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Opening the black box of economy-wide material and energy flows: from accounting to stock modeling and input-output analysis

DISSERTATION

submitted in fulfilment of the requirements for the degree of

Doktor der Naturwissenschaften

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Fakultät für Interdisziplinäre Forschung und Fortbildung

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Mag. Dominik Wiedenhofer Vienna, Austria; 21.6.2017

Personal foreword

My personal journey with environmental issues took off, when I started studying for my Bachelors' degree in Environmental- and Bio Resource Management at the University of Life Sciences (BOKU). However, it took me until a selective class during my undergraduate studies, to discover a way of thinking about the world, which really intrigued me. In this class I got a first glimpse into systems theory, based on the insights from the World3 model, the Limits to Growth and critical intervention points from Donella Meadows (Meadows, 1999; Meadows et al., 2004). Discovering these theories relieved me from over-focusing on individual green consumerism and technology. Although it took me a while to grasp the implications and challenges of the systems approach, I knew that I was hooked.

Luckily, I discovered the Institute of Social Ecology, which not only embraces systems approaches, but also goes beyond it. There I found another piece to my personal puzzle: a rigorous drive towards a systematic, quantitative and comprehensive perspective on society-nature interactions, grounded in biophysical realities and open to various theoretical complexities of both natural and social science.

During the long journey towards my PhD, which took me from February 2012 to June 2017, when this was submitted, I have met and worked with many inspiring and wonderful people, whom I cannot mention all. I still have to thank Julia and Manfred for sharing their time and ideas with me during my master's thesis. My research visit to the University of Sydney was personally and professionally very important for me. I want to thank my supervisor Prof. Fridolin Krausmann, who has gently nudged me in the right moments and otherwise gave me space to develop my ideas. I am immensely grateful to Marina, Willi, Fridolin and Nina for giving me the opportunities to work with them, experience how research is organized, and learn about a diverse set of topics. I am still amazed and very thankful, that Dabo, whom I have only met two times in person, put so much effort into our collaboration. Furthermore, I have to thank Hiroki and Fridolin for giving me the chance to work at Nagoya University for one summer, which was a very enriching and important experience for me. I also want to thank the entire Institute of Social Ecology for maintaining such a friendly, constructive and positively challenging working atmosphere. The mix of different people, topics and methods, all connected via an overarching paradigm, really opened my eyes and made me realize how many interesting aspects sustainability science has to offer.

Finally, I also want to thank my family and friends for supporting me through all these years and putting up with my regularly (over-) excited or sometimes disheartened stories. Finally yet most importantly, I am deeply grateful to have met Maria on this journey, who is such a wonderful, congenial and brilliant partner and played a big role for me to actually finish this journey and submit this dissertation.

“Systems folks would say one way to change a paradigm is to model a system, which takes you outside the system and forces you to see it whole. We say that because our own paradigms have been changed that way. […] I don't think there are cheap tickets to system change. You have to work at it, whether that means rigorously analyzing a system or rigorously casting off paradigms. In the end, it seems that leverage has less to do with pushing levers than it does with disciplined thinking combined with strategically, profoundly, madly letting go.” (Meadows, 1999, p. 13)

Summary

Investigating society-nature interactions and providing policy-relevant information on the progress towards sustainability is a challenging and complex task. Sustainability itself is a multi-dimensional concept, incorporating social, environmental, economic and intergenerational considerations. Evaluating progress requires a theoretically grounded, robust and appropriate monitoring framework.

The concept of socioeconomic metabolism postulates, that societies continuously require material and energy inputs to reproduce their biophysical structures (buildings, infrastructure, manufactured capital, livestock and the population). Thereby, these biophysical inputs are ultimately transformed into wastes and emissions. A minimum precondition of sustainability can then be formulated as the continuous ability to maintain and adapt the biophysical structures of society via material and energy flows (the socioeconomic metabolism), under conditions of global environmental change, such as a warming climate.

Economy-wide material and energy flow accounting (ew-MEFA) is an established methodology, in which the concept of the socioeconomic metabolism has been operationalized. It enables the systematic and comprehensive observation of material and energy flows crossing well-defined boundaries between nature and socioeconomic systems. So far, ew-MEFA has treated the socioeconomic system as a black box and focused on establishing principles to account for flows crossing said boundaries.

Two important research frontiers, the rapid globalization of trade and resource flows as well emerging strategies towards a more circular economy, have made it necessary to go beyond the state-of-the-art in ew-MEFA. Opening the black box enables adequately investigating supply chains and stock dynamics, to provide appropriate policy-relevant indicators on these issues.

For this dissertation, I included six peer-reviewed research articles and grouped them into two research objectives. All articles apply and evaluate different modeling strategies to open the black box. Building on the established strengths of ew-MEFA and expanding the monitoring framework using modeling, generates new insights into society-nature interactions and prospects for sustainability. Environmentally-extended input-output analysis can provide important consumption-based views into the role of trade for national and international resource use and emissions. Dynamic material flow analysis and stock modeling enables a closer investigation of the temporal dynamics of the biophysical structures of society and their respective inputs and outputs, thereby opening up new perspectives on the circular economy.

This dissertation contributes to a growing knowledge base on the limits and potentials for a more sustainable socioeconomic metabolism. Modeling can help identifying opportunities and challenges for critical interventions and leverage points. Ultimately, the systemic perspective of ew-MEFA on the socioeconomic metabolism and the biophysical structures of society comprehensively informs the monitoring of progress towards and trade-offs between the multi-dimensional concept of sustainability.

Zusammenfassung

Nachhaltigkeit ist in aller Munde, gleichzeitig ist es eine große wissenschaftliche Herausforderung fundierte Aussagen dazu zu treffen. Nachhaltigkeit beinhaltet sowohl soziale, ökologische, ökonomische als auch intergenerationale Dimensionen. Ob sich eine Gesellschaft auf einem nachaltigen Pfad befindet, bedarf theoretisch fundierter, robuster und nachvollziehbarer Monitoring – Systeme und Indikatoren.

Das Konzept des gesellschaftlichen Stoffwechsels, bzw des sozialen Metabolismus postuliert, dass Gesellschaften stetig Material und Energie benötigen, um ihre biophysischen Strukturen zu reproduzieren. Besagte Strukturen beinhalten die Bevölkerung, alle Nutztiere, sowie Gebäude, Infrastrukturen, Maschinen und sonstige Artefakte. Im Zuge dieser Reproduktion werden nun alle genutzten Materialien und Energieträger früher oder später in Emissionen und Abfälle umgewandelt. Eine Minimal-Voraussetzung für Nachhaltigkeit ist daher, dass Gesellschaften trotz regionaler bis globaler Umweltveränderungen, in der Lage bleiben, ihre biophysischen Strukturen zu erhalten, zu ernähren, zu betreiben und zu adaptieren.

Die Material und Energiefluss – Bilanzierung (ew-MEFA) ist eine etablierte Methode, mit der das Konzept des gesellschaftlichen Metabolismus operationalisiert wurde. Es ermöglicht die systematische Beobachtung und Quantifizierung von Material- und Energieflüssen, welche von Gesellschaften aus der Natur entnommen und genutzt werden. Bisher lag der Schwerpunkt darauf, die Prinzipien einer Erfassung von Material- und Energieflüssen welche die Grenzen zwischen Natur und Gesellschaften überschreiten, zu entwickeln und zu harmoniseren. Dabei wurde das sozioökonomische System bisher als ‚black box' vereinfacht.

Zwei wichtige Themen haben es nun notwendig gemacht, diese black box zu öffnen. Die Globalisierung von Produktion und Konsum ist verknüpft über stetig wachsenden Handel. Dies führt zur Verschiebung von nationaler Ressourcen-Nutzung entlang von Produktionsketten. Somit benötigt es innovative Indikatoren, um beobachtbar zu machen, was dies für nationale Nachhaltigkeits-Politik bedeutet. Im nationalen Kontext gewinnt außerdem die Idee einer Kreislaufwirtschaft immer mehr an Bedeutung. Hier werden große Nachhaltigkeitspotentiale vermutet. Daher benötigt es systematische Untersuchungen und robuste Indikatoren um fakten-basierte Empfehlungen aussprechen zu können.

In dieser Dissertation beschäftigte ich mich somit damit, anhand verschiedener Modellierungs-Ansätze besagte black box zu öffnen und somit die Möglichkeiten der Material- und Energiefluss–Bilanzierung zu erweitern. Die umwelt-erweiterte input-output Analyse, biophysische Handelsmodellierungen, sowie verschiedene Modellierungsstragien zur Quantifizierung der biophysischen Strukturen wurden eingesetzt und kritisch evaluiert. Basierend auf meinen Arbeiten, ergeben sich nun klare Möglichkeiten, verbesserte Indikatoren und systematische Einsichten über die Potentiale und Limitationen von wachsendem Handel, sowie einer Kreislaufwirtschaft, für eine nachhaltigere Entwicklung zu liefern.

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1. Investigating the biophysical dynamics of society-nature interactions as contribution to sustainability science

Recent advances in Earth Systems Science have shown that the current global socioeconomic system is driving humanity against Planetary Boundaries of the Earth System (Rockström et al., 2009; Steffen et al., 2015). Beyond these boundaries, irreversible and potentially catastrophic global environmental change becomes ever more likely, ultimately triggering tipping points in the Earth System which would lead to a completely different state of the environment than in which humanity evolved (O'Neill et al., 2017; Rockström et al., 2009; Steffen et al., 2015). Four of the nine boundaries that Steffen et al., 2015 identify, are at risk or are already trespassed: anthropogenic climate change, land-system changes, loss of biodiversity, ranging from genetic to functional ecosystem diversity and fourthly changes to global biochemical cycles of nitrogen and phosphorus.

These fundamental changes in the Earth System have prompted a debate on defining a new geological epoch, the Anthropocene, which is characterized by humanity having achieved the level of a natural force in shaping, changing and influencing planetary bio-geo-chemical cycles (Crutzen, 2002; Lenton et al., 2016; Steffen et al., 2011). These global environmental changes are ultimately driven by the exponential increase in the scale and extent of socioeconomic uses of energy, materials, land and subsequent emissions and wastes required to sustain humanity and all of its structures. From 1900 to 2010 global material extraction increased 11 fold, total primary energy supply 12 fold, carbon emissions from fossil fuels and cement production by factor 16, socioeconomic in-use stocks of buildings, infrastructure and other manufactured capital by factor 23, while global population 'only' increased by factor four (Fischer-Kowalski et al., 2014; Hoekstra and Wiedmann, 2014; Krausmann et al., 2017; Lenton et al., 2016).

Debates on environmental problems and planetary or ecological limits to human activity have a long and conflictual history because these issues are not restricted to 'the environment'. Much more importantly, the way societies biophysically interact with nature (e.g. socioeconomic use of resources and land, resulting emissions and wastes) becomes central (Fischer-Kowalski and Haberl, 2007; Haberl et al., 2016). This opens up the fundamental question, how these interactions are governed via norms and institutions, how the growth dynamics of the current socioeconomic system drive these interactions, what the societal outcomes are and how adaptation to socio-ecological change happens (Fischer-Kowalski and Haberl, 2007; Haberl et al., 2016; Singh et al., 2013; Vatn, 2016).

Research on these topics is conducted under the umbrella of Sustainability Science, which has been described as a unified and increasingly recognized scientific field since around the year 2000 (Bettencourt and Kaur, 2011; Kates, 2011). This growing field is characterized by high geographical and disciplinary diversity, as well as an integrative commitment to interdisciplinary collaboration across social, natural and technical sciences (Bettencourt and Kaur, 2011). *"[…] sustainability science is a different kind of science that is primarily use-inspired, […] with significant fundamental and applied knowledge components, and commitment to moving such knowledge into societal action"* (Kates, 2011, p. 19450). Inter- and transdisciplinary research requires building theories across disciplines, by drawing on epistemologies and methods from different fields and fusing them into new concepts, which is a time-consuming and challenging endeavor (Fischer-Kowalski and Weisz, 1999; Haberl et al., 2016; Kates, 2011; Moran and Lopez, 2016).

Increasingly an interdisciplinary systems perspective on the interlinkages between human and natural systems is recognized as being essential for sustainability science (Liu et al., 2015). Only then can trade-offs between conflicting societal and ecological goals be properly assessed, especially against the potentials for spillovers and problem-shifting across temporal and spatial scales (Haberl et al., 2016; Lenton et al., 2016; Liu et al., 2015; Singh et al., 2013). Within the different branches of sustainability science, Industrial Ecology is especially poised to provide important insights towards such a systems perspective, due to its analytical focus and rich inventory of methods to investigate the biophysical basis of societies and the relevant physical exchanges between social and natural systems (Clift and Druckman, 2016; Pauliuk and Hertwich, 2015; Weisz et al., 2015).

1.1. An analytical systems approach to the biophysical basis of society: the concept of socioeconomic metabolism and the framework of material and energy flow accounting

Within Industrial Ecology, the concept of a socioeconomic metabolism[1] has quickly gained prominence in guiding research on the biophysical exchanges between society and nature across temporal and spatial scales (Ayres and Simonis, 1994; Fischer-Kowalski and Haberl, 2007; Haberl et al., 2016; Pauliuk and Hertwich, 2015; Schandl et al., 2015). *"Socioeconomic metabolism constitutes the self-reproduction and evolution of the biophysical structures of human society. It comprises those biophysical transformation processes, distribution processes, and flows, which are controlled by humans for their purposes. The biophysical structures of society ('in use stocks') and socioeconomic metabolism together form the biophysical basis of society"* (Pauliuk and Hertwich, 2015, p. 85). Biophysical structures of society not only include the in-use material stocks of manufactured capital such as buildings, infrastructure and machinery, but also the human population and it's livestock (Fischer-Kowalski and Erb, 2016; Fischer-Kowalski and Weisz, 1999; Haberl et al., 2004; Pauliuk and Hertwich, 2015).

Based on this an approach, a minimum condition for sustainability can be formulated as the continuous ability of society to reproduce it's biophysical structures via their socioeconomic metabolism,

[1] Commonly used variations include industrial, social, societal or urban metabolism.

while being able to adapt to socio-ecologically induced and naturally occurring changes in the biogeosphere (Fischer-Kowalski, 2011; Fischer-Kowalski and Weisz, 1999; Haberl et al., 2011, 2004).

Economy-wide material and energy flow accounting[2] (ew-MEFA) is a prominent method in which the concept of the socioeconomic metabolism has been operationalized. It allows to systematically investigate biophysical exchanges between society and its natural environment and provides robust and policy relevant indicators (Fischer-Kowalski et al., 2011; Haberl et al., 2004). Ew-MEFA, as a comprehensive monitoring framework, has been substantially advanced since the seminal studies of the World Resources Institute were conducted in the late 1990s (Adriaanse et al., 1997; Matthews et al., 2000). After a phase of standardization in the early 2000s (Eurostat, 2001; OECD, 2008), ew-MEFA was implemented into official statistical reporting and policy processes in Japan, the EU and by the OECD and UNEP from the mid 2000's onwards.

Currently global ew-MEFA databases cover biophysical information on extraction, trade and use of biomass, metal ores, fossil fuels and non-metallic minerals disaggregated into 45-50 material categories. Data is available on the national level from 1950 (Schaffartzik et al., 2014b) and 1970 onwards (Lutter et al., 2016b; UNEP, 2016), globally also for the entire 20th century (Krausmann et al., 2009). The main indicators currently derived from ew-MEFA are the physical trade balance (PTB), domestic extraction (DE), and domestic material consumption (DMC). DMC is widely used as headline policy indicator for national apparent material consumption and resource efficiency (GDP/DMC) (Fischer-Kowalski et al., 2011; Weisz et al., 2006).

By utilizing comprehensive long-term ew-MEFA data, important insights have been generated. The relationships between development, economic growth, trade and materials consumption has been widely explored (Behrens et al., 2007; Giljum et al., 2014; Krausmann et al., 2009; Schaffartzik et al., 2014b; Steinberger et al., 2013). The requirements and potentials for decoupling resource use from economic activity have been clearly articulated (OECD, 2015; UNEP, 2016, 2011). The rapid transformation of Asian-Pacific economies has been documented (Schandl and West, 2010). The global and national prospects for decoupling and resource efficiency from a production-based and a consumption-based perspective are currently debated (OECD, 2015; UNEP, 2016).

Based on the comprehensive information generated in economy-wide material flow accounts, energy flow accounting assumes an energetic viewpoint. This means transforming material flows, usually measured as mass flow (tons), into their gross calorific contents (joules). Thereby food and feed biomass are counted as energy carriers, in addition to technical energy such as coal, oil or hydro and nuclear power

[2] The closely related approach of material and substance flow analysis also draws on the concept of socioeconomic metabolism, but usually takes a more flexible approach in terms of which materials or substances covered, which spatial and temporal scales are investigated and how the system boundaries are defined (Baccini and Brunner, 2012; Brunner and Rechberger, 2017; Chen and Graedel, 2012).

(Haberl, 2001). This enables the study of the energetic socioeconomic metabolism and the importance of societal energy uses as the prime movers of the metabolism and the reproduction of the biophysical structures of society (Haberl, 2001; Smil, 2008; Warr et al., 2010). This view provided important insights into the long-term socio-ecological transitions between socio-metabolic regimes and their different energetic base, land-use requirements and subsequent sustainability problems (Fischer-Kowalski, 2011; Fischer-Kowalski and Schaffartzik, 2015; Krausmann et al., 2008; Sieferle et al., 2006; Wiedenhofer et al., 2013a).

Ew-MEFA is closely aligned to the system of national environmental-economic accounts and therefore to other policy relevant socioeconomic indicators such as GDP or employment (Eurostat, 2001; OECD, 2008). Ew-MEFA indicators constitute a so-called production-based or territorial perspective (Bringezu, 2015; Fischer-Kowalski et al., 2011; Moriguchi, 2007). Subsequently, ew-MEFA corresponds to the national and international political sphere of action (Fischer-Kowalski et al., 2011). For this framework an interesting co-evolution of scientific knowledge and policy interest can be observed, both pushing and pulling each other (Bringezu, 2015). Increasingly, the comprehensive knowledge base on annual material and energy flows becomes relevant for the development of governance efforts for sustainable natural resource use at national and international scales (Bringezu et al., 2016).

Growing political awareness of environmental-economic interrelations and the usefulness of ew-MEFA indicators are exemplified by the United Nations' Sustainable Development Goal 12 on ensuring sustainable consumption and production patterns (UN Economic and Social Council, 2016), OECD reporting on green growth and material productivity (OECD, 2015) and the International Resource Panel of the UNEP (UNEP, 2016, 2011). In a number of countries, ew-MFA indicators are used in policies, such as the EU2020 Flagship Initiative for a Resource Efficient Europe (European Commission, 2011); the EU Circular Economy Action Plan (European Commission, 2015); China's circular economy plans (Mathews and Tan, 2016; Su et al., 2013); and Japanese 3R policies (Ministry of the Environment Japan, 2016; Takiguchi and Takemoto, 2008).

1.2. State-of-the-art in ew-MEFA: the black box approach to the socioeconomic system

Ew-MEFA is designed as a comprehensive monitoring framework to systematically observe the dynamics and composition of socio-economic flows of material and energy resources and thereby to assess progress towards sustainability (Adriaanse et al., 1997; Fischer-Kowalski et al., 2011; Haberl et al., 2004; Matthews et al., 2000). Ew-MEFA builds upon data from statistical reporting (e.g. agricultural, mining or energy statistics, trade statistics) provided by national statistical offices and international organizations (E.g. International Energy Agency, Food and Agricultural Organization of the United Nations etc.). This is the case, because the main sources of information:

a) has to be aligned to the system of national accounts;
b) needs to be readily available, robust and politically acceptable;
c) all flows should be accountable using mass-balance principles, to ensure a thermodynamically correct representation of society-nature interactions in the form of material and energy flows.

Cleary defined and harmonized system boundaries are another strength of the ew-MEFA framework, so that long-term and cross-sectional comparability at any spatial scale can be achieved (Figure 1) (Fischer-Kowalski et al., 2011; Haberl et al., 2004). For domestic extraction of resources and the corresponding domestic processed outputs of wastes and emissions, it is the society-nature boundary, which is defined based on the level of control and management exerted by society (Fischer-Kowalski and Weisz, 1999; Pauliuk and Hertwich, 2015). For biophysical imports and exports, it is the administrative and political boundary with other socioeconomic systems. Also (net) additions to stocks of artefacts such as buildings, infrastructure and machinery, as well as livestock and the human population need to be accounted for, as these flows are crossing the boundary between the biophysical structures of society and the socioeconomic metabolism (Fischer-Kowalski et al., 2011; Fischer-Kowalski and Weisz, 1999; Pauliuk and Hertwich, 2015).

Finally, an important conceptual difference in ew-MEFA is drawn between direct flows physically crossing said boundaries, and the ‘indirect’ or ‘embodied’ flows used along supply chains to deliver these flows of goods and services (Figure 1) (Fischer-Kowalski et al., 2011). These standardized accounting conventions and derived indicators are summarized in Figure 1, which already shows that socioeconomic processes within the ‘national economy’ are treated as a black box in the MEFA framework and the current emphasis lies on flows crossing said boundaries.

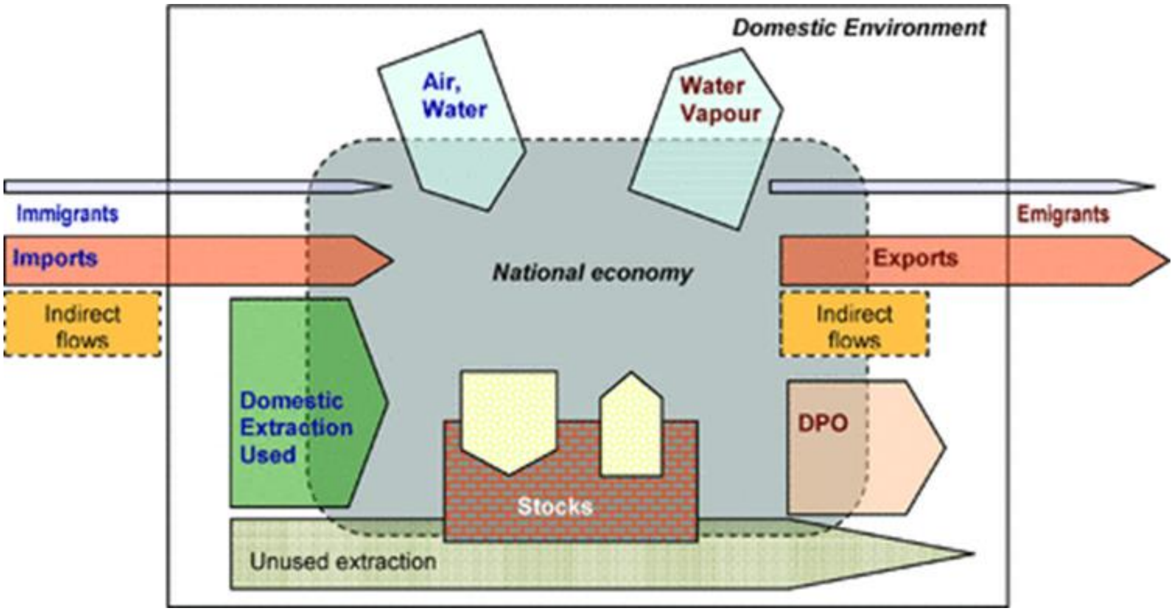

Figure 1: The Harmonized framework of economy-wide material and energy flow accounting (Eurostat, 2001; Fischer-Kowalski et al., 2011; OECD, 2008). The national socioeconomic system is treated as a black box and flows are covered when they enter or leave the system.

Ew-MEFA databases so far have focused on direct flows and available databases mainly report data on domestic extraction, imports and exports (Fischer-Kowalski et al., 2011; Giljum et al., 2014; Krausmann et al., 2009; Lutter et al., 2016b; Schaffartzik et al., 2014b; UNEP, 2016). This allows for the calculation of input and consumption indicators such as domestic material consumption (DMC = DE + Imp – Exp) or domestic material inputs (DMI = DE + Imp). Outflows (DPO, domestic processed outputs), indirect flows as well as stock related flows (gross & net additions to stock) have so far been less frequently addressed. The corresponding accounting methods and indicators are therefore less elaborate and standardized (Fischer-Kowalski et al., 2011; Kovanda, 2017; Kovanda et al., 2007; Lutter et al., 2016a). Studies which provide full balances of all input and output flows of socio-economic systems are rare (Adriaanse et al., 1997; Hashimoto et al., 2007; Hashimoto and Moriguchi, 2004; Kovanda, 2017; Kovanda et al., 2007; Matthews et al., 2000; Moriguchi, 2007). Given the prevalent emphasis on the successful establishment of accounting principles and procedures for flows crossing system boundaries in ew-MEFA, the socioeconomic system itself has so far been treated as a black box (see grey box 'national economy' in Figure 1).

The main reason for this black box simplification is that for many socioeconomic processes the statistical databases on energy and material flows are incomplete, relatively dispersed and not harmonized. This limits the possibilities of an accounting approach. For wastes and emissions, with the exception of carbon emissions (e.g. CO2 from fossil fuels and cement production), national accounts are substantially less well developed than for trade and extraction (coverage of the indicator domestic processed outputs

(DPO), Figure 1) (Moriguchi and Hashimoto, 2016). Furthermore, flows related to the biophysical structures of society (stock related flows, additions to stock), are also not directly available in most statistical databases. This is due to structural differences in how data on materials, energy, wastes and emissions are classified and how well all these flows are actually measured (Fischer-Kowalski et al., 2011; Kovanda, 2017; Matthews et al., 2000; Moriguchi and Hashimoto, 2016). To account for ‘indirect’ upstream flows highly-detailed, commodity-specific, bi-lateral trade models are required to trace material and energy use across supply chains. While a complete tracing of materials throughout the socioeconomic system has successfully been reported for a number of specific materials (Chen and Graedel, 2012), doing so for the entirety of materials and energy carriers covered in ew-MEFA, is quite challenging.

Conceptually, on the other hand, all materials and energy carriers extracted and traded are processed throughout economic sectors and then either accumulate in biophysical structures of society (in-use stocks of buildings, infrastructure and other artefacts, as well as livestock and the population), are dissipated (i.e. fertilizer), or end up as wastes and emissions. This clearly means that input flows need to explicitly linked to stock dynamics and output flows. One important challenge is then, that economic sectors and in-use stocks need to be operationalized to trace these flows and that some of these processes and especially stock dynamics need to be modelled (see following two sections) (Pauliuk and Müller, 2014; Schiller et al., 2016; Wiedmann et al., 2015). This constitutes an important next step beyond the existing emphasis on accounting procedures in ew-MEFA.

Overcoming these limitations and taking the next steps towards fulfilling the conceptual premises of ew-MEFA, is now required to deal with pressing new research frontiers and policy needs (Clift and Druckman, 2016; Fischer-Kowalski et al., 2011; Pauliuk and Müller, 2014; Schandl et al., 2015). Two of these frontiers are introduced below, which are also the topic of this dissertation.

1.3. Production- vs consumption-based perspectives in ew-MEFA: time to open the black box to investigate supply chains

The first frontier stems from the increasing fragmentation of supply chains and globalizing production systems. This has led to growing concerns about the impact of national consumption on the global environment due to economic activities in other countries and world-regions. Increasing spatial disconnects between production and consumption as well as between environmental pressures and consumptive benefits are often discussed due to their potential for problem- and burden shifting. These disconnects have been shown to undermine national and international policy efforts in reducing pressures on the environment (Bais et al., 2015; Kanemoto et al., 2014; Peters et al., 2011; Tukker et al., 2016; Wiedmann et al., 2015). To investigate these spatial disconnects it is necessary to trace so-called 'indirect' material and energy flows through economic sectors and across international supply chains.

For ew-MEFA, this is a major challenge, because the framework has been built along a territorial or production-based perspective (Fischer-Kowalski et al., 2011; Kovanda et al., 2012; Weisz et al., 2006). This means that trade is accounted for in physical terms when goods actually cross national borders and domestic extraction is allocated to the economy where these activities physically take place. Therefore, current headline indicators are aligned to the system of national environmental-economic accounts and subsequently to the national political sphere of action (Fischer-Kowalski et al., 2011).

Providing robust and policy relevant information on global resource use and emissions indirectly occurring along international supply chains, makes it necessary to develop complementary consumption-based indicators. These are termed 'indirect flows', 'raw material equivalents' or 'material footprints' (Fischer-Kowalski et al., 2011; Kovanda and Weinzettel, 2013; Schaffartzik et al., 2016; Schoer et al., 2012; Wiedmann et al., 2015). A consumption-based perspective effectively shifts the system boundaries of ew-MEFA, because it allocates all material and energy flows along global supply chains required to deliver goods and services to their final consumer (Peters, 2008). Effectively, tackling this frontier requires opening up the black box of the socioeconomic system, by modeling inter-sectoral economic relationships along supply chains, their related environmental pressures and the links to consumption of households, governments, capital formation or entire nations (Giljum et al., 2016; Hoekstra and Wiedmann, 2014; Minx et al., 2009; Tukker et al., 2016).

Tracing material and energy flows along supply chains is a data and modeling intensive effort. Early efforts used information from life cycle inventories (LCI) and other process specific factors (Bringezu et al., 2003). However, it quickly became clear that environmentally-extended input-output analysis (IO) provides a more comprehensive approach based on officially available national data in the form of national IO tables (Hertwich, 2005; Kovanda and Weinzettel, 2013; Lutter et al., 2016a; Munksgaard et al., 2005; Schaffartzik et al., 2014a; Suh et al., 2004; Suh and Nakamura, 2007). Some attempts at using physical input-output tables were conducted (Hoekstra and van den Bergh, 2006; Weisz and Duchin, 2006), although data requirements for a complete coverage of ew-MEFA are substantial and hard to overcome. Very quickly, various so-called hybrid IO-LCI approaches were developed to estimate consumption-based indicators for the ew-MEFA approach (Kovanda and Weinzettel, 2013; Schaffartzik et al., 2014a; Schoer et al., 2012). Due to various methodological challenges, limitations and assumptions involved in combining national IOs with process-specific LCI information (Majeau-Bettez et al., 2011; Schaffartzik et al., 2014a; Schoer et al., 2013), increasingly also multi-regional input-output models (MRIO) were developed (Wiedmann, 2009; Wiedmann et al., 2011).

Nowadays there is a range of different MRIOs available. They comprise consistent information on global inter-sectoral supply chains based on national input-output tables in monetary terms, which can be extended with biophysical information (Inomata and Owen, 2014; Wiedmann et al., 2015). Additionally, also approaches based on purely physical trade data, using information on process chains and trade flows have been developed, although mainly for biomass flows (Kastner et al., 2014, 2011). These methods are advancing rapidly and are gaining significance in Industrial Ecology.

However, these new methodological approaches require careful and critical reflection on the interpretability of derived indicators. MRIOs have been shown to yield contradictory results than physical trade approaches, which requires further study and clarification (Giljum et al., 2016; Inomata and Owen, 2014; Kastner et al., 2014; Lutter et al., 2016a; Owen et al., 2017). The ew-MEFA approach utilizes principles of biophysical accounting, aligned with standard socioeconomic national accounting. Biophysical estimates are ‘independent’ of monetary information or proxies (Steinberger et al., 2010). This ensures that ew-MEFA indicators are directly relatable to socioeconomic indicators, statistically speaking they remain independent from them. Modeling material and energy flows along inter-sectoral monetary information in IO frameworks then introduces new challenges in keeping these useful properties.

Overall, consumption-based approaches open up new and interesting avenues to investigate resource use and emissions along international supply chains through socioeconomic systems. They, for example, enabling linking indirect flows related to consumption patterns, lifestyles, human development and social progress (Kastner et al., 2012; Lamb et al., 2014; Lenzen and Cummins, 2011; Lenzen and Peters, 2010; Steinberger et al., 2012; Wilson et al., 2013). Additionally, they open up new perspectives on national resource use requirements, efficiency, decoupling and the effects of policy (Afionis et al., 2017; Minx et al., 2009; Schoer et al., 2012; Wiedmann et al., 2015). Ultimately, they provide information on telecoupling between systems (Liu et al., 2015). Advancing these approaches in a way that is consistent with ew-MEFA is therefore one objective of this dissertation.

1.4. The Circular Economy as emerging sustainability strategy: opening the black box to develop appropriate monitoring and indicators

An important second research frontier for monitoring progress towards sustainability has emerged, alongside the role of international trade in displacing environmental burdens and resource use. Recently, growing national policy efforts and business momentum around the Circular Economy (CE) concept as a new sustainability strategy have become prominent. Europe, China and Japan already have explicit policies on this issue and a large number of corporations and smaller business are rallying around the approach popularized by the Allen McArthur Foundation (Webster, 2017).

In this line of reasoning, the current economic system is described as linear, where resources are extracted, used and then disposed of. In contrast, a circular economy should consist mainly of closed material loops, where end-of-life products are re-used or recycled and only those materials, which ecosystems can absorb, are returned to natural systems (UNEP, 2012; Webster, 2017). This improved circularity is supposed to lead to reduced demand for primary resources, decreasing environmental pressures, improved resource efficiency and ultimately, progress towards sustainability (European Commission, 2015; Mathews and Tan, 2016; Su et al., 2013). These ideas are closely connected to the widely known 3R's (reduce, re-use, recycle) developed in the 1980s (Moriguchi and Hashimoto, 2016). Appropriate comprehensive monitoring and policy relevant indicators are required to evaluate national and global progress towards sustainability.

While ew-MEFA can in principle be used as a tool to assess the socio-metabolic impacts of circular economy strategies at an economy-wide level, certain methodological advancements are required to provide appropriate indicators (Hashimoto and Moriguchi, 2004). At the moment, ew-MEFA databases provides data on materials extraction and physical trade, but not for recycling flows within the socioeconomic system and no consistent information on domestic processed outputs (wastes, emissions) to nature (Fischer-Kowalski et al., 2011; Giljum et al., 2014). Therefore, ew-MEFA currently can only monitor parts of the goals put forward by circular economy strategies.

Opening the black box then becomes necessary, to follow extracted, traded, processed, used and recycled materials to where they ultimately end up, in a mass-balanced and thermodynamically correct manner. This can be either in the socioeconomic system as biophysical structures of society (in-use stocks, livestock, population), or when the material and energy flows are returned to natural systems in the form of waste and emissions. However, such information is simply not directly available from statistical data sources (Fischer-Kowalski et al., 2011; Moriguchi and Hashimoto, 2016). Therefore, major advances beyond accounting become necessary. Modeling approaches, which are fully compatible with system boundaries, definitions and concepts of ew-MEFA need to be developed (see section 1.2). In particular, it becomes necessary to move from only investigating annual flows crossing system boundaries, towards empirically including the in-use stocks of the biophysical structures of society and economy-internal recycling loops.

In-use stocks of artefacts, people and livestock are conceptually covered in ew-MEFA and (net) additions to stocks have sometimes been estimated (Augiseau and Barles, 2016; Fischer-Kowalski et al., 2011; Kovanda et al., 2007; Pauliuk and Müller, 2014; Tanikawa et al., 2015). Modeling stocks is a rapidly advancing research area in Industrial Ecology and beyond (Augiseau and Barles, 2016; Chen and Graedel, 2015; Müller et al., 2014; Pauliuk and Müller, 2014; Tanikawa et al., 2015). Bottom-up, top-down, static and dynamic approaches, as well as various combinations have been presented. In-use stocks and demand for their services drive current and future resource use for maintenance, ongoing expansion, but equally end-of-life outputs and therefore potentials for recycling and a circular economy (Hashimoto et al., 2007; Müller, 2006; Pauliuk et al., 2013a; Pauliuk and Müller, 2014; Wang et al., 2014; Weisz et al., 2015). Which of these methods is most appropriate for an opening of the black box in ew-MEFA, to provide macro-level monitoring of the CE, is therefore a pertinent question for this dissertation.

Comprehensive information on stock dynamics would enable an important step towards empirically closing the mass balances from the input to the output side of the metabolic system, thereby establishing consistent economy-wide accounts from extraction, to trade, uses, transformations to products, stock accumulation and ultimately wastes and emissions. This knowledge base would then provide systematic long-term information on the biophysical basis of society, covering all socio-metabolic flows as well as biophysical structures (Pauliuk and Hertwich, 2015; Pauliuk and Müller, 2014). Comprehensive indicators on progress towards a circular economy could be provided. Important information for a systems perspective on the interlinkages between coupled human and ecological systems would then become possible (Liu et al., 2015).

2. Research objectives and questions: modeling the insides of the black box of the socioeconomic system in ew-MEFA

The following two research objectives and their underlying research questions (a-h) motivated and guided my research contributions included into this dissertation:

Objective Nr. 1: From territorial accounting in ew-MEFA, towards a consumption-based perspective:

a. How can the black box of the socioeconomic system in ew-MEFA be opened, to systematically investigate international supply chains and link resource use and emissions across economic sectors to final consumption?
b. What are upstream consumption-based implications of final consumption of economies and their population, compared to a territorial perspective?
c. Which modeling approaches are appropriate to systematically complement current production-based territorial information in ew-MEFA with consumption-based indicators?
d. Which new insights into progress towards sustainability emerge from a consumption-based perspective?

Objective Nr. 2: Integrating biophysical structures into ew-MEFA, to monitor progress towards a more sustainable circular economy:

e. How can the biophysical structures of society (artefacts, livestock and population), be explicitly operationalized and quantified to open up the black box of ew-MEFA?
f. What are the specific methodological challenges for estimating in-use stocks of artefacts?
g. How can a fully mass-balanced view on material and energy flows and stocks, throughout the biophysical basis of society, be achieved for the ew-MEFA framework?
h. What are important insights and next steps towards a comprehensive monitoring of progress towards a more sustainable circular economy?

To address these two main research objectives of this dissertation, I selected six peer-reviewed research articles from my scholarly outputs, which contribute to the above discussed research frontiers and offer building blocks towards advancing the methodological framework and policy relevance of ew-MEFA. A number of different projects and funders supported these research efforts; consulting projects led by BIOIS for the European Directorate General for the Environment (DG Env), two large projects in the EU Commissions FP7 program, a basic research project funded by the Austrian Science Fund FWF, to a series of smaller research and consulting projects for the Austrian Federal Ministry for Agriculture and the Environment. Finally, I invested my personal time to finalize the research summarized herein.

In first three articles of this dissertation, Input-Output Analysis was applied and evaluated as a modeling strategy to cover inter-sectoral relationships across supply chains within and across national economies, enabling an assessment of the material, land and carbon footprints of final demand (Table 1). We evaluated biophysical modeling and environmentally-extended input-output approaches to account for upstream biomass and land footprints, highlighting the importance of operationalizing land-use intensity to comprehensively account for the land requirements of consumption (Schaffartzik et al., 2015). We compared several multi-regional and hybrid IO-LCA input-output models for their robustness and discussed the required steps to provide policy-relevant consumption-based indicators on ‘indirect flows’ in the ew-MEFA framework (Eisenmenger et al., 2016). Finally, I led an investigation of the role of inequality for household consumption in China and their respective carbon footprints from direct and indirect emissions from fossil fuels and cement production (Wiedenhofer et al., 2017). China is one of the most dynamic and due to its scale, highly important countries for global sustainability. This makes it highly relevant to open the black box for not only supply chains, but also equally for a more specific understanding of the role of consumption, inequality, urbanization and prospects towards absolute reductions of emissions.

In the other three articles, different approaches monitoring the circular economy were tested and developed. All involved explicitly modeling in-use stocks and extending the ew-MEFA framework. I developed a stock-driven, bottom-up modeling of flows to evaluate recycling potentials and policy goals towards improved circularity of non-metallic construction minerals, also linking the modeling to ew-MEFA accounts (Wiedenhofer et al., 2015). In a second paper, we developed an extended ew-MEFA framework introducing new indicators and sustainability criteria to evaluate progress towards a circular economy at the global and European scale (Haas et al., 2015). Finally, we squarely built upon ew-MEFA principles and accounts, to develop an input-driven, top-down dynamic stock model to investigate the global accumulation of materials in manufactured capital and its implications for energy use, emissions and decoupling (Krausmann et al., 2017).

Table 1: Overview on research contributions included in this thesis

	Authors	Title	Modeling …			Journal
			Method	Approach	Scale	
Research objective Nr. 1: Production- and consumption-based perspectives	Schaffartzik, Haberl, Kastner, **Wiedenhofer**, Eisenmenger, Erb	Trading Land: A Review of Approaches to Accounting for Upstream Land Requirements of Traded Products: A Review of Upstream Land Accounts	Review of biophysical accounts and Environmentally Extended Input-Output Analysis	Static, for 2004 and 2007	Several country results	*Journal of Industrial Ecology (2015)*
	Eisenmenger, **Wiedenhofer**, Schaffartzik, Giljum, Bruckner, Schandl, Wiedmann, Lenzen, Tukker, Koning	Consumption-Based Material Flow Indicators — Comparing Six Ways of Calculating the Austrian Raw Material Consumption Providing Six Results	Single-regional, multi-regional, and hybrid IO-LCA models	Static, for 2007	Austria	*Ecological Economics (2016)*
	Wiedenhofer, Guan, Liu, Meng, Zhang, Wie	Unequal household carbon footprints in China	Chinese IO model linked to global multi-regional IO model	Static, for 2007 and 2012	China	*Nature Climate Change (2017)*
Research objective Nr. 2: Integrating stock-flow relations into ew-MEFA	**Wiedenhofer**, Steinberger, Eisenmenger, Haas	Maintenance and Expansion: Modeling Material Stocks and Flows for Residential Buildings and Transportation Networks in the EU25	Bottom-up modeling of stocks and flows, links to ew-MEFA	Dynamic, 2004-2009 Scenario until 2020	EU25	*Journal of Industrial Ecology (2015)*
	Haas, Krausmann, **Wiedenhofer**, Heinz	How Circular Is the Global Economy?: An Assessment of Material Flows, Waste Production, and Recycling in the European Union and the World in 2005	Extended economy-wide material and energy flow accounting (ew-MEFA)	Static, for 2005	Global and EU27	*Journal of Industrial Ecology (2015)*
	Krausmann, **Wiedenhofer**, Lauk, Haas, Tanikawa, Fishman, Miatto, Schandl, Haberl	Global socioeconomic material stocks rise 23-fold over the 20th century and require half of annual resource use	Top-down, input-driven stock model, coupled into economy-wide material and energy flow accounts (ew-MEFA)	Dynamic, 1900-2010 Scenarios until 2030	Global and three world regions	*Proceedings of the National Academy of Sciences (2017)*

The following sections provide an overview of my research contributions. I discuss more closely how they contribute to answering the specific research topics raised in this dissertation and also outline my personal contribution in the case of multi-authored papers.

2.1. Trading Land: A Review of Approaches to Accounting for Upstream Land Requirements of Traded Products: A Review of Upstream Land Accounts

The ew-MEFA framework has been conceptualized as integrated monitoring tool for socioeconomic uses of material, energy and land (Haberl et al., 2004). Due to increasing global socio-economic trade of biomass-based products, it became clear that consistent and robust indicators on so-called land footprints, i.e. the land use required to produce traded products, are also needed (question b). Ideally, these would also be compatible with the biophysical accounting of biomass flows in ew-MEFA, to enable systematic comparison and evaluation (question a - c) (Schaffartzik et al., 2015). I contributed to the design of this review and to the analysis by evaluating the MRIO based approaches. I also made major contributions to writing the manuscript.

For this research, we evaluated existing methods for land footprinting, focusing on two main methodological challenges. (1) How can land be allocated to products traded and consumed and (2) which metrics are required to account for differences in land quality and land-use intensity (Schaffartzik et al., 2015). Two families of accounting approaches have been evaluated: biophysical, factor-based versus environmentally-extended input-output analysis. While biophysical approaches capture a large number of products and different land uses, they suffer from a truncation problem. Economic input-output approaches overcome this truncation problem, but are hampered by the higher aggregation of sectors and products. Despite conceptual differences, the overall similarity of results generated by both types of approaches is remarkable. Diametrically opposed findings for some of the world’s largest producers and consumers of biomass-based products, make interpretation difficult and make methodological improvements necessary. We finally discuss possible reasons and remedies for these methodological challenges (Schaffartzik et al., 2015).

While the focus of this study is on land, the links to ew-MEFA are obvious, since from a methodological perspective, socio-economic biomass flows are underlying the land requirements analyzed (questions a). Furthermore, for land use there are physical accounting approaches available, as well as mixed monetary biophysical input-output approaches, to estimate consumption-based indicators (questions b). This enables a wider evaluation of the differences and similarities in these methods and contributes to the further advancement of these methods (question c). It became quite clear that practically, there are important differences between IO approaches, which use relatively aggregate monetary inter-sectoral relationships to allocate land-use requirements along supply chains, and biophysical approaches directly estimating land requirements related to more detailed traded biomass products. Which method is more appropriate therefore strongly depends on the research questions asked (questions c). Importantly, to monitor sustainability, only monitoring area extents is not enough and indicators of land-use intensity are required (question d).

2.2. Consumption-Based Material Flow Indicators — Comparing Six Ways of Calculating the Austrian Raw Material Consumption Providing Six Results

Due to the need for consumption-based indicators in ew-MEFA, a comparative evaluation of existing and widely used modeling approaches to calculate consumption-based indicators complementing the existing territorial ew-MEFA indicators was conducted (Eisenmenger et al., 2016) (questions a - c). I contributed to the design of the research and implemented one of the multi-regional input-output approaches (WIOD) to estimate material footprints and evaluated the MRIO literature; thereby I provided insights into the workings and assumptions of MRIO-based research. I substantially contributed to writing the manuscript.

In this article, we evaluated six modeling approaches to calculating the Austrian material footprint for the year 2007, using 3 multi-regional input–output (MRIO) and 3 hybrid life-cycle analysis-IO approaches. Five of these resulted in total raw material consumption (RMC), or material footprint, higher than the territorial indicator domestic material consumption (DMC). One hybrid LCA-IO approach delivered an RMC lower than DMC. For specific material categories, results between models diverge by 50% or more. Additionally, it became clear that the consumption-based indicators on the raw material equivalents of physical imports and exports, as estimated with an MRIO, are not based on the same system boundary definition as in an hybrid IO-LCA approach or in the territorial production-based ew-MEFA. Therefore clear definitions and delineations of the underlying boundaries and their interpretability are developed in this article. Due to the policy relevance of the RMC and DMC indicators it is paramount that their robustness is enhanced, which needs both data and method harmonization (Eisenmenger et al., 2016).

With this study, an important step has been made to reflect on the methodological and conceptual challenges of production- and consumption-based perspectives for ew-MEFA. It enabled closer insights into the differences between hybrid IO-LCA vs MRIO approaches, as well as the biophysical accounting in ew-MEFA (questions a – c). This research constituted an imported input into the ongoing evaluation of ew-MEFA and the increasing demand for upstream indicators by policy makers (questions d).

2.3. Unequal household carbon footprints in China

China has become a major driver of global environmental change. Increasing affluence and rapidly decreasing poverty are important successes, however, the environmental consequences are also substantial (Spangenberg, 2014). Utilizing a consumption-based MRIO approach, I investigated the dynamics and unequal contributions of different household income groups to overall emissions footprints from fossil fuel combustion and cement production (question a - d) (Wiedenhofer et al., 2017). This work was inspired by some of my previous research on the energy requirements of household consumption (Wiedenhofer et al., 2013b). I designed this research with contributions from Dabo Guan and performed the necessary calculations, utilizing national and global emissions statistics, the Chinese IO and the GTAP-MRIO. I analyzed the results, discussed the findings with my co-authors and wrote the manuscript. Dabo Guan and the team of Chinese co-authors provided data and contributed to writing the manuscript.

In this research, distributional focused carbon footprints for Chinese households were investigated and a carbon-footprint-Gini coefficient was used to quantify inequalities (Wiedenhofer et al., 2017). In 2012 the urban very rich, comprising 5% of population, induced 19% of the total carbon footprint related to final consumption in Chinese households, with 6.4 tCO_2/cap, while the average Chinese household footprint remains comparatively low (1.7 tCO_2/cap). In contrast, the carbon footprints of the rural population and urban poor, comprising 58% of population, are far below the Chinese average 0.5–1.6 tCO_2/cap. From 2007 to 2012, the total household footprint increased by 19%, with 75% of the increase due to growing consumption of the urban middle class and the rich households. These findings suggest, that the transformation of Chinese lifestyles away from the current trajectory of carbon-intensive consumption patterns, will require policy interventions to improve living standards and encourage sustainable consumption (Wiedenhofer et al., 2017).

In this study, the population of a China and its consumption are conceptualized as drivers of resource use and emissions across the international economy. Therefore, it was important to take into account the international supply chains involved in delivering goods and services consumed by different households, to more accurately depict the overall resource use and emissions requirements of consumption (questions b, d). Startign from robust emissions accounts (Z. Liu et al., 2015), I coupled a high resolution national IO model with 135 sectors, to the global GTAP-MRIO with 57 sectors, to mitigate the impact of widely discussed aggregation errors (questions a, c) (Lenzen, 2011; Steen-Olsen et al., 2014). The findings highlight that not everyone benefits equally from resource use and emissions, and that inequality between households matters substantially for an analysis of their contributions to global change (question b, d). For ew-MEFA, I find that a differentiation of the material and energy requirements of different lifestyles, income groups or other actors is important for sustainability research (question d).

2.4. Maintenance and Expansion: Modeling Material Stocks and Flows for Residential Buildings and Transportation Networks in the EU25

Turning to the second research objective of this dissertation, specific interest for the possibility of improved recycling and ‘loop closing’ as a national strategy voiced by the European Commission in a consulting project, motivated the development of a bottom-up, stock-driven dynamic MFA model (questions e - h). This model was applied to residential buildings and the road and rail network in the EU25, to investigate the potentials towards closing material loops due to full implementation of the European Waste Framework Directive and specifically, the goals on improved recycling of construction & demolition waste until 2020 (Wiedenhofer et al., 2015). I designed this research with contributions from Julia K. Steinberger. Subsequently I developed the modeling approach, compiled the necessary data and drafted the manuscript. The co-authors contributed to the analysis of the model results and to writing the manuscript.

In this article, we quantified and compared the magnitude of material requirements for expansion versus those for maintenance of existing in-use stocks of residential buildings, road and rail networks. We discussed the findings in relation to economy-wide consumption of non-metallic minerals (Wiedenhofer et al., 2015). We further assessed the recycling potentials by comparing the magnitudes of estimated input, waste, and recycling flows from 2004 – 2009. In a trend scenario until 2020, we assessed the potential impacts of the European Waste Framework Directive and its specific recycling goals on material flows. In the EU25 a large share of material inputs are directed at maintaining existing stocks, especially for the road network. Improved management of existing transportation networks and residential buildings is therefore crucial for the future quantity and composition of non-metallic minerals required. Even with substantially improved recycling, fully closing loops could only be achieved, if the continued expansion of in-use stocks would be stopped (Wiedenhofer et al., 2015).

The main challenge for this research was to develop a macro-scale dynamic bottom-up stock accounting approach, relying on a variety of data sources and estimation procedures, for the EU25 (question e, f). While this approach enabled a closer view on the dynamics of different stocks and the respective flows, poor data availability on non-residential buildings and other infrastructures, as well as incomplete data coverage of residential buildings severely limited this approach (questions e - g). Still, this bottom-up modeling provided important insights into stock-flow relations and stock dynamics, which substantially contributed to the next two research contributions. Furthermore, this study showed that modeling efforts can successfully complement existing ew-MEFA indicators to better inform national policy goals (questions e, h).

2.5. How Circular Is the Global Economy? An Assessment of Material Flows, Waste Production, and Recycling in the European Union and the World in 2005

The circular economy is promoted as a strategy to reduce inputs of primary materials and outputs of wastes and emissions, by closing economic and ecological loops of resource flows. However, comprehensive indicators are still lacking. In this research we extended the ew-MEFA framework to provide appropriate indicators for a first assessment of the biophysical circularity of the economy on the global and European level (questions e – h) (Haas et al., 2015). I contributed to the design of the research and the conceptual developments required for adapting the ew-MEFA approach, especially in regards to operationalizing stock dynamics. I provided data and information concerning input and recycling of mineral materials and stock estimation. Furthermore, I contributed to writing the manuscript.

Our calculations show that globally 4 gigatonnes of waste materials per year (Gt/yr) are recycled, but that these flows are moderate compared to 62 Gt/yr of processed materials and total domestic processed outputs of 41 Gt/yr (Haas et al., 2015). We identify two main reasons for the low degree of circularity: Firstly, 44% of processed materials are used to provide energy (technical energy, food and feed) and are thus – for thermodynamic reasons - not available for recycling. Secondly, socioeconomic in-use stocks are growing at a high rate, with annual net additions of 17 Gt/yr. Despite considerably higher end-of-life recycling rates, in the EU the overall degree of circularity is even lower than the global average, because of large inputs and growing stocks. Our results indicate that mainly focusing on the 'output' side (end-of-pipe) will yield limited opportunities for a more sustainable circular economy. However, a shift to renewable energy from fossil fuel use, a significant reduction of societal stock growth, and decisive eco-design of new products going into use, are required to substantially advance towards a more circular economy (Haas et al., 2015).

For this research, we systematically opened the black box and traced material flows throughout the socioeconomic metabolism (questions e, g). To estimate outflows from stocks we used data from the previous research (Wiedenhofer et al., 2015) and used a simple delayed outflow approach, based on fixed lifetimes of past inflows (question f) (van der Voet et al., 2002). In this manner, we were able to make a first step towards closing mass-balances from economy-wide extraction, consumption, recycling, to domestic processed outputs (question g). We conceptually explored under which conditions the circular economy would contribute to progress towards sustainability, which is often implicitly assumed, but not automatically the case (question h). Furthermore, we also opened the black box by specifically evaluating the circularity potentials of much more detailed materials flows, than usually done in ew-MEFA studies (question k, l). From this research, I conclude that a proper integration of in-use stocks into ew-MEFA requires dynamic modeling, especially to be able to conduct long-term monitoring and dynamic scenario assessments (question e – h).

2.6. Global socioeconomic material stocks rise 23-fold over the 20th century and require half of annual resource use

As a final step towards opening the black box, I developed a top-down, input-driven dynamic stock model, which is fully compatible with ew-MEFA (questions e - g) (Krausmann et al., 2017). For this work, I developed the concept of the dynamic model with contributions from Fridolin Krausmann, and practically implemented the model with support by Tomer Fishman (Fishman et al., 2014) during a research stay at Nagoya University, Japan. I further developed a comprehensive treatment of uncertainty, using Monte-Carlo Simulations and sensitivity analysis (question f). After implementating additional modules to estimate recycling and downcycling flows, I provided estimates of global in-use stock dynamics for the 20th century, based on data gathered by the co-authors and me. Together with Fridolin Krausmann, I then wrote the manuscript, supported by the other co-authors.

In this article, we could show that globally during the 20th century, an increasing share of extracted materials are used to build in-use stocks of manufactured capital, including buildings, infrastructure, machinery and equipment (Krausmann et al., 2017). Approximately half of materials extracted globally by humans each year are used to build up or renew in-use stocks of materials. From 1900 to 2010 global material stocks increased 23-fold, reaching 792 Pg (± 5%) in 2010. Despite efforts to increase recycling rates worldwide, continuous stock growth precludes closing material loops. Recycling currently only contributes 12% of material inflows to stocks. Our estimates indicate that stocks are likely to continue to grow, driven by large infrastructure and building requirements in emerging economies. A convergence of material stocks at the level of industrial countries would lead to a fourfold increase in global stocks, and CO_2 emissions exceeding climate change goals. Reducing future increases of material and energy demand and greenhouse gas emissions will require decoupling of services from stocks and flows of materials. Examples include a more intensive utilization of existing stocks, longer service lifetimes and more efficient design of new stocks. The configuration and quantity of stocks determine future waste flows and recycling potential and are therefore key to closing material loops and reducing waste and emissions in a circular economy (Krausmann et al., 2017).

With this study, we opened up a comprehensive way forward for the systematic quantification and integration of in-use stocks into the ew-MEFA framework (questions e - g). The dynamic stock model I developed can be extended to world-regional and national scales, as well as for different functional stock types (question h). By combining the insights from the previous article on the circular economy with this dynamic model, we showed a way towards fully linking extraction, processing, uses, stock accumulation, end-of-life outflows, recycling and waste production (question g, h). In this manner, systematic and comprehensive insights into the development of the socioeconomic metabolism and the prospects towards a more circular economy can be produced (question h).

3. Discussion

The concept of socioeconomic metabolism has proven to be a fruitful interdisciplinary systems approach and has inspired a growing body of research on society-nature interactions across temporal and spatial scales (Brunner and Rechberger, 2017; Fischer-Kowalski and Haberl, 2007; Haberl et al., 2016; Pauliuk and Hertwich, 2015; Pauliuk and Müller, 2014). One important operationalization of this concept has been achieved in economy-wide material and energy flow accounting (ew-MEFA) (Fischer-Kowalski et al., 2011; Haberl et al., 2004). So far, this methodological framework has been focused on comprehensively accounting for all biophysical exchange processes between socioeconomic systems and their domestic environments, making them potentially manageable via robust and harmonized indicators for environmental-economic policy (section 1.1). The socioeconomic system itself has so far been treated as a black box, where material and energy flows are accounted for when they cross clearly defined system boundaries, but the specific processes and transformations of materials within the socioeconomic system and across supply chains were rarely systematically linked (section 1.2).

However, the emergence of new research frontiers in regards to international trade and the circular economy (sections 1.3 & 1.4), has made it necessary to expand the ew-MEFA approach. This requires opening up this black box and tracing extracted and traded materials throughout their uses, their processing across economic sectors and their accumulation as in-use stocks within the socioeconomic system, towards their ultimate fate as wastes and emissions (section 2). With this dissertation, a number of modeling approaches were applied and evaluated, which enable an opening of the black of the socioeconomic system within the established methodological accounting principles of ew-MEFA, to provide robust and relevant information and indicators on two important research frontiers.

3.1. Objective one: from territorial accounting towards a consumption-based perspective

Currently, ew-MEFA provides territorial production-based indicators covering the biophysical economy within a given territory. The growing empirical evidence on economy wide material flows indicates that production-based indicators need to be complemented by consumption-based counterparts to get a full picture of resource consumption, resource productivity and the decoupling of economic growth and resource use. This is an important and complex step, because it adds an entirely new perspective to ew-MEFA: starting from monetary final consumption and tracing the upstream or indirect material and energy requirements along international supply chains, follows a different system boundary than a production-based perspective, which accounts for flows physically crossing territorial system boundaries.

The contributions to the first research objective included herein investigate these upstream flows and evaluate different methods to open up the black box of the socioeconomic system (research questions a, b, c). It became clear that the investigation of supply chains and inter-sectoral dependencies requires modeling approaches complementing the accounts efforts prevailing in ew-MEFA research. Environmentally-extended input-output analysis seems generally well suited to approximate the spatial disconnects between production and consumption across national boundaries (questions a, b, c) (Eisenmenger et al., 2016; Schaffartzik et al., 2015) (sections 2.2 & 2.1). Interestingly, this method also enables a closer look into the socioeconomic system, for example by linking household consumption to the upstream resource and emissions footprints (Wiedenhofer et al., 2017) (section 2.3, research question b, d).

From the two methodological reviews included into this dissertation I conclude, that two modeling approaches are capable of quantifying upstream resource uses: multi-regional input-output models and bilateral physical trade models (questions a, b, c) (Eisenmenger et al., 2016; Schaffartzik et al., 2015) (sections 2.2 & 2.1). Depending on the research goals, the choice of modeling approach might differ, but due to the much larger data availability and more systematic coverage of the socio-economic system, multi-regional input-output models seem particularly well suited to provide consumption-based indicators for the ew-MEFA framework (question c). The research included herein also contributes to the necessary critical reflection on limitations and differing coverage of specific indicators within the framework, informing the needs for further methodological harmonization (questions c, d) (Eisenmenger et al., 2016; Schaffartzik et al., 2015) (sections 2.2 & 2.1).

My research on this topic corroborates, that the next steps for input-output modeling need to include addressing the main concerns of aggregation biases and mass balance violations in existing applications (question c) (Majeau-Bettez et al., 2016; Steen-Olsen et al., 2014; Többen, 2017). Additionally, the implications and assumptions of how the IO framework is being extended with environmental information, by linking specific material flows to specific monetary inter-sectoral relationships, also needs to be more carefully evaluated (Eisenmenger et al., 2016; Owen et al., 2017) (section 2.2). In the light of the relevance of capital formation as a driver of emissions and resource use (Krausmann et al., 2017; Södersten et al., 2017) (section 2.6), it would be important to critically reflect on how annual investments should be handled and eventually close the IO model for capital (Pauliuk et al., 2015). Finally, the specific policy implications of such a consumption-based approach need to be developed further, because these indicators do not directly relate to the national political sphere of action, as production-based indicators do (question d) (Eisenmenger et al., 2016; Schaffartzik et al., 2015). While such consumption-based indicators are increasingly useful to evaluate policy efforts for mere problem-shifting (Afionis et al., 2017), they also pose limitations because international supply chains cannot be governed nationally and directly targeting final demand is politically challenging (question d).

In this dissertation, the implications of urbanization, rising household incomes and increasing inequality for the carbon footprints of consumption have been investigated, for one of the currently most dynamic and highly interesting countries: China (question b, d) (Wiedenhofer et al., 2017) (section 2.3). For this work it was quite useful to start with robust accounts on carbon emission from fossil fuels and cement production (Z. Liu et al., 2015). Based on this physical information, links to the Chinese national IO and the global multi-regional Input-Output model GTAP could be established (questions a, b, c). This enabled a comprehensive investigation of the possibilities and limitations of sustainable consumption in the context of rapidly growing affluence in urban areas versus large swaths of low-income households in urban and rural areas (question d) (Wiedenhofer et al., 2017) (section 2.3). Interestingly, IO modeling enables a better understanding of how material and energy use and subsequently emissions within and across socioeconomic systems are connected to economic sectors, households, government consumption and capital formation. This enables linking national-level information typically presented in ew-MEFA, to more specific parts of the socioeconomic system.

3.2. Objective two: Monitoring progress towards a more sustainable circular economy

For the second objective, the established territorial perspective in ew-MEFA provides the starting point to investigate the dynamics of material and energy use within socioeconomic systems (question e). So far, ew-MEFA has focused on the accounting of material and energy flows when they cross the specific system boundaries of the national economy. However, to comprehensively monitor and evaluate the circular economy and other resource use strategies, opening the black box of the socioeconomic system within ew-MEFA becomes necessary, utilizing compatible and appropriate modeling approaches (questions e-g). Consistent integration of in-use stocks of materials into ew-MEFA is of specific importance, as my research shows (question h). Thereby, the potentials of a more circular economy as a strategy towards reducing socio-economic resource flows and a more sustainable social metabolism have been investigated (questions e – h) (Haas et al., 2015; Krausmann et al., 2017; Wiedenhofer et al., 2015) (section 2.4, 2.5, 2.6).

My research contributions included herein have all utilized different modeling approaches to linking existing information from ew-MEFA to their fate throughout the socioeconomic system (questions e - g). It turned out to be fruitful to shift from the purely material perspective of ew-MEFA accounts, to a use driven differentiation of material flows, in particular to operationalize the fundamental difference between energetic vs. material uses, because this determines any further potential for closing material loops (question e – g) (Haas et al., 2015) (section 2.5). While this fundamental difference already existed in ew-MEFA, so far material and energetic perspectives were applied separately: ether material flows in tons were investigated, or all energy carriers were transformed into their energetic contents, plus energy forms not captured in MFA such as hydropower and nuclear heat, measured in joules.

Another step towards a combined perspective has now been put forward in this dissertation: after resources are extracted, traded and processed, we allocate them to different types of use in the socioeconomic system. These range from accumulation in in-use stocks of buildings, infrastructure and other artefacts, to energetic uses such as digestion as food and feed, or thermal conversion of fossil energy carriers or other dissipative uses (question g) (Haas et al., 2015; Krausmann et al., 2017) (sections 2.5 & 2.6). Clearly, possibilities for closing the loops and the necessity for absolute reductions due to sustainability considerations differ according to the use-types, and can now be investigated more systematically (question h).

The findings of my research strongly support the insight, that in-use stocks of buildings, infrastructure and other artefacts are very important, as they already require more than half of annual global resource extraction (question h) (Krausmann et al., 2017) (section 2.6). Combined information from ew-MEFA, various other sources and a dynamic top-down stock modeling exercise yielded important insights into the dynamics of the socioeconomic metabolism and the biophysical structures of society, because of steadily increasing global material extraction and growing in-use stocks of artefacts (buildings, infrastructure, machinery, …) (Krausmann et al., 2017) (section 2.6). Clearly, the accumulation of in-use stocks is a major driver of global resource extraction and processing, due to ongoing urbanization, increasing demands for living space, growing transportation infrastructures and the overall expansion of manufactured capital. Again China emerges as an important driver of global dynamics, due to the rapid socio-ecological transformation during currently ongoing industrialization and urbanization processes (Krausmann et al., 2017; Spangenberg, 2014) (section 2.6).

My research on the circular economy also shows, that in-use stocks need to be explicitly covered in an extended ew-MEFA monitoring. This is due to their long-term dynamics in determining end-of-life outflows, recycling potentials and ongoing material and energy requirements for their construction, maintenance and operation (questions e-g) (Haas et al., 2015; Krausmann et al., 2017; Wiedenhofer et al., 2015) (sections 2.4, 2.5, 2.6). In my research, I also investigated stock dynamics using a bottom-up modeling approach, which enabled the differentiation between material flows for the maintenance and replacement of existing stocks, vs the material requirements of ongoing expansions of housing and road infrastructure (questions e - g) (Wiedenhofer et al., 2015) (section 2.4). Thereby I could show that in-use stocks of residential buildings and the road and rail network in the EU25 are still growing. Even substantially improved recycling and waste management, as enacted in the European Waste Framework Directive 2008/98/EC, can therefore only partially contribute to a closing of material loops (questions d). The reason is, that material requirements of housing stock expansion are substantially larger than the effective recycling potentials from end-of-life materials (Wiedenhofer et al., 2015) (section 2.4). However, such bottom-up efforts are difficult to compile and require large amounts of country specific data, which are often not available (question e, f) (Schiller et al., 2016; Wiedenhofer et al., 2015) (section 2.4). If such a knowledge base can be generated, important next steps would be a sub-national regionalized modeling efforts to inform waste management strategies, because transporting materials is the major environmental and economic constraint for recycling of bulk materials (question d) (Schiller et al., 2017, 2016; Tanikawa et al., 2015).

The ew-MEFA framework is directly related to material and substance flow analysis, which both are used to investigate the material dynamics of the socioeconomic metabolism and the biophysical structures of society (Brunner and Rechberger, 2017; Chen and Graedel, 2012; Pauliuk and Hertwich, 2016; Pauliuk and Müller, 2014). In material and substance flow analysis, specific materials or substances are traced from their cradle to the grave and through various socioeconomic processes in a stock-flow consistent manner. They often inform specific resource management strategies and identify issues such as the accumulation of problematic substances in specific media or unmanaged losses to the environment (Brunner and Rechberger, 2017; Chen and Graedel, 2012). Due to this more detailed focus and higher flexibility in terms of systems definition, material and substance flow analysis also has well developed concepts for handling complexity and uncertainty across socioeconomic processes (Augiseau and Barles, 2016; Brunner and Rechberger, 2017; Laner et al., 2015; Müller et al., 2014; Pauliuk et al., 2013b; Rechberger et al., 2014). A number of studies utilizing material and substance flow analysis of specific systems provided important information as well as calibration and validations for the comprehensive ew-MEFA perspectives included in this dissertation (Haas et al., 2015; Krausmann et al., 2017) (section 2.5 & 2.6). When opening the black box of the socio-economic system in ew-MEFA, an important learning opportunity between both closely related approaches therefore exists (questions e- h).

4. Conclusions and outlook

With this dissertation, the black box in ew-MEFA can now be fully opened. Input-output modeling and physical trade models provide robust and relevant information on resource use and emissions along supply chains. Thereby they contribute to a consumption-based complement to the existing production-based indicators of ew-MEFA (Eisenmenger et al., 2016; Schaffartzik et al., 2015) (section 2.2 & 2.1). Additionally, input-output modeling is well suited to link different categories of final demand to their upstream resource and emissions requirements (Wiedenhofer et al., 2017) (section 2.3).

Also, an extended ew-MEFA has been introduced to monitor progress towards a circular economy (Haas et al., 2015) (section 2.5). Due to certain limitations encountered for a bottom-up stock modeling approach (Wiedenhofer et al., 2015), a dynamic top-down stock model fully compatible with ew-MEFA has been developed (Krausmann et al., 2017) (section 2.6). The next steps will be combining the extended ew-MEFA and derived circularity indicators (Haas et al., 2015) (section 2.5), with the dynamic top-down stock model (Krausmann et al., 2017) (section 2.6). This will enable important new insights into the dynamics of the biophysical basis of society and the prospects towards a more sustainable circular economy.

The most interesting outcome of this dissertation for me is the explicit integration of the biophysical structures of society into ew-MEFA, effectively combining accounting with modeling. The stocks of human population and livestock can be covered using statistical bottom-up data. While these stocks can increase/decrease over time in terms of weight and numbers, they mainly transform energetically used materials into outflows (i.e. feed and food are digested into emissions as well as liquid and solid waste). In-use stocks of infrastructure, buildings and other artefacts accumulate slowly and are used for years or even decades. Therefore, end-of-life outputs, recycling potentials and waste flows are subject to the service lifetimes of in-use stocks. To fully capture the socioeconomic processes and stock dynamics transforming inputs of materials and energy into outputs of wastes and emissions, it seems useful to differentiate the uses of these material and energy flows and to consistently model stock dynamics (Haas et al., 2015; Krausmann et al., 2017) (section 2.5 & 2.6).

In this way, the temporal dynamics of the biophysical structures of society and their interaction with the socioeconomic metabolism can be modelled, and a fully dynamic ew-MEFA could be developed. Linking the dynamics of these structures with material and energy inputs and outputs over time, makes the influence of the systems' past on the present and future explicit (Müller et al., 2011; Pauliuk and Hertwich, 2016; Pauliuk and Müller, 2014). While the biophysical structures of society were (re)produced by resource use in previous times, these in-use stocks drive current input requirements and material outputs of wastes and emissions related to their maintenance and use (Müller, 2006; Pauliuk and Müller, 2014). Furthermore, material cycles and energy flows are coupled (Baynes and Müller, 2016). Therefore, stock dynamics drive material and energy requirements, making their explicit modeling crucial for prospective assessments and scenario development (Baynes and Müller, 2016; Pauliuk and Hertwich, 2016; Pauliuk and Müller, 2014). Herein, the biophysical requirements and recycling potentials of reproducing and adapting existing biophysical structures need to be explicitly taken into account (Wiedenhofer et al., 2015). Stock dynamics strongly constrain and shape the potentials for rapid absolute reductions of materials, energy and emissions required to stay within Planetary Boundaries. This could constitute an interesting starting point to investigate path-dependencies and lock-ins with a dynamic ew-MEFA approach.

Following on this, it would be interesting to investigate how in-use stocks of artefacts, in interaction with material and energy flows, provide services to society (Haberl et al., 2017; Krausmann et al., 2017; Müller, 2006; Pauliuk and Müller, 2014). Examples for such services include heated living space or safe and swift mobility, but this should be extended to a more systematic coverage of the Stocks-Flows-Services nexus, including a systematic definition and operationalization of 'services' (Haberl et al., 2017). Decoupling services from materials and energy use and subsequently emissions, but also stock requirements, are increasingly discussed as key strategies towards absolute reductions and sustainability (Akenji et al., 2016; Haberl et al., 2017; Krausmann et al., 2017; Pauliuk and Müller, 2014). Ultimately, resource use and in-use stock patterns are merely means, while on a societal level human development and wellbeing are the desired ends, which are dependent on the availability and access to these services (Lamb et al., 2014; Mayer et al., 2017; Steinberger et al., 2013). However, the unequal distribution of environmental pressures associated with the consumption of and access to these services, across different lifestyles, income groups and nations need to be taken into account, to assess fair and just progress towards sustainability in the Anthropocene (Pichler et al., 2017; Wiedenhofer et al., 2017).

Such an extended dynamic ew-MEFA approach, potentially even including induced indirect flows, could substantially inform a broader systems integration of coupled social and ecological systems (Haberl et al., 2016; Liu et al., 2015). This requires systematically covering telecoupling between systems, to identify critical interactions or nexuses between socio-ecological material and energy flows (Bleischwitz et al., 2017; Liu et al., 2015). Ultimately, trade-offs and option spaces for ecological, social and economic aspects of sustainability could then be made more explicit (Erb et al., 2016; Liu et al., 2015; López et al., 2017; Mayer et al., 2017). Such an understanding of the dynamics of the biophysical basis of society could help identify critical intervention and leverage points towards socio-ecological sustainability within Planetary Boundaries (Meadows, 1999; Pauliuk and Hertwich, 2016; Steffen et al., 2015).

> *"The highest leverage of all is to keep oneself unattached in the arena of paradigms, to realize that NO paradigm is "true," that even the one that sweetly shapes one's comfortable worldview is a tremendously limited understanding of an immense and amazing universe"* (Meadows, 1999, p. 13).

5. Literature

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6. Full research articles for objective one: From territorial accounting in ew-MEFA, towards a consumption-based perspective

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FORUM

Trading Land

A Review of Approaches to Accounting for Upstream Land Requirements of Traded Products

Anke Schaffartzik, Helmut Haberl, Thomas Kastner, Dominik Wiedenhofer, Nina Eisenmenger, and Karl-Heinz Erb

Keywords:

environmental accounting
environmental input-output analysis
industrial ecology
land footprint
land use
trade

Supporting information is available on the *JIE* Web site

Summary

Land use is recognized as a pervasive driver of environmental impacts, including climate change and biodiversity loss. Global trade leads to "telecoupling" between the land use of production and the consumption of biomass-based goods and services. Telecoupling is captured by accounts of the upstream land requirements associated with traded products, also commonly referred to as land footprints. These accounts face challenges in two main areas: (1) the allocation of land to products traded and consumed and (2) the metrics to account for differences in land quality and land-use intensity. For two main families of accounting approaches (biophysical, factor-based and environmentally extended input-output analysis), this review discusses conceptual differences and compares results for land footprints. Biophysical approaches are able to capture a large number of products and different land uses, but suffer from a truncation problem. Economic approaches solve the truncation problem, but are hampered by the limited disaggregation of sectors and products. In light of the conceptual differences, the overall similarity of results generated by both types of approaches is remarkable. Diametrically opposed results for some of the world's largest producers and consumers of biomass-based products, however, make interpretation difficult. This review aims to provide clarity on some of the underlying conceptual issues of accounting for land footprints.

The Need to Account for Upstream Land Requirements

Researchers and policy makers alike are responding to the challenge posed by the global fragmentation of supply and use chains. In environmental accounting, the need to account for upstream resource requirements associated with traded goods has been identified. As indicators are developed for environmental pressures and impacts, no matter where they occur, associated with a given level of consumption, questions arise as to how to allocate responsibility for global resource use. As a contribution to the ongoing debate, this article provides a review of approaches to accounting for upstream land requirements of traded products.[1] Upstream land refers to the land globally required to produce the goods and services for a given level of final demand. Upstream land consists of direct (e.g., cropland used to grow wheat for export) and indirect requirements (e.g., land used to grow oil crops for the production of lubricant for agricultural machinery used in the harvest of wheat for export). Biologically productive land is a key resource for humans as well as ecosystems. Land use is a pervasive driver of climate change, biodiversity loss, and other aspects of global environmental change (Foley et al. 2005).

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DOI: 10.1111/jiec.12258
Editor managing review: Daniel Müller

Volume 00, Number 0

Environmental policies and accounts, such as the United Nations Framework Convention on Climate Change (UNFCCC), operate from a production-based perspective (Peters 2008), holding countries accountable for the emissions that occur on their territory. In some cases, however, policies aimed at reducing domestic emissions lead to increased emissions elsewhere. In order to curb anthropogenic global warming, it is necessary to avoid this so-called leakage of greenhouse gases (GHGs) (Munksgaard and Pedersen 2001). Owing to the high relevance of emission leakage for global climate-change policy, research is more advanced for upstream emissions than for other forms of resource use (Galli et al. 2013; Fang et al. 2014; Čuček et al. 2012). The issue of upstream land requirements, however, is closely related to that of upstream emissions, which include emissions from land-use change (Gavrilova et al. 2010; Saikku et al. 2012). Leakage has also been observed for land-use policies. Prohibiting or limiting land-use expansion, for example, for nature conservation, in one country may lead to increased imports or decreased exports of biomass products (Rudel et al. 2009) unless consumption levels decrease. Protection of forests, as envisioned in the UNFCCC REDD+ Program (FAO/UNDP/UNEP 2008), can lead to increased imports of wood and wood products, which may, in turn, be associated with deforestation or forest degradation in other countries. If lower technical efficiency or environmental standards apply in these countries, aggravated impacts may be the result (Mayer et al. 2005). Stricter environmental protection legislation in developed countries could cause displacement of production to areas of the world where it is more environmentally harmful owing to the required intensification and/or extensification of land use (West et al. 2010). The regrowth of tropic forest cover in Vietnam can be linked to (partially illegal) logging and reduction of forest cover in other countries of South East Asia and in China (Meyfroidt and Lambin 2009). In many cases, forest transitions, that is, the return of forests in area and density after periods of deforestation (Kauppi et al. 2006), coincide with considerable displacement of forest harvest or even deforestation to other countries (Kastner et al. 2011a; Meyfroidt et al. 2010).

Displacement effects occur not only within, but also across land-use types. Tropical deforestation is significantly driven by the expansion of agricultural land for export-oriented production (DeFries et al. 2013; 2010; Hosonuma et al. 2012; Karstensen et al. 2013). The GHG savings achieved by substituting bioenergy for fossil fuels may be reduced or even negated through the associated indirect land-use change (Bird et al. 2013; Lapola et al. 2010) and the GHG emissions it causes (Chum et al. 2011; Haberl 2013; Searchinger et al. 2008). Land is increasingly recognized as a scarce resource and competition between the different possible uses of land is already, as expected (Haberl et al. 2014; Lambin and Meyfroidt 2011; Weinzettel et al. 2013), leading to conflicts between the different stakeholders involved (Gerber 2011; Peluso and Lund 2011).

In order to be effective, policies for sustainable governance of the earth's biologically productive land must consider the connection (or coupling) of developments across spatial distances. Trade is one of the central mechanisms mediating these connections: Changes in the final demand of one region are often directly and/or indirectly linked to land-use change elsewhere. Within land-system science, these insights have motivated the analysis of "teleconnections" or "telecouplings" (e.g., Güneralp et al. 2013; Haberl et al. 2009; Meyfroidt et al. 2013; Seto et al. 2012). Taking into account direct and indirect land requirements along global supply and use chains is paramount to understanding issues such as land-use displacement or land-related leakage. The currently emerging indicators of upstream land requirements of traded products push hard on the frontiers of socioeconomic metabolism research. New methods must be developed for the calculation of such indicators. These approaches challenge existing system boundary definitions and allocation principles in environmental accounting. The central challenges lie in developing an accounting principle by which land use can be allocated from a consumption perspective that reflects specific natural productivity and land-use intensity. Two main families of approaches currently exist that allow for the estimation of the share of a country's production that is dedicated to trade (also see Henders and Ostwald 2014). One can be characterized as economic modeling and most commonly takes the form of environmentally extended input-output analysis (EEIOA) based on the work of Leontief (1970). The other is based on biophysical accounting. Kastner and colleagues (2014b) have pointed out that these two types of approaches may produce diametrically opposed results for the land requirements associated with one country's final demand. The reasons why different results are generated are currently being investigated (Liang and Zhang 2013; Owen et al. 2014).

This review focuses on the conceptual differences among approaches to accounting for upstream cropland requirements of traded products. An a priori political decision as to how responsibility for land use ought to be allocated globally must be made in choosing the appropriate approach to use in calculating upstream land requirements. The results of these approaches are discussed in light of their conceptual differences.

Allocating Responsibility for Land Use

Whether production- or consumption-based land-use accounts are required depends on the research or policy question. Upstream land requirements that are related to domestic final demand, but occur in another sovereign country, are subject to that country's legislation and jurisdiction. European politicians, for example, cannot pass laws to alter agricultural production in Brazil. In order to inform domestic policy, it is necessary to use production-based accounts on land use. Where land-use decisions increasingly respond to foreign, rather than domestic, final demand, the potential impact of national policies in regulating land use may be limited (Lambin and Meyfroidt 2011). In order to curb global deforestation, reducing domestic consumption associated with a high upstream land requirement may be defined as a political goal. Corresponding policies may aim for the reduction of food waste or introduce disincentives to the import of bioenergy directly or indirectly linked to tropical deforestation. Information on imported and exported upstream land requirements may additionally be required in guiding policy.

The choice of either a production- or a consumption-based approach simultaneously reflects a political decision on the allocation of responsibility for land use. In economic terms, production is associated with added value, from capital and labor, within an economy. A country that exports land-based products receives revenues in return. France, for example, designates valuable agricultural land to the production of wine, one of the world's most expensive agricultural commodities, for export and receives significant income in return (FAO 2014). Under a production-based perspective, the argument might be made that a country is responsible for the income from its factors of production. In contrast, the two main families of consumption-based approaches (economic modeling and biophysical accounting) allocate responsibility by either economic spending or biophysical use: In simplified terms, the environmentally extended input-output approach distributes land use to monetary final demand according to the direct and indirect monetary inputs required in the production process. The biophysical approaches translate consumption in biophysical units (most commonly tonnes [t]) into the land required in production by using product-specific factors (most commonly in tonnes per hectare [t/ha]).

The consumption-based allocation of land requirements raises issues that form part of the long-standing debate on the allocation of environmental burdens to products in the life cycle assessment (LCA) community (Reap et al. 2008; Finnveden et al. 2009). Where one production process yields more than one product, environmental burdens may either be allocated to the dominant product (see Huppes 1994) or to all of the coproducts according to their share in monetary value of production (e.g., Fargione et al. 2008) or according to their share in the total mass, energy, or exergy expended in production. The current International Organization for Standardization (ISO) standards on LCA (ISO 2006) prescribe subdivision or expansion of the system boundaries of analysis by allocating to each product of a multiproduct process those environmental burdens that would occur in the corresponding single-product process (Azapagic and Clift 1999; Kim and Dale 2002).

Comparative assessments of producer and consumer responsibility (Munksgaard and Pedersen 2001; Muradian et al. 2002) have formed the basis for distinguishing allocation of responsibility by considering (economic) benefits or (ecological) burdens (Ferng 2003). It has been shown that even though a form of shared responsibility might be most appropriate, its definition is not trivial (Bastianoni et al. 2004; Jakob and Marschinski 2012). The concept of shared producer and consumer responsibility (Gallego and Lenzen 2005; Lenzen et al. 2007) is one manner of dealing with these issues. Thereby, all factor use is shared between the sectors along the supply chain, downstream sectors, and final consumers. It can either be formulated on a simple 50:50 allocation in each step through the supply chain or by using information on value added generated by production step, allocating responsibility for factor use by profit generated. Owing to the complexity and data requirements of such an approach, it has not been widely applied.

Next to the choice between the production- and consumption-based perspective and the different possibilities of allocating responsibility under the latter, upstream land requirements could be allocated in a number of different manners (Eder and Narodoslawsky 1999), depending on whether or not indirect effects of land-use change across spatial and temporal scales are taken into account. The approaches reviewed in this article are discussed in terms of the principles according to which they allocate land use to final consumption.

Measuring Land

In addition to differences in the principles according to which responsibility for land use is allocated, approaches to accounting for upstream land requirements differ conceptually in the metrics they employ. When land is measured in units of area such as hectares or square kilometers, no information about the productivity of that land or the intensity of its use can be conveyed. Land is used for agricultural production, as cropland, grazing land, and pasture, for forestry and as built-up land for human settlements, buildings, and infrastructure. These land-use types have very different impacts on ecosystems. Further, it is not straightforward to correctly represent land used for multiple purposes (e.g., agroforestry or forest grazing) in environmental statistics and accounting. Measuring upstream land requirements in terms of area extent aggregates land of different qualities, potentially making the results of such accounts difficult to interpret (Erb 2004; Haberl et al. 2004).

In accounts of upstream land requirements, trade flows (in biophysical units such as tonnes or monetary units such as Euros) are commonly converted into an area equivalent based on assumed yields, that is, on the mass or the economic value obtained per unit of area at the point of origin of the trade flow. A particular challenge in assessing upstream land requirements across land-use types (e.g., cropland, grassland, or forestry) is that land itself is an extremely heterogeneous resource in terms of quality. Land shows a vast gradient in natural productivity (declining in general terms from the equator to the poles owing to the temperature gradient, but strongly modified by other climatic factors, in particular, precipitation), in soil fertility (depending on many parameters, such as chemical composition of subsoil, microorganisms, depth, and so on), topography, and other factors. These differences in quality are often mirrored in the way land is used in agriculture, which may be labor- and/or energy-intensive (Erb et al. 2013; Kuemmerle et al. 2013). Grazing often occurs on marginal land (Asner et al. 2004; Erb et al. 2007), whereas high-value market crops will usually concentrate on the most fertile, productive plots.

Even within the same land-use type, productivity differences can be substantial owing to differences in land quality and/or management intensity. For upstream land accounts, this also raises the question of how multicropping (i.e., multiple annual

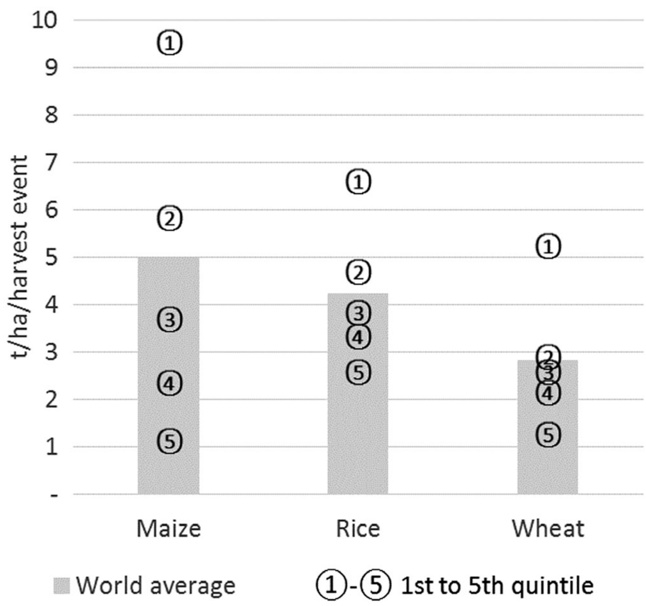

Figure 1 Yields in tonnes per hectare and harvest event in 2007 (t/ha/harvest event) for maize, rice, and wheat by country quintiles and world average in 2007, based on FAOSTAT data (FAO 2014). Quintiles each represent approximately 20% of global area harvested for the respective crops (as closely as possible using data at the national level), ordered by average country-level yields. Quintile yields are averages calculated from total production of the crop on the area. t/ha = tonnes per hectare.

harvests on the same plot of land) are translated into units of area. Between the quintile of countries with the highest maize yield and those with the lowest, for example, yields (in t/ha/harvest event) differ by a factor of approximately 9 (approximately 3 for rice and 4 for wheat; see figure 1) in 2007.[2] Yields on, in this case, cropland differ not only between, but also within countries (Monfreda et al. 2008). In terms of estimating upstream land requirements associated with traded products, this can become relevant if, for example, crops for export are produced on high-yielding productive areas whereas crops for domestic consumption are harvested from less-productive land. In this case, the use of the national average yield would lead to an overestimation of the land dedicated to production for export and an underestimation of the land required to satisfy domestic final demand.

The ecological footprint (EF) (Wackernagel and Rees 1998) is an approach to estimating upstream land requirements, which addresses different productivity levels of land by distinguishing types of land use. Land in ha of varying productivity is converted to global hectares (gha). This measure reflects the area that would be needed to produce a given harvest on land of global average productivity in a specific reference year (Kitzes et al. 2009; Wackernagel et al. 2002). The transformation from ha to gha allows for comparison of the results with the threshold of global biocapacity. The global or national "overshoot," that is, the extent to which resource demand exceeds potential resource supply (biocapacity), provides a strong indication of unsustainability. In order to account for the vast differences in average productivity of different land-use types, the footprint approach applies equivalence factors. These factors reflect the variation of the productivity of a given land-use type (at the global scale) from the global average productivity. Once they have been transformed to gha as a standardized measure of productivity, land areas of different quality and under different use can be aggregated (Kitzes et al. 2009). However, in expressing upstream land requirements in gha, the relationship to the land area actually available or used within each country is lost (Erb 2004; van den Bergh and Grazi 2014; van den Bergh and Verbruggen 1999). In particular, the global average productivity estimates for each land-use type cannot distinguish between the impact of natural fertility and agricultural management on yields (Wackernagel et al. 2004).

An approach that takes a different route in tackling these intricacies is the embodied human appropriation of net primary production (eHANPP) approach (Erb et al. 2009). The eHANPP concept is an extension of the human appropriation of net primary production (HANPP), an indicator of the changes in ecological energy flows associated with land use. HANPP is defined as the difference between the potential net primary production (NPP; i.e., the biomass production of green plants) of a defined land area and the amount of NPP remaining in the ecosystem after harvest. HANPP includes two separate processes: (1) alterations of NPP resulting from land use ($HANPP_{luc}$) and (2) harvest ($HANPP_{harv}$) (Haberl et al. 2007). eHANPP considers the differences in productivity potentials of land in trading countries, as well as the differences in land-use intensity across all types of land use (cropland, grazing, forestry, and built-up land) and between countries (also see the Supporting Information on the Web). eHANPP refers to the NPP of ecosystems and assesses the amount of ecological energy (or carbon) flows appropriated in providing biomass products. In contrast to land-use or footprint accounts, which are measured in area units, eHANPP is measured in t of carbon or dry-matter biomass.

A central advantage of this approach is that, whereas land can be used multiