling of stocks from energy and CO_2 emissions (Figure 12). Stocks, energy use and emissions grew by factors of 5.4, 3.1, 2.3 respectively from 1949 to 2017. Interestingly, CO_2 emissions declined from 2009 to 2017, while material stocks and energy use continued to grow (absolute decoupling). Factors for declining CO_2 emissions during this period include mild weather in some years and increasing shares of natural gas and renewables in primary energy consumption (EPA 2019, EIA 2018a). Parts of declining CO_2 emissions during this period can also potentially be attributed to rising net imports of emission-intensive products (UNEP 2019, Wiedmann et al. 2015; Figure SI.65).

4.4 The role of stocks in reconciling societal wellbeing and ecological sustainability

It appears that stocks present a dilemma for the USA. On the one hand, stocks deliver services, which are crucial for the societal wellbeing in the USA. On the other hand, the production and use of these stocks are responsible for more than three quarters of the GHG emissions of the USA, which must be significantly reduced to keep global warming below 1.5 degrees until the end of the 21st century (IPCC 2014, 2018). The key challenge for the USA is therefore to decrease its stock-related emissions while ensuring a sufficient amount of services for an increasing population (Bureau of the Census, 2018).

Changing stock-production patterns and making services provided from existing in-use stocks more energy efficient appear as possible pathways. Retrofitting (increasing the energy efficiency of existing buildings) can lead to substantial reductions in energy use and thus CO_2 emissions (IPCC, 2014). Recycling of materials significantly reduces energy requirements for the production of many stock types. Producing a tonne of secondary aluminum, copper, iron/steel or paper for example requires 5 %, 25-37 %, 61-84 and 53 % of the energy needed for producing a tonne from raw materials respectively (Bureau of International Recycling 2008, 2016). The circularity of the economy of the USA is still quite low: 16 % of EoL outflows were recycled in 2015 and 13 % of inputs to stock were recycled materials (excluding aggregates). Recycling rates are quite high for paper and iron/steel but are still low for aluminum, copper, other metals, glass, asphalt and plastics. One explanation for the low recycling rates for asphalt are legal requirements, which only allow certain shares of recycled asphalt in asphalt production (e.g. 15 % in surface courses in Alabama, 30 % in Arizona and 70 % in Arkansas; Federal Highway Administration, 2016). Large exports of copper scrap have contributed to a lower domestic recycling rate. This is also the case for aluminum, for which the USA have been a scrap net exporter from 1980 to 2017 (USGS 2019c, Chen 2013). Other factors that possibly contribute to low recycling rates include product designs which make separation of materials difficult or impossible, a low awareness for loss of resources, missing recycling incentives due to low values of discarded materials and a lack of recycling infrastructures or technologies (UNEP, 2011a). Implementing policies that increase recycling (e.g. mandatory product designs, landfill restrictions, public campaigns; Plastics Europe 2018) are therefore of key importance for a circular economy and climate change mitigation.

Furthermore, iron/steel (3.5 Gt) aluminum (0.15 Gt) and copper (0.08 Gt) accumulated in in-use stocks are currently greater than domestic natural reserves of the USA (0.76 Gt, 0.01, 0.05 Gt respectively; USGS, 2018). Additionally, large amounts of metals (a total of 2.4 Gt of iron/steel, aluminum, copper final waste from 1870 to 2017) went to controlled and uncontrolled landfills, other treatment facilities, export or remain in place as hibernating stocks. At least a part of these materials could potentially be recovered and recycled in the future. These figures indicate a promising potential for urban mining (Chen and Graedel, 2012). Information on the exact whereabouts of final waste is, however, usually scarce and often unavailable (Powell and Chertow 2019, Fishman et al. 2014). Chen (2013) for example estimates that the location of 76 % of aluminum waste produced from 1900 to 2003 in the USA is unknown. Deeper knowledge about the location of urban and hibernating stocks through "enhanced government support for data acquisition and analysis, recycling technologies research, and other research and development efforts" is thus "a key strategy in moving towards sustainable metal supply" (UNEP 2011b: 17-29) and important for climate change mitigation and resource conservation.

However, making stock production and use less energy intensive is possibly not sufficient to achieve climate change goals. Krausmann et al. (2017) for example estimate that a convergence of global stocks at the level of per capita stocks prevailing in industrialised countries, coupled with a strong decline of the energy intensity of stocks (-52 % between 2010 and 2050) would result in cumulative emissions of 1987 Gt CO_2 from 2010 to 2050. In the event of full decarbonization until 2050, which is considered unlikely e.g. by the International Energy Agency, cumulative CO_2 emissions would amount to 1111 Gt CO_2. A convergence of global stocks at 132 t/cap along with a 52 % decrease in the energy intensity of material stocks would induce emissions of 1107 Gt CO_2 or, in the event of full decarbonisation, 689 Gt CO_2. Only this last scenario would be compatible with the globally remaining carbon budget from 2018 (580 Gt CO_2) for keeping global warming below 1.5 degrees with a 50 % chance (IPCC, 2018).

A convergence of global stocks at a significantly lower level of 132 t/cap, however, would also imply a considerable reduction of large stocks in industrialised nations (Krausmann et al., 2017). Halving the per capita stock of the USA, while ensuring a sufficient amount of services for a growing population likely requires far reaching socio-ecological transformations of stock use, including more intensive use of stocks and stock redistribution (e.g. redistributing living space, increasing the average household size) as well as stock transformation (e.g. building smaller residential housing units, expanding public transportation). The USA, for example, have a comparatively high endowment of housing space compared to other industrialised nations. There are 2.4 rooms/person in the USA which is the 2nd highest value for OECD countries (OECD, 2019). Housing space in the USA, however, is also unequally distributed: households with an annual household income of below 20000 $/yr in 2015 e.g. had an average floor space of 56 m^2/household member, households with an income above 140000 $/yr had an average floor space of 95 m^2/household member (EIA, 2015). Moreover, 0.2 % of the population in the USA was homeless in 2017 of which 30 % were unsheltered (SPI, 2019). Stock redistribution could thus contribute to ensuring both societal wellbeing and ecological sustainability at the same time.

Future research could use the MISO-model to assess the socio-ecological effects of efficiency improvements, stock redistribution and transformation on resource use, stocks, services, outflows, waste and emissions. The key challenge in this regard is to determine a both socially and ecologically sustainable level of stocks and stock use. Answering this question requires a more functional perspective on in-use stocks (i.e. how many tonnes of concrete are stocked in different functional stock types such as buildings or roads) to systematically asses the link between stocks, energy consumption and GHG emissions and services. It would also be interesting to analyse the spill-over effects between different strategies. Changing mobility patterns could e.g. increase the lifetimes of streets, leading to lower maintenance flows and a reduction of resource use. If stocks are reduced, discarded stocks (e.g. iron/steel, aluminum in motor vehicles) also become available as secondary materials for recycling.

5 Conclusion

Over the last 150 years, the in-use material stock of the USA has increased 160-fold from 0.6 Gt in 1870 to 96 Gt in 2017. The current material stock provides many services and is the basis for economic activity. Simultaneously, the production, use, maintenance and disposal of stocks requires large amounts of annual material and energy flows. Presently, 35 % of domestic material consumption (1.9 Gt/yr) is used for building and maintaining stocks. About 67 % of primary energy consumption (69 EJ/yr is used for providing services from stocks, which contributes 55 % to the GHG emissions of the USA (3.6 Gt CO_2-eq./yr). The energy and carbon intensities of stocks have strongly declined from 1949 to 2017, which has resulted in a long-term relative decoupling of stocks, energy and emissions. Further stock efficiency improvements alone, however, are likely insufficient to keep global warming below 1.5°C. Stocks possibly need to be reduced, transformed and redistributed to ensure both a sufficient amount of services and ecological sustainability. Further research can contribute to changes in stock use- and production patterns by further linking stocks and flows to services and exploring biophysically meaningful pathways to sustainability transformations (Wiedenhofer et al. 2019, Haberl et al. 2017).

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Supplementary Information

Material stocks and sustainable resource use in the United States of America from 1870 to 2017

Quirin Dammerer

Summary of contents:

The MISO-Model tracks the evolution of **M**aterial **I**nputs, (In-Use) **S**tocks and **O**utputs within predefined spatial and temporal system boundaries. In this report we describe all the data, sources, assumptions and calculation procedures used for quantifying the material flows and parameters (processing and manufacturing losses, re- and downcycling rates, lifetimes) necessary to estimate in-use stocks, End-of-Life outflows, re- and downcycling and final waste for the USA from 1870 to 2017. Furthermore, we extensively compare and validate modelling results against the available literature.

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1 General Notes on the MISO-model

The MISO-model requires data for the following exogenous input parameters: stock-building materials or primary inputs to stock, processing and manufacturing losses, average lifetimes, recycling rates/flows and downcycling rates/flows to estimate values for five endogenous variables: in-use stocks, End-of-Life (EoL) outflows, recycling flows/rates, downcycling flows/rates and final waste. We distinguish **4 material groups** and *14 stock types*. Each stock type is produced from one or more stock-building materials. Solidwood, for example, is produced from industrial roundwood, asphalt from bitumen and sand and gravel and flat glass from limestone, soda ash and silica sands (see Table SI.1).

- **Biomass**: *Solidwood, Paper and Paperboard*
- **Metals**: *Iron/Steel, Aluminum, Copper, Other Metals*
- **Non-metallic Minerals**: *Concrete, Asphalt, Bricks (incl. Stones and Tiles), Sand and Gravel in sub-base and base-course layers (primary and downcycled), Container Glass, Flat Glass*
- **Fossil Energy Carriers**: *Plastics*

The model follows a two-step estimation procedure. First, processing losses (e.g. loss of bark, moisture content) are deducted from stock-building materials as depicted in economy-wide material flow accounts (ew-MFA; e.g. industrial roundwood, clay) to obtain primary inputs to stock (e.g. solidwood, bricks). Second, primary inputs to stock, manufacturing losses, average lifetimes and re- and downcycling rates or flows are then used to estimate in-use material stocks, EoL outflows, re- and downcycling rates or flows and final waste flows of the socio-economic system.

Technically, the first step of the procedure is not necessary to estimate the endogenous variables if information on primary inputs to stock is available from sources. However, to ensure ew-MFA consistency of the model, we establish an ew-MFA consistent link between primary inputs and stock-building materials, e.g. by adding processing losses to primary inputs. Primary copper production as metal content, for example, is a primary input to stock and available from the database of the United States Geological Survey (USGS; Kelly and Matos, 2014). However, in ew-MFA accounts, copper extraction (ew-MFA code: A.2.3.1, Table SI.1) must be reported in gross ore, i.e. including waste rock, which is not reported in the USGS database. To estimate gross ore, we used information on the average ore grade of mined copper in the USA and added waste rock (i.e. processing losses) to copper metal contents. Note that for each stock type, processing losses are added to the production of primary inputs to stock, whereas traded materials are accounted for with the mass at the point of time where the borders of the USA are crossed. Stock-building materials consumption is calculated as domestic extraction plus imports minus exports (Krausmann et al., 2018). An overview of sources for material inputs and the development of inputs over time are shown in Tables SI.1-2 and Figures SI.14-16.

Average lifetimes are estimated by multiplying end-use shares for inputs to stock by the lifetimes of their respective end-uses. End-use shares for stock types are shown in Figures SI.17-24, a summary of assumed lifetimes for different end-uses is shown in Table SI.3. For flat and container glass, sand and gravel in sub-base and base-course layers, other metals and asphalt, no end-use shares were available, and thus treated as a single-use category. The development of lifetimes for all stock types over time is shown in Figure SI.25, for an overview of lifetimes in 2017 see Table SI.3. The lifetime distribution in the MISO-model is normal (symmetric), lifetimes are thus given with a mean and a standard deviation. For further information on the chosen lifetime distribution, see Wiedenhofer et al. (2019).

The MISO-model either uses reported EoL recycling flows to calculate rates, or reported EoL recycling rates to estimate flows, depending on the available data. Recycling rates for materials are calculated from EoL outflows, estimated with the MISO-model, and the reported amount of (exogenous) EoL recycling flows. EoL recycling flows are calculated by multiplying EoL outflows by their corresponding (exogenous) EoL recycling rates from the literature. Downcycling flows depend on the supply of aggregates by downcycled EoL outflows of asphalt, concrete and bricks, stones and tiles and the demand for aggregates in sub-base and base-course layers. Demand which cannot be met by downcyced materials is fulfilled by primary (newly extracted) and recycled aggregates. Downcycling of materials other than asphalt, concrete and bricks, stones and tiles is not considered in this study. An overview of the development of re- and downcycling flows and rates is shown in Figures SI.26-28.

The MISO-model estimates the endogenous variables for predefined spatial and temporal system boundaries. The temporal boundaries are based on a spin-up period which is necessary to derive robust initial values for stocks, outflows, waste and re-/downcycling flows in the starting year of the actual study period. The length of the spin-up period is based on the material with the longest mean lifetime estimated within the model (Krausmann et al., 2017). In this study, aggregates have the longest lifetime (80 years). Here we begin the spin-up period in 1800 and analyse results for the period 1870 to 2017. Good data are available for many material inputs to stock already for the spin-up period.

This is the first study which estimates the total stock of manufactured capital, outflows, re-/downcycling and final waste for the USA by material stock type. Although there are no similar studies available to compare estimates with, several studies are available which estimated stocks of specific materials, EoL outflows, re-/downcycling flows and final waste. To check for the robustness of MISO-results we systematically compared the estimates of the MISO-model against the available literature.

In Section 2 of the SI we describe all the data, sources, assumptions and calculation procedures used for quantifying all parameters necessary to estimate stocks, EoL outflows, re-/downcycling and final waste for the USA for 1870-2017. In Section 3 we compare and validate results against the literature.

2 Data for Material Flows and Parameters

2.1 Biomass

Paper and Paperboard

Stock-Building Materials, Processing Losses and Primary Inputs to Stock

Paper and paperboard (P&B) primary inputs to stock are calculated as P&B primary production plus imports minus exports. Data for P&B total production (primary and secondary), imports and exports is available in the online database of the United States Geological Survey (USGS; Kelly and Matos, 2014) for 1900-2014 and from FAOSTAT (2019) for 1961-2017. There is a good fit between both sources in the overlapping period (Figure SI.1), hence we use USGS data for 1900-2014 and expand the series to 2017 with data from FAOSTAT. Because data on primary production was only available for 1965-2014, we subtract assumed recycling flows from total P&B production to obtain primary inputs before 1965 and after 2014 (for recycling calculations see below). For data prior to 1900 we followed the approach suggested by Krausmann et al. (2017: SI-10) and extrapolated total inflows by multiplying per capita consumption from 1900 with population data for 1800-1899 from the Maddison Project (2018).

To ensure ew-MFA consistency over the entire period, we add processing losses (e.g. moisture content, loss of bark) to primary production. The corresponding stock-building material for P&B in ew-MFA is industrial roundwood (Table SI.1) which is available for 1961-2017 in FAOSTAT. The USGS online database on the other hand reports data for processed primary (excluding secondary P&B production) wood products for 1965-2014. Figure SI.2 shows that the development of both flows is very similar, hence we assume that the differences between the values are processing losses. Wood production is reported in ew-MFA including bark, which has to be added to production data from FAOSTAT (Krausmann et al., 2018). Based on information from Krausmann et al. (2018: 31) for loss of bark (10 % of industrial roundwood) and data from USGS and FAOSTAT we estimate processing losses for every year for the period 1961-2014 according to formula 1 and add them to primary P&B production. For 2015-2017 we keep processing losses constant at the level of 2014. For the period 1800-1965 we estimate the average processing losses for 1965-2014 (54 %) and add them to primary P&B production.

$$(1)\ Processing\ Losses\ (\%)_{IndustrialRoundwood} = 1 - \frac{Primary\ Forestry\ Production}{Industrial\ Roundwood\ Production * 1.1}$$

Manufacturing Losses

Since the online databases of USGS and FAOSTAT already depict finished paper production, imports and exports, we assume that manufacturing losses are already included in processing losses.

Lifetimes

To calculate average lifetimes, we calculated the shares of “printing and writing paper” and “other (short-lived) paper types” (e.g. newsprint, wrapping papers, household and sanitary papers) consumption based on FAOSTAT data for 1961-2017 (Figure SI.17). Prior to 1961, we keep shares constant at the level of 1961. End-use shares for P&B were multiplied with lifetimes for paper products (9 years for “printing and writing paper”, 1 year for “other (short-lived) paper types”) by Skog and Nicholson as reported in IPCC (2003). Weighted average lifetimes were 3±2 years in 2017 (Table SI.3).

Recycling

EoL recycling flows for P&B (“secondary production”) are available in the USGS online database from 1965 to 2014 (Kelly and Matos, 2014). According to Wernick et al. (1996), the share of recycled P&B production in total P&B production was approximately constant from 1900 to 1965. Before 1965 we thus keep the share of secondary production in total P&B production constant at the level of 1965 (26 %). For 2015 to 2017, we keep the share of secondary production constant at the level of 2014 (64 %).

Solidwood

Stock-Building Materials, Processing Losses and Primary Inputs to Stock

Because solidwood recycling is assumed to be negligible (see below), solidwood primary inputs to stock are calculated as solidwood production (i.e not "primary production") plus imports minus exports. Data for solidwood production and trade is available from the USGS online database (Kelly and Matos, 2014) for the period of 1900 to 2014. "Lumber", "Plywood and veneer", "Wood panel products" and "Other industrial wood products" were identified as solidwood primary inputs to stock. For the period of 1800 to 1899, lumber production was taken from the Historical Statistics of the United States[10] (HSUS; Bureau of the Census 1975; L98-112).[11] Additionally, since "Other industrial wood products"[12] were already a significant input to stock in 1900 (50 % of lumber consumption) we keep this ratio constant before 1900 and add it to the reported lumber consumption. Values for primary inputs to stock from 2015 to 2017 are based on growth rates for industrial roundwood production reported in FAOSTAT (2019). To obtain stock-building materials we add processing losses to solidwood production. The corresponding stock-building material for solidwood in ew-MFA is industrial roundwood. Processing losses are thus calculated in the same way as for P&B production (see calculations above).

Manufacturing Losses

Manufacturing losses are assumed at 5 %, based on scrapped wood products during construction (e.g. due to over-purchasing, customizing materials) displayed in Cochran and Townsend (2010).

Lifetimes

Average lifetimes were calculated based on end-use shares for solidwood (lumber, plywood/veneer, wood panel products and other industrial wood products) from the USGS online database (Kelly and Matos, 2014) and the HSUS (Bureau of the Census 1975, Figure SI.18). Lifetimes (75 years for lumber and plywood/veneer and 25 for wood panel products and other industrial wood products) are from Cochran and Townsend (2010). Lumber is used for construction purposes with a long lifetime (75 years) and non-construction purposes with a shorter lifetime (e.g. consumer goods such as furniture, EPA 2015). However, no information on the exact distribution of lumber end-uses over time could be found. Based on EPA (2015) we assume that 20 % of lumber consumption is used for non-construction applications (assumed lifetime: 25 years) and 80 % is used for construction purposes (lifetime: 75 years) for 1800-2017. Weighted average lifetimes were calculated at 63±20 years for 2017 (Table SI.3).

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[10] We linearly interpolated lumber consumption between available data points from 1799 to 1899.

[11] The unit for lumber production in HSUS is board feet. Conversions between board feet and metric tonnes depend on the distribution of wood types, due to specific densities of coniferous and non-coniferous wood (Krausmann et al., 2018). Based on estimations with USGS/HSUS data, we assume a conversion factor of 0.0011 metric tonnes for 1 board foot for 1800-1900.

[12] Other industrial wood products include "cooperage logs, poles and piling, fence posts, hewn ties, round mine timbers, box bolts, excelsior bolts, chemical wood, shingle bolts and miscellaneous items" (USGS, 2014).

Recycling

Solidwood recycling is assumed to be negligible and recycling rates set to 0 % (Krausmann et al., 2017).

2.2 Metals

Iron/Steel

Stock-Building Materials, Processing Losses and Primary Inputs to Stock

Iron/steel primary inputs to stock are calculated as steel primary production plus net trade (imports minus exports) of steel products. Total (primary and secondary) production data was taken from the USGS online database for 1900-2015 (Kelly and Matos, 2014) and the Mineral Commodity Summary of 2018 for 2016-2017 (USGS, 2018) and cross-checked with data from the World Steel Association (WSA 2019; Figure SI.3).[13] Overall there is a very good fit between both databases. For 1864-1899 we use data from the HSUS (Bureau of the Census, 1975; P231-300). Because total production always includes both primary and recycled inputs to stock, we subtract recycling flows from total production (for the calculation of recycling flows see below). Prior to 1864 we followed the approach suggested by Krausmann et al. (2017: SI-10) and extrapolated total inflows by multiplying per capita consumption from 1864 with population data for 1800-1863 from the Maddison Project (2018). For iron/steel net trade, data from the USGS online database was cross-checked with statistics from the United Nations (UN) Comtrade (2019) database. Trade in both databases includes semi-finished products but excludes final end-use products (e.g. vehicles, machinery), which we do not account for. Overall there is a good fit between both sources (Figure SI.4), hence we use USGS data for iron/steel trade for 1913-2017. Trade data for iron/steel was non-existent before 1913, thus we assume that production equals consumption before 1913. Net trade was 6 % of steel production annually on average for 1914-1950, the impact on results should thus be small. The exclusion of final products possibly provides greater distortions but cannot be accounted for at this point in time (Gierlinger and Krausmann, 2012).

Iron/steel is produced from iron ore, which is reported as gross ore (i.e. including waste rock) in the USGS online database and the Mineral Commodity Summary of 2018 for the period of 1900-2017. Apparent consumption of iron/steel in ew-MFA accounts is calculated as iron ore production plus iron ore imports minus iron ore exports plus steel imports minus steel exports. Prior to 1900, we assume iron ore grades of 50 % (Krausmann et al., 2018), while losses occurring during other processing steps of steel production (steelmaking, casting, rolling/forming) are assumed at 15 % (Cullen et al., 2012). To ensure ew-MFA consistency before 1900 we add these processing losses to primary production of iron/steel. Processing losses from iron ore to iron/steel are thus assumed at 58 % from 1800 to 1899.

Manufacturing Losses

According to Cullen et al. (2012), fabrication losses occurring during the global production of steel end-use goods are 15 %. Fabrication losses are almost entirely recycled by

[13] We compare crude steel production between the two sources but use steel product shipments from USGS as inputs to stock.

manufacturers, but 15 % are lost during the recycling process of preconsumer scrap. Manufacturing losses are thus assumed at 2.2 %.

Lifetimes

Average lifetimes were calculated based on end-use shares (containers, transportation, construction, steel service centres and other) from the USGS online database for 1979-2003 (Kelly and Matos, 2014) and Mineral Commodity Summaries for 2004-2017 (USGS, 2019a). Prior to 1979, we keep end-use shares constant at the level of 1979 (Figure SI.19). Lifetimes for different end-uses were taken from Müller et al. (2011).[14] Weighted average lifetimes were calculated at 34±12 years for 2017 (Table SI.3).

Recycling

The USGS online database (Kelly and Matos, 2014) reports iron and steel scrap consumption for 1939-2015. However, the database does not specifically report End-of-Life recycling flows ("old scrap"). Old scrap recycling flows for 1998-2017 were therefore obtained by multiplying shares of post-consumer (old) scrap from the Mineral Commodity Summaries of 1998 to 2017 (USGS, 2019a) by total scrap consumption from the USGS online database. For 1969-1990 we assumed that "receipts of scraps" in Brown (2013) includes old and prompt scrap (the latter is produced during the manufacturing process). We then calculated the share of "receipts of scrap" in total production of scrap (also including home scrap, which is recirculating scrap from current operations). We assumed that 23 % of total scrap is prompt scrap based on USGS (2019a) and subtracted the share of prompt scrap from the combined share of old and prompt scrap. We then multiplied e share of old scrap by the total amount of scrap in the USGS database. For 1939-1968 and 1991-1997 we kept the share of old scrap constant at the level of 1969 and 1990 respectively. Since steel recycling started in the 1900s (Wernick et al., 1996) and the USGS database reports scrap consumption from 1939, we assume that the share of EoL recycling flows in total iron/steel production linearly increases from 0 % in 1900 to 21 % in 1939. While data for the period of 1998-2017 is therefore of good quality, recycling inputs prior to 1998 are rough estimates.

Aluminum

Stock-Building Materials, Processing Losses and Primary Inputs to Stock

Aluminum primary inputs to stock are calculated as aluminum primary production plus imports minus exports. Data for aluminum primary production was taken from the USGS online database for 1900-2015 (Kelly and Matos, 2014), the Mineral Commodity Summary of 2018 (USGS, 2018) for 2016-2017 and the HSUS (Bureau of the Census, 1949: 152) for 1886-1900. Prior to 1886, aluminum production was non-existent (Aluminum Leader, 2018). Concerning aluminum trade, USGS data was cross-checked with statistics from the UN Comtrade (2019) database. Trade in both databases includes imports and exports of crude aluminum and semimanufactures but excludes final end-use products, which we do not account for. Overall there is a very good fit between both databases (Figure SI.5). Therefore, we use USGS data for aluminum net trade. Prior to 1911, no trade data is reported in USGS/HSUS, and we assume that aluminum production is equal to consumption. The USA

[14] Life times for the category "other" were assumed to be the average of "machinery" and "others" in Müller et al. (2011). The life time of the end-use category "Service centers and distributors" was assumed to be the average of the life times of the other four categories (construction, transportation, containers and other).

became a major net importer of aluminum only from the 1980s, impacts on results should thus be small (Chen and Graedel, 2012).

Aluminum is either produced from bauxite or from alumina, for which data is available from 1900 to 2017 in the USGS online database and the Mineral Commodity Summaries of 2018. Stock-building materials for aluminum are equal to bauxite production plus bauxite imports minus bauxite exports plus alumina imports minus alumina exports plus aluminum imports minus aluminum exports. Prior to 1900 we estimate bauxite consumption based on USGS (2018: 31) which states that "As a general rule, 4 tons of dried bauxite is required to produce 2 tons of alumina, which, in turn, produces 1 ton of aluminium". Based on this information we assume that processing losses are 75 % from 1886 to 1900 and estimate bauxite consumption by multiplying aluminum primary production by the factor 4.

Manufacturing Losses

Manufacturing losses are based on Chen and Graedel (2012). Losses during the production of semi-finished and final products are 3 and 1 % respectively. Combined manufacturing losses are thus assumed at 3.7 %. Technically, losses for semimanufactures should not be deducted from net imports of semimanufactures. However, net trade of semimanufactures were fairly small compared to primary production for 1900-2017 (Liu et al., 2011). We therefore consider any excess losses to be small.

Lifetimes

Average lifetimes were calculated based on end-use shares (construction, consumer durables, containers and packaging, electrical, machinery and equipment, transportation, other) from Chen and Graedel (2012) for 1960-1975, the USGS database for 1975-2003 (Kelly and Matos, 2014) and Mineral Commodity Summaries for 2004-2017 (USGS, 2019a). Prior to 1960, shares were held constant (Figure SI.20). Lifetimes were taken from Chen (2013). We use their mid-values for mean lifetimes and standard deviations. Weighted average lifetimes were calculated at 22±8 years for 2017 (Table SI.3).

Recycling

End-of-Life recycling flows for aluminum ("old scrap secondary production") were taken from the USGS online database for the period of 1939 to 2015 (Kelly and Matos, 2014) and from the Mineral Commodity Summary of 2018 for 2016-2017 (USGS, 2018). Prior to 1939 (the last year for which old scrap secondary production is reported in the USGS database) we set recycling of old scrap to zero.

Copper

Stock-Building Materials, Processing Losses and Primary Inputs to Stock

Copper primary inputs to stock are calculated as copper primary production plus imports minus exports. Data for copper primary production was taken from the USGS online database for 1900-2015 (Kelly and Matos, 2014), the Mineral Commodity Summary of 2018 for 2016-2017 (USGS, 2018) and the HSUS (Bureau of the Census, 1949: 150-151) for 1845-1900. Prior to 1845 (102 metric tonnes/year), copper production was small and thus set to zero. Concerning copper trade, data from the online database of USGS was cross-checked with trade statistics from the UN Comtrade (2019) database. Overall there is a very good fit between both sources in terms of trade development (Figure SI.6). However, exports and imports are substantially lower in USGS, since Comtrade not only includes refined copper, but also several other semi-fished copper products. Therefore, we use trade data from Comtrade for the period of 1961-2017 to estimate total copper consumption. For 1900-1961 we use USGS trade data and for 1870-1900 we use copper trade statistics from Gierlinger and Krausmann (2012). Prior to 1870, no trade data is available, and we assume that production equals consumption. Comprehensive trade data for final goods was not available and thus not considered in this study.

Since copper consumption is reported as metal content and copper extraction must be reported in gross ore in ew-MFA, we assume declining ore grades from 2.5 % in 1845 to 0.5 % in 2017 (Wang et al. 2015, Gierlinger and Krausmann 2012). Apart from separating copper from waste rock, an additional amount of copper is lost during other processes (e.g. smelting; Graedel et al., 2002). These losses are omitted, due to their comparatively small size. Processing losses are thus assumed at 97.5–99.5 %.

Manufacturing Losses

Manufacturing losses are assumed at 1 %, based on Glöser et al. (2013) and include losses during the fabrication of semi-finished products and final goods (Figure S15 in the SI of Glöser et al.). Technically, losses during the fabrication of semimanufactures should not be deducted from net imports of semimanufactures. Losses are, however, quite small and we consider any excess losses to be small.

Lifetimes

Average lifetimes for 1845-1999 are based on end-use shares reported by Spatari et al. (2005) and for 2000-2017 on end-use shares by the USGS online database and its Mineral Commodity Summaries (USGS 2019a, Kelly and Matos 2014; Figure SI.21). Lifetimes for different end-uses were taken from Spatari et al. (2005).[15] Weighted average lifetimes were calculated at 34±20 years for 2017 (Table SI.3).

Recycling

End-of-Life recycling flows for copper ("old scrap secondary production") were taken from the USGS online database for the period of 1906-2015 (Kelly and Matos, 2014) and from the Mineral Commodity Summary of 2018 for 2016-2017 (USGS, 2018). Recycling flows were very low in 1906 (2 % of total production). We therefore assume that recycling is insignificant before 1906 and set recycling to zero.

Other Metals

Stock-Building Materials, Processing Losses and Primary Inputs to Stock

Primary inputs to stock for other metals are calculated as other metals primary production plus imports minus exports. Other metals include nickel, lead, zinc, tin, gold, silver, platinum-group metals, thorium, arsenic, chromium, lithium, magnesium metal, manganese, mercury, strontium, tungsten, antimony, bismuth, cadmium, cobalt, gallium, indium, molybdenum,

[15] Since we use normal lifetime distributions, mean lifetimes by Spatari et al. (2005) were recalculated to be symmetric

niobium (columbium), rare earths, selenium, tantalum, titanium, vanadium and zirconium. Primary production data for these metals was taken from the USGS online database for 1900-2015 (Kelly and Matos, 2014) and the Mineral Commodity Summaries of 2018 for 2016-2017 (USGS, 2018). Trade data was cross-checked with the UN Comtrade (2019) database for nickel, lead, zinc and tin, while data for all other metals was of poor quality. Overall there is a good fit between both sources (Figures SI.7-10). Therefore, we use trade data from USGS to estimate other metals consumption. Since no production and trade data was available prior to 1900 for most of the metals included, we use data from the HSUS (Bureau of the Census, 1975; M221-255) for manganese, lead and zinc for the period of 1801 to 1900, as they made up about 90 % of other metals consumption in 1900. To obtain gross ore for ew-MFA consistency, we use the average ore grade of other metals (6 %; Krausmann et al. 2017). Similar to copper, we exclude processing losses other than waste rock as these losses are quite diverse for the various kinds of metals and these kinds of losses are comparatively small. Processing losses for other metals are thus assumed at 94 %.

Lifetimes/Manufacturing Losses

Since by weight the largest amount of metals in this group is used in alloys with metals such as iron/steel, aluminum and copper (Krausmann et al. 2017, Kelly and Matos 2014, UNEP 2011, Gerst and Graedel 2008), we use the average of the lifetimes and manufacturing losses assumed for iron/steel, aluminum and copper (Krausmann et al., 2017). Manufacturing losses for other metals were thus calculated at 2.3 % and average lifetimes were calculated at 30±15 years for 2017 (Table SI.3).

Recycling

End-of-Life recycling flows for other metals ("old scrap secondary production") were taken from the USGS online database for the period of 1906 to 2015 (Kelly and Matos, 2014) and from the Mineral Commodity Summaries of 2018 for 2016-2017 (USGS, 2018). Prior to 1906 (137 metric tonnes/year), reported secondary production levels of other metals in USGS is small and thus set to zero.

2.3 Non-Metallic Minerals

Bricks, Stones and Tiles

Stock-Building Materials, Processing Losses and Primary Inputs to Stock

Data for miscellaneous and ball clay and dimension stone production and net trade was taken from the USGS online database (Kelly and Matos, 2014) and USGS (2019a) for the years 1970-2017 and 1900-2015, respectively. Miscellaneous and ball clay were included because they are used for the production of bricks and tiles (Kelly and Matos, 2014). For 2016-2017, the mass of stones is assumed constant at the level of 2015. Data for bricks consumption from 1869 to 1970 was taken from the HSUS (Bureau of the Census, 1975; P231-300).[16] We assume that 550 bricks are equal to 1 tonne of clay (Cochran and Townsend 2010). For data prior to 1869 we followed the approach suggested by Krausmann et al. (2017: SI-10) and extrapolated total inflows by multiplying per capita consumption from 1869 with population data for 1800

[16] We linearly interpolated bricks consumption for 1869-1879, 1879-1889, 1889-1895 and 1940-1943

to 1868 from the Maddison Project (2018). Processing losses for clays (loss of moisture content during bricks production) are assumed at 26 % and added to primary inputs for 1869-1970 because we used data for bricks production for this period. For 1970-2017, clay stock-building materials are equal to primary inputs to stock. Processing losses for stones are assumed at 0 %, primary inputs for stones are thus equal to stock-building materials (Krausmann et al., 2017).

Manufacturing Losses

Manufacturing losses for bricks, stones and tiles are assumed at 4 %, based on scrapped bricks and other clay products during construction processes (e.g. due to over-purchasing or customizing materials to fit specific applications) displayed in Cochran and Townsend (2010).

Lifetimes

Lifetimes are based on end-use shares (bricks, stones, tiles, Figure SI.22) from the Bureau of the Census (1975), Kelly and Matos (2014), USGS (2019a) and lifetimes from Cochran and Townsend (2010). Bricks and stones were assumed to have the same lifetime. Lifetimes were 74±25 in 2017 (Table SI.3).

Recycling and Downcycling

Nowadays, bricks recycling is insignificant (USGS 2018, 2019a). We assume that recycling rates for Bricks /Stones/Tiles decrease from 15 % in 1800 to 0 % in 1960. Furthermore, we assume a downcycling rate of 35 % for 1800-1920, which decreases to 10 % in 1970 and then grows to 47 % in 2017 (Figure SI.28). Assumptions for re- and downcycling are very rough and based on Krausmann et al. (2017).

Concrete

Stock-Building Materials, Processing Losses and Primary Inputs to Stock

Concrete primary inputs to stock are equal to concrete production, as concrete is usually not traded. Concrete production can be estimated based on cement consumption (Cochran and Townsend, 2010). To do so, we assume that concrete contains 11 % of cement, cement has a density of 3150 kg/m³ and concrete has a density of 2300 kg/m³ (Cochran and Townsend, 2010). Based on these assumptions we then estimate concrete production according to Formula 2. Cement consumption (cement production plus imports minus exports) is available in the USGS online database (Kelly and Matos, 2014) for 1900-2015 and in the Mineral Commodity Summary of 2018 for 2016-2017 (USGS, 2018). USGS data on cement consumption was cross-checked with data from Cembureau (2019). Overall there is a good fit between both sources (Figure SI.11), hence we use USGS data for our estimations.[17] For 1818-1900, cement consumption was taken from the HSUS (Bureau of the Census, 1975; M188-204). Prior to 1818 (4275 metric tonnes/year), cement consumption is small and we set concrete inputs to zero.

$$(2)\ Concrete\ Production\ (t) = \frac{Cement\ Consumption\ (t)}{0.11} * \frac{2300}{3150}$$

[17] We include both portland and masonry cement in our estimations. Concrete production from masonry cement was calculated in the same way as concrete production from portland cement.

Raw materials for cement production (80 % limestone, 20 % clay and other raw materials) as reported in ew-MFA accounts were estimated based on Formula 3 (Kapur et al., 2009). Data for sand and gravel (S&G) in concrete production was estimated by using formula 4 from Krausmann et al. (2018: 51). Processing losses for cement production are 42 % and 0 % for sand and gravel (Krausmann et al., 2017).

$$(3)\ Raw\ materials\ for\ Cement\ Production\ (t) = Cement\ Production\ (t) * 1.7$$

$$(4)\ Sand\ and\ Gravel\ for\ Concrete\ Production\ (t) = Cement\ Consumption\ (t) * 6.1$$

Manufacturing Losses

Manufacturing losses are assumed at 4 %, based on scrapped concrete during construction (e.g. due to over-purchasing, customizing materials) displayed in Cochran and Townsend (2010).

Lifetimes

Average lifetimes were calculated based on end-use shares (buildings, roads/bridges, other) and lifetimes from Cochran and Townsend (2010).[18] We linearly interpolate shares between available data points (1902, 1927, 1952, 1962, 1977, 1979, 1982, 2002) from Cochran and Townsend and keep shares constant at the level of 2010 for 2011-2017 (Figure SI.23). For 1818-1900 we keep shares constant at the level of 1900. Weighted average lifetimes were calculated at 52±17 years for 2017 (Table SI.3).

Recycling and Downcycling

We assume that concrete recycling started in 1970 (similar to asphalt) and increased to 3 % in 1996, after which we assume that it kept on increasing linearly to 5 % in 2017 (Krausmann et al. 2017, Kelly 1998, Wilburn and Goonan 1998). Regarding downcycling, Wilburn and Goonan (1998) report that in 1996, 50 % of concrete EoL waste was landfilled. The rest was used for cement concrete (3 %), asphaltic concrete (4 %) or road base and others (43 %; Kelly, 1998). Sandler (2003) states that estimates have placed re- and downcycling rates for concrete between 50 and 57 %.[19] Based on these studies we assume a constant downcycling rate of 10 % before 1970, which linearly increases to 55 % in 2017.

Container Glass and Flat Glass

Stock-Building Materials, Processing Losses and Primary Inputs to Stock

Container and flat glass production were estimated based on soda ash consumption and coefficients from Ruth and Dell´Anno (1997). For the production of 1 kg of container (flat) glass, 0.22 (0.23) kg of soda ash, 0.65 (0.73) kg industrial sand and 0.19 (0.24) kg of limestone are needed. For the production of container glass, additionally 0.11 tons of Feldspar are used. We estimate glass consumption based on soda ash consumption. Soda ash is not only used for glass, but also for e.g. chemicals and soaps. End-use shares for soda ash consumption are available for 1980-2003 in the USGS online database (Kelly and Matos, 2014). End-use shares

[18] Since we use normal lifetime distributions, mean lifetimes for roads/bridges and other structures were recalculated to be symmetric.

[19] Sometimes the terms recycling and downcycling are used interchangeably. Here we assume that Sandler refers to combined re- and downcycling rates.

of soda ash consumption for container and flat glass production were held constant before 1980 and after 2003. To perform a cross-check, total glass production was also estimated based on industrial S&G consumption (silica sands). End-use shares for glass production from industrial S&G are available for 1975-2015 in USGS and USGS (2019a). For 1900-1975 shares of silica sands for glass production were held constant at the level of 1975.

Overall there is a good fit between estimates from soda ash and silica sands consumption (Figure SI.12). Glass production estimated from silica sands also include fibre and other types of glass. For this reason (among others), estimates from silica sands generally tend to be higher than the estimates from soda ash. Both estimations also agree well with the values for glass production shown in Ruth and Dell´Anno (1997). We use glass estimations from soda ash, since we can explicitly distinguish between container and flat glass. For data prior to 1900 we followed the approach suggested by Krausmann et al. (2017: SI-10) and extrapolated total inflows by multiplying per capita consumption from 1900 with population data for 1800 to 1899 from the Maddison Project (2018). Trade data for container and flat glass was either of insufficient quality (UN Comtrade, 2019) or not accessible for this study (e.g. Glass Packaging Institute, 2018) at this point in time. Ideally, this kind of data will be available in future studies.

Stock-building materials for flat and container glass (soda ash, silica sands, limestone and feldspar) were estimated based on soda ash consumption and production coefficients. For 1 kg of container and flat glass, 1.17 kg and 1.2 kg materials are needed, respectively. Processing losses are assumed at 17 % for container glass and 20 % for flat glass (Ruth and Dell´Anno, 1997).

Manufacturing Losses

Since most losses occurring during the manufacturing of glass are internally recycled, manufacturing losses are assumed at 0 % (Butler and Hooper, 2011).

Lifetimes

Lifetimes for container and flat glass are based on Wiedenhofer et al. (2019). Container glass lifetimes decrease from 5 years in 1900 to 1.5 years in 1970 and then increase to 3 years in 2010. Flat glass lifetimes decrease from 50 years in 1900 to 30 years in 1970 and then remain constant (Figure SI.25).

Recycling

Recycling rates for container glass are based on data for glass municipal solid waste and recycling from EPA (2018a). Recycling rates increased from 2 % in 1960 to 33 % in 2015. For 2016-2017 we keep recycling rates constant, prior to 1960 we assume that recycling is negligible and set recycling rates to 0 %. We assume that flat glass is generally not recycled (Krausmann et al. 2017; EPA 2015, 2018a).

Aggregates (Primary)

Primary Inputs to Stock

Aggregates for sub-base and base-course layers in built infrastructures are estimated in the MISO-model. For buildings, we assume that “an average of 70 kg of aggregate […] per Mg of concrete and 45 kg per Mg of bricks” (Krausmann et al. 2017: SI-9) is used for sub-base layers

(Figure SI.34). For sub-base and base-course layers in roads, we estimate aggregates demand according to formula 5.

*(5) Aggregate_base_course = Asphaltcons * base_course_multiplier*

Information on asphalt consumption (*Asphaltcons)* is taken from USGS and the International Energy Agency (2018; see section on Asphalt below). To calculate the term *base_course_multiplier*, two kinds of coefficients are needed: 1) the overall ratio of asphalt to sand and gravel stocks in roads and 2) the proportion of asphalt used each year for new roads. These coefficients are estimated as follows.

1) Overall proportion of Asphalt to Sand and Gravel in Roads:

Information on road width, length and depth for different road types in the USA are given by Miatto et al. (2017). They distinguish between 7 road types of which 4 are asphaltic. These 4 types are "low type pavement" (LTP), "intermediate pavement" (IP), "high flexible pavement" (HFP) and "high composite pavement" (HCP). Based on this information and specific densities for asphalt and sand and gravel (Krausmann et al., 2018) we calculated the amount of asphalt and sand and gravel contained in each road type (Table SI.4). We then calculated multipliers for each road type, according to formula 6.

$$(6)\ Multiplierroadtypex = \frac{Sand\ and\ Gravelroadtypex}{Asphaltroadtypex}$$

To estimate the overall multiplier for combined road types, information on the development of the total road kilometres for asphaltic road types is needed. Miatto et al. (2017) show road kilometres for all road types for 1905-2015 (Figure SI.29). Four time periods can broadly be distinguished: 1905-1940, 1940-1980, 1980-2008 and 2008-2015. In the first period, growth for mileage of all road types was almost the same. From 1940 on, growth for high composite pavement roads stopped while the three remaining road types experienced strong growth. For 1980-2008, low type and intermediate pavement kilometres first stopped to grow and then grew slightly until 2008, while the high flexible pavement road type continued to grow until 2008. From 2008, expansion of asphaltic roads almost stopped.

For 1905 to 1940, we calculated the share of each road type (LTP, IP, HFP, HCP) in total new road construction (Figure SI.30). We then estimated the total multiplier for each year according to formula 7. Prior to 1905 we keep the multiplier constant at the level of 1905.

*(7) MultiplierTotalyearx= ShareNewRoadConstructionLTPyearx * MultiplierLTP+ ShareNewRoadConstructionIPyearx * MultiplierIP + ShareNewRoadConstructionHFPyearx * MultiplierHFP + ShareNewRoadConstructionHCPyearx * MultiplierHCP*

For 1941 to 1980, we calculated the share of the road types LTP, IP and HFP in total new road construction (Figure SI.31). We then estimated the total multiplier according to formula 8.

*(8) MultiplierTotalyearx= ShareNewRoadConstructionLTPyearx * MultiplierLTP + ShareNewRoadConstructionIPyearx * MultiplierIP + ShareNewRoadConstructionHFPyearx * MultiplierHFP*

For 1981-2017 we assume that the multiplier is equal to the multiplier of the high flexible pavement type, since the growth of kilometres of this road type is the main driver of asphaltic road construction. The development of the total multiplier for asphaltic roads (*MultiplierTotal*) can be seen in Figure SI.32.

2) The proportion of asphalt used each year for new roads

To obtain coefficients for the proportion of asphalt used each year in new roads, we use information on the usage of materials for new road construction by Miatto et al. (2017). They state that "the share of material requirements has progressively shifted from new construction to maintenance. In the early years, we expect that about 70% of the yearly material inflows would have been required for expanding the network. This share progressively diminished, dropping on average below 50% in 1966, and arriving […] at an average share of 20% in 2015" (SI. 11). Based on this information we assume that in 1870 (when use of asphalt started, NAPA 2019a) asphalt consumption was used entirely for new roads, while in the years 1905/1966/2015, 70/50/20 % of asphalt was used for new road construction, respectively. Between these years, we linearly interpolated shares of asphalt for new roads (Figure SI.33). For 2016-2017 we keep the share of asphalt for new roads constant at the level of 2015.

Finally, we multiply the *MultiplierTotal* by the share of asphalt used for new roads (formula 9) to obtain the *base_course_multiplier*, which we use to calculate aggregates consumption for sub-base and base-course layers of roads (formula 5). The development of this multiplier can be seen in Figure SI.34.

*(9) Base_course_multiplieryearx= MultiplierTotalyearx * Share of Asphalt for new road constructionyearx*

Stock-Building Materials, Processing and Manufacturing Losses, Lifetimes and Recycling

Processing and manufacturing losses for aggregates in sub-base and base-course layers are assumed at 0 %, average lifetimes are assumed at 80±72 years and recycling rates are assumed constant at 60 % for the entire period (1800-2017) based on Wiedenhofer et al. (2019) and Krausmann et al. (2017).

Asphalt

Stock-Building Materials, Processing Losses and Primary Inputs to Stock

Asphalt primary inputs to stock are equal to asphalt production, as asphalt is usually not traded. Asphalt concrete is typically composed of 5 % bitumen and 95 % sand and gravel (S&G, excluding air and other small components) and can thus be estimated based on these materials (Miatto et al. 2017, Cochran and Townsend 2010). Data for S&G consumption for asphalt production and other applications is available in Kelly and Matos (2014) and USGS (2019a) for 1975-2017. However, about 50 % of total S&G consumption is reported as unspecified use. Estimating asphalt from S&G consumption could thus likely result in an underestimation of total asphalt consumption. On the other hand, data for bitumen production/consumption is available since the early 20th century from different sources: the UN Industrial Commodity Production Statistics (UNICPS)/the UN Energy Statistics Database (UNSD, 2019), the International Energy Agency (IEA, 2019) and the online database of USGS (Kelly and Matos, 2014). Miatto et al. (2017) also estimated bitumen consumption for road construction in the USA, using a bottom-up approach and distinguishing between virgin and recycled asphalt production. Apart from Miatto et al. (2017), all sources agree well with each other (Figure SI.13). The estimates of Miatto et al. (2017) are significantly smaller because exact construction standards (thickness of layers, width) for certain road types in the USA are unknown and this distorts bitumen consumption estimates by the bottom-up approach of

Miatto et al. (2017). Therefore, we decided to use official statistics on bitumen consumption from USGS ("Asphalt and Road Oil")[20] for 1905-2013 and IEA data for 2014-2015. For 2016-2017 we keep Asphalt consumption constant at the level of 2015. Since asphalt usage began in the 1870s (NAPA, 2019a), we assume that asphalt production linearly increased from 0 tonnes in 1869 to 807500 tonnes in 1905. We assume that 85 % of all bitumen is used for road construction, the rest being used for roofing (asphalt shingles) and other uses (Pyshyev et al., 2016). We exclude bitumen used in these (comparatively small) applications from our analysis and focus on bitumen in asphalt. Asphalt consumption is estimated based on formula 10.

$$(10)\ Asphalt\ Production\ (t) = \frac{Bitumen\ consumption\ (t) * 0.85}{0.05}$$

Processing losses for bitumen and sand and gravel are assumed at 0 %, based on Krausmann et al. (2017). Stock-building materials are therefore equal to primary material inputs to stock for asphalt.

Manufacturing Losses

Manufacturing losses are assumed at 0 % based on Cochran and Townsend (2010).

Lifetimes

Average lifetimes are based on Cochran and Townsend (2010). Since we use normal lifetime distributions, mean lifetimes were recalculated to be symmetric. Lifetimes were 23±11 years in 2017.

Recycling and Downcycling

Data on asphalt recycling was available for the years 1993 (USDT, 1993), 1996 (Wilburn and Goonan, 1998) and 2009 to 2017 (NAPA, 2019b). Since the first sustained recycling efforts began in the 1970s (FHA, 2016), we calculate the share of recycled to new asphalt of available data points and linearly interpolate between them, starting from 0 % in 1970. Regarding downcycling, Wilburn and Goonan report that 8 % of asphalt pavement debris was downcycled in 1996. According to NAPA (2019b), 4-12 % of reclaimed asphalt pavement (RAP) from roads and parking lots was downcycled annually from 2009 to 2017. Based on this information we assume a downcycling rate of 10 % in 1970, which linearly decreases to 5 % in 2017. Prior to 1970, asphalt downcycling rates were assumed constant at 10 %.

2.4 Fossil Energy Carriers

Plastics

Stock-Building Materials, Processing Losses and Primary Inputs to Stock

Data for plastics production is published on an annual basis by the American Chemistry Council (The Resin Review, ACC 2019) and currently covers the period from 1973 to 2018. However, this data was not accessible for this study. Therefore, we use a different approach to estimate plastics production in the USA: Global plastics (polymer resin and fiber) production for the

[20] Asphalt and Road Oil is included in the "Organics (nonrenewable)" section of USGS.

period of 1950 to 2015 is reported by Geyer et al. (2017). Plastics Europe (2019) additionally shows production shares by different regions and countries for 2006-2016. Although Plastics Europe does not explicitly show production shares for the USA, they show production shares for the NAFTA region. We held shares of plastics production prior to 2006 for the NAFTA region constant at the level of 2006[21] and multiplied the relative share of US GDP in the NAFTA region with the amount of plastics produced by the whole NAFTA region to obtain total plastics production values for the USA. Relative shares of GDP of the USA, Canada and Mexico were calculated based on data from the Maddison Project (2018). Prior to 1950 we assume that plastic production was negligible and thus set the primary inputs to stock to zero. Values for 2016 and 2017 on the other hand are based on the average annual growth level of 2010-2015. Since data for plastics net trade is currently not available to us, we assume that production equals consumption for 1950-2017. Since information on processing losses are currently not available, we assume that stock-building materials are equal to primary inputs to stock (Wiedenhofer et al. 2019, Krausmann et al. 2017).

Manufacturing Losses

Manufacturing losses are estimated based von a substance flow analysis study by Van Eygen et al. (2016) for Austria. According to Van Eygen et al. (2017), 10 % of plastics production is lost during the manufacturing process. No information on the internal recycling rates of manufacturing losses for the USA could be found. Based on EoL recycling rates for 2015 from EPA (2018a), we assume that 9 % of manufacturing losses are recycled internally. Manufacturing losses are thus assumed constant of 9 %.

Lifetimes

Since end-use shares for plastics for the USA were not accessible for this study, we calculated lifetimes based on end-use shares for Europe (Plastics Europe 2019, Figure SI.24), because end-use shares are relatively similar for Europe and the USA (Geyer et al., 2017). Since Plastics Europe production shares do not include fibre, we assumed a constant share of 11.5 % for Textiles based on Geyer et al. (2017). All other plastic shares were decreased by 2.3 % to take this into account. Lifetimes for different end-uses were taken from Geyer et al. (2017) and Bento et al. (2016). Weighted average lifetimes were calculated at 10±6 years for 2017 (Table SI.3).

Recycling

Recycling rates for plastics are based on EPA (2018a) and are calculated as recycled plastics divided by total plastics municipal solid waste. We linearly interpolated recycling rates between available data points (1980, 1990, 2000, 2005, 2010-2015). Plastics recycling started in 1980 (0.3 %) and increased to 9 % in 2015. Recycling rates for 2016 and 2017 were kept constant at the level of 2015.

[21] For 2017, we keep the share constant at the level of 2016.

3 Comparisons with Results of other Studies

3.1 Biomass

Paper and Paperboard/Solidwood

The biomass stock estimates of the MISO-model are on average 46 % smaller than the stock estimates by Fishman et al. (2014) but exhibit a very similar development over time (Figure SI.35). Fishman et al. (2014) start from industrial roundwood as reported in ew-MFA because they use ew-MFA data from Gierlinger and Krausmann (2012) to estimate stocks. Fishman et al. (2014) deduct 10 % losses from industrial roundwood to obtain inputs to stock. This means they only take loss of bark into account but ignore other losses in the wood processing industries (e.g. saw mills), including changes in moisture content (Krausmann et al., 2018). Here we use data on primary inputs for paper and paperboard and solidwood (lumber, plywood and veneer, wood panel products and other industrial wood products) from Kelly and Matos (2014) to estimate biomass stocks. We estimate that the processing losses from industrial roundwood to primary inputs to stock are significantly larger (54 % on average from 1964-2014) than the losses assumed by Fishman et al. (2014). For these reasons, we arrive at a lower biomass stock. Because of the smaller stocks, final waste estimates (defined as End-of-Life outflows minus re- and downcycling flows) of the MISO-model are also smaller than the estimates of Fishman et al. (2014, Figure SI.54). Additionally, final waste dynamics are different from each other, as in our estimation’s lifetimes change during the studied period, while Fishman et al. (2014) keep lifetimes constant and omit recycling in their calculations of biomass stocks (we include recycling for paper and paperboard).

Concerning EoL outflows and recycling rates, our estimates for paper and paperboard (P&B) and solidwood agree well with estimates by EPA (2015, 2016, 2018a, 2018b). Values for P&B EoL outflows and recycling rates show a similar trend over time but MISO-estimates are generally higher (Figures SI.47, SI.58). P&B results of the MISO-model are likely higher because EPA uses data from the American Forest & Paper Association (EPA, 2014) to estimate outflows and recycling, while we use data from Kelly and Matos (2014). MISO-results for solidwood EoL outflows are likely slightly higher because we include lumber, plywood and veneer, wood panel and industrial wood products in our estimations (Figure SI.48) while EPA uses a material flow approach to estimate demolition debris of buildings, roads and bridges in the USA and therefore does not include industrial wood products (EPA, 2015).

3.2 Metals

Iron/Steel

For iron/steel, the stock estimates by Rauch (2009) and Müller et al. (2006, 2011) agree very well with our estimates. Our results are 1-8 % smaller (Table SI.5). The estimates of Brown (1954) and Sullivan (2003) which are 51 % and 27 % higher than our results respectively only refer to quotes from Gerst and Graedel (2008), as the original studies were not accessible and estimation details not available. Furthermore, we cannot identify the methodology used by Sullivan (2005; 28 % higher stock). As a result, we cannot explain the larger differences between these studies´ estimates and our results.

Time-series data for iron/steel stocks were published by Fishman et al. (2014), Pauliuk et al. (2013) and Müller et al. (2011). Per capita stocks from Müller et al. (2011) agree well with our results (see Figure SI.36 and Figure 4 in Müller et al. 2011). Pauliuk et al. use (2013) three modelling approaches to estimate stocks for steel. Figure SI.37 shows approach (a). In this case per-capita stocks of Pauliuk et al. (2013) are still growing, which is at odds with our results, because our results show signs of per capita stock saturation. However, when looking at their approaches (b) and (c), which are more similar to our approach, results agree well with each other (see Figure SI.36 and Fig. 4 in Pauliuk et al. 2013).

Concerning the results of Fishman et al. (2014), the iron/steel stock estimates in this study are substantially larger than their stock results (Figure SI.38). Fishman et al. (2014) start from iron ore as reported in MFA accounts because they use ew-MFA data from Gierlinger and Krausmann (2012) to estimate iron stocks. Fishman et al. (2014) deduct 80 % losses from iron ore to obtain inputs to stock. This study on the other hand uses data on steel consumption from Kelly and Matos (2014) to estimate iron/steel stocks. Since our results generally agree well with the results of other studies, we assume that Fishman et al. (2014) have underestimated iron stocks (because their assumed loss factor was too high). Since stocks are substantially larger in our study, this also explains the larger final waste flows (defined as EoL outflows minus re- and downcycling flows, Figure SI.55) compared to the results of Fishman et al. (2014). Additionally, final waste dynamics are quite different over the observed period, as lifetimes in this study (29-35 years) are smaller than the lifetimes assumed by Fishman et al. (2014; 50 years) and we include recycling in our calculations of iron/steel stocks. The estimated EoL recycling rates for iron/steel in this study agree quite well with recycling rates in the literature (Table SI.8).

Aluminum

For aluminum the results of this study generally agree well with other sources (range of ±30 %, Table SI.6). Studies have found that aluminum stocks are not very sensitive to different lifetime distribution models but are sensitive to variations in mean lifetimes, which can differ between studies (Liu and Müller 2013, Chen and Graedel 2012). Therefore, we consider variations of ±30 % as an acceptable range (Liu and Müller, 2013). McMillan et al. (2010) arrive at a significantly lower stock (-57 %), which may be a result of longer lifetimes assumed by our study (Chen and Graedel, 2012). Time-series data for stocks and EoL outflows by Liu and Müller (2013), Chen and Graedel (2012) and Liu et al. (2011) agrees quite well with our results for the twentieth century (for stocks see Figure SI.40 and Figure 3 in Liu et al. (2011)/Fig. 10a in Chen and Graedel (2012) and Figure SI.39 for a comparison between stocks from Liu and Müller (2013) and this study; for EoL outflows see Figure SI.49 and Fig. 6b in Chen and Graedel 2012). Differences in values for the 20th century may arise out of variations in lifetimes. From 1990 to 2009, stock estimates of this study grew slower than the stocks calculated by Chen and Graedel (2012) and Liu et al. (2011), possibly because we do not account for trade of final products, which constitute a significant and increasing share of net imports from 1990 to 2008 (Liu et al., 2011). The growth of aluminum stocks is thus likely stronger during this period than our results indicate. The estimated EoL recycling rates agree quite well with EoL recycling rates in the literature (Table SI.8).

Copper

Copper in-use stocks agree well with results of other studies (range of ±30 %, Table SI.7). The estimates of Ingalls (1935), Merrill (1949) and Sullivan (2003) were obtained from Gerst and

Graedel (2008) as the original studies were not accessible. Additionally, the calculation method used by Gerst and Graedel (2008) to determine in-use stocks for Nathan Associates (2004) could not be identified. As a result, we cannot explain the larger differences between these results and our estimates. The estimated EoL recycling rates of this study agree quite well with EoL recycling rates in the literature (Table SI.8).

Other Metals/Aluminum/Copper

In-use stocks for the aggregate of other metals as defined in this study (i.e. including all metals other than iron, aluminum and copper) are not available in the literature.[22] However, Fishman et al. (2014) have estimated stocks for aluminum, copper and other metals as a group (which they define as "other metals"). Figure SI.41 shows that the development of non-ferrous (i.e. excluding iron) metals stock estimates by Fishman et al. is very similar to the development of non-ferrous metals in this study. However, the results of Fishman et al. (2014) are on average 80 % larger than our results. This also impacts final waste flows, which are substantially bigger than the estimates in this study (Figure SI.56).

Fishman et al. (2014) start from "other metals" ores (i.e. excluding iron) as reported in MFA accounts because they use ew-MFA data from Gierlinger and Krausmann (2012). Fishman et al. (2014) deduct 90 % processing and manufacturing losses from gross ore to obtain actual inputs to stock. This study on the other hand uses data on primary inputs for aluminum, copper and other metals to estimate stocks. We estimate processing losses of stock-building materials of 75 % for aluminum, 97.5-99.5 % for copper and 94 % for other metals and deduct manufacturing losses of 3.7 %, 1 % and 2.3 % from aluminum, copper and other metals primary inputs respectively. Additionally, lifetimes for other metals assumed by Fishman et al. (2014; 50 years) are substantially higher than the lifetimes assumed in this study (20-40 years). Given that we use more detailed information on primary inputs to stock for aluminum, copper and other metals and our results for aluminum and copper agree well with the results of other studies, we assume that Fishman et al. (2014) have overestimated non-ferrous metals stocks.

3.3 Non-Metallic Minerals

Concrete

To compare concrete in-use stocks, we use estimates of cement in-use stocks by Kapur et al. (2008) and Cao et al. (2017) and transform them to concrete in-use stocks by applying Formula 2 (see section 2.3). Overall, the estimates of these studies agree well with our results in terms of stock development (Figures SI.42-43). For the period of 1950 to 1975, our estimates are higher than the results by Cao et al. (2017), likely because their starting point of concrete in-use stock estimations is the year 1931, while our starting point is the year 1818. Values reported in Kapur et al. (2008) are almost equal from 1900 to 1950. However, estimates from Cao et al. (2017) and Kapur et al. (2008) start to overtake the results estimated in this study from 1970 and 1950; respectively. A possible explanation for this result is an increasing amount of downcycled cement/concrete EoL outflows. These are included in the cement stock estimates of Kapur et al. (2008) and Cao et al. (2017), while we include downcycled concrete in the stock of aggregates in sub-base and base-course layers. Other discrepancies are likely

[22] In-use stock estimates are available for gold, lead, tungsten, zinc at the national level and stocks for chromium and nickel are available at the state level (Gerst and Graedel, 2008).

the result of variations in recycling rates and lifetimes. Concrete EoL outflows also agree well with estimates by EPA (2015, 2016, 2018b; Figure SI.50) in terms of size and development over time. Our results are slightly higher as we include both portland and masonry cement in our estimations.

Bricks/Stones/Tiles and Container Glass/Flat Glass

In-use stock data is not available for bricks, stones and tiles and container and flat glass, but EPA (2015, 2016, 2018a, 2018b) have estimated EoL outflows and recycling flows for bricks and total glass. Overall our results agree well with these estimates in terms of size and development over time (Figures SI.51-52, SI.59), which give us confidence that our stock results are also robust.

Aggregates and Asphalt

To compare asphalt stocks with those reported by Miatto et al. (2017), we have transformed their results for bitumen stocks to asphalt by assuming that asphalt contains 95 % sand and gravel and 5 % bitumen.[23] Additionally, we have calculated the stock of aggregates in sub-base and base-course layers in Miatto et al. (2017) by subtracting S&G needed for asphalt and concrete from total S&G stock per road km, multiplying them with total road length of depicted road types. Overall, the development of asphalt and aggregates stocks is similar over time, but the results of Miatto et al. (2017) are substantially smaller than our estimates (Figures SI.44-45). According to Miatto et al. (2017), the actual depth/width of road layers for many road types is unknown. At the same time, these are likely the roads with the highest material intensities and road length. Therefore, we assume that the bottom-up approach of Miatto et al. (2017) has resulted in an underestimation of asphalt and aggregates stocks.

Non-Metallic Minerals

Overall, non-metallic minerals stocks and final waste flows agree with the results by Fishman et al. (2014) for the period of 1930 to 1970 (Figures SI.46, SI.57). However, from 1970 to 2005, their results start to diverge from those presented in this study. Fishman (2014) use ew-MFA data by Gierlinger and Krausmann (2012) to estimate stocks. Gierlinger and Krausmann (2012) assume that for every tonne of asphalt consumed annually, demand for aggregates as filling material in roads is 0.5 tonnes. However, the assumed demand for aggregates per tonne of asphalt in this study, although decreasing over time, is always higher than 0.5 tonnes. Therefore, we tried to replicate the figures of Gierlinger and Krausmann (2012) for the domestic material consumption (extraction plus imports minus exports) of sand and gravel for concrete and asphalt production and for aggregates used as filling material. To obtain sand and gravel for concrete, cement consumption was multiplied by the factor 6.1. To obtain sand and gravel for asphalt, bitumen consumption was multiplied by the factor 20. The asphalt estimate was further increased by 50 % to account for sand and gravel as filling material. Figure SI.63 shows a comparison between estimated and reported figures for sand and gravel consumption provided by Gierlinger and Krausmann (2012). We find that the estimated figures for sand and gravel consumption are substantially lower than the reported figures by Gierlinger and Krausmann (2012). We cannot explain these differences. However, since

[23] NAPA (2019c) has also estimated the asphalt in-use stock in roads and arrives at a total stock of 16.3 Gigatonnes (18 billion short tons). However, as they do not describe the methodology according to which this number is calculated (and the year for this value), we cannot explain the differences between their estimates and our results.

Gierlinger and Krausmann (2012) estimate a much higher consumption for sand and gravel than we do in this study and this gap is also strongly increasing from 1980, we conclude that these differences are a very important factor for differing non-metallic stocks (see also Figure SI.64) between Fishman et al. (2014) and our results from 1970 to 2005.

3.4 Fossil Energy Carriers

Plastics

In-use stock data for the USA is not available for plastics. However, EPA (2018a) have estimated plastics EoL outflows and EoL recycling flows, which agree quite well with our results in terms of size and development of flows over time (Figures SI.53, SI.60). It appears that our methodology for estimating inputs to stock and other parameters (e.g. lifetimes, manufacturing losses) for plastics gives reasonable results, which also gives us confidence in the robustness of our material stock estimates for plastics.

4 Figures and Tables

4.1 Data for Material Flows and Parameters

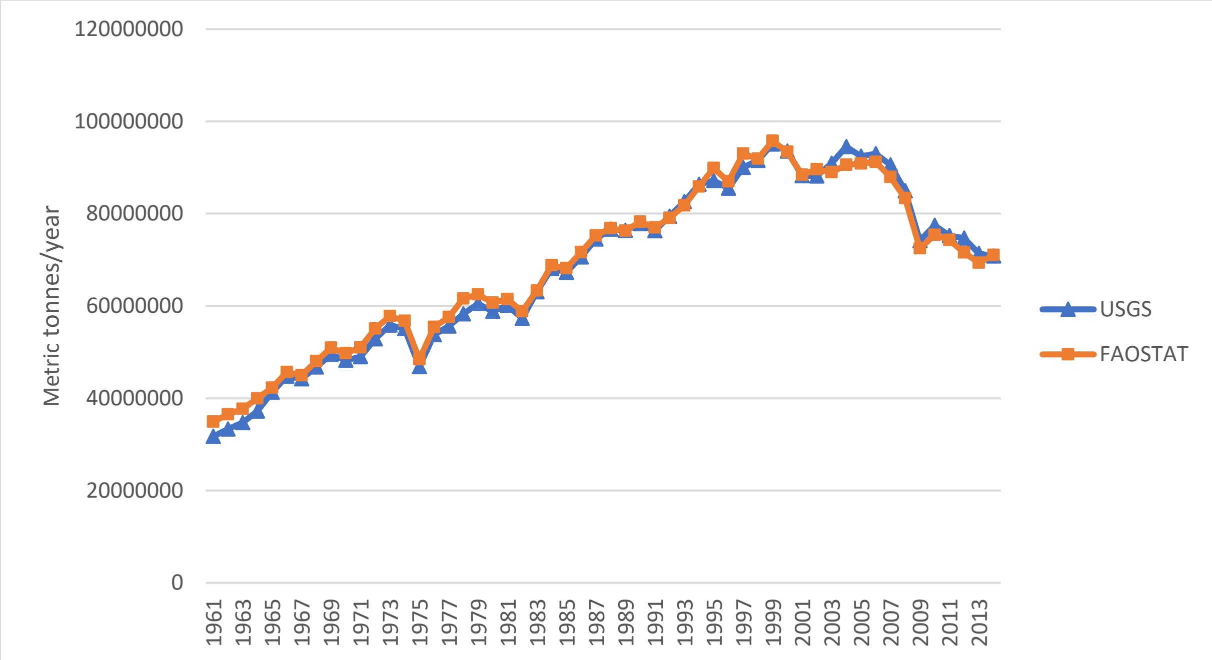
4.1.1 Material Flows

Table SI.1: Data sources for stock-building materials and material inputs to stock in the MISO-model

Stock-Building Materials (ew-MFA)	ew-MFA Code (Extraction)	Material Inputs to Stock (MISO)	Data Sources (incl. cross checks)	Time Period available in Data
Industrial Roundwood	A.1.4.1	Solidwood	Bureau of the Census (1975), FAOSTAT (2019), Kelly and Matos (2014)	1800/1900-2017
Industrial Roundwood	A.1.4.1	Paper and Paperboard	FAOSTAT (2019), Kelly and Matos (2014), Wernick et al. (1996)	1900-2017
Iron Ore	A.2.1	Iron/Steel	Brown (2013), Bureau of the Census (1975), USGS (2018, 2019a), Kelly and Matos (2014), Wernick et al. (1996), WSA (2019)	1864-2017
Bauxite	A.2.2	Aluminum	Bureau of the Census (1949), UN Comtrade (2019), Kelly and Matos (2014), USGS (2018)	1886-2017
Copper Ore	A.2.3.1	Copper	Bureau of the Census (1949), UN Comtrade (2019), Gierlinger and Krausmann (2012), Kelly and Matos (2014), USGS (2018)	1845-2017
Other Metal Ores	A.2.3.2-A.2.3.8	Other Metals	Bureau of the Census (1975), UN Comtrade (2019), Kelly and Matos (2014), USGS (2018),	1801-2017
Stones/Clays	A.3.1/A.3.5	Bricks, Stones and Tiles	Bureau of the Census (1975), Kelly and Matos (2014), USGS (2018)	1869/1900-2017
Limestone, Clays (for Cement) + Sand and Gravel	A.3.1 + A.3.2 + A.3.4	Concrete	Bureau of the Census (1975), Cembureau (2019), Kelly and Matos (2014), USGS (2018)	1818-2017
Limestone + Silica Sands + Soda Ash	A.3.2 + A.3.4 + A.3.8	Flat Glass	Kelly and Matos (2014), USGS (2018, 2019a), Ruth and Dell´Anno (1997)	1900-2017
Limestone + Silica Sands + Soda Ash + Feldspar	A.3.2 + A.3.4 + A.3.8. + A.3.8	Container Glass	Kelly and Matos (2014), USGS (2018, 2019a), Ruth and Dell´Anno (1997)	1900-2017
Sand and Gravel	A.3.4	Sand and Gravel for sub-base and base-course Layers	MISO-Model Estimations based on Miatto et al. (2017)	-
Sand and Gravel + Crude Oil (Bitumen)	A.3.4 + A.4.3	Asphalt	IEA (2018), Kelly and Matos (2014), Miatto et al. (2017), UNSD (2019)	1905-2015
Crude Oil	A.4.3	Plastics	Geyer et al. (2017), Plastics Europe (2019), Maddison Project (2018)	1950-2016

Notes: Aluminum/Concrete/Copper/Other Metals/Plastics consumption was very small/non-existent before 1886/1818/1845/1801/1950, respectively. For all other materials, we assumed constant per capita use prior to the earliest available year and multiplied per capita material flows with population data from the Maddison Project (2018), except for Asphalt where we assume that production linearly increased from 0 tonnes in 1869 to 807500 tonnes in 1905. For solidwood, lumber consumption data was available from 1800-2014 (2014-2017 estimated based on growth rates for Industrial Roundwood) while Other Industrial Wood Products were estimated before 1900. For Clays/Stones, data was available for 1869-2017/1900-2015. Plastics Production in the USA was estimated for all years, based on global plastics production data by Geyer et al. (2017), production shares of the NAFTA region by Plastics Europe (2019) and GDP data from the Maddison Project (2018).

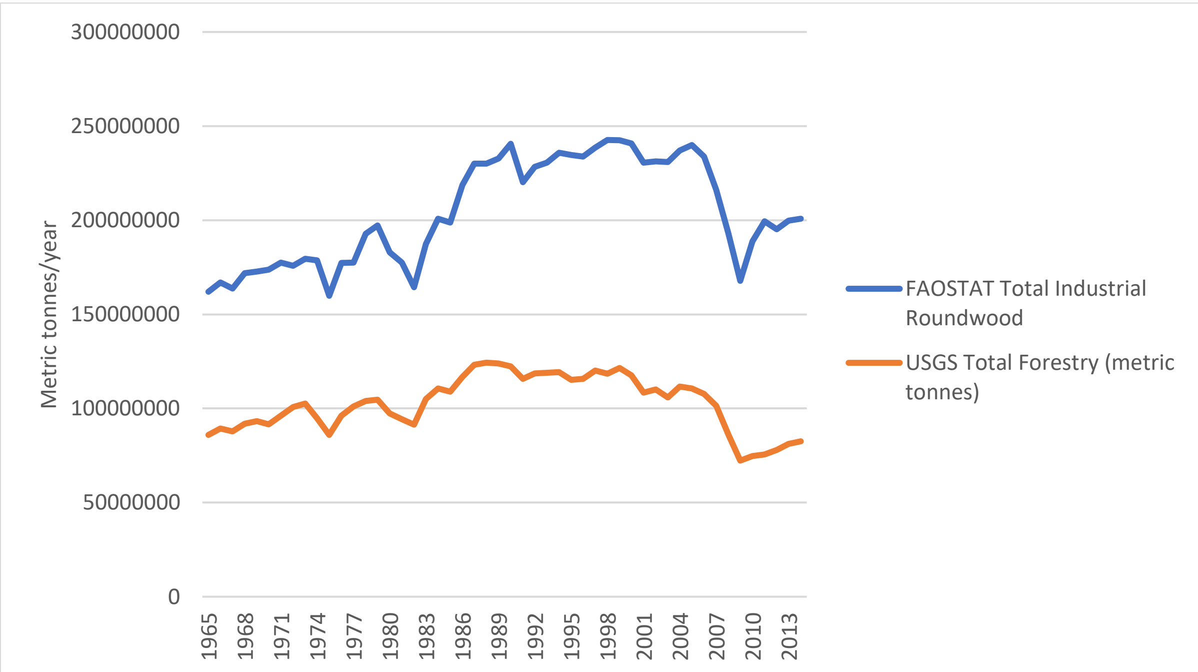
Figure SI.1: Apparent consumption of paper and paperboard in the USA, 1961-2014: Comparison of databases



Notes: Paper and paperboard consumption equals domestic production plus imports minus exports.

Sources: USGS data is from Kelly and Matos (2014), FAOSTAT data is from FAOSTAT (2019).

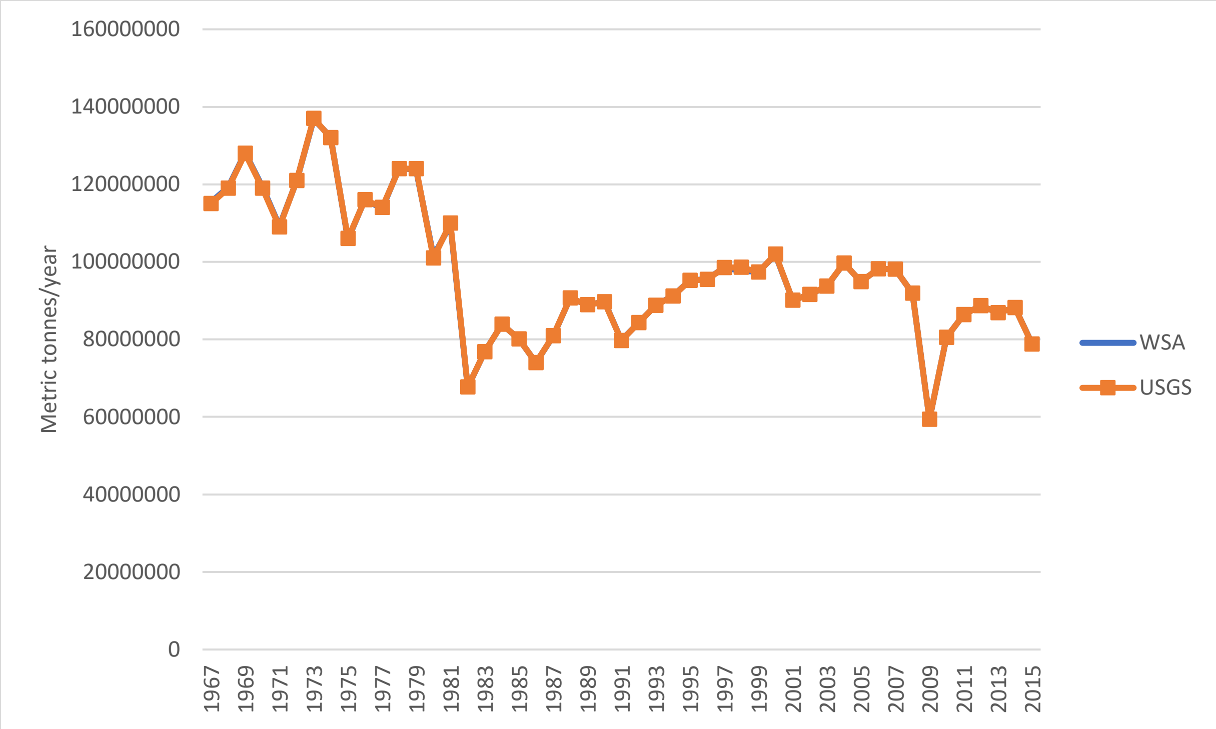
Figure SI.2: Forestry production and industrial roundwood production in the USA, 1965-2014



Notes: Data for industrial roundwood is given in m^3 coniferous and non-coniferous wood in FAOSTAT, we used conversion factors of Krausmann et al. (2018) to convert FAOSTAT data into metric tonnes/year. Data for USGS forestry production includes production of lumber, other industrial wood products, plywood and veneer, wood panel products and primary paper and paperboard production.

Sources: FAOSTAT data on industrial roundwood production is from FAOSTAT (2019), USGS data on forestry production is from Kelly and Matos (2014).

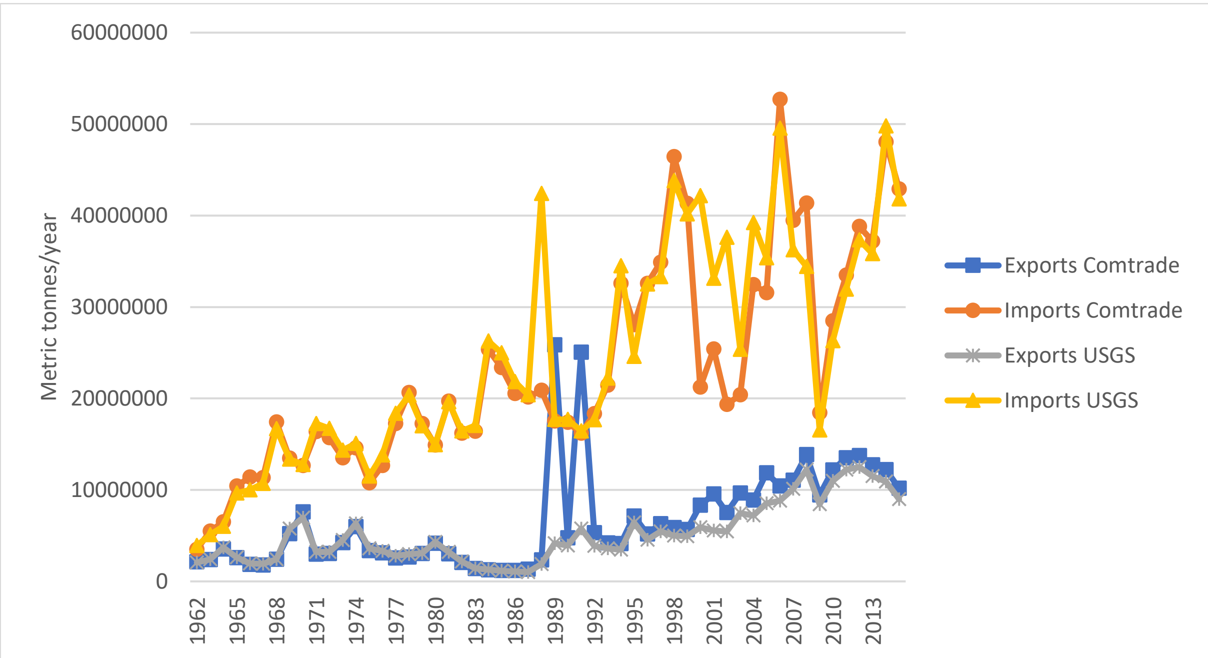
Figure SI.3: Crude/raw steel production in the USA, 1967-2015: Comparison of databases



Note: Crude/raw steel production is steel in the first solid state.

Sources: WSA data is from the World Steel Association (WSA, 2019), USGS data is from Kelly and Matos (2014).

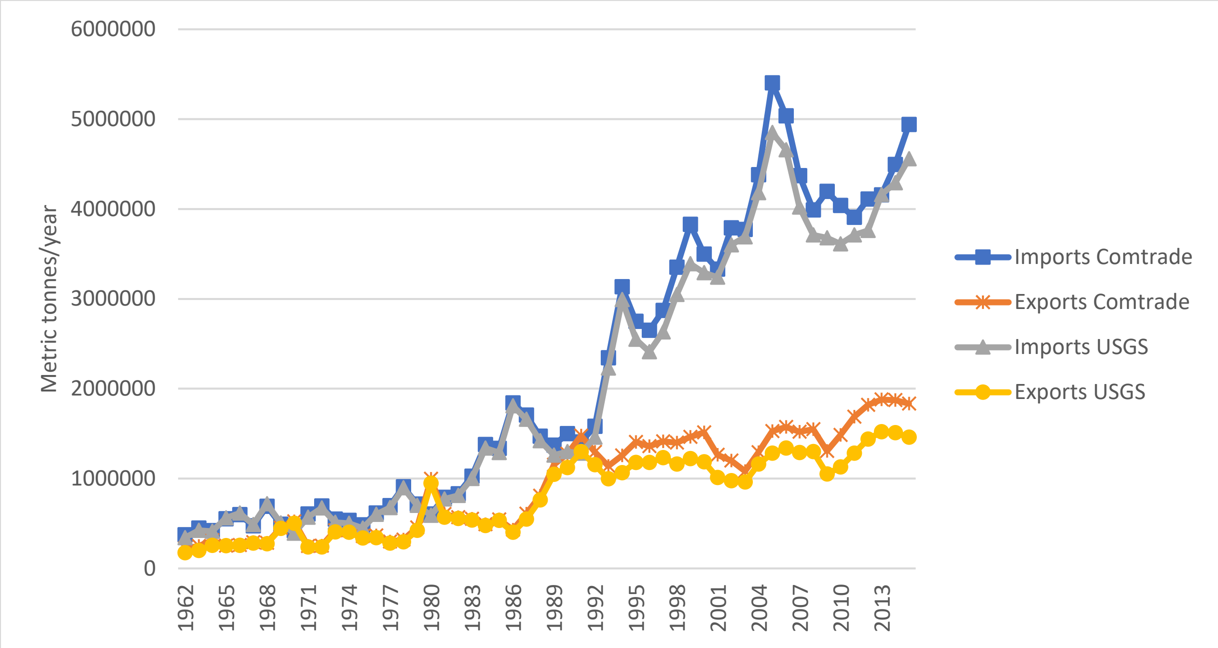
Figure SI.4: Iron/steel imports and exports in the USA, 1962-2015: Comparison of databases



Notes: Comtrade imports and exports includes SITC Rev. 3 Commodity Codes 671-679 (including semifinished products such as iron and steel bars, wires and tubes). USGS imports and exports includes steel mill products (semifinished products and other trade of iron/steel).

Sources: Comtrade data is from UN Comtrade (2019), USGS data is from Kelly and Matos (2014).

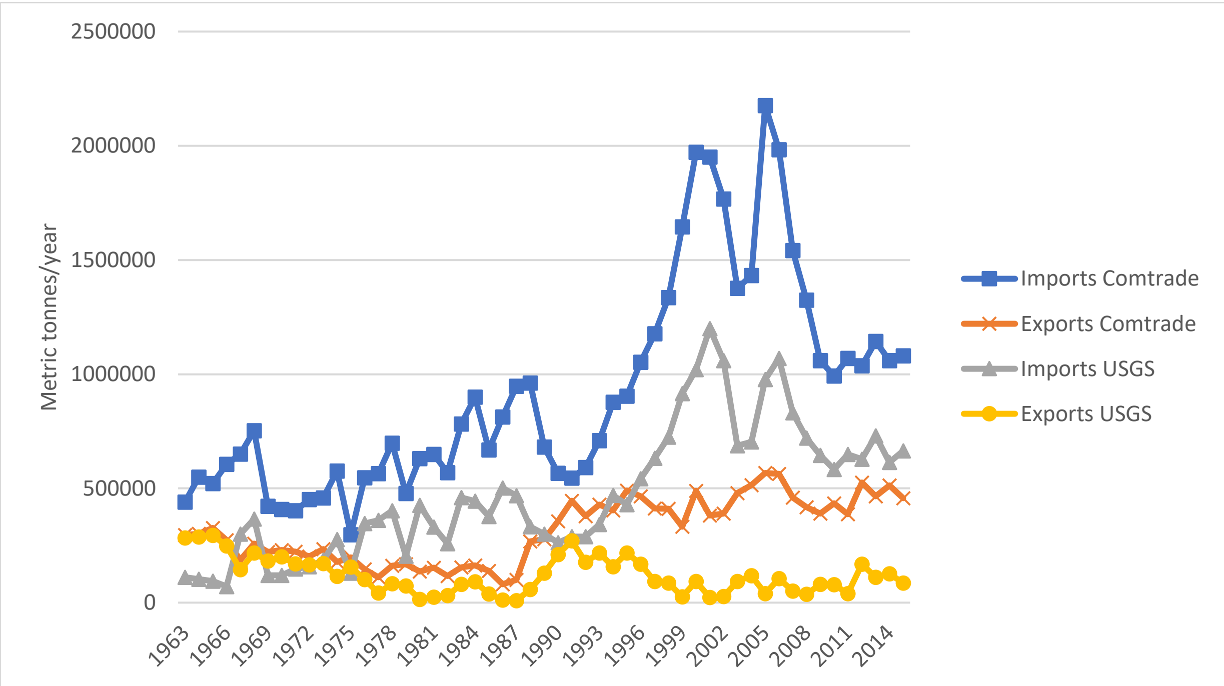
Figure SI.5: Aluminum imports and exports in the USA, 1962-2015: Comparison of databases



Notes: Comtrade imports and exports include SITC Rev. 3 Commodity Code 684 (aluminum and aluminum alloys unwrought and worked). USGS imports and exports include crude aluminum and semimanufactures. Information from Mineral Yearbooks (USGS, 2019b) was used to subtract exported scrap from exports displayed in USGS.

Sources: Comtrade data is from UN Comtrade (2019), USGS data is from Kelly and Matos (2014).

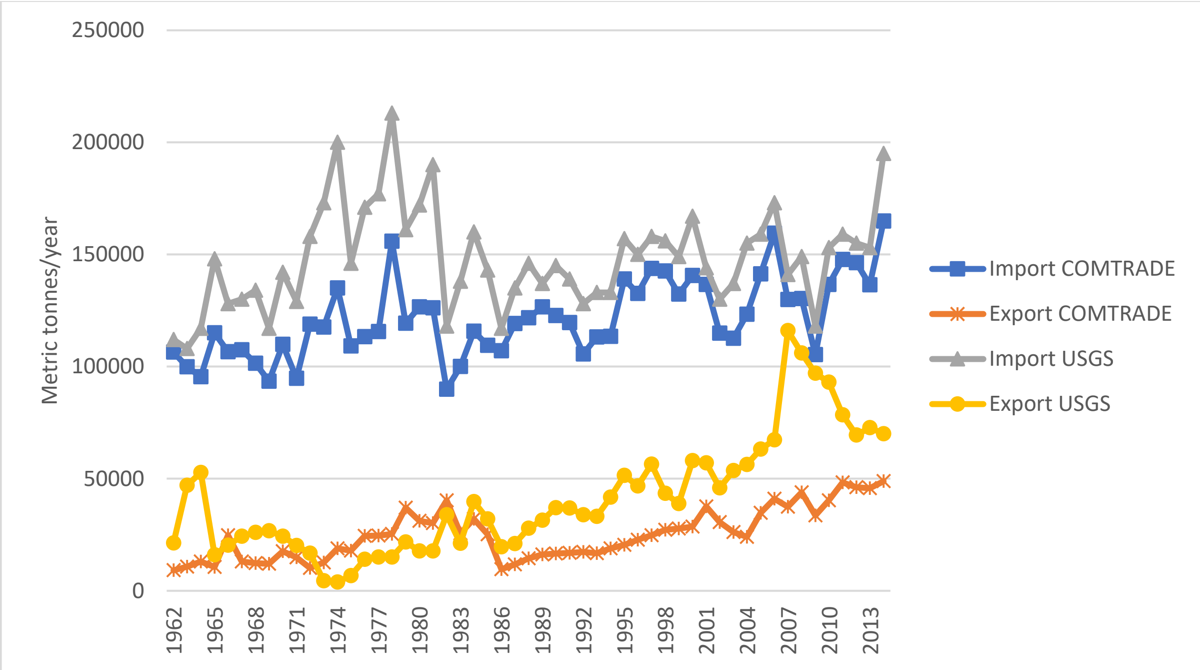
Figure SI.6: Copper imports and exports in the USA, 1963-2015: Comparison of databases



Notes: Comtrade imports and exports include SITC Rev. 3 Commodity Code 682 (e.g. refined and unrefined copper, semi-finished products like bars, tubes, plates and wire). USGS imports and exports include refined copper and excludes semifabricated and manufactured copper products.

Sources: Comtrade data is from UN Comtrade (2019), USGS data is from Kelly and Matos (2014).

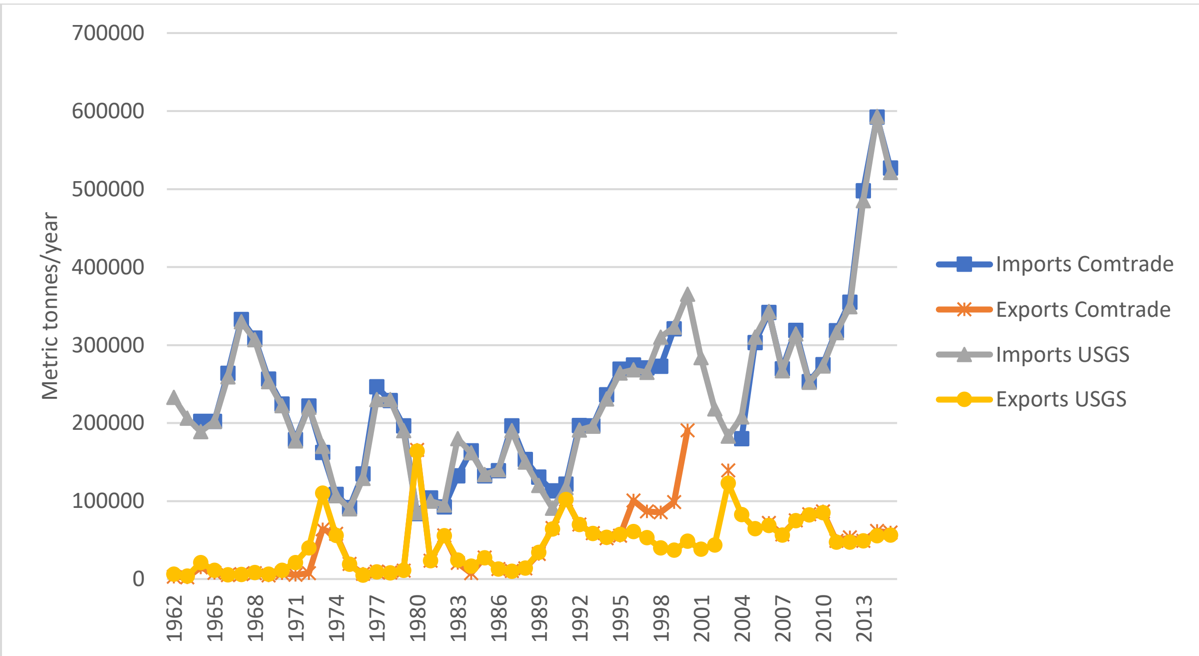
Figure SI.7: Nickel imports and exports in the USA, 1962-2014: Comparison of databases



Notes: Comtrade imports and exports include SITC Rev. 3 Commodity Code 683 (nickel and nickel alloys unwrought and worked, excluding electroplating anodes). USGS imports and exports include a variety of products, including metallurgical and chemical-grade oxides, ferronickel, and plating salts (from 1988 onwards trade also includes secondary nickel) and excludes steel mill products, castings, and downstream manufactured products which contain nickel-bearing steel.

Sources: Comtrade data is from UN Comtrade (2019), USGS data is from Kelly and Matos (2014).

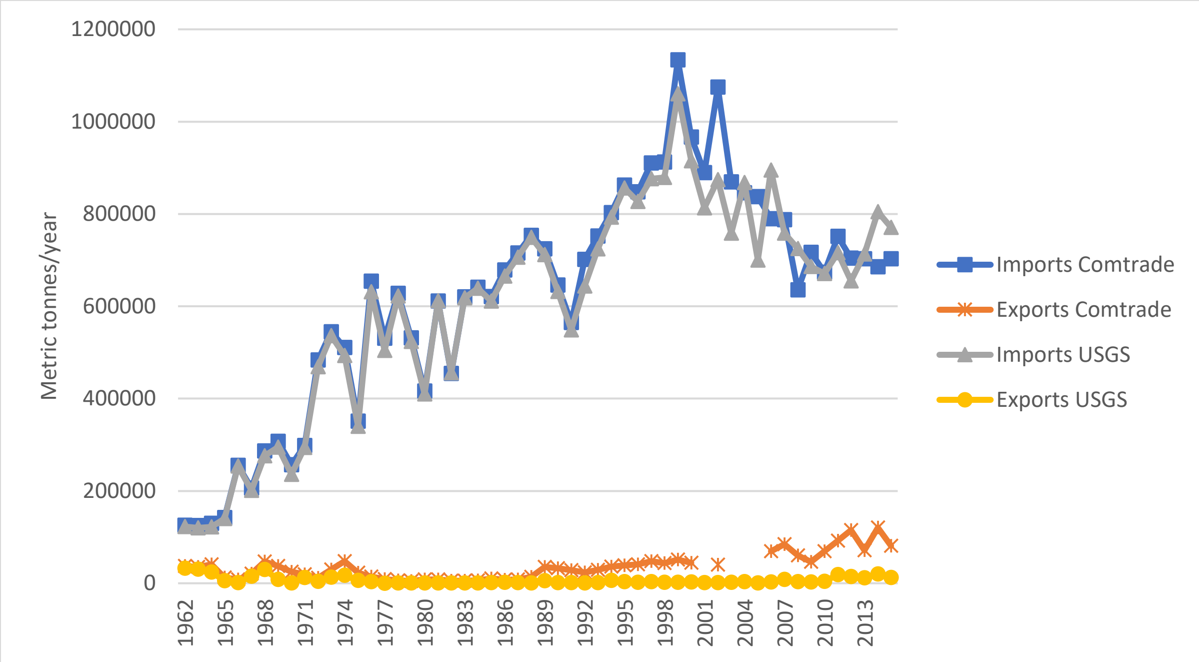
Figure SI.8: Lead imports and exports in the USA, 1962-2015: Comparison of databases



Notes: Comtrade imports and exports include SITC Rev. 3 Commodity Code 685 (lead and lead alloys unwrought and worked). USGS imports and exports include refined lead in various shapes and forms and excludes manufactured lead products.

Sources: Comtrade data is from UN Comtrade (2019), USGS data is from Kelly and Matos (2014).

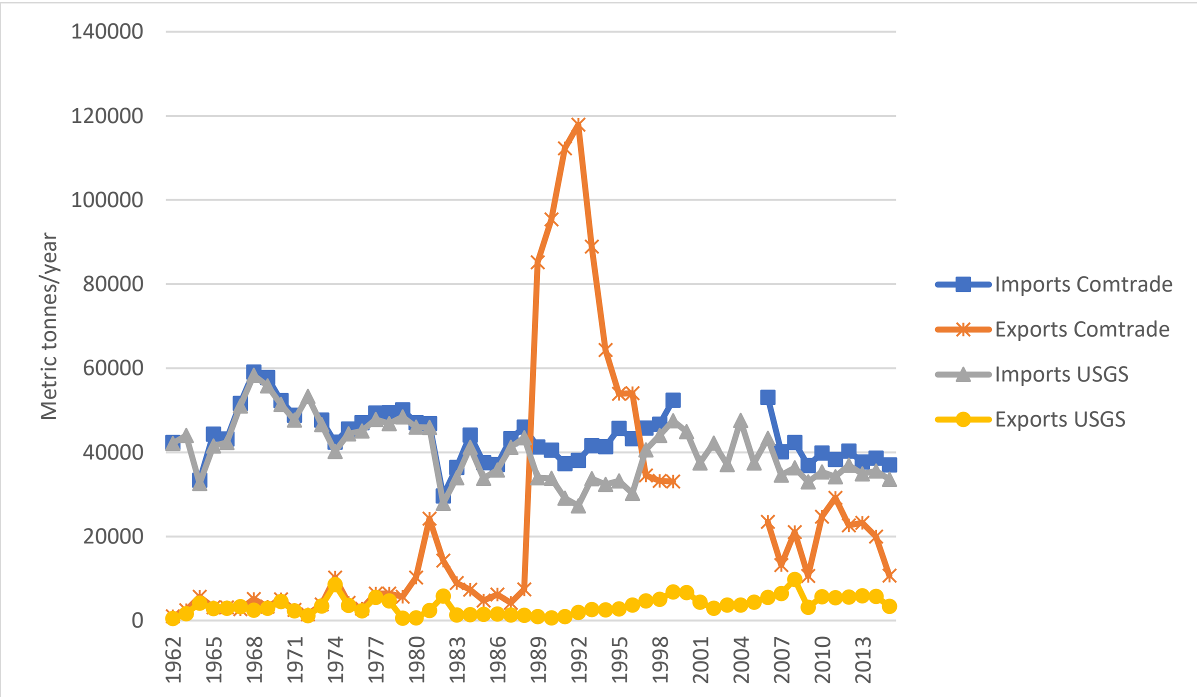
Figure SI.9: Zinc imports and exports in the USA, 1962-2015: Comparison of databases



Notes: Comtrade imports and exports include SITC Rev. 3 Commodity Code 686 (zinc and zinc alloys unwrought and worked). USGS imports and exports include refined zinc in various shapes and forms and excludes manufactured zinc products.

Sources: Comtrade data is from UN Comtrade (2019), USGS data is from Kelly and Matos (2014).

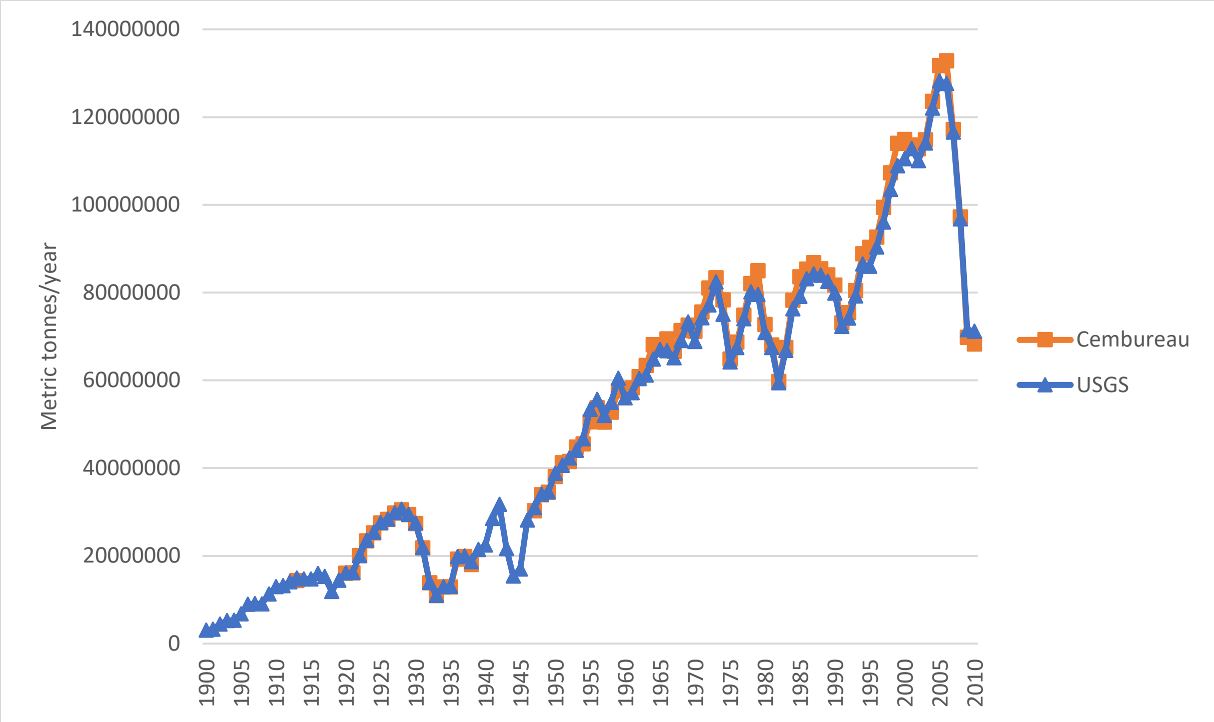
Figure SI.10: Tin imports and exports in the USA, 1962-2015: Comparison of databases



Notes: Comtrade imports and exports include SITC Rev. 3 Commodity Code 687 (tin and tin alloys unwrought and worked). USGS imports and exports include refined tin in various shapes and forms and excludes manufactured tin products.

Sources: Comtrade data is from UN Comtrade (2019), USGS data is from Kelly and Matos (2014).

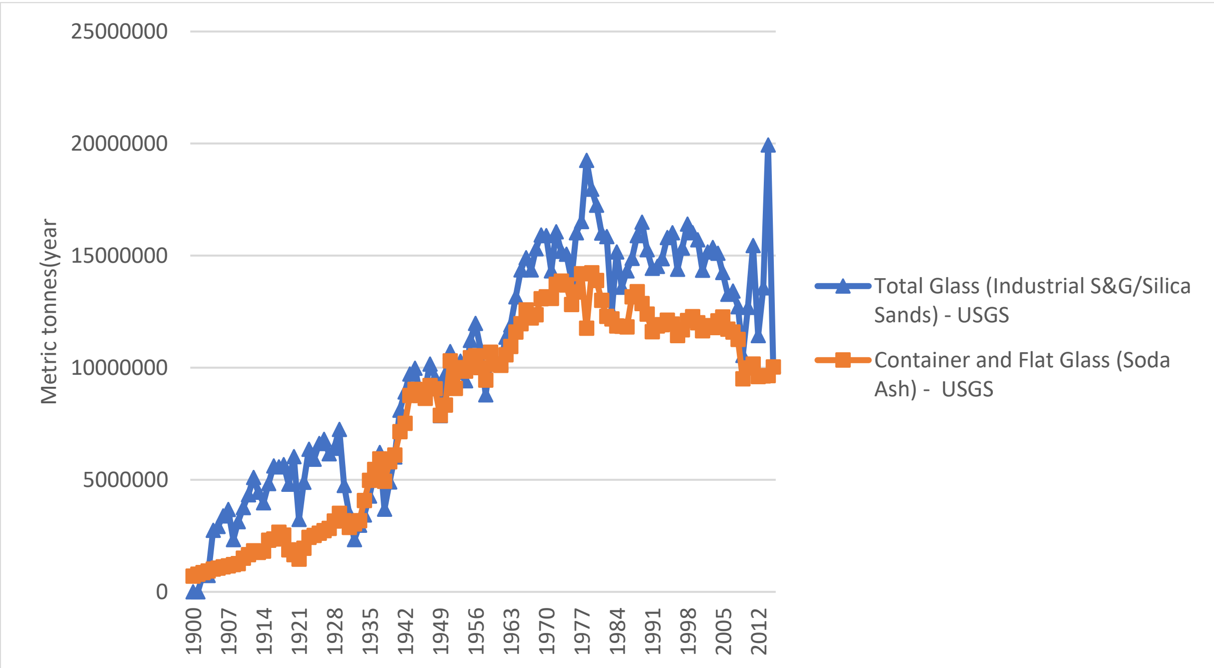
Figure SI.11: Apparent consumption of cement in the USA, 1900-2010: Comparison of databases



Notes: Cement consumption is equal to domestic production of cement plus imports minus exports.

Sources: Cembureau data is from Cembureau (2019), USGS data is from Kelly and Matos (2014).

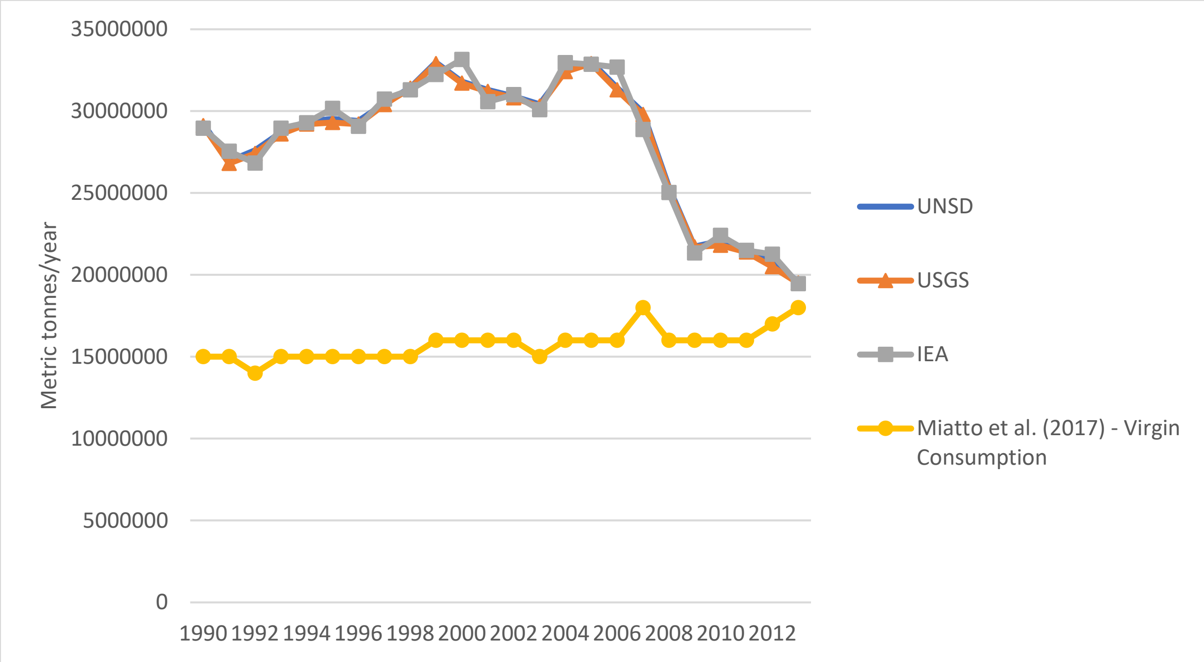
Figure SI.12: Estimated glass production in the USA, 1900-2015



Notes: Glass production was estimated based on soda ash and silica sands consumption and glass production coefficients.

Sources: Data on silica sands and soda ash consumption is from Kelly and Matos (2014) and USGS (2019a), production coefficients are from Ruth and Dell´Anno (1997).

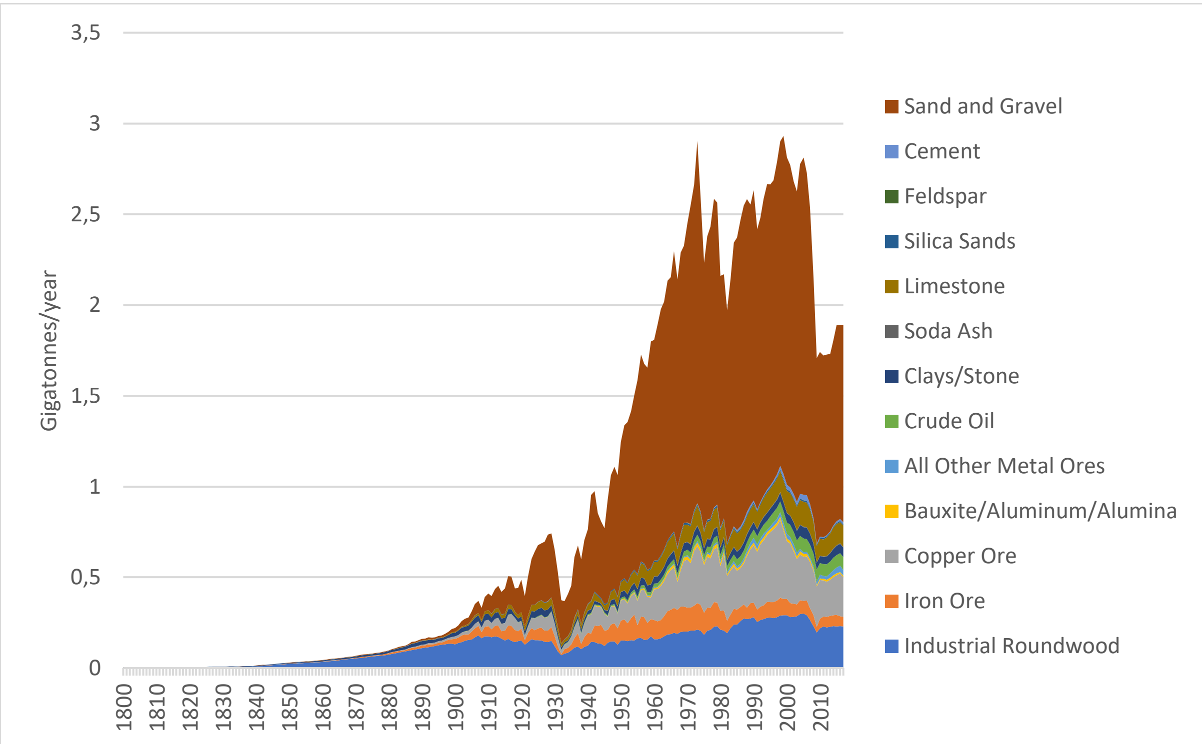
Figure SI.13: Apparent consumption of bitumen in the USA, 1990-2013: Comparison of databases



Notes: Bitumen consumption equals domestic production of bitumen plus imports minus exports.

Sources: USGS data is from Kelly and Matos (2014), UNSD data is from UNSD (2019), IEA data is from IEA (2019), Miatto et al. (2017) – Virgin consumption data is from Miatto et al. (2017).

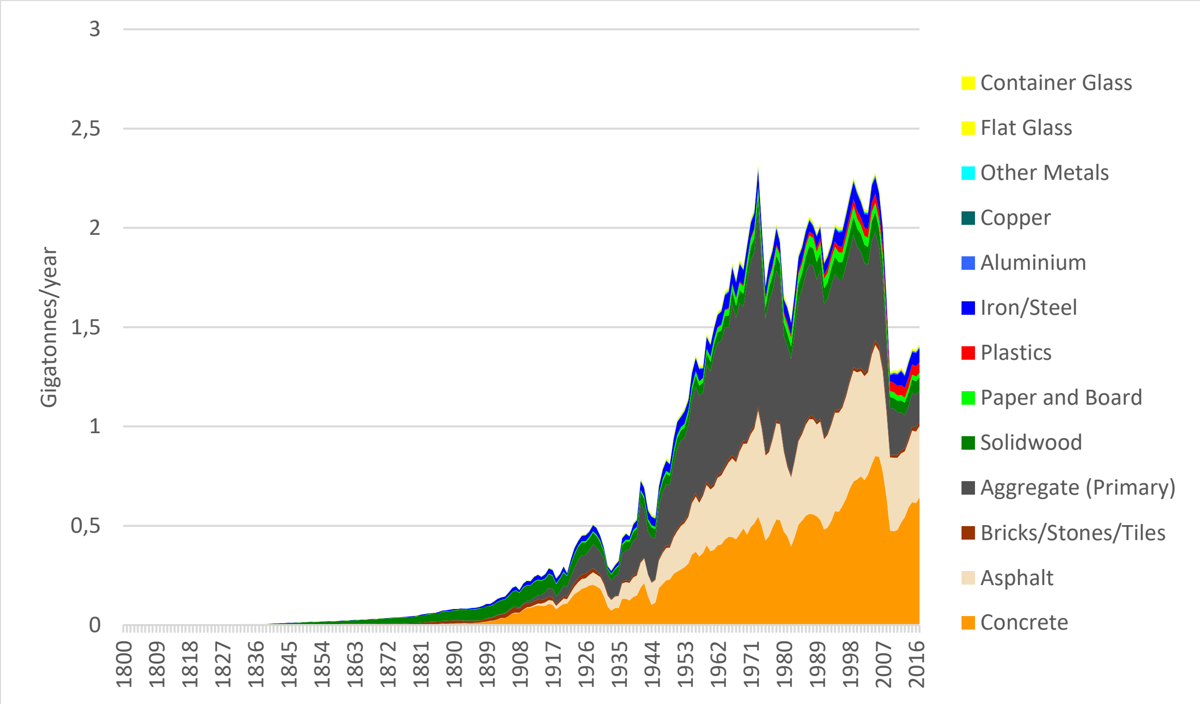
Figure SI.14: Stock-building materials consumption in the USA, 1800-2017



Notes: Stock-building materials include all materials in ew-MFA which are not used for either energy conversion or other dissipative uses and thus stay in the socio-economic system on average for longer than one year.

Sources: Own calculations

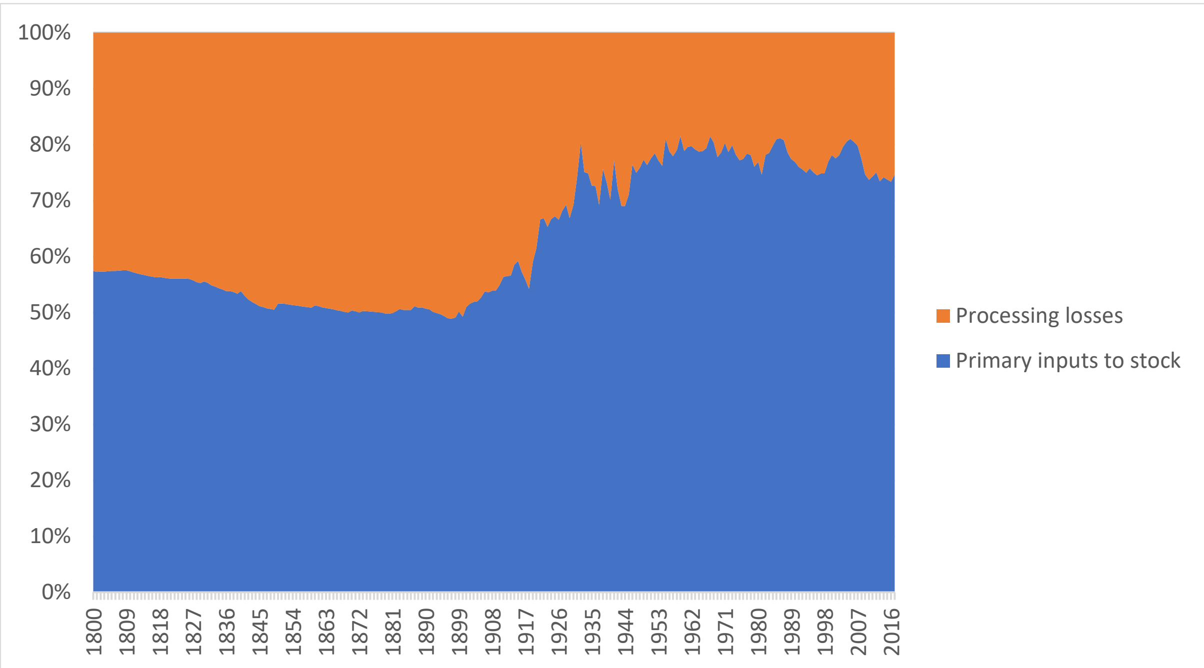
Figure SI.15: Primary inputs to stock in the USA, 1800-2017



Notes: Primary inputs to stock are equal to stock-building materials minus processing losses (e.g. loss of moisture content during bricks production, CO_2 emissions during cement production or waste rock from ores processing).

Sources: Own calculations

Figure SI.16: Shares of primary inputs to stock and processing losses in stock-building materials in the USA, 1800-2017



Notes: Stock-building materials are equal to primary inputs to stock plus processing losses (e.g. loss of moisture content during bricks production, CO_2 emissions during cement production or waste rock from ores processing).

Sources: Own calculations

4.1.2 Processing and Manufacturing Losses

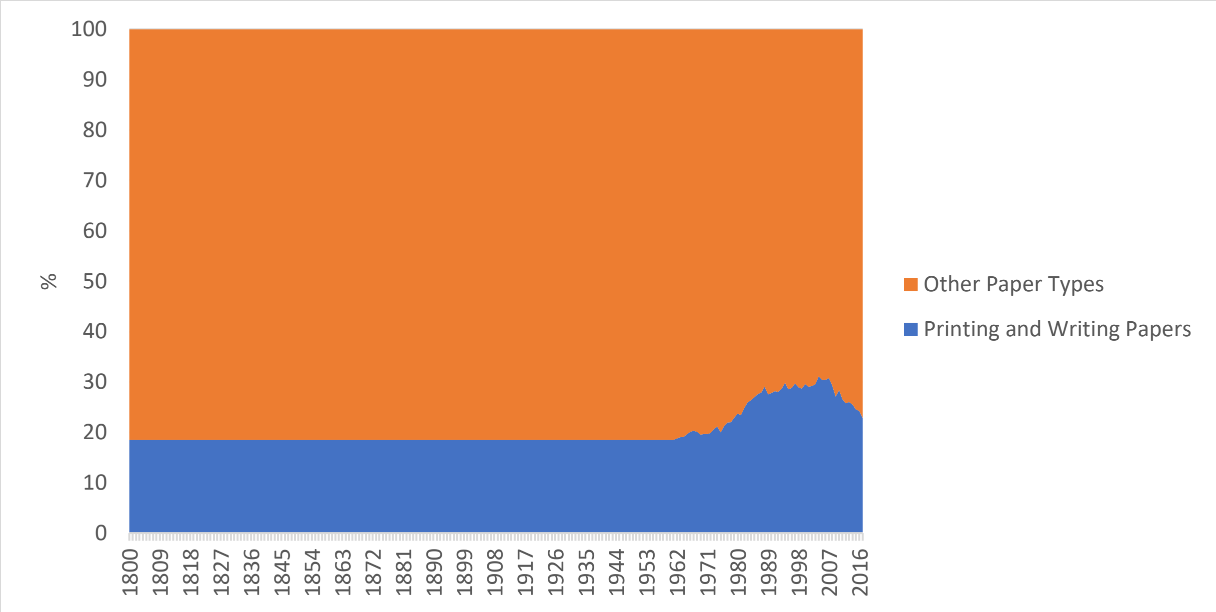
Table SI.2: Processing and manufacturing losses used in the MISO-model

Material Inputs to Stock	**Processing Losses**	**Manufacturing Losses**	**Sources (Processing Losses/Manufacturing Losses)**
Solidwood	54 %	5 %	FAOSTAT (2019), Kelly and Matos (2014), Krausmann et al. (2018) / Cochran and Townsend (2010)
Paper and Paperboard	54 %	0 %	FAOSTAT (2019), Kelly and Matos (2014), Krausmann et al. (2018) / Kelly and Matos (2014)
Iron (Steel)	58 %	2.2 %	Cullen et al. (2012), Krausmann et al. (2018) / Cullen et al. (2012)
Aluminum	75 %	3.7 %	Kelly and Matos (2014), USGS (2018) / Chen and Graedel (2012)
Copper	97.5-99.5 %	1 %	Gierlinger and Krausmann (2012), Wang et al. (2015) / Glöser et al. (2013)
Other Metals	94 %	2.3 %	Krausmann et al. (2017) / Gerst and Graedel (2008), Kelly and Matos (2014), Krausmann et al. (2017)
Bricks and Tiles/Stones	26/0 %	4 %	Krausmann et al. (2017)/ Cochran and Townsend (2010)
Concrete	42/0 %	3 %	Kapur et al. (2009), Krausmann et al. (2017) / Cochran and Townsend (2010)
Container Glass	15 %	0 %	Ruth and Dell´Anno (1997) / Butler and Hooper (2011)
Flat Glass	20 %	0 %	Ruth and Dell´Anno (1997) / Butler and Hooper (2011)
Sand and Gravel for sub-base and base-course Layers	0 %	0 %	Krausmann et al. (2017) / Krausmann et al. (2017)
Asphalt	0 %	0 %	Krausmann et al. (2017) / Cochran and Townsend (2010)
Plastics	0 %	9 %	Krausmann et al. (2017) / EPA (2018a), Van Eygen et al. (2017)

Notes: Processing losses for solidwood and paper and paperboard are average processing losses for the period 1965-2014, processing losses for aluminum are for aluminum production from Bauxite and processing losses for concrete are 42 % for cement production and 0 % for sand and gravel.

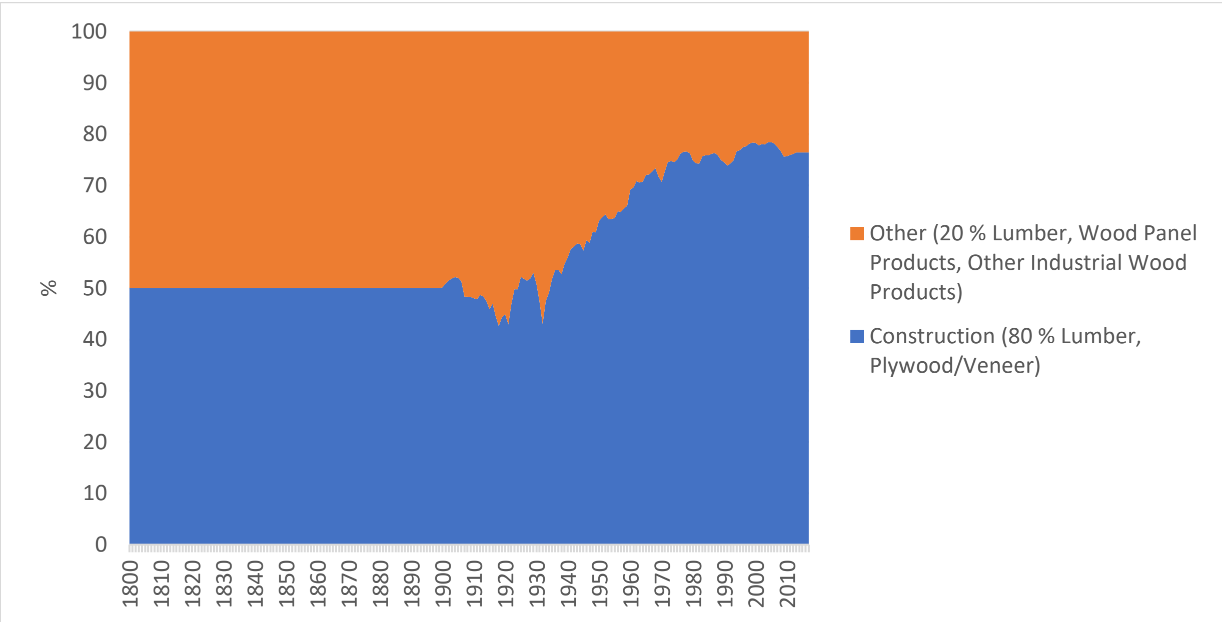
4.1.3 Lifetimes

Figure SI.17: End-use shares for paper and paperboard primary inputs to stock in the USA, 1800-2017



Sources: FAOSTAT (2019), own calculations

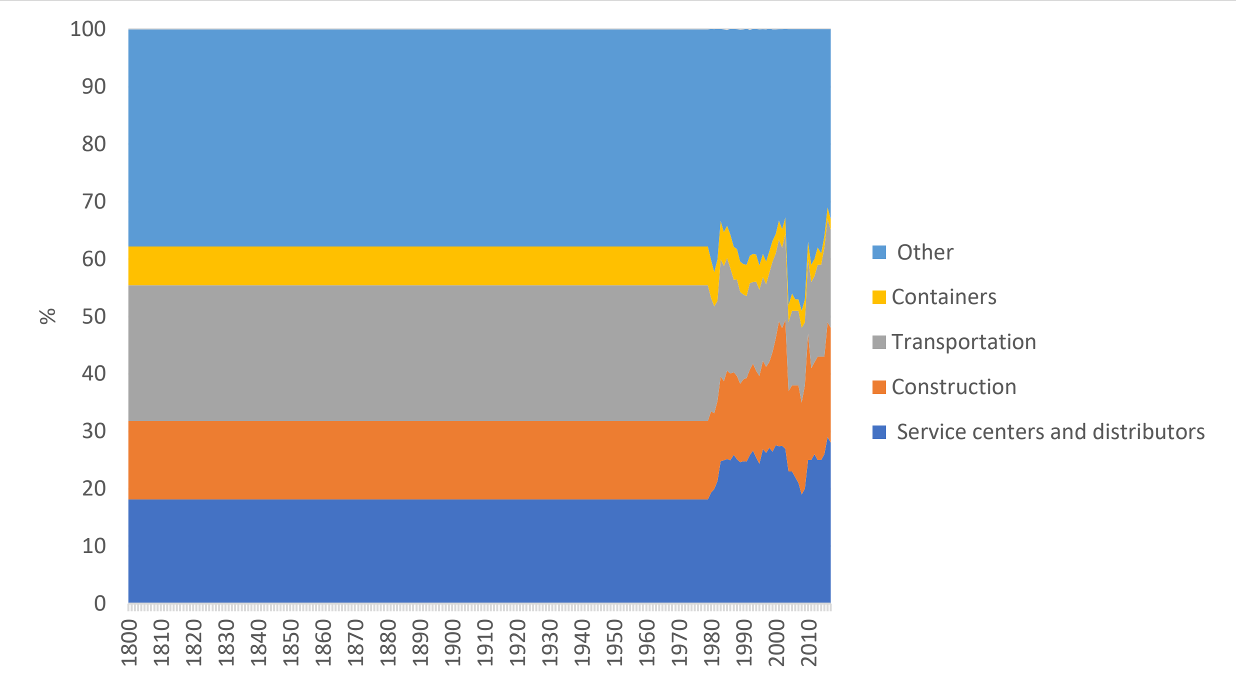
Figure SI.18: End-use shares for solidwood primary inputs to stock in the USA, 1800-2017



Notes: Of all lumber, 80 % is assumed to be used for construction purposes (lifetime: 75 years) and 20 % is assumed to be used for non-construction applications (included in other uses, lifetime: 25 years).

Sources: Bureau of the Census (1975), Kelly and Matos (2014), own calculations.

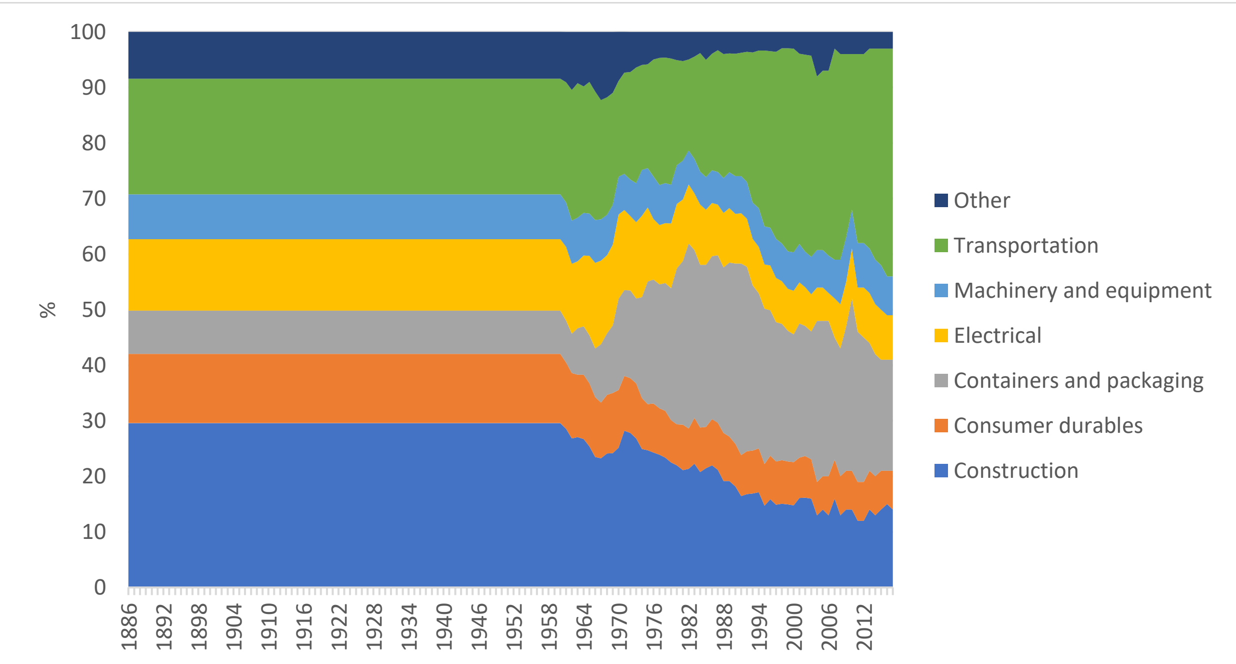
Figure SI.19: End-use shares for iron/steel primary inputs to stock in the USA, 1800-2017



Notes: Steel service centers are "situated between the steel mills that make the finished steel and the manufacturers of steel products [...] Service centers buy steel and process it by using burning units, cut-to-length lines, edgers, grinders, levelers, plasma tables, saws, shears, and slitters before reselling it to manufacturers, or they distribute it without additional processing" (Fenton 2005: 20).

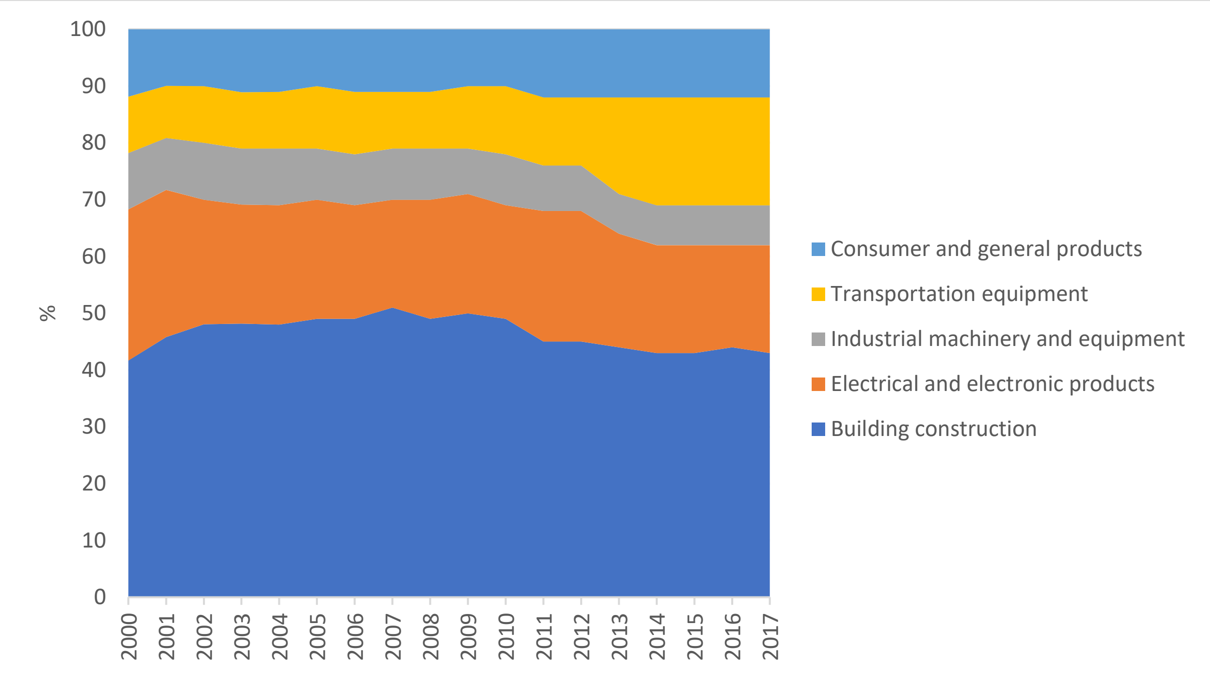
Sources: Kelly and Matos (2014), USGS (2019a), own calculations.

Figure SI.20: End-use shares for aluminum primary inputs to stock in the USA, 1886-2017



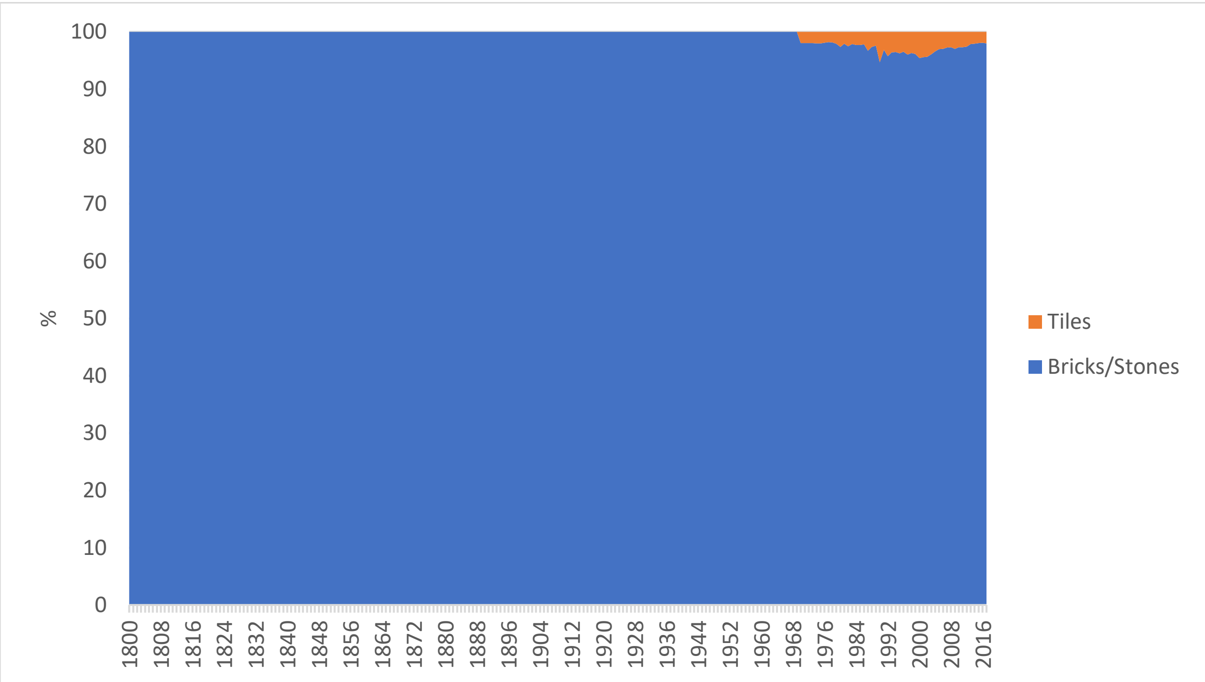
Sources: Chen and Graedel (2012), Kelly and Matos (2014), USGS (2019a), own calculations

Figure SI.21: End-use shares for copper primary inputs to stock in the USA, 2000-2017



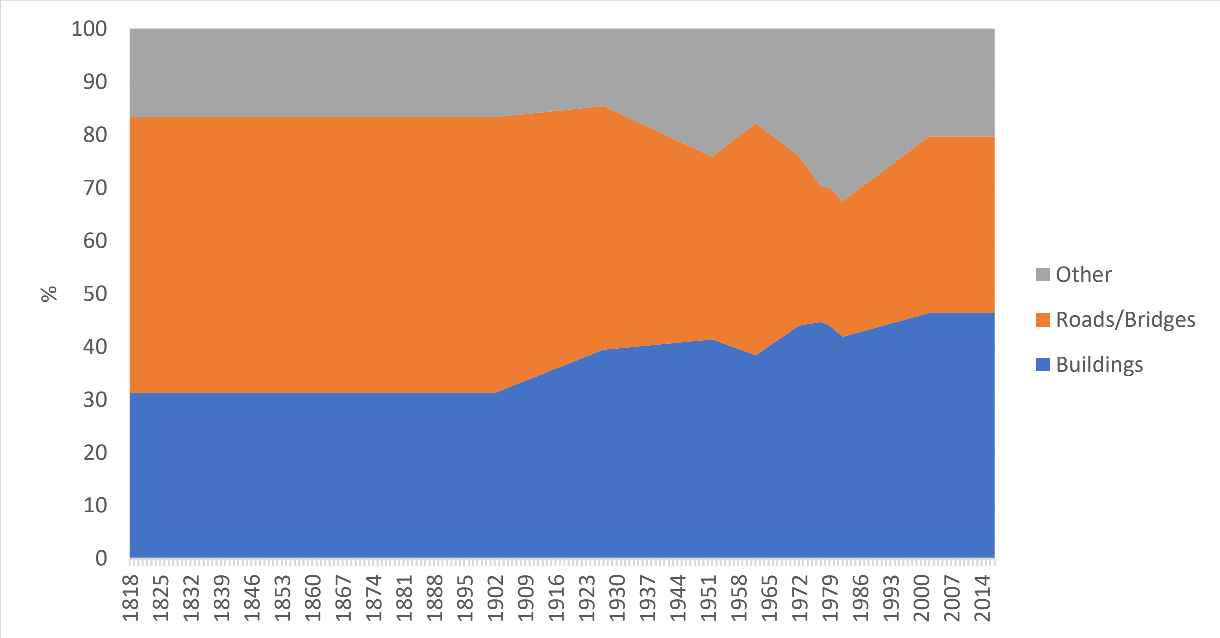
Sources: Kelly and Matos (2014), USGS (2019a), own calculations

Figure SI.22: End-use shares for bricks, stones and tiles primary inputs to stock in the USA, 1800-2017



Sources: Bureau of the Census (1975), Kelly and Matos (2014), USGS (2019a), own calculations

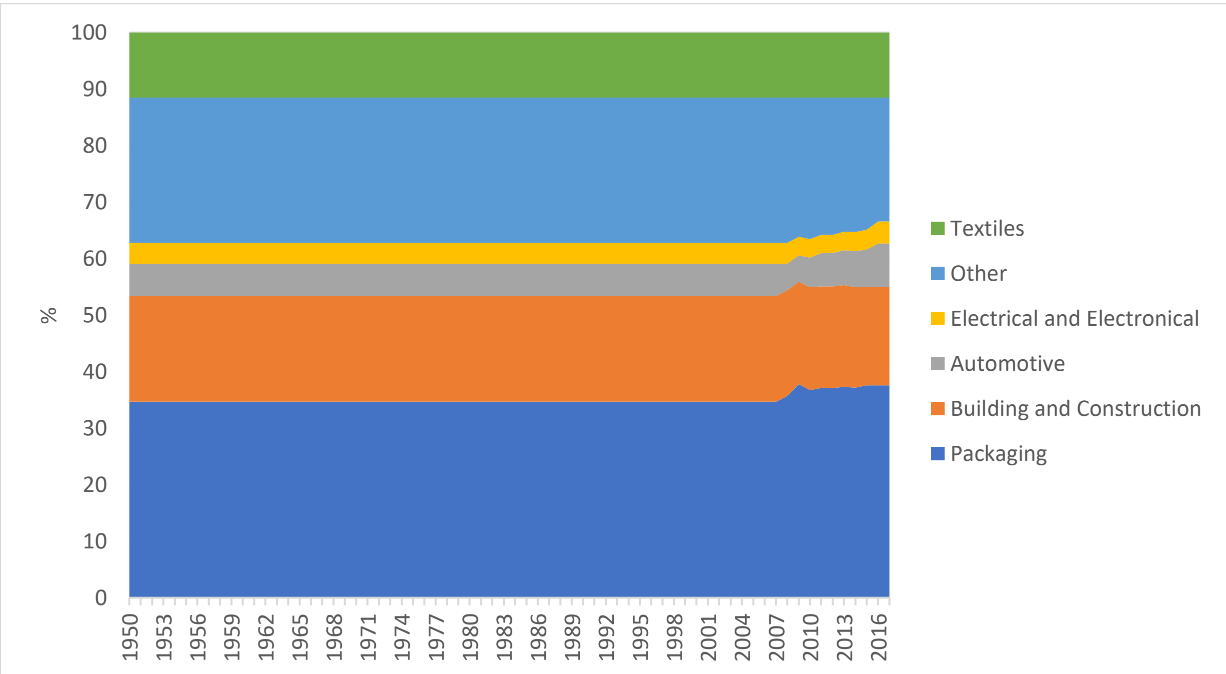
Figure SI.23: End-use shares for concrete primary inputs to stock in the USA, 1818-2017



Notes: End-use shares were linearly interpolated between available data points (1902, 1927, 1952, 1962, 1977, 1979, 1982, 2002).

Sources: Cochran and Townsend (2010), own calculations

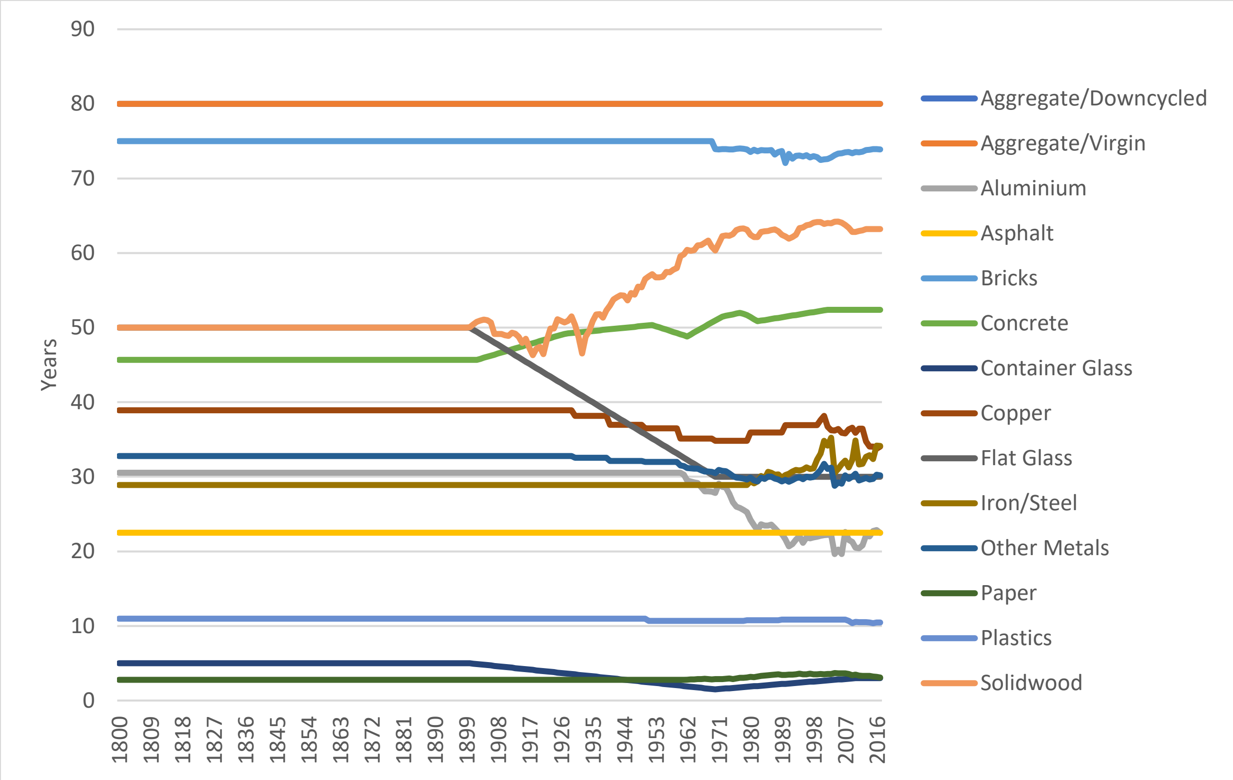
Figure SI.24: End-use shares for plastics primary inputs to stock in the USA, 1950-2017



Notes: End-use shares are for Europe which are relatively similar for the USA (Geyer et al., 2017).

Sources: Geyer et al. (2017), Plastics Europe (2019), own calculations

Figure SI.25: Mean lifetimes for material inputs to stock in the MISO-model, 1800-2017



Sources: Own calculations

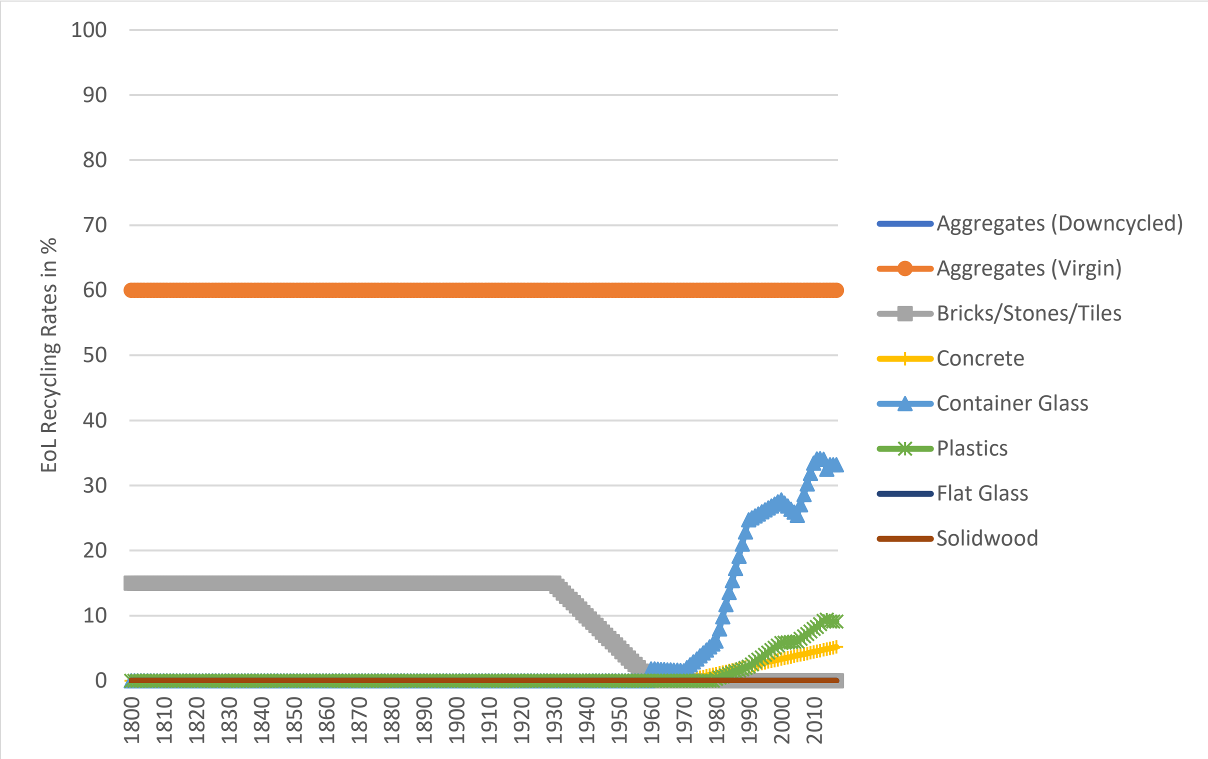
Table SI.3: End-use shares, lifetimes for end-uses and weighted lifetimes in 2017 for material inputs to stock in the MISO-model

Material Inputs to Stock	Application (mean lifetime in years)	Lifetimes 1900 (years)	Lifetimes 2017 (years)	Sources (End-Use Shares/Lifetimes)
Solidwood	Construction: Lumber and Plywood/Veneer (75), Other: Lumber, Wood Panel Products and Other Industrial Wood Products (25)	50±15	63±20	Bureau of the Census (1975), Kelly and Matos (2014), EPA (2015)/ Cochran and Townsend (2010)
Paper and Paperboard	Printing and Writing Papers (9), Other: e.g. Newspapers, Household Sanitary Papers (1)	3±1	3±2	FAOSTAT (2019)/ IPCC (2003)
Iron/Steel	Construction (75), Containers (1), Other (23), Service Center and Distributors (30), Transportation (23)	29±10	34±12	Kelly and Matos (2014), USGS (2019a)/ Müller et al. (2011)
Aluminum	Construction (55), Consumer Durables (15), Containers and Packaging (1), Electrical (40), Machinery and Equipment (25), Transportation (20), Other (12)	31±13	22±8	Chen and Graedel (2012), Kelly and Matos (2014), USGS (2019a)/ Chen (2013)
Copper	On-Site Waste (1), Plumbing (55), Wiring (45), Built-in Appliances (20), Industrial Electric and Electronic Products (20), Consumer Electric and Electronic Products (13), Infrastructure (65), Motor Vehicles (13), Other Transport (30)	39±23	34±20	Spatari et al. (2005), Kelly and Matos (2014), USGS (2019a)/ Spatari et al. (2005)
Other Metals	Various Applications (Alloys, Batteries, ...): Other Metals are treated as one category where lifetimes are the average of the lifetimes of iron/steel, aluminum and copper	33±16	30±15	Single Use-Category/ Gerst and Graedel (2008), Kelly and Matos (2014), Wiedenhofer et al. (2019)
Bricks, Stones and Tiles	Bricks and Stones for Construction (75), Tiles (25)	75±25	74±25	Bureau of the Census (1975), Kelly and Matos (2014)/ Cochran and Townsend (2010)
Concrete	Buildings (75), Roads/Bridges (32), Other (35)	46±15	52±17	Cochran and Townsend (2010)/ Cochran and Townsend (2010)
Flat Glass	Flat Glass applications (e.g. Windows, Glass Doors, …) (30-50)	50±15	30±9	Single Use-Category/ Wiedenhofer et al. (2019)
Container Glass	Container Glass applications (e.g. Bottles, Jars, …) (1.5-5)	5±2	3±1	Single Use-Category/ Wiedenhofer et al. (2019)
Sand and Gravel for sub-base and base-course layers	Buildings, Roads, Infrastructures (80)	80±72	80±72	Single Use-Category/ Wiedenhofer et al. (2019)
Asphalt	Roads (23)	23±11	23±11	Single Use-Category/ Cochran and Townsend (2010)
Plastics	Packaging (1), Building and Construction (35), Automotive (14-17), Electrical and Electronical (8), Textiles (5), Other (8)	11±7	10±6	Geyer et al. (2017), Plastics Europe (2019)/ Geyer et al. (2017), Bento et al. (2016)

Notes: Lifetimes are shown with a mean (left value) and three standard deviations (right value), 99.7 % of stocks reach the end of their lifetime between the lower and the upper bound of the depicted lifetimes (e.g. 14 and 54 years for copper). For 2000-2017, lifetimes of Spatari et al. were allocated to end-use shares by Kelly and Matos/Bureau of the Census as follows: Building Construction: On-Site Waste, Plumbing, Wiring, Built-in Appliances (average lifetimes, assuming constant shares of 1999); Infrastructure: Electrical and Electronic Products; Industrial Electric and Electronic Products: Industrial Machinery and Equipment; Motor Vehicles and other Transport: Transportation Equipment (average lifetimes, assuming constant shares of 1999); Consumer Electric and Electronic Products: Consumer and general Products.

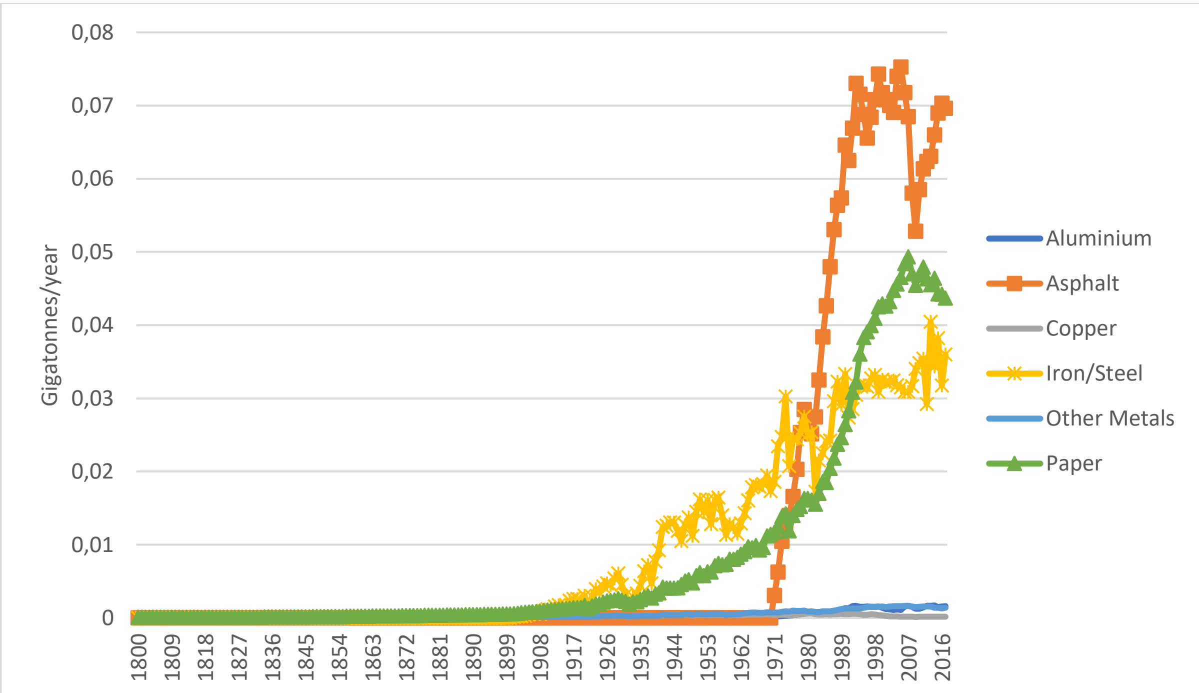
4.1.4 Recycling and Downcycling

Figure SI.26: End-of-Life recycling rates assumed in the MISO-model, 1800-2017



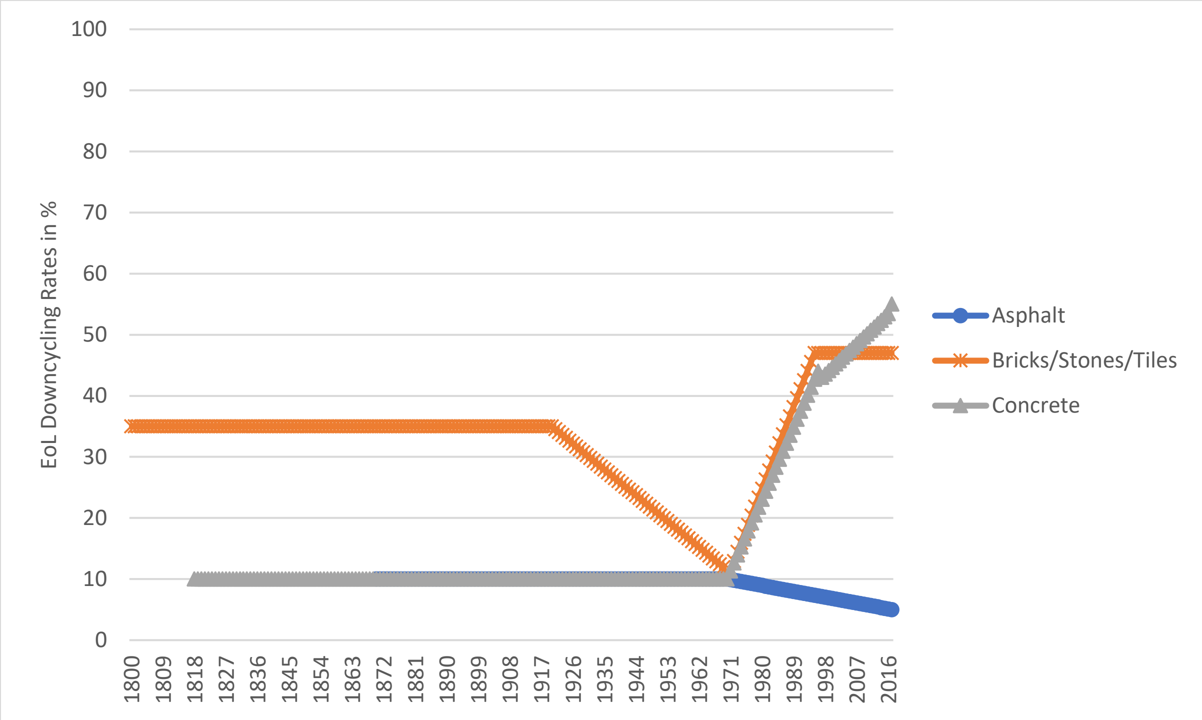
Sources: Wilburn and Goonan (1998), Kelly (1998), Sandler (2003), Krausmann et al. (2017), EPA (2018a), Wiedenhofer et al. (2019), own calculations

Figure SI.27: End-of-Life recycling flows assumed in the MISO-model, 1800-2017



Sources: USDT (1993), Wernick et al. (1996), Wilburn and Goonan (1998), Brown (2013), Kelly and Matos (2014), FHA (2016), NAPA (2019a), own calculations

Figure SI.28: End-of-Life downcycling rates in the MISO-model, 1800-2017



Sources: Wilburn and Goonan (1998), Sandler (2003), Krausmann et al. (2017), NAPA (2019a), Wiedenhofer et al. (2019), own calculations

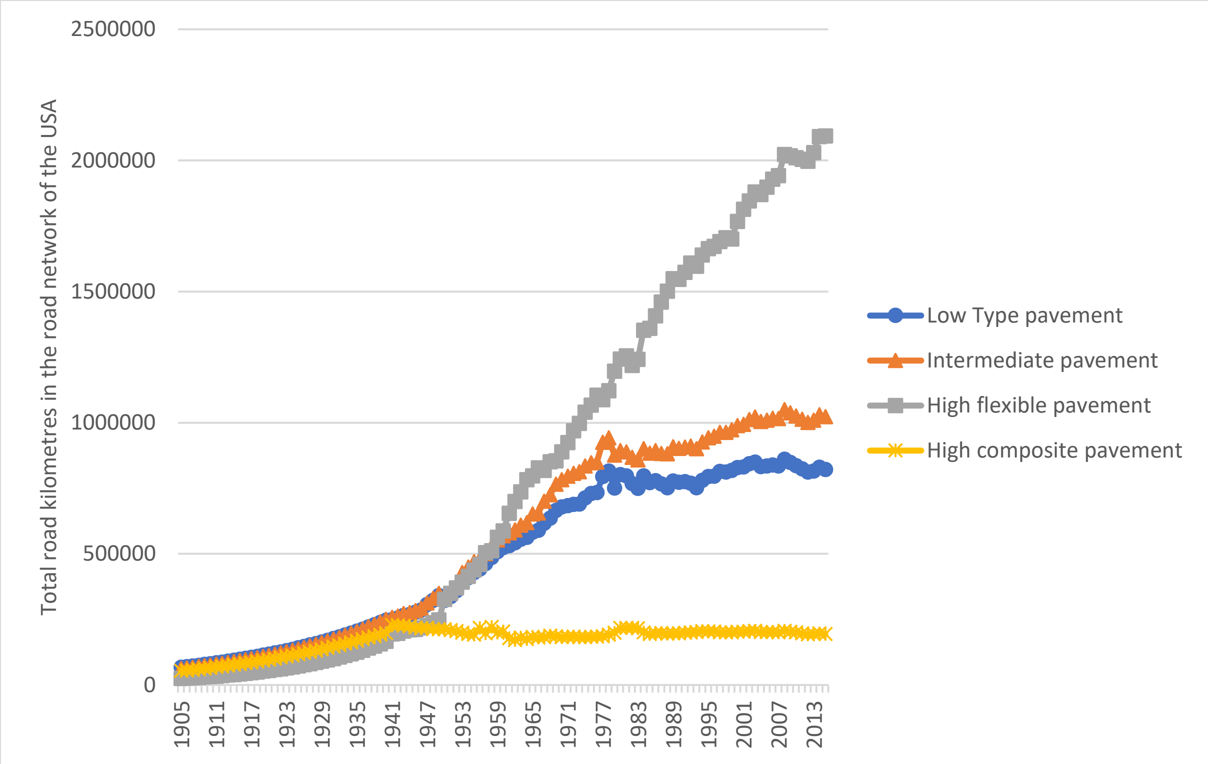
4.1.5 Multipliers

Table SI.4: Multipliers for asphaltic road types

	Depth (m)	Length (m)	Width (m)	Density (t/m³)	Asphalt (t)	Sand and Gravel (t)	Multiplier
Low Type pavement	0.02/0.05	1000	3.75	2.24/2.19	168	410	2.4
Intermediate pavement	0.035/0.135	1000	6	2.24/2.19	470.4	1770	3.8
High flexible pavement	0.05/0.21	1000	10.5	2.24/2.19	1176	4818	4.1
High composite pavement	0.05/0.03	1000	12	2.24/2.19	1344	787	0.6

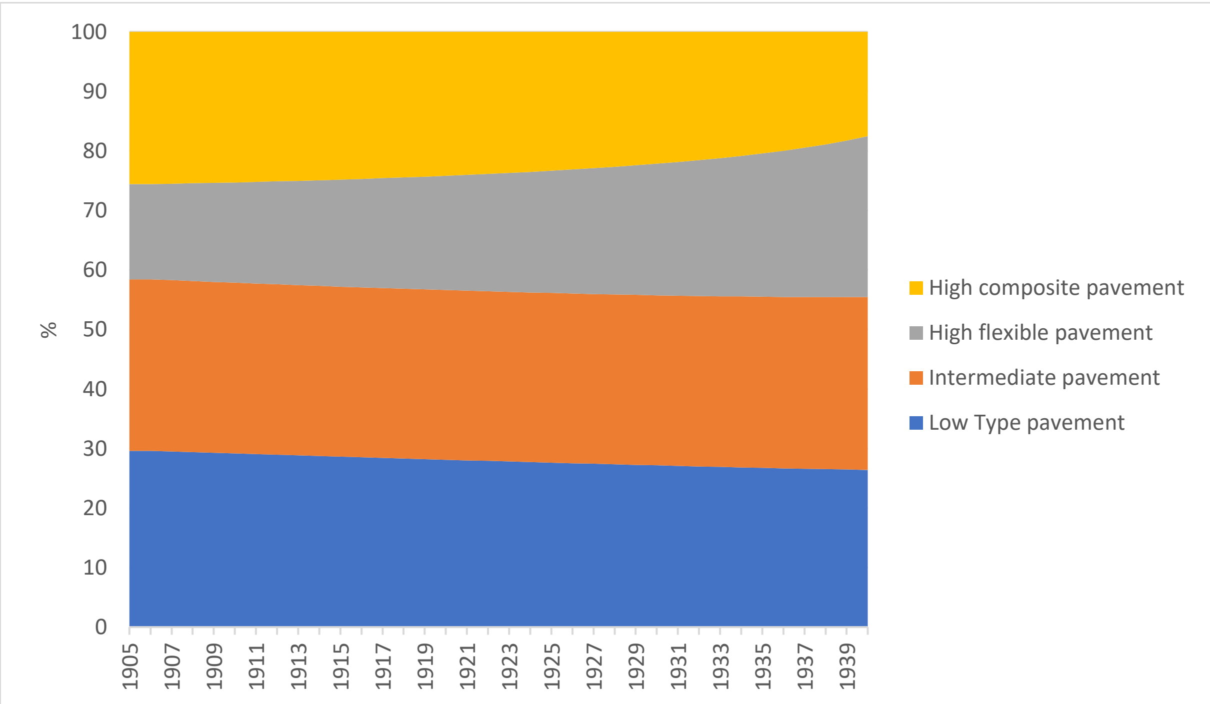
Notes: Depth/length/width of roads is based on Miatto et al. (2017), densities are based on Krausmann et al. (2018). Width for low type pavement/high flexible pavement are the averages for combined rural and urban roads in Miatto et al. (2017). The left value in the depth column is for asphalt, the right value in the depth column is for sand and gravel. The amount of asphalt (in tonnes) and sand and gravel (in tonnes) needed per road kilometre is equal to depth*length*width*density. The multiplier is equal to sand and gravel in asphaltic road types divided by asphalt needed in asphaltic road types.

Figure SI.29: Total road kilometres for asphaltic road types in the USA, 1905-2015



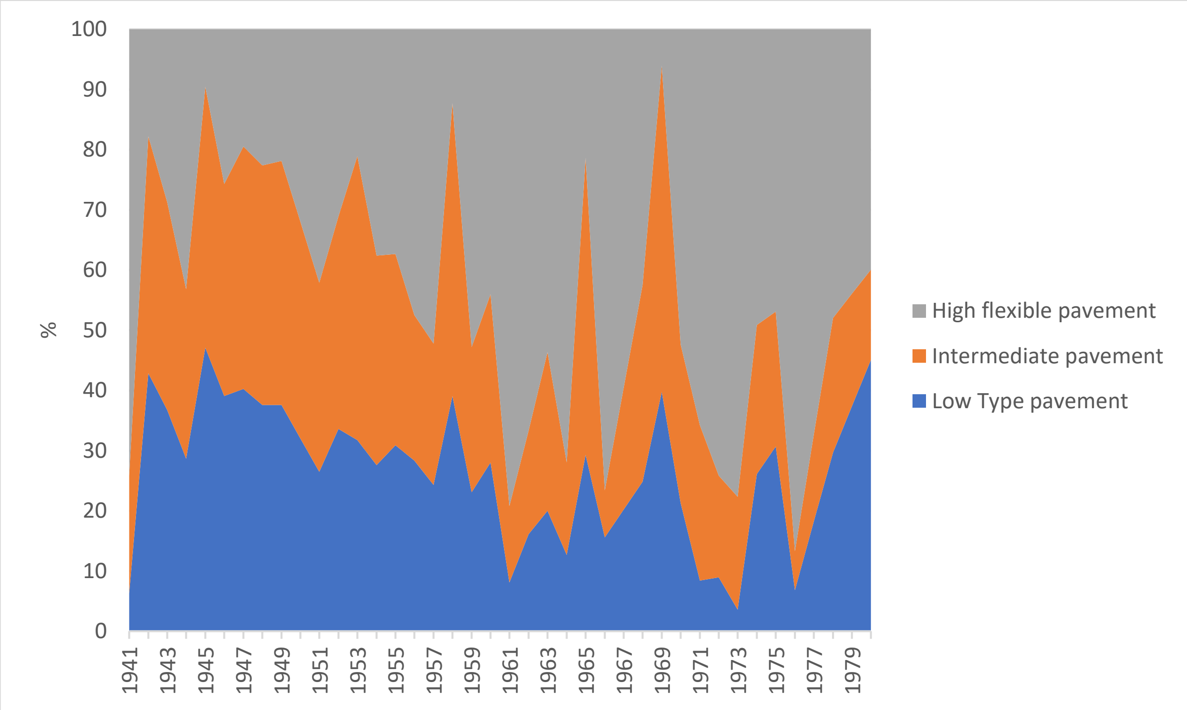
Source: Miatto et al. (2017), own calculations

Figure SI.30: Shares of road types in total new asphaltic road construction in the USA, 1905-1940



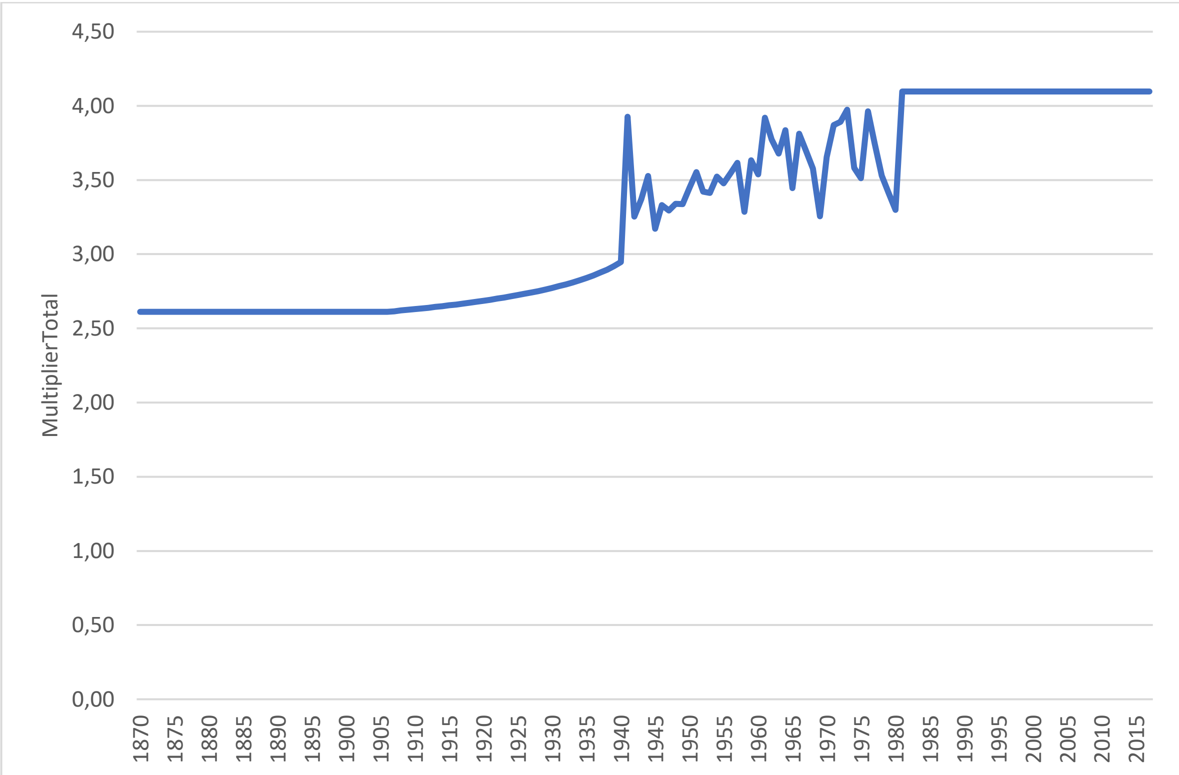
Source: Miatto et al. (2017), own calculations

Figure SI.31: Shares of road types in total new asphaltic road construction in the USA, 1941-1980



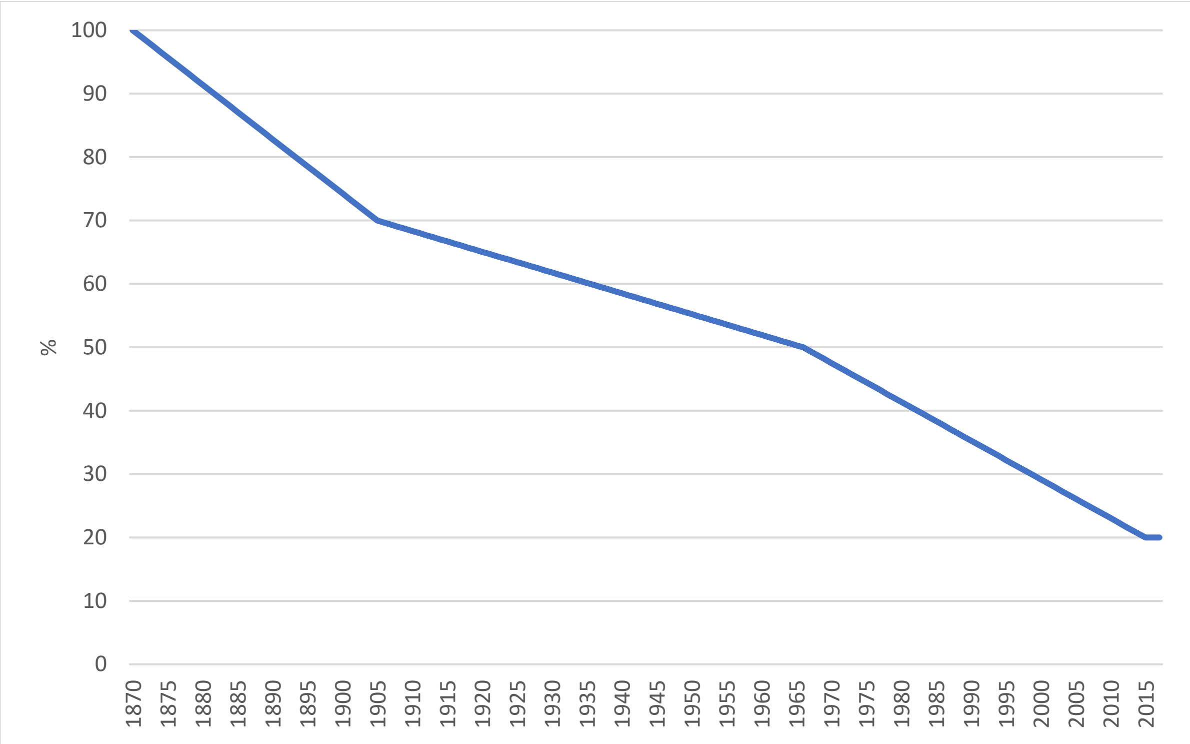
Source: Miatto et al. (2017), own calculations

Figure SI.32: Calculated average MultiplierTotal for asphaltic roads, 1870-2017



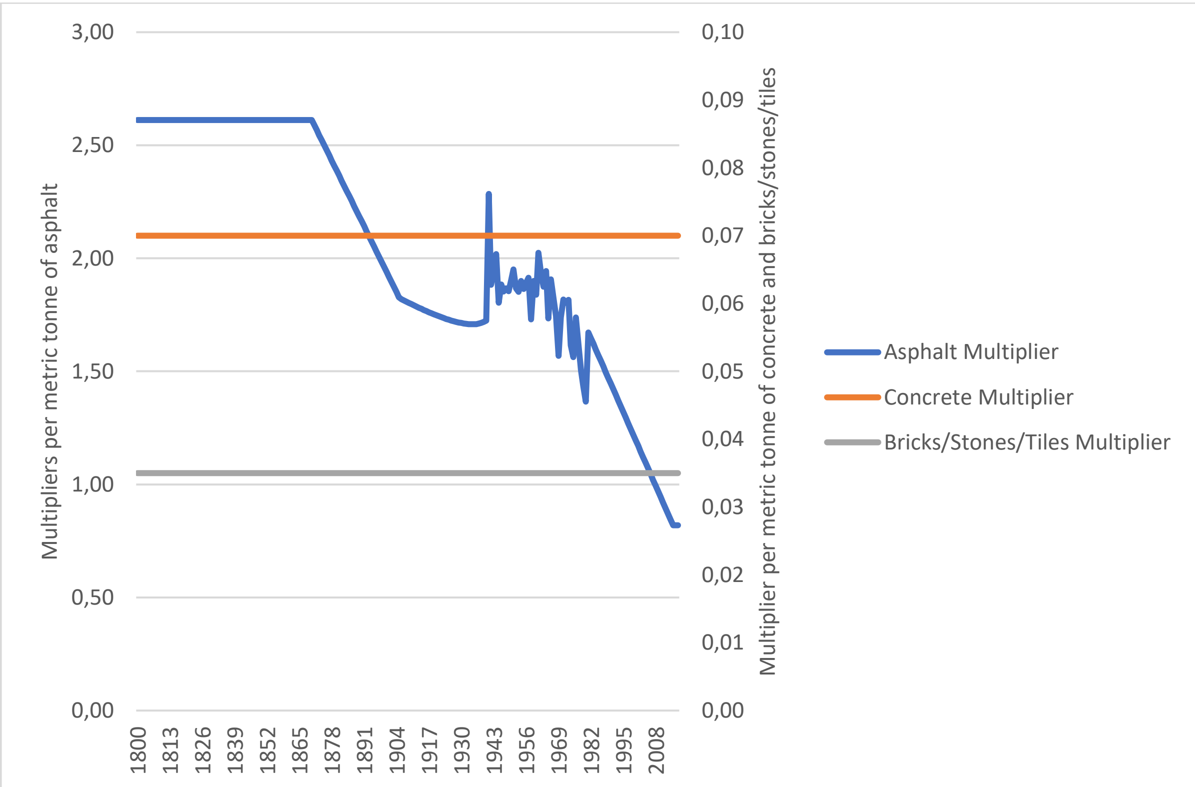
Source: Miatto et al. (2017), own calculations

Figure SI.33: Assumed shares of asphalt used for new road construction in the USA, 1870-2017



Source: Miatto et al. (2017), own calculations

Figure SI.34: Multipliers for Sub-Base and Base-Course Layers 1800-2017

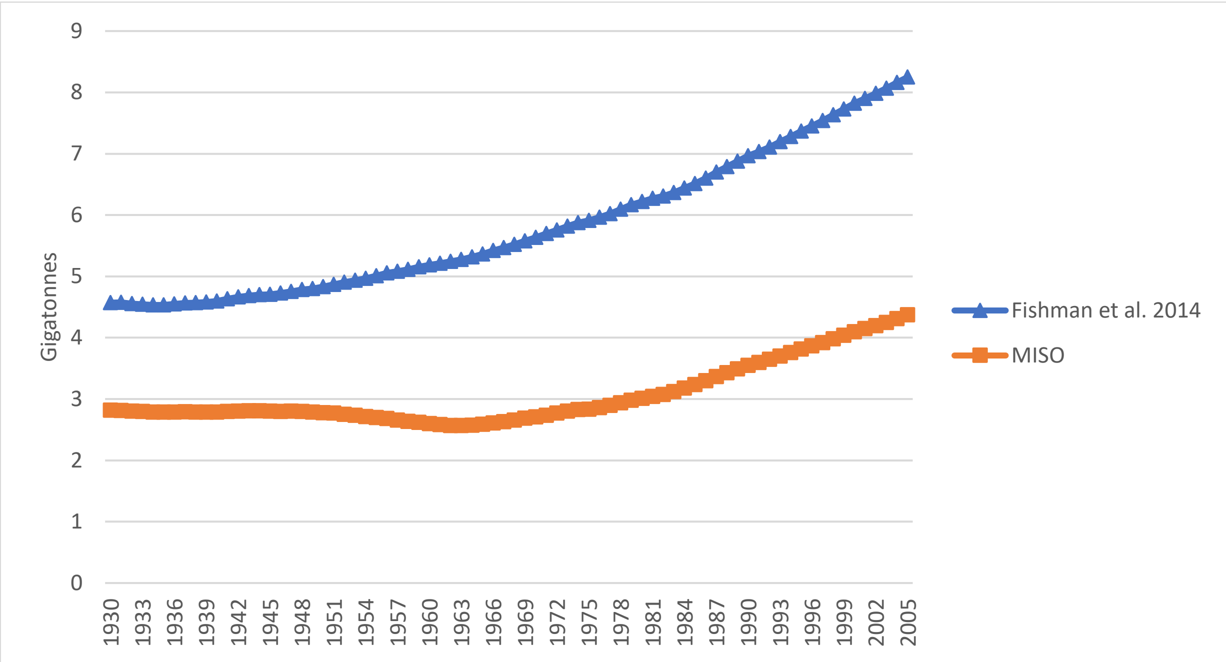


Source: Miatto et al. (2017), own calculations

4.2 Comparison with Results of other Studies

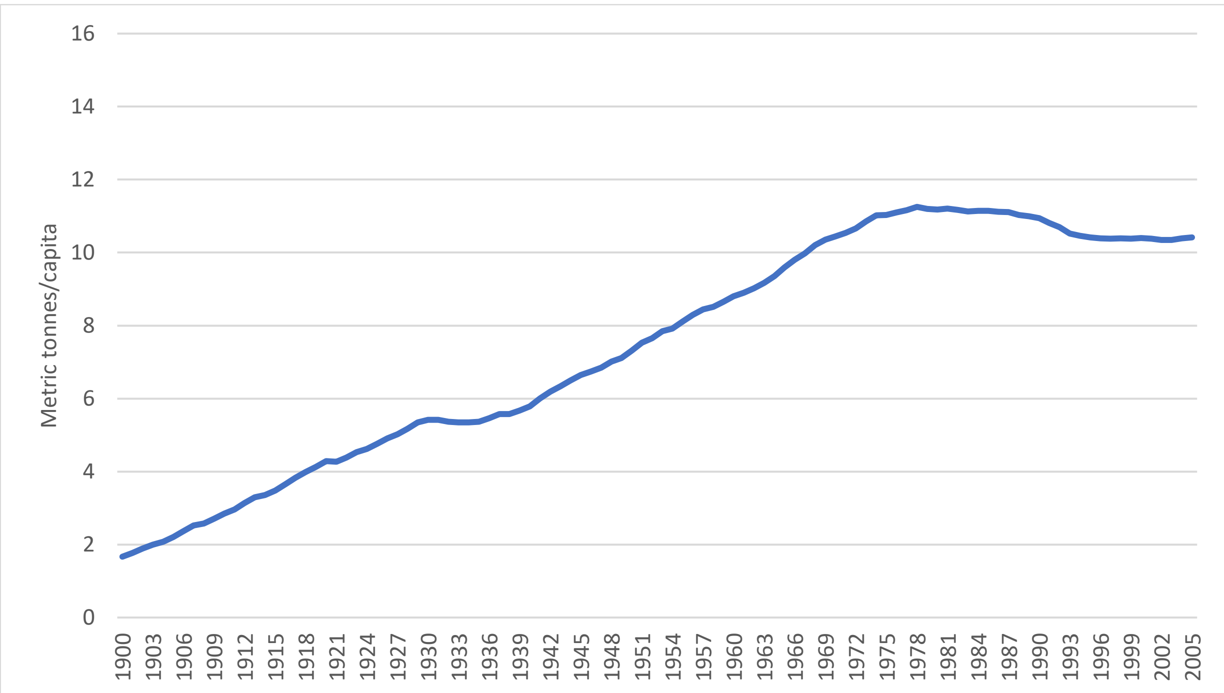
4.2.1 In-Use Stocks

Figure SI.35: Comparison of MISO stock estimates for biomass in the USA for 1930-2005



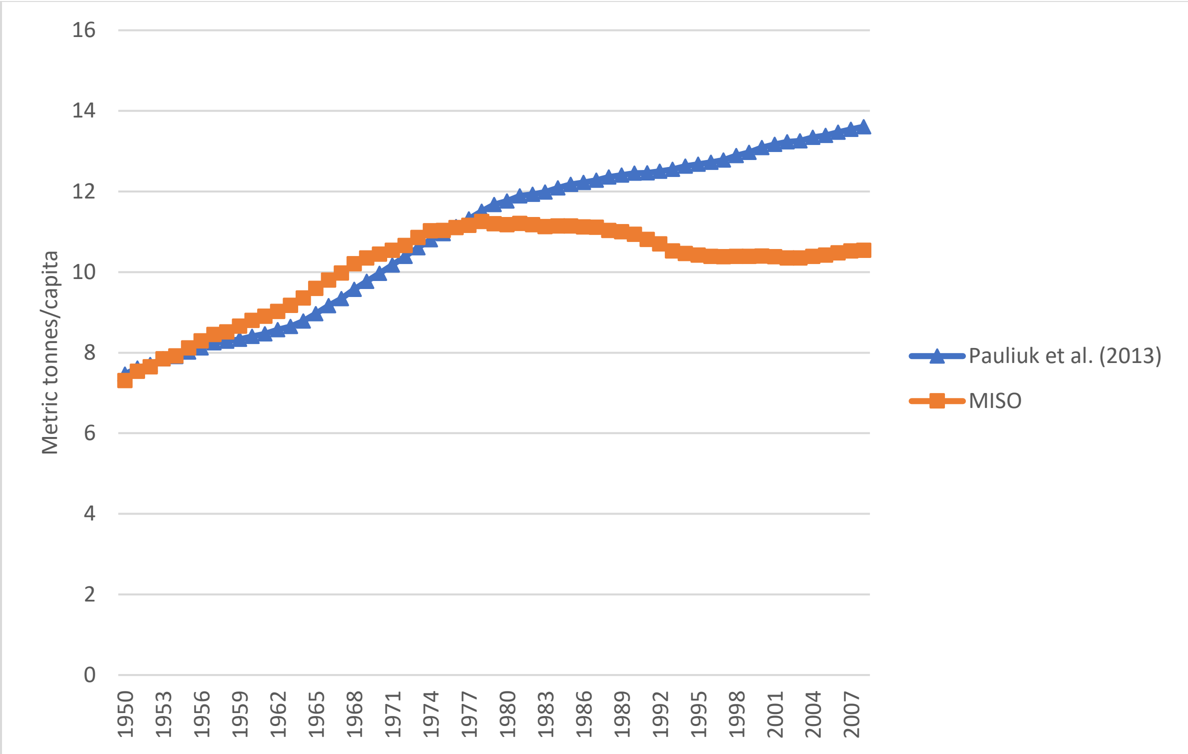
Notes: MISO estimates include stocks for solidwood and paper and paperboard. Stocks estimates from Fishman et al. (2014) include timber.

Figure SI.36: MISO per capita stock estimates for iron/steel in the USA, 1900-2005



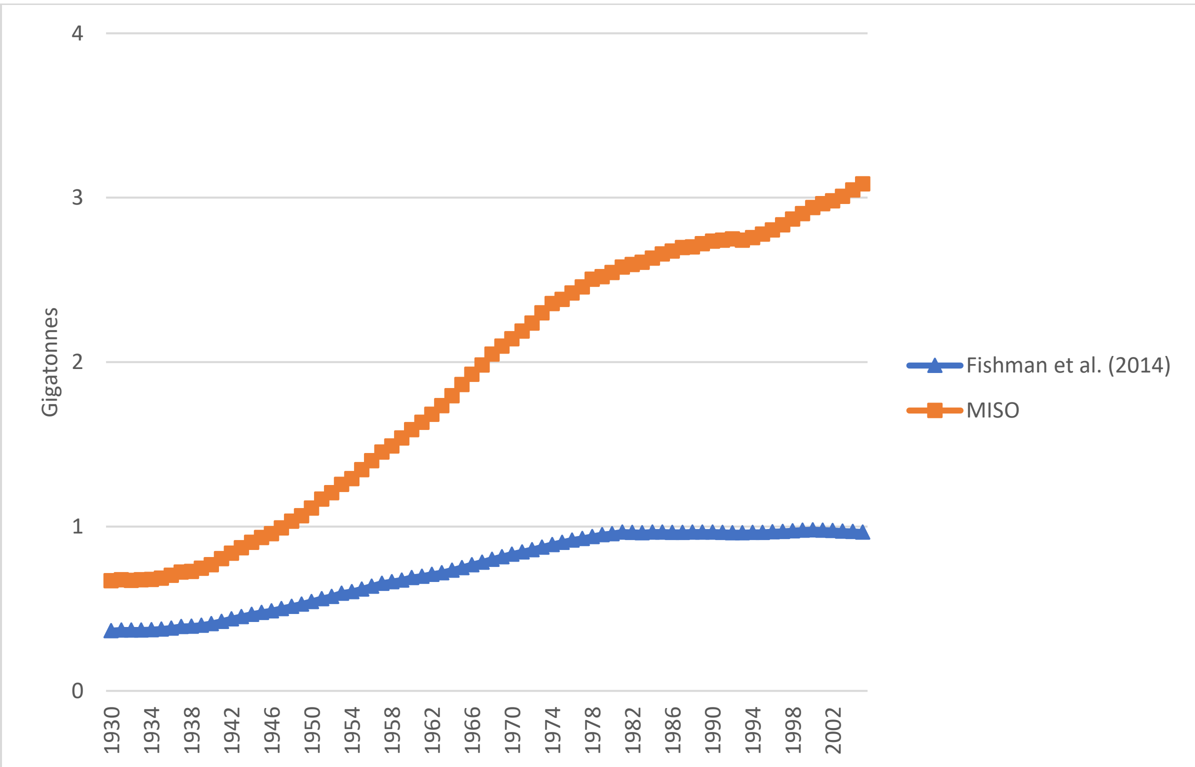
Notes: MISO stocks are from own calculations and were divided by population data from the Maddison Project (2018) to obtain per capita stocks.

Figure SI.37: Comparison of per capita stock estimates for iron/steel in the USA for 1900-2008



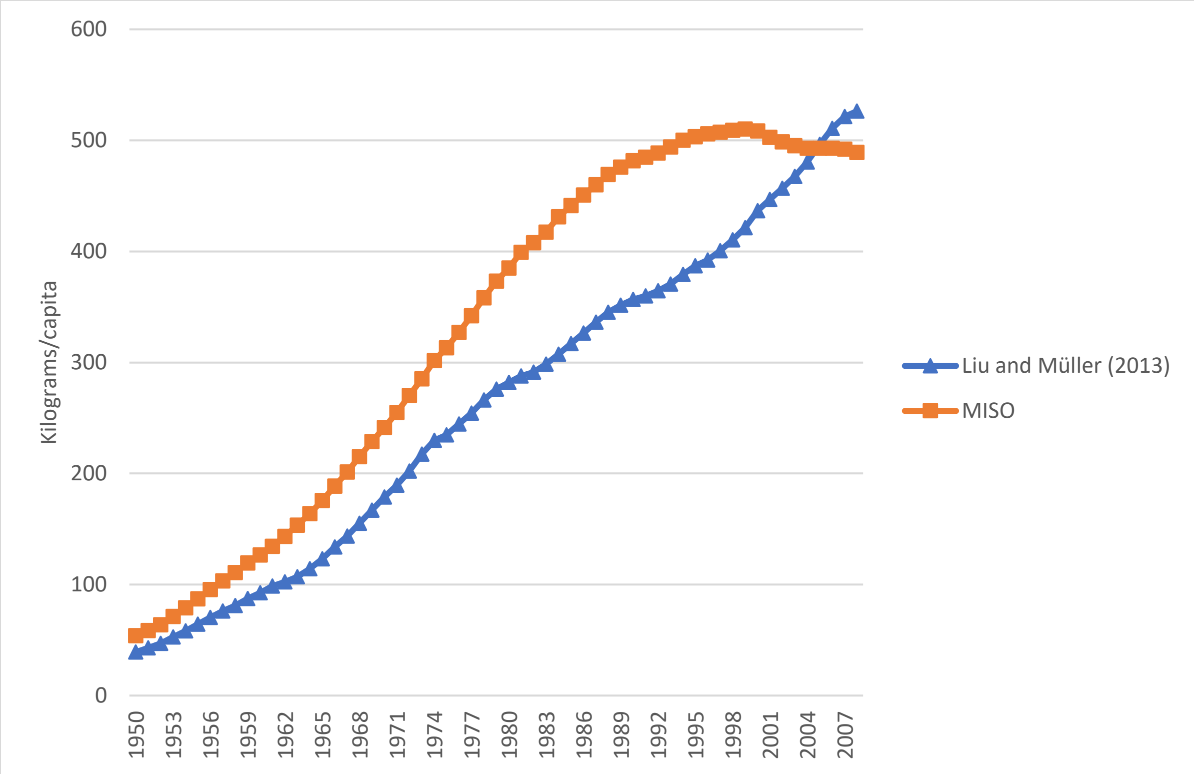
Notes: MISO-Stocks are from own calculations and divided by population data from the Maddison Project (2018) to obtain per capita stocks.

Figure SI.38: Comparison of stock estimates for iron/steel in the USA for 1930-2005



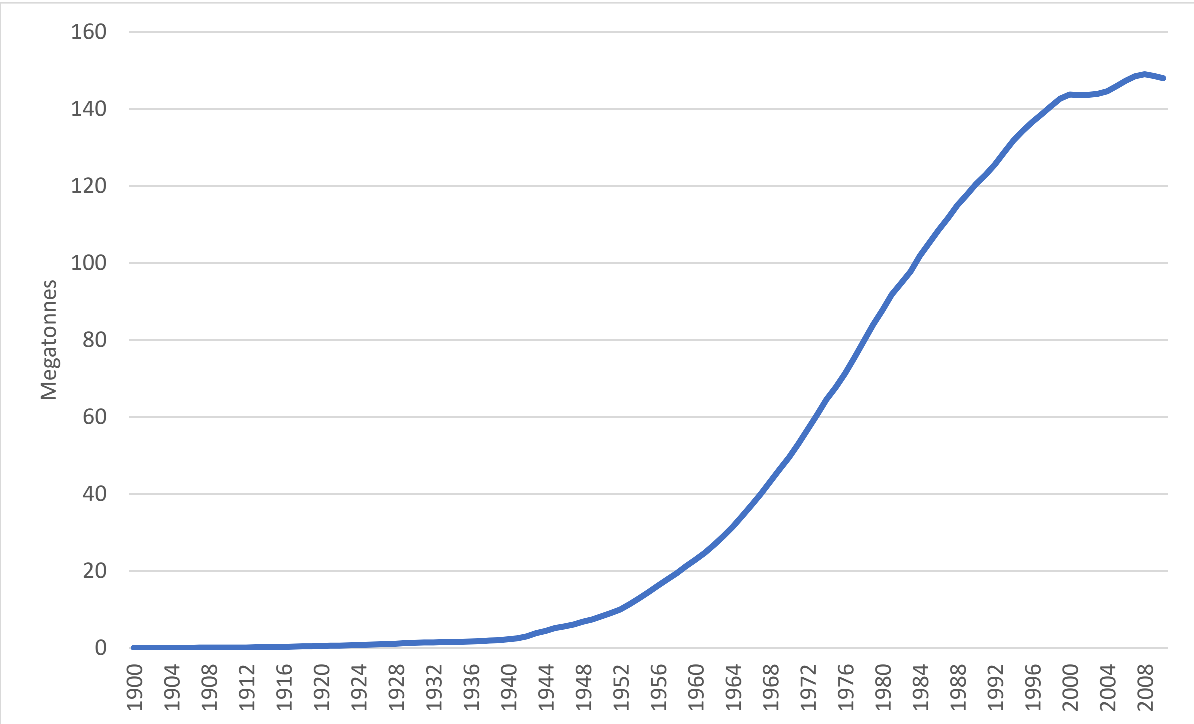
Notes: MISO estimates include stocks for iron/steel. Stock estimates from Fishman et al. (2014) include iron.

Figure SI.39: Comparison of per capita stock estimates for aluminum in the USA for 1950-2008



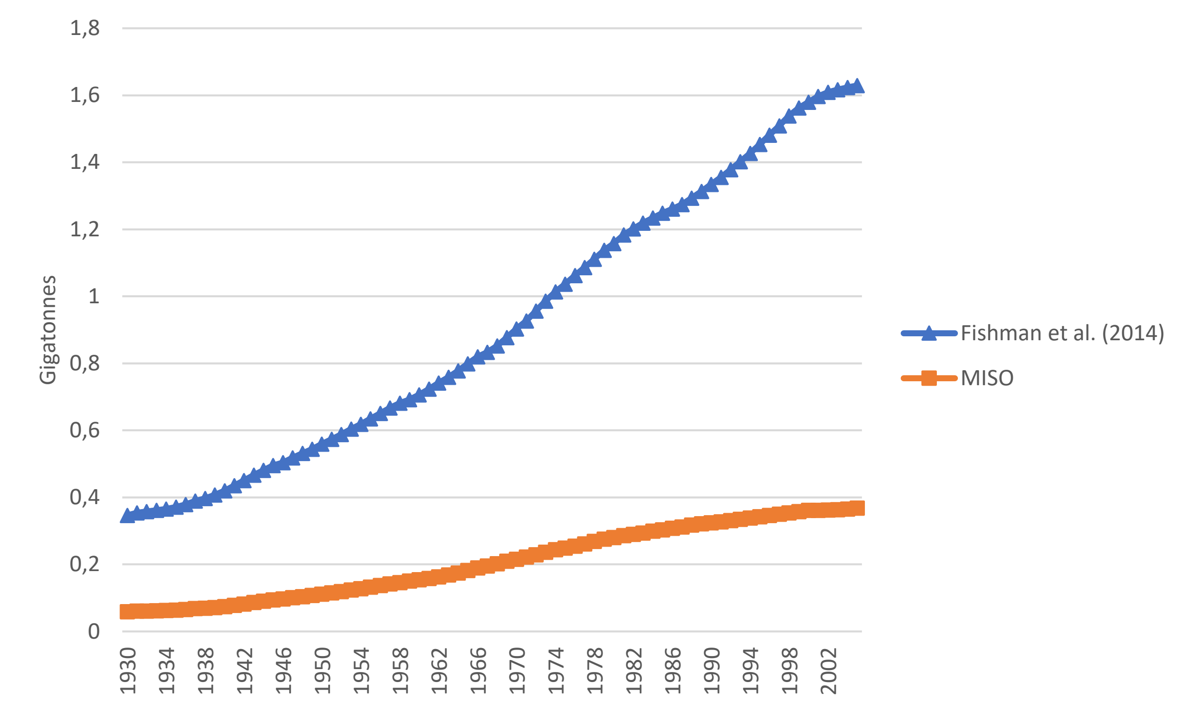
Note: MISO-Stocks are from own calculations and divided by population data from the Maddison Project (2018) to obtain per capita stocks.

Figure SI.40: MISO stock estimates for aluminum in the USA, 1900-2010



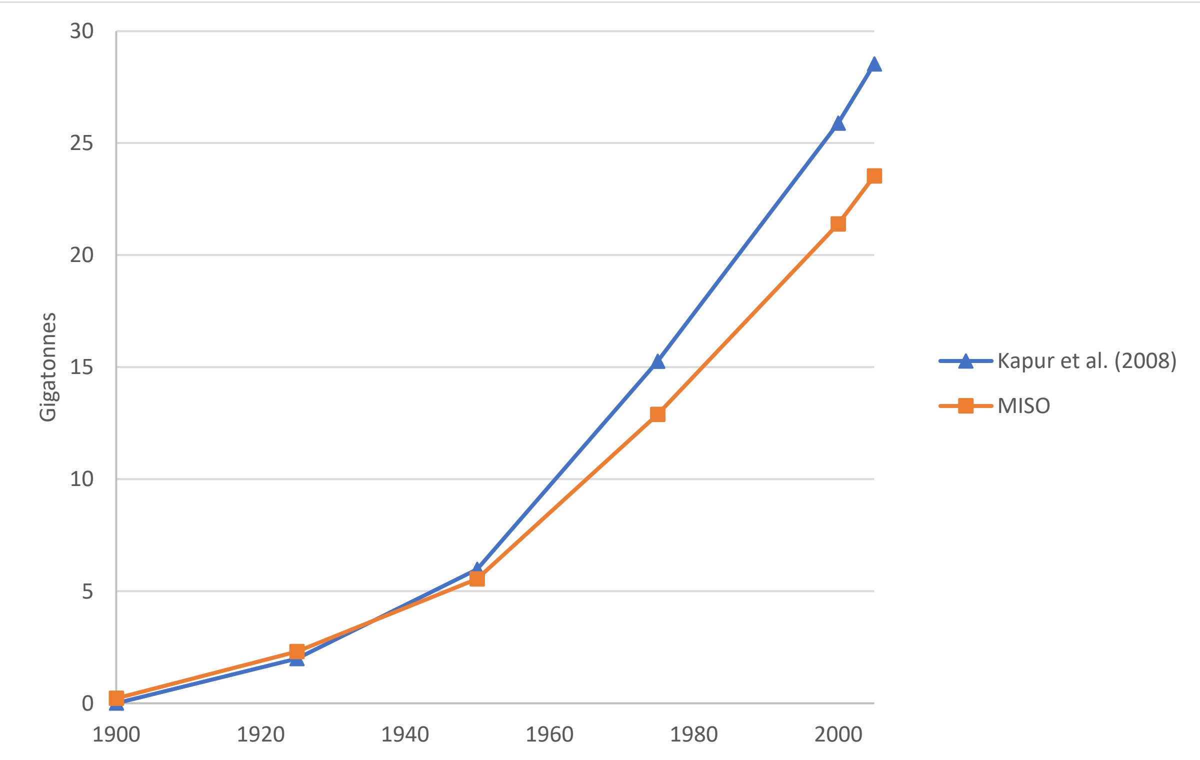
Notes: Estimates are from own calculations

Figure SI.41: Comparison of stock estimates for non-ferrous metals in the USA for 1930-2005



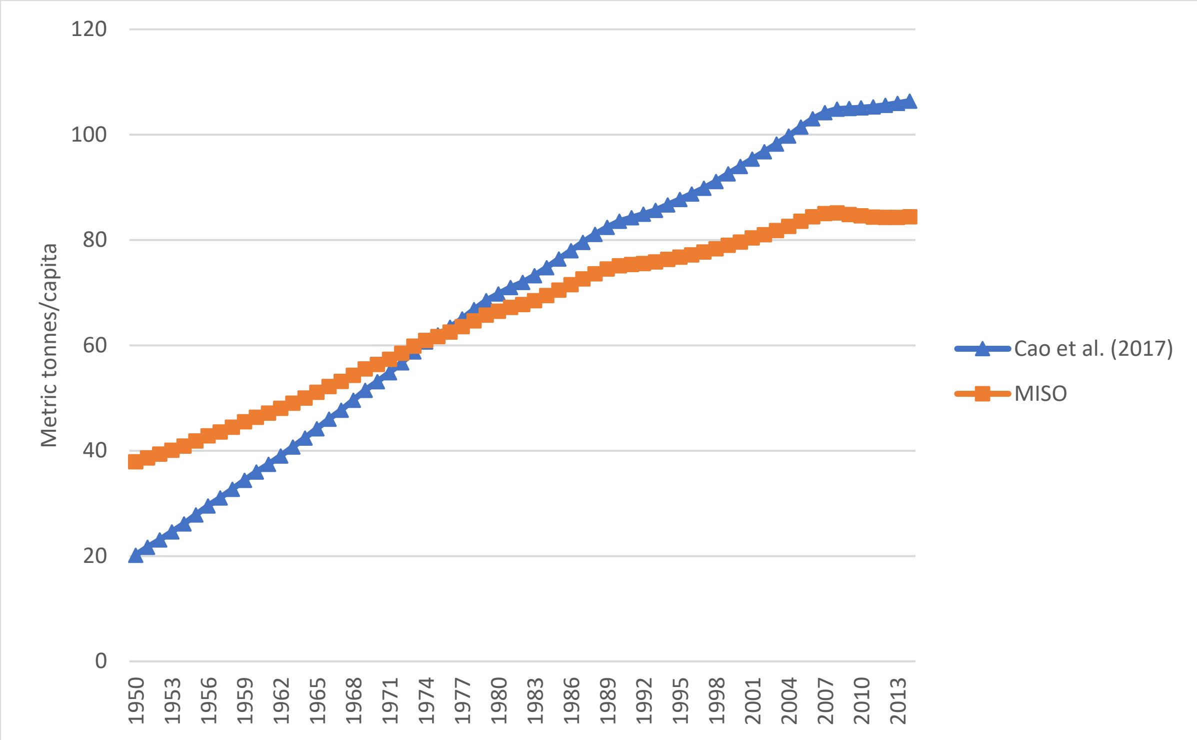
Notes: MISO estimates include stocks for aluminum, copper and other metals. Stock estimates from Fishman et al. (2014) include all metals other than iron.

Figure SI.42: Comparison of stock estimates for concrete in the USA for 1900-2005



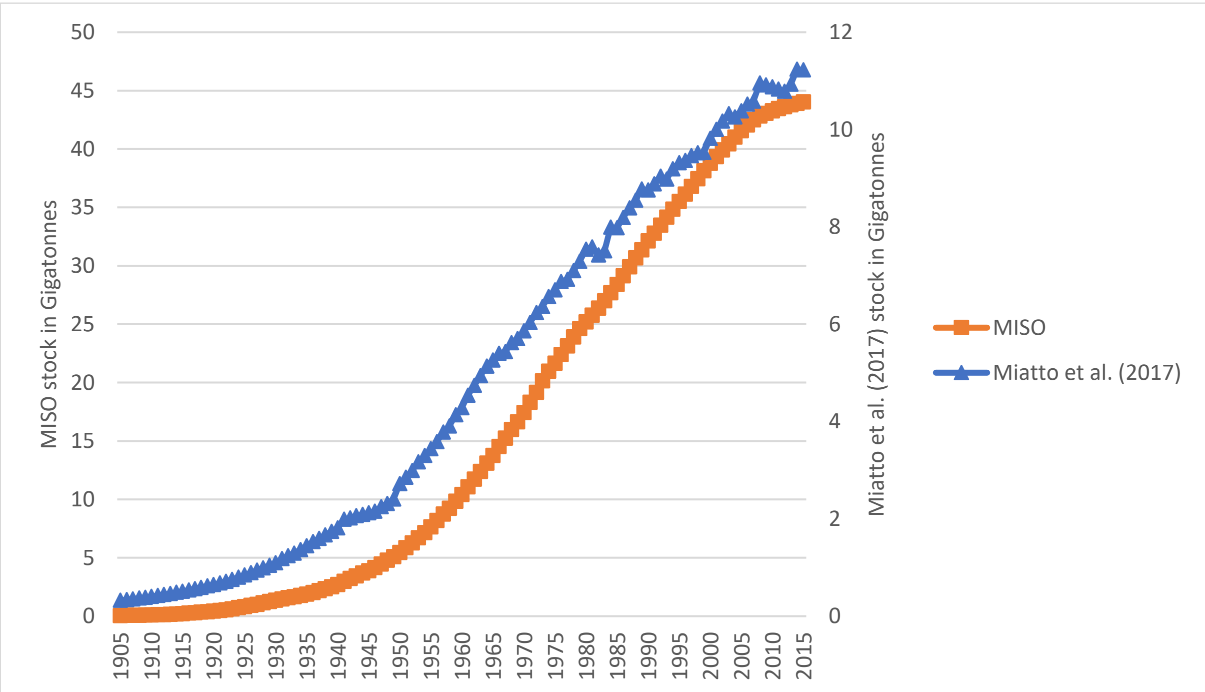
Notes: Values for Kapur et al. (2008) were calculated based on their results for cement in-use-stocks and coefficients from Cochran and Townsend (2010) which we also used to calculate concrete production values for the MISO-model (formula 2).

Figure SI.43: Comparison of per capita stock estimates for concrete in the USA for 1950-2014



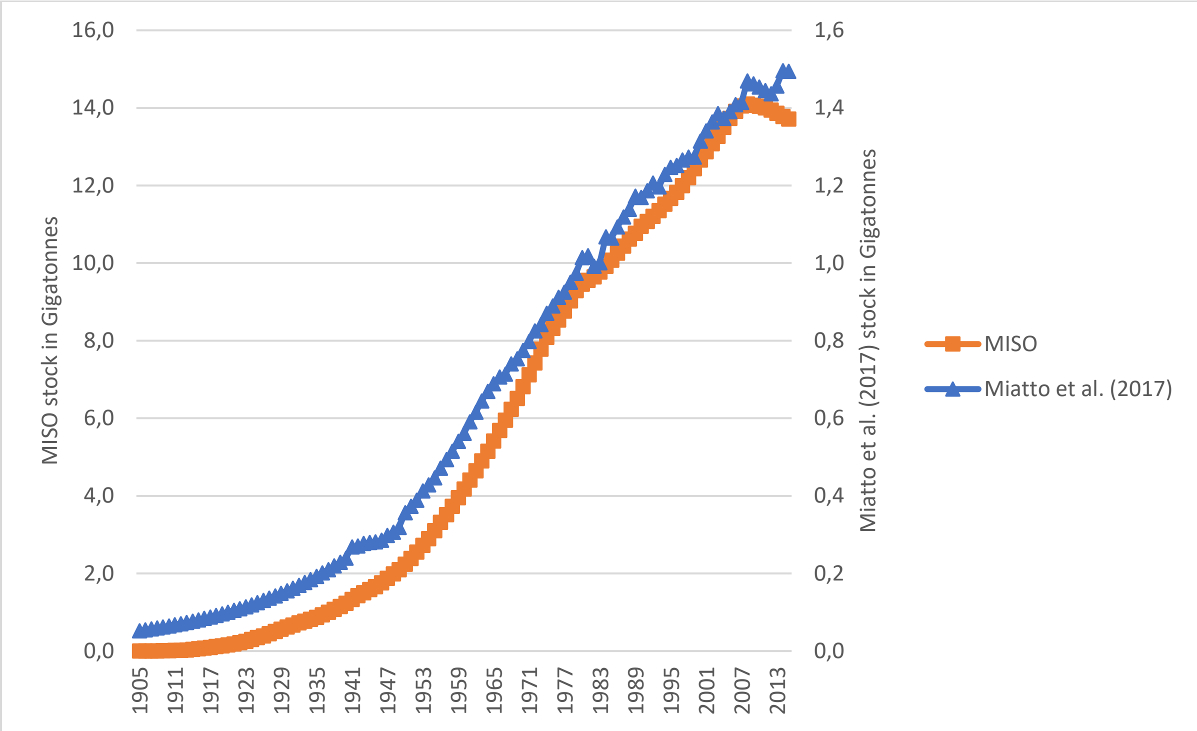
Notes: Values for Cao et al. (2017) were calculated based on their results for cement in-use-stocks and coefficients from Cochran and Townsend (2010) which we also used to calculate concrete production values for the MISO-model (formula 2). MISO-Stocks were divided by population data from the Maddison Project (2018) to obtain per capita stocks.

Figure SI.44: Comparison of stock estimates for aggregates in sub-base and base-course layers in the USA for 1905-2015



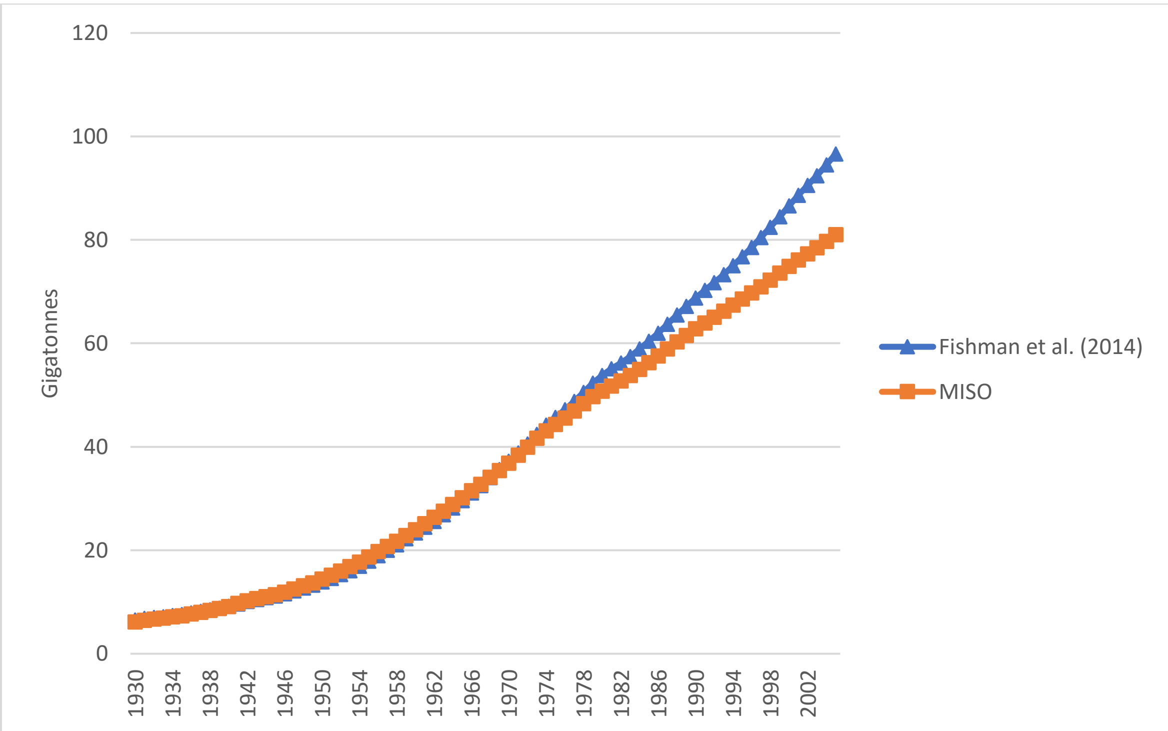
Note: Values for Miatto et al. (2017) include aggregates in sub-base and base-course layers of roads, MISO-model estimations also include a small fraction of aggregates in sub-base and base-course layers of other infrastructure such as buildings. MISO results are depicted on the left axis, the results of Miatto et al. (2017) are depicted on the right axis.

Figure SI.45: Comparison of stock estimates for asphalt in the USA for 1905-2015



Note: Values for Miatto et al. (2017) were calculated based on their results for bitumen stocks and the assumption that asphalt consists of 95 % sand and gravel and 5 % bitumen. MISO results are depicted on the left axis, the results of Miatto et al. (2017) are depicted on the right axis.

Figure SI.46: Comparison of stock estimates for non-metallic minerals in the USA for 1905-2015



Notes: MISO estimates include stocks for concrete, asphalt, aggregates in sub-base and base-course layers, bricks, stones and tiles, container glass and flat glass. Stock estimates from Fishman et al. (2014) include non-metallic minerals.

Table SI.5: Comparison of stock estimates for iron/steel in the USA for various years

Author	Year	Estimate (Gt)	MISO-Estimate (Gt)	% Difference
Brown (1954)*	1950	2,3	1,1	-51 %
Sullivan (2003)*	2000	4,0	2,9	-27 %
Rauch (2009)	2000	3,2	2,9	-8 %
Sullivan (2005)	2002	4,1	3,0	-28 %
Müller et al. (2006)	2004	3,1	3,0	-1 %
Müller et al. (2011)	2005	3,2	3,1	-4 %

Notes: *Values were taken from Gerst and Graedel (2008); whenever per-capita stocks were depicted in the literature, they were multiplied with population data from the Maddison Project (2018) to obtain total stocks.

Table SI.6: Comparison of stock estimates for aluminum in the USA for various years

Author	Year	Estimate (Gt)	MISO-Estimate (Gt)	% Difference
Recalde et al. (2008)*	2000	0,11	0,14	34 %
Liu and Müller (2013)	2000	0,12	0,14	16 %
Sullivan (2003)**	2000	0,14	0,14	5 %
Rauch (2009)	2000	0,11	0,14	32 %
Sullivan (2005)*	2002	0,14	0,14	1 %
Sullivan (2005)	2002	0,14	0,14	1 %
Hatayama et al. (2009)*	2003	0,12	0,14	20 %
Liu and Müller (2013)	2005	0,15	0,15	-1 %
Liu et al. (2011)*	2006	0,15	0,15	1 %
McMillan et al. (2010)***	2007	0,09	0,15	57 %
Chen and Graedel (2012)	2009	0,15	0,15	-2 %
Liu and Müller (2013)	2010	0,16	0,15	-10 %

Notes: *Values were taken from Liu and Müller (2013) **Values were taken from Gerst and Graedel (2011) ***Average value of 0.911-0.976 was taken for McMillan et al. (2010); whenever per-capita stocks were depicted in the literature, they were multiplied with population data from the Maddison Project (2018) to obtain total stocks.

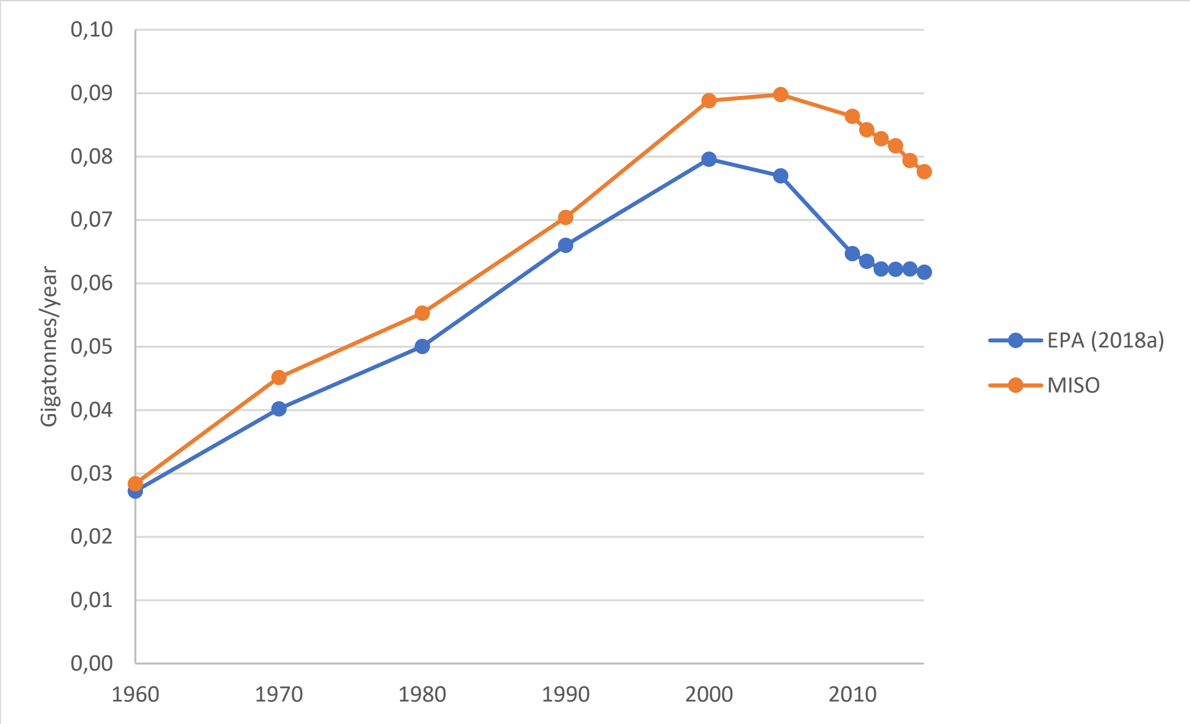
Table SI.7: Comparison of stock estimates for copper in the USA for various years

Author	Year	Estimate (Gt)	MISO-Estimate (Gt)	% Difference
Ingalls (1935)*	1932	0,01	0,02	116 %
Merrill (1949)*	1948	0,02	0,03	66 %
Merrill (1959)*	1957	0,03	0,04	49 %
McMahon (1965)*	1960	0,04	0,04	11 %
Merrill (1964)*	1961	0,05	0,04	-12 %
Sousa (1981)*	1979	0,07	0,06	-5 %
Jolly (1999)*	1990	0,07	0,07	6 %
Zeltner et al. (1999)*	1990	0,07	0,07	-3 %
Ayres et al. (2003)*	1998	0,07	0,08	12 %
Gordon et al. (2006)*	1999	0,07	0,08	20 %
Sullivan (2003)*	2000	0,11	0,08	-27 %
Rauch (2009)	2000	0,07	0,08	21 %
Sullivan (2005)	2002	0,12	0,08	-29 %
Nathan Associates (2004)*	2003	0,05	0,08	64 %

Notes: *Values were taken from Gerst and Graedel (2008); whenever per-capita stocks were depicted in the literature, they were multiplied with population data from the Maddison Project (2018) to obtain total stocks.

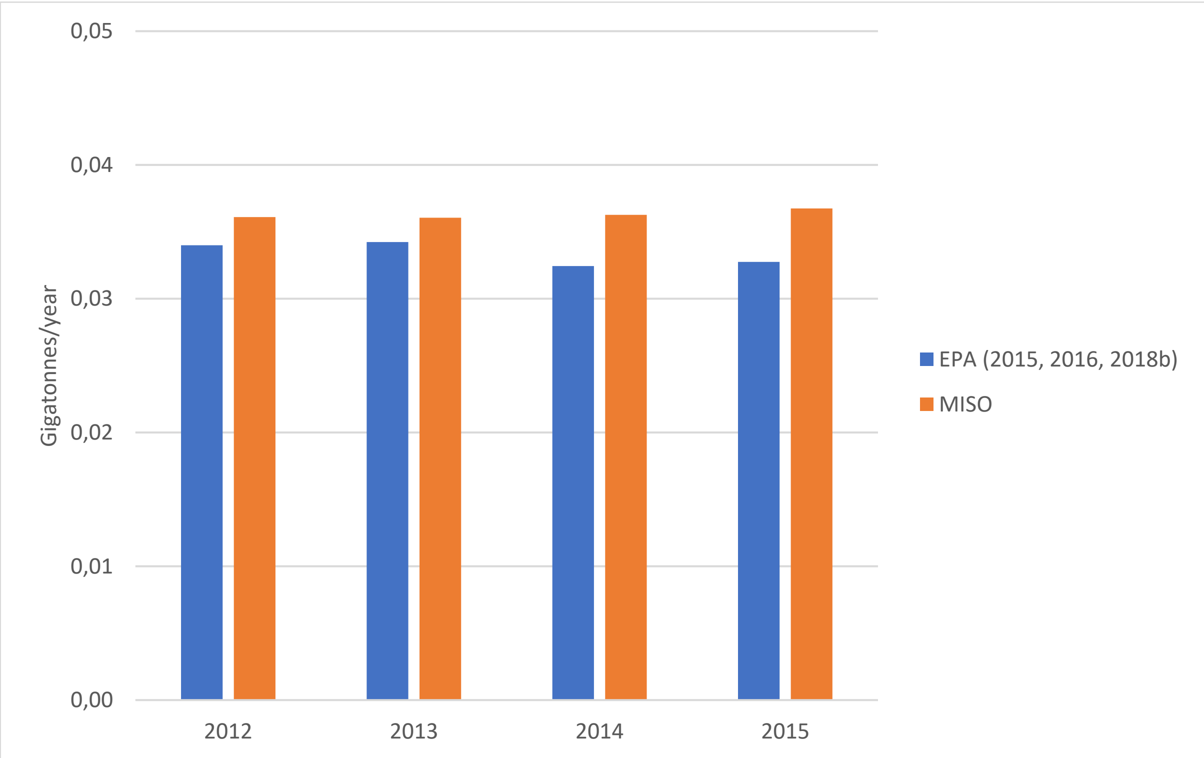
4.2.2 End-of-Life Outflows

Figure SI.47: Comparison of End-of-Life outflows estimates for paper and paperboard in the USA for 1905-2015



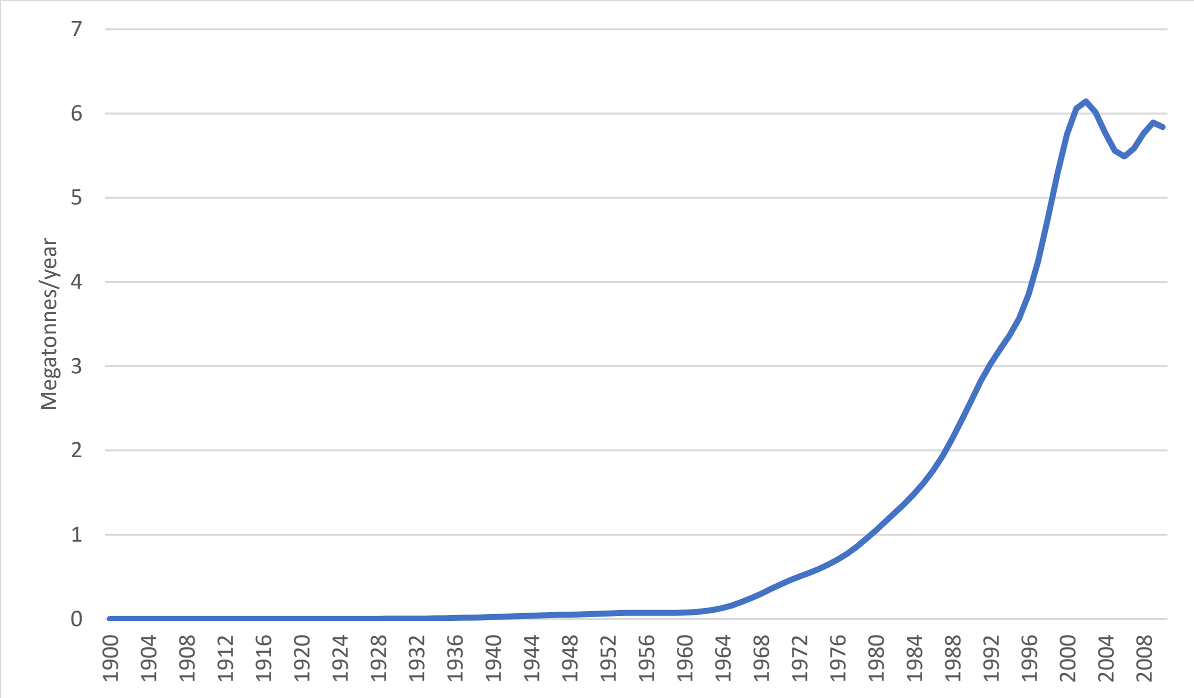
Note: We classified municipal solid waste of paper and paperboard from EPA as End-of-Life outflows.

Figure SI.48: Comparison of End-of-Life outflows estimates for solidwood in the USA for 2012-2015



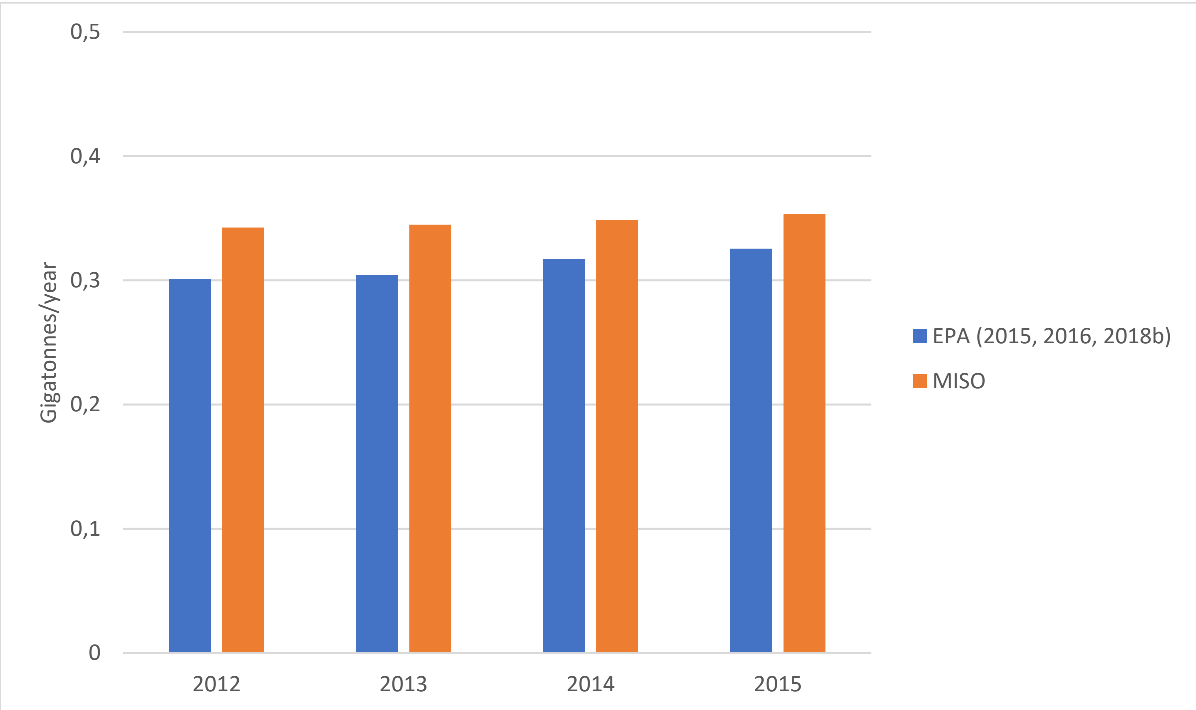
Notes: We classified demolition waste of wood products from EPA as End-of-Life outflows for solidwood.

Figure SI.49: MISO End-of-Life outflows estimates for aluminum in the USA, 1900-2009



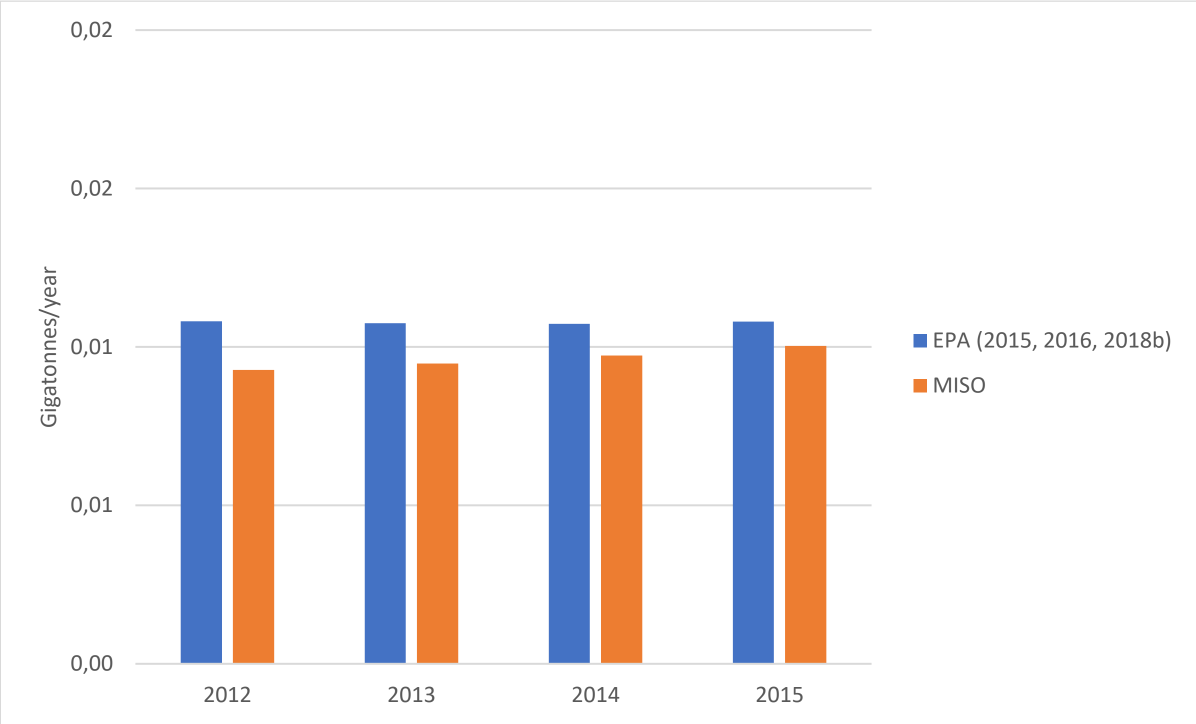
Notes: Estimates are from own calculations.

Figure SI.50: Comparison of End-of-Life outflows estimates for concrete in the USA for 2012-2015



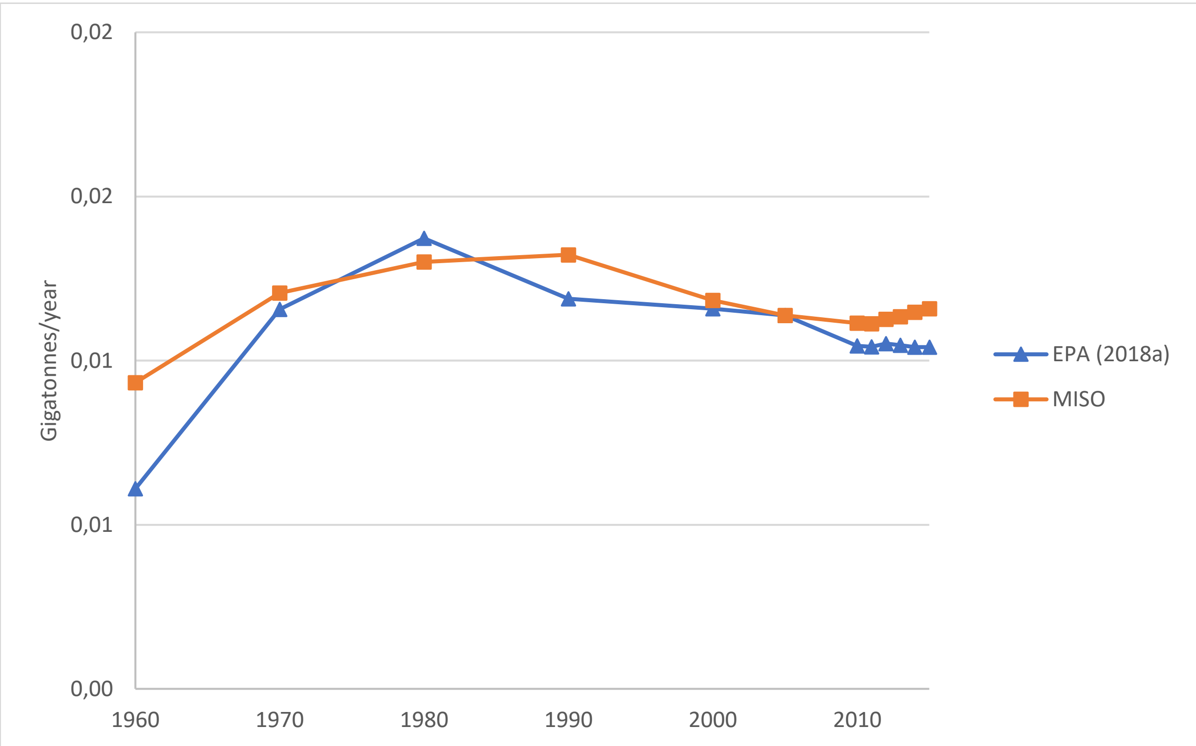
Notes: We classified demolition waste of portland cement concrete from EPA as End-of-Life outflows for concrete.

Figure SI.51: Comparison of End-of-Life outflows estimates for bricks, stones and tiles in the USA for 2012-2015



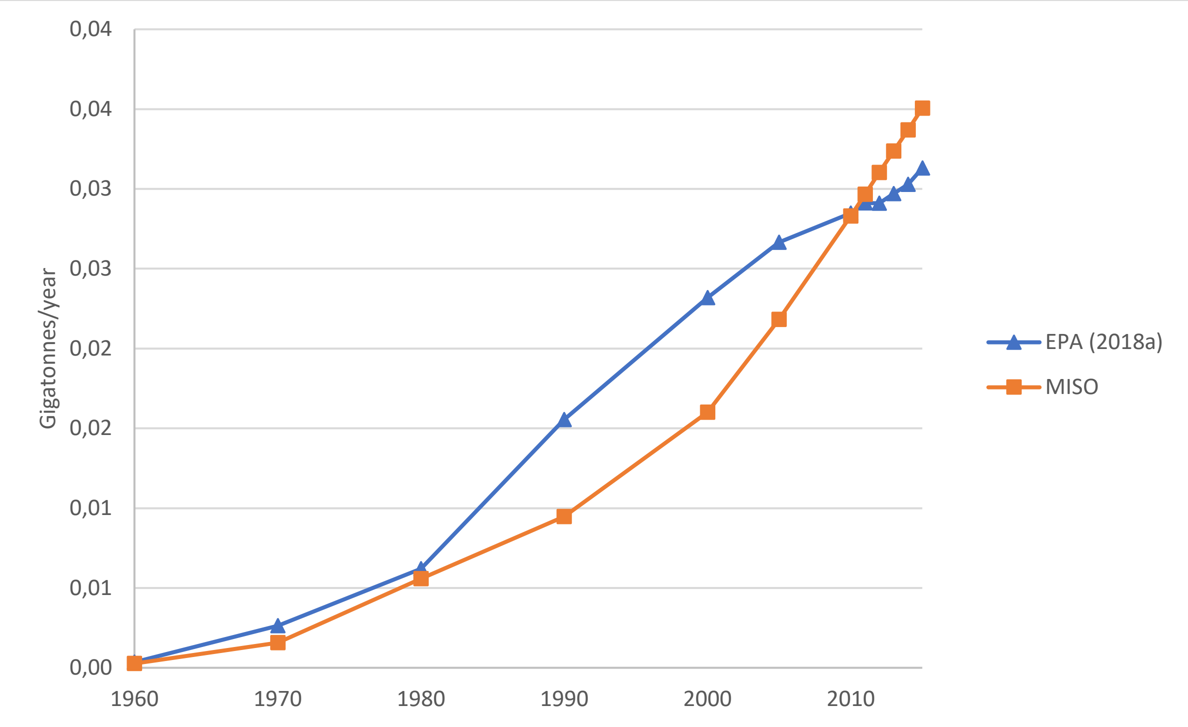
Notes: We classified demolition waste of brick and clay tile from EPA as End-of-Life outflows for bricks and tiles. End-of-Life outflows for stones are included in MISO estimations while EPA does not include End-of-Life outflows for stones.

Figure SI.52: Comparison of End-of-Life outflows estimates for glass in the USA for 1960-2015



Notes: We classified municipal solid waste of glass from EPA as End-of-Life outflows. MISO estimates include both End-of-Life outflows for container and flat glass.

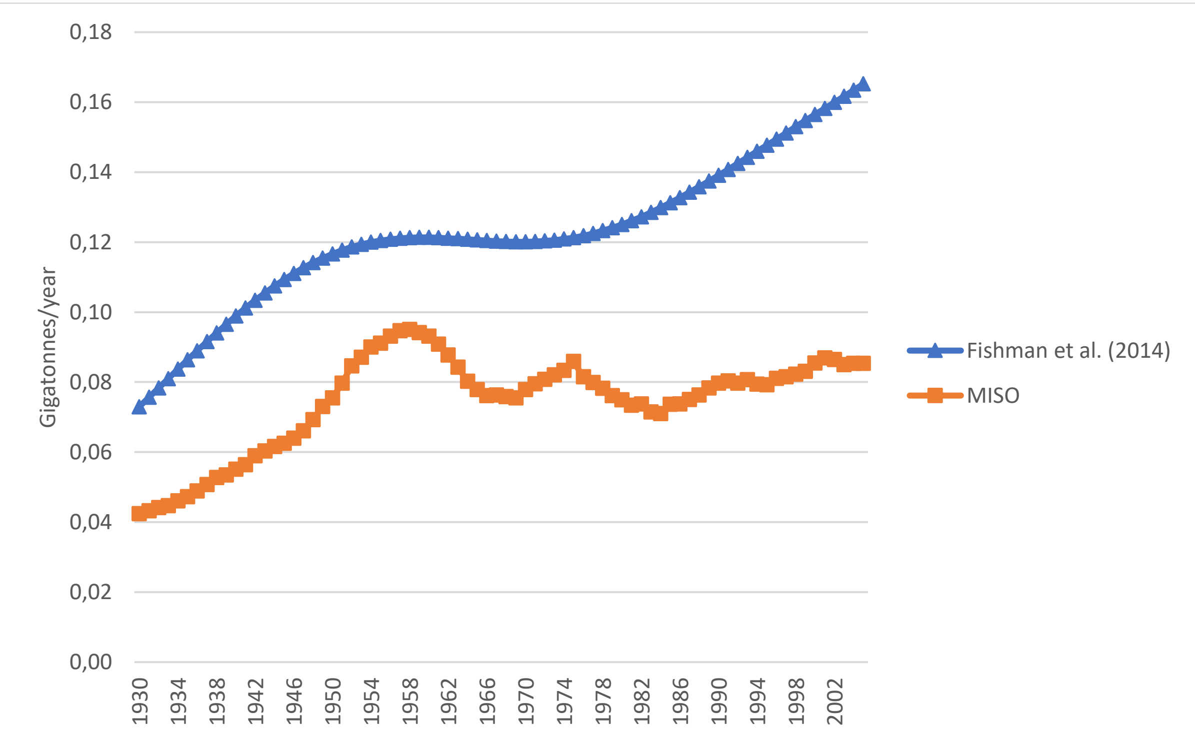
Figure SI.53: Comparison of End-of-Life outflows estimates for plastics in the USA for 1960-2015



Notes: We classified municipal solid waste of plastics from EPA as End-of-Life outflows.

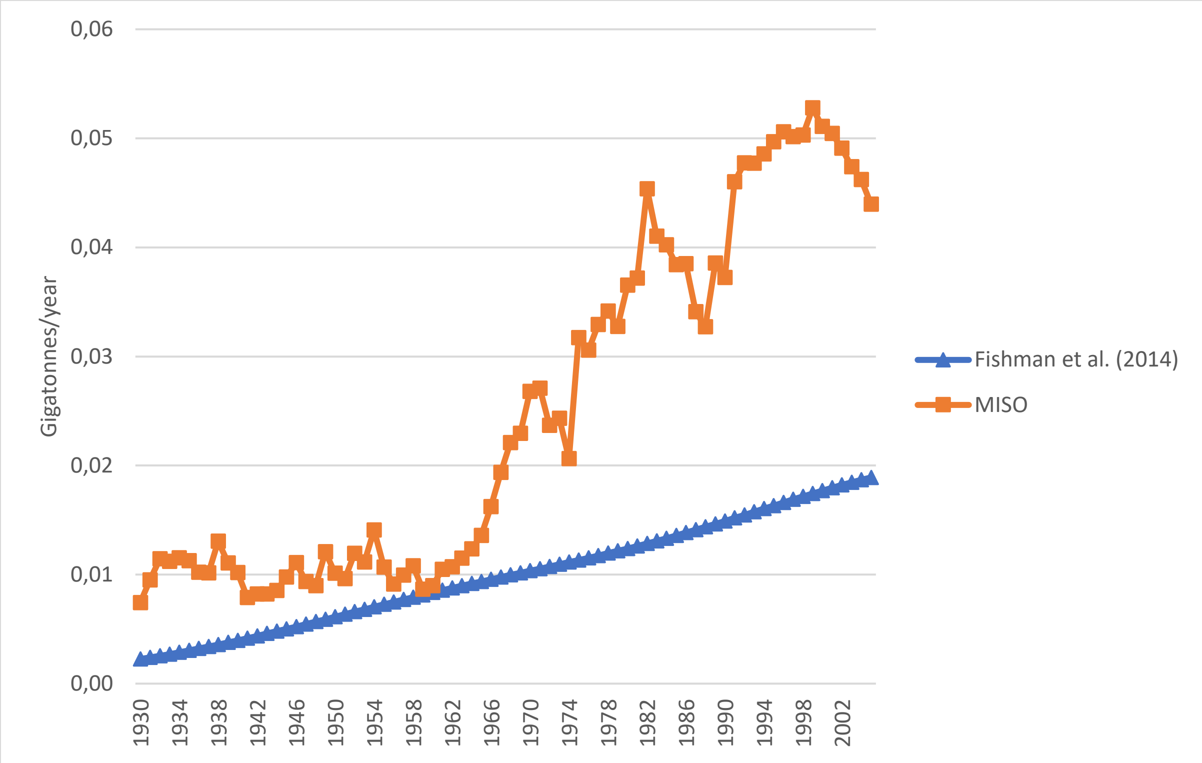
4.2.3 Final Waste

Figure SI.54: Comparison of final waste estimates for biomass in the USA for 1930-2005



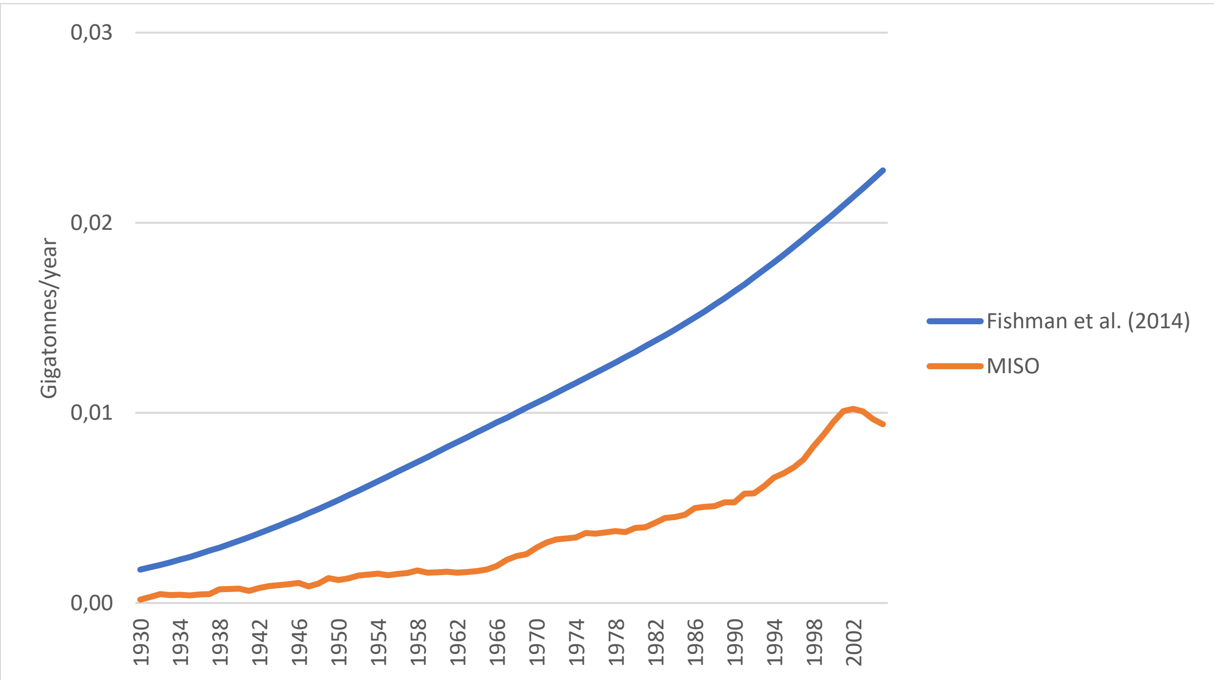
Notes: MISO estimates include final waste for solidwood and paper and paperboard. Final waste estimates from Fishman et al. (2014) include timber.

Figure SI.55: Comparison of final waste estimates for iron/steel in the USA for 1930-2005



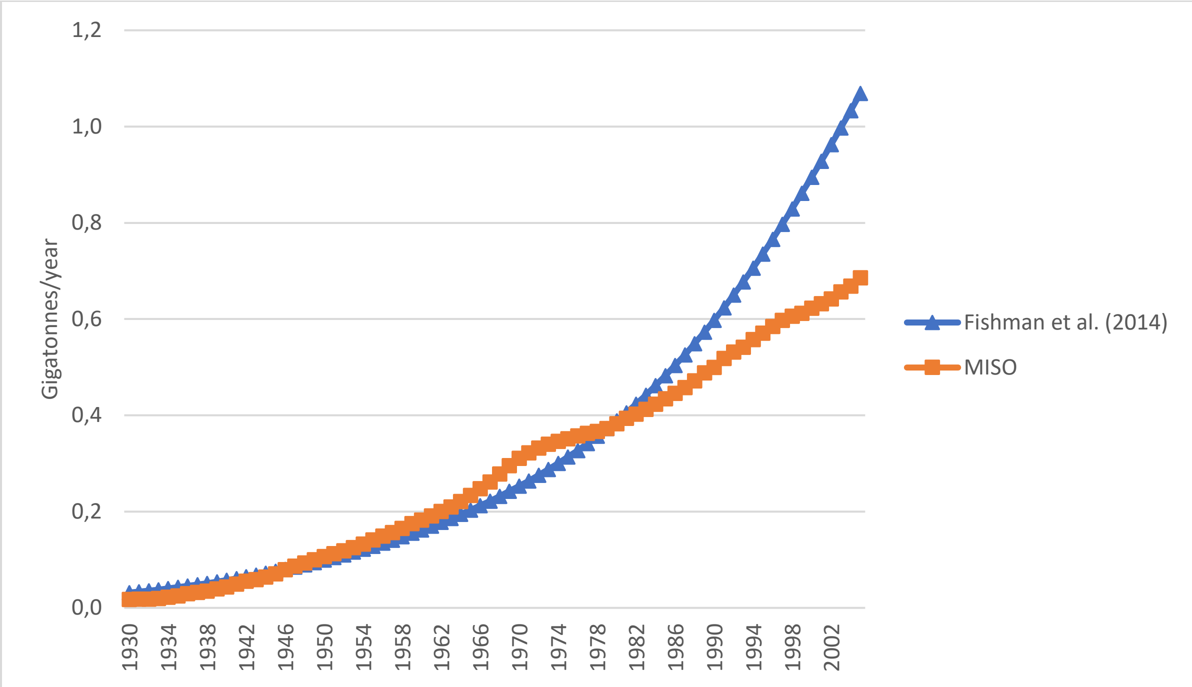
Notes: MISO estimates include final waste for iron/steel. Final waste estimates from Fishman et al. (2014) include iron.

Figure SI.56: Comparison of final waste estimates for non-ferrous metals in the USA for 1930-2005



Notes: MISO estimates include final waste for aluminum, copper and other metals. Final waste estimates from Fishman et al. (2014) include all metals other than iron.

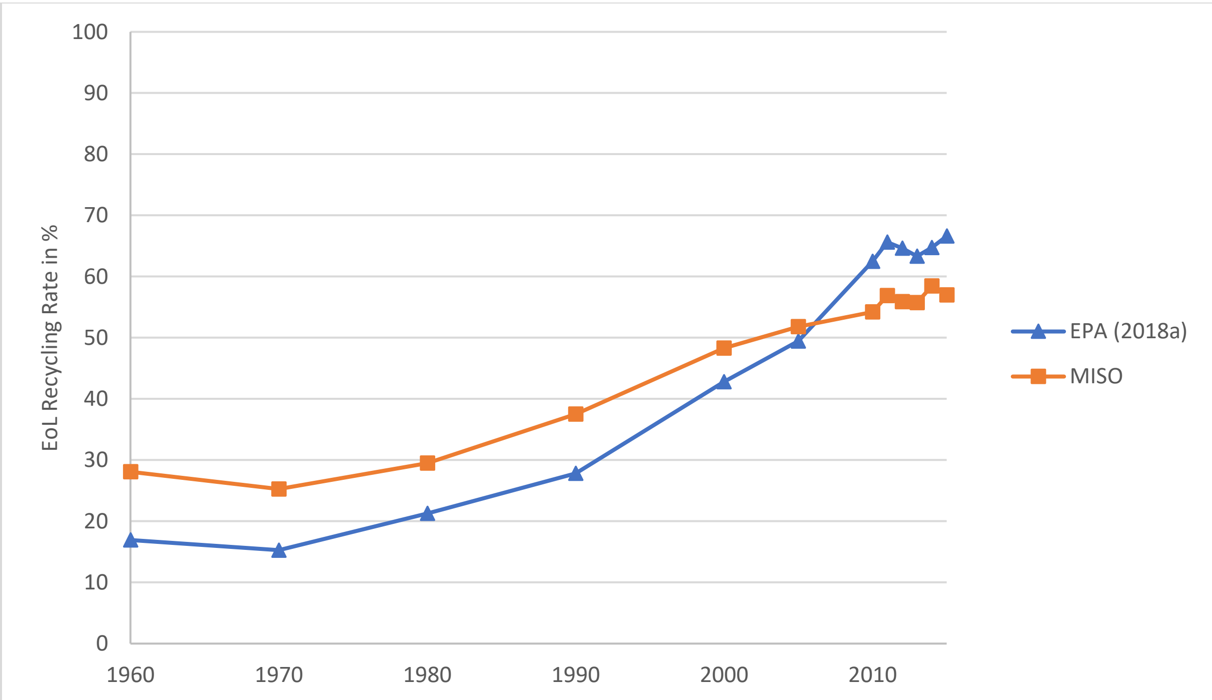
Figure SI.57: Comparison of final waste estimates for non-metallic minerals in the USA for 1930-2005



Notes: MISO estimates include final waste for concrete, asphalt, aggregates in sub-base and base-course layers, bricks, stones and tiles, container glass and flat glass. Final waste estimates from Fishman et al. (2014) include non-metallic minerals.

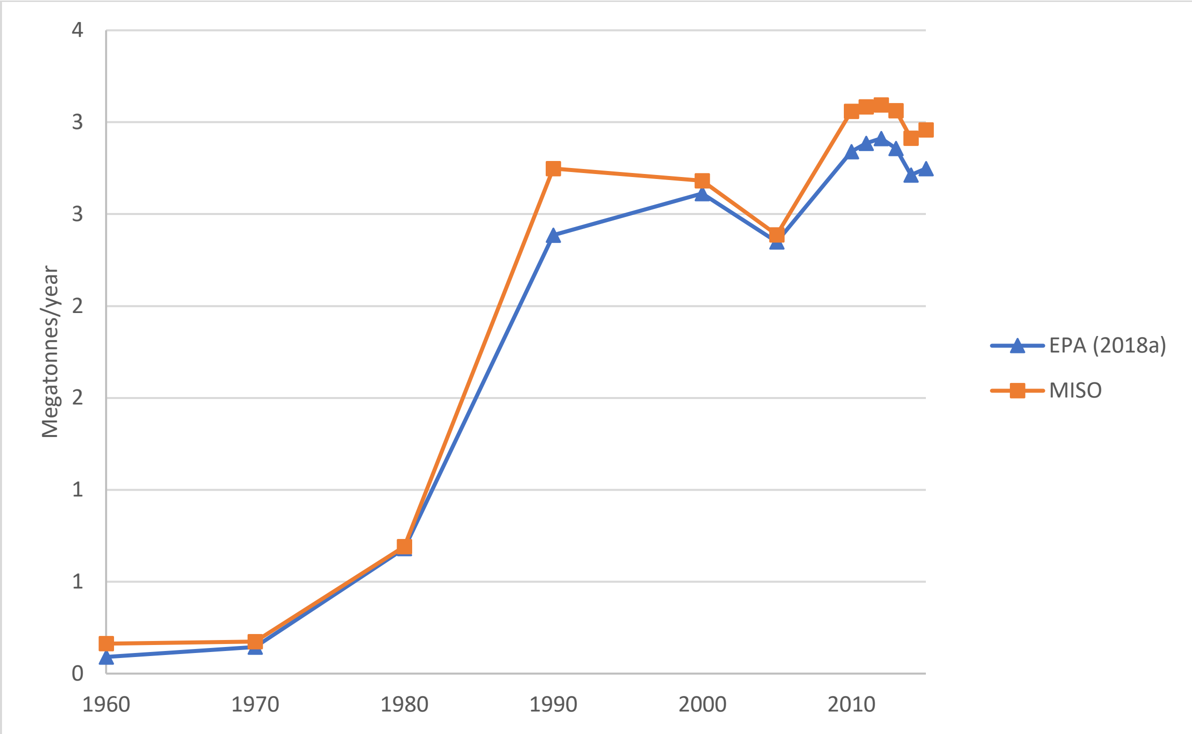
4.2.4 Recycling

Figure SI.58: Comparison of estimates for paper and paperboard recycling rates in the USA for 1960-2005



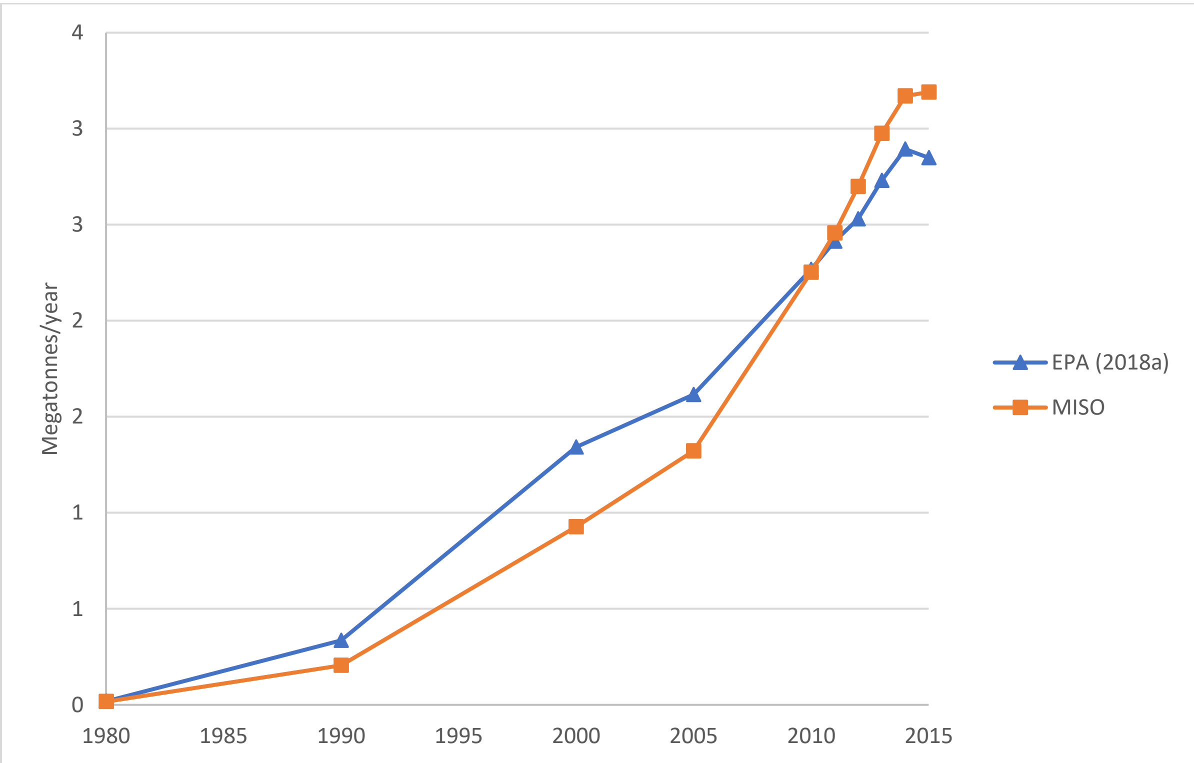
Notes: Recycling rates for EPA are calculated as total paper and paperboard recycled flows divided by municipal solid waste. MISO estimates are calculated as total paper and paperboard recycling flows divided by total End-of-Life outflows

Figure SI.59: Comparison of estimates for glass recycling flows in the USA for 1960-2005



Notes: Values from EPA are recycled municipal solid waste of glass. MISO estimates include container and flat glass.

Figure SI.60: Comparison of estimates for plastics recycling flows in the USA for 1980-2015



Notes: Values from EPA are recycled municipal solid waste of plastics.

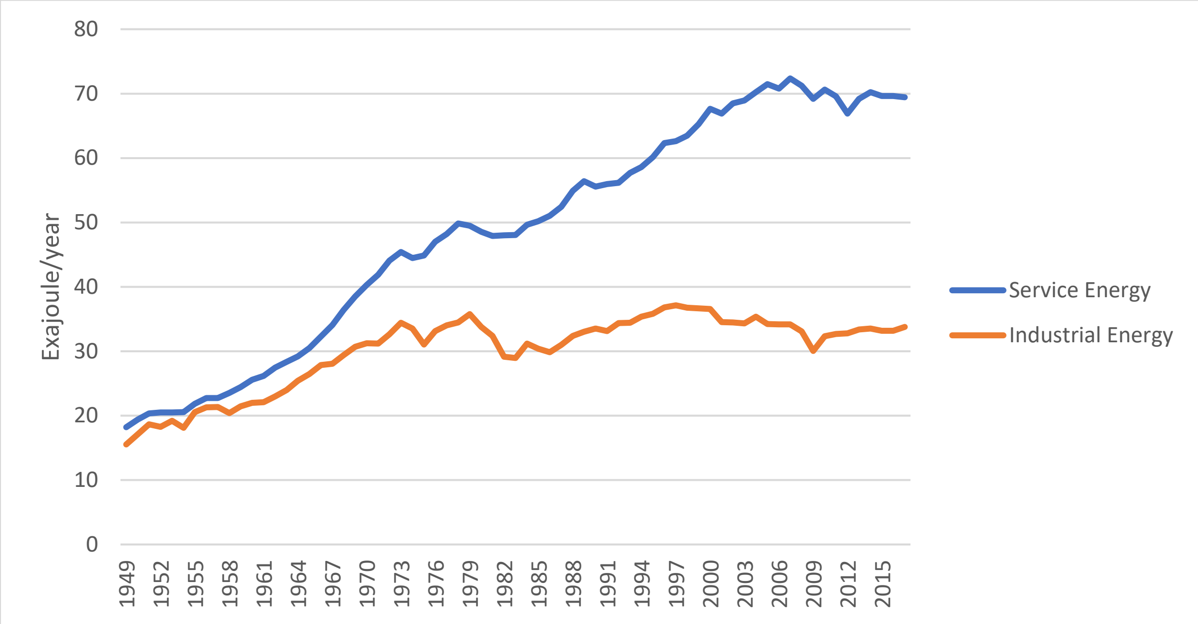
Table SI.8: Comparison of End-of-Life recycling rates for iron/steel, aluminum and copper in the USA for various years

Material	Author	Year	Estimate	MISO-Estimate
Iron/Steel	Fenton (2004)	1998	46.6 %	41.0 %
Iron/Steel	Wang et al. (2007)	2000	61.3 %	40.2 %
Aluminum	Plunkert (2006)	2000	34.3 %	23.8 %
Aluminum	Chen (2013)	1992/2009	49/24 %	53.3/21.4 %
Copper	Graedel et al. (2004)	1994	33.8 %	26.1 %
Copper	Goonan (2009)	2004	11.5 %	9.3 %

Notes: Values for Fenton (2004: 3), Plunkert (2006: W6) and Goonan (2009: X3) were calculated as old scrap generated divided by old scrap consumed. The values of Graedel et al. (2004: Supporting Information p. 21) was calculated as old scrap/(old scrap + landfilled waste and dissipated plus + trade of old scrap). For Wang et al. (2007: 5123) we assume that industrial scrap is completely recycled and calculate the EoL recycling rate as (purchased scrap - industrial scrap)/(EoL discards). Values for Chen (2013: 933-934) depict their reported domestic EoL recycling rate (i.e. excluding exported scrap).

4.3 Additional Figures

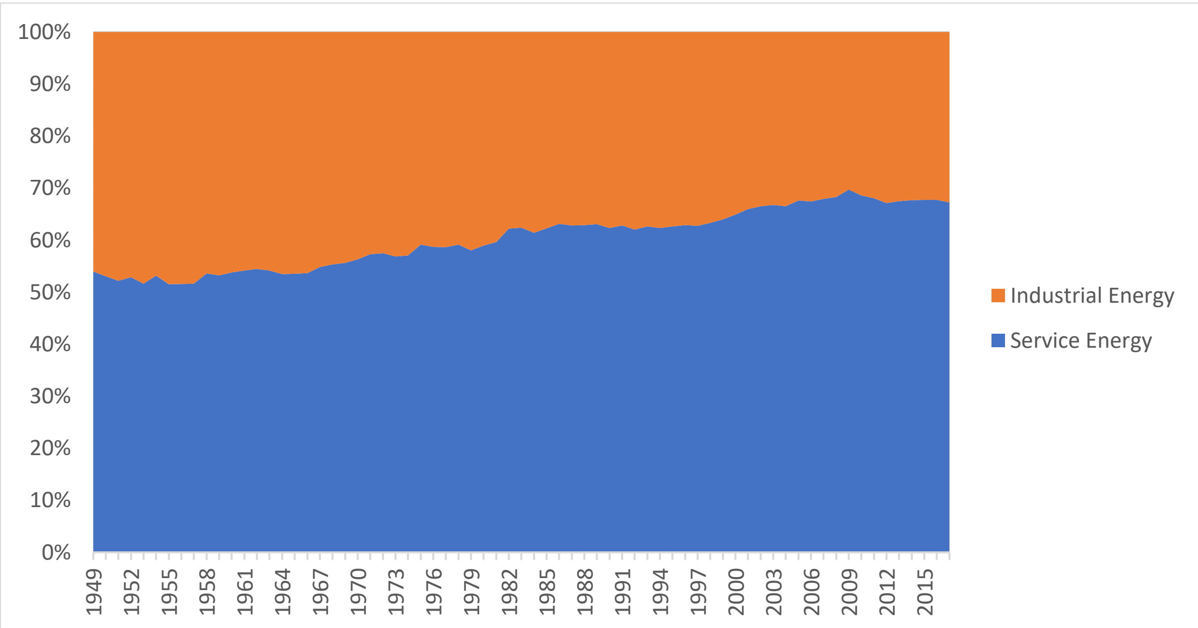
Figure SI.61: Service and industrial primary energy consumption in the USA, 1949-2017



Notes: Service energy includes the sectors transport, commercial and residential. Industrial energy includes energy consumption from the industry sector. Primary energy consumption of the energy sector was allocated to the end-use sectors industry, transport, commercial and residential based on electricity end-use shares.

Sources: EIA (2019), own calculations

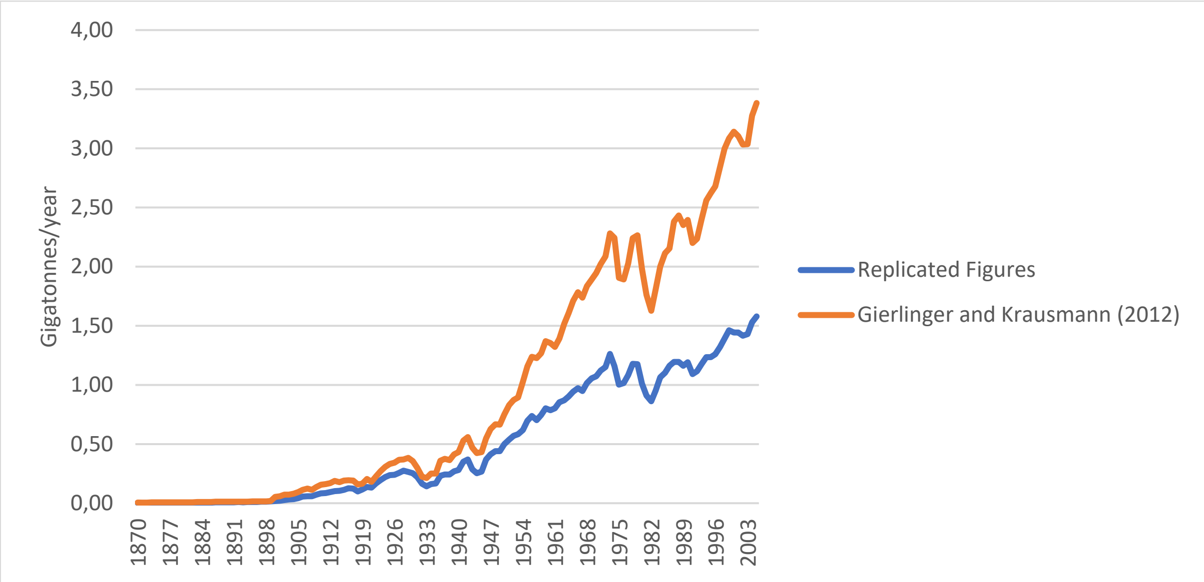
Figure SI.62: Share of industrial and service energy in total primary energy consumption in the USA, 1949-2017



Notes: Service energy includes the sectors transport, commercial and residential. Industrial energy includes energy consumption from the industry sector. Primary energy consumption of the energy sector was allocated to the end-use sectors industry, transport, commercial and residential based on electricity end-use shares.

Sources: EIA (2019), own calculations

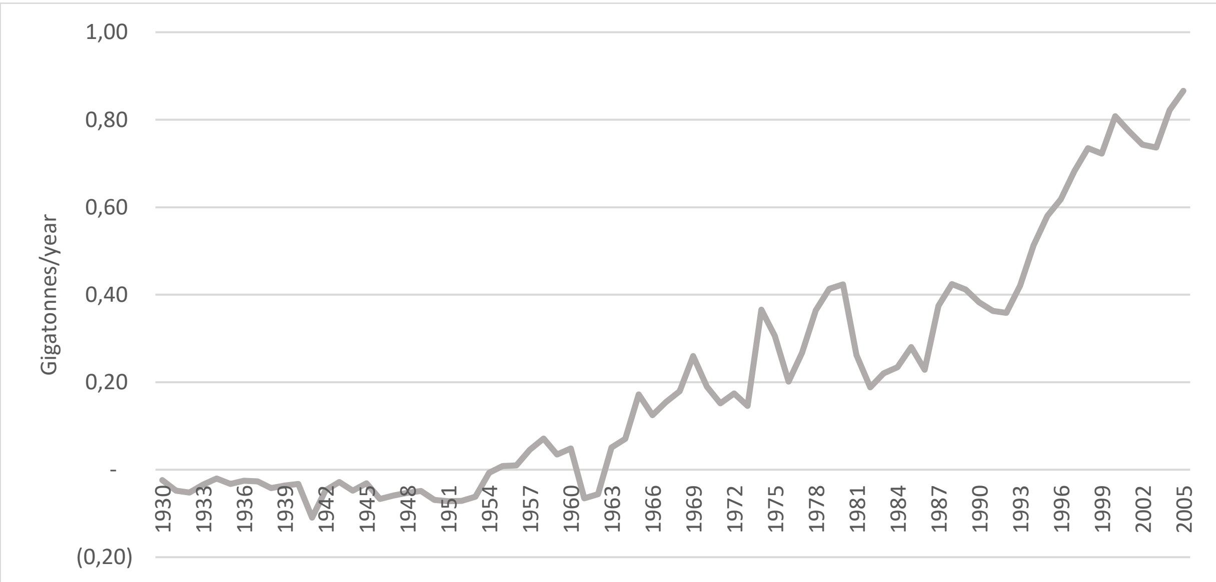
Figure SI.63: Comparison of sand and gravel consumption reported by Gierlinger and Krausmann (2012) and replicated figures in the USA for 1870-2005



Notes: To obtain sand and gravel for concrete, cement consumption was multiplied by the factor 6.1. To obtain sand and gravel used for asphalt, bitumen consumption was multiplied by the factor 20. The asphalt estimate was further increased by a factor of 1.5 to account for sand and gravel as filling material. Methods are depicted in Gierlinger and Krausmann (2012). The figures of Gierlinger and Krausmann (2012) are the reported amounts of sand and gravel consumption of their study. The figure shows that the reported values for sand and gravel are significantly larger than the replicated figures.

Sources: Data for cement and bitumen consumption is from the Bureau of the Census (1975), Kelly and Matos (2014), USGS (2019a) and the IEA (2019).

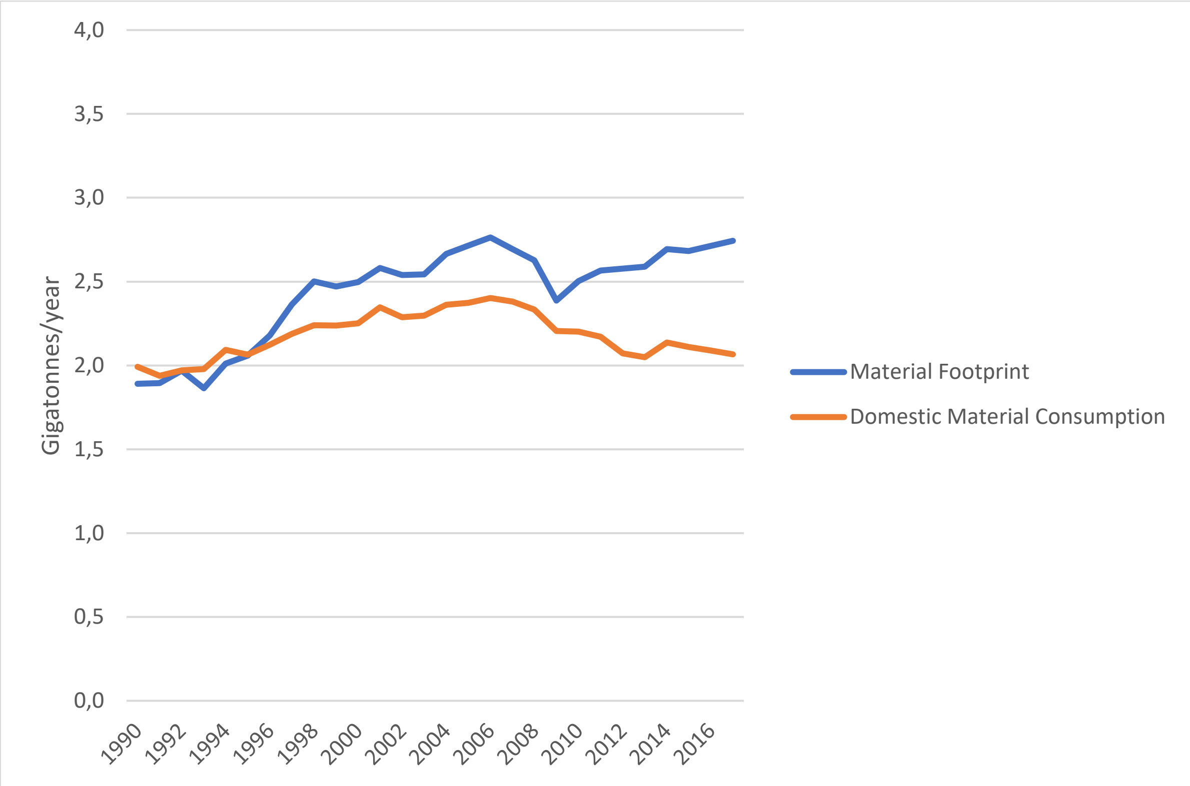
Figure SI.64: Difference of net additions to stock for non-metallic minerals between Fishman et al. (2014) and the MISO-model in the USA for 1930-2005



Notes: Differences are calculated as net additions to stock for non-metallic minerals from Fishman et al. (2014) minus net additions to stock for non-metallic minerals from the MISO-model. Non-metallic minerals in the MISO-model includes concrete, sand and gravel in asphalt (95 %), bricks, stones and tiles, aggregates, flat glass and container glass.

Sources: Fishman et al. (2014), own calculations

Figure SI.65: Domestic material consumption and material footprint for fossil energy carriers in the USA for 1990-2017



Notes: Domestic material consumption (DMC) of fossil energy carriers is equal to domestic material extraction of fossil energy carriers plus imports minus exports. The DMC (for fossil energy carriers) is thus "limited to the amount of materials directly used by an economy […] It does not include the upstream raw materials related to imports and exports originating from outside of the focal economy" (Wiedmann et al. 2015: 6271). The material footprint (MF) is a consumption-based indicator and includes all raw material extraction (in this case for fossil energy carriers) needed to fulfil the final demand of an economy, i.e. it includes all upstream materials. For 1995-2017, the MF for fossil energy carriers was higher than the DMC of the USA.

Sources: UNEP (2019), own calculations

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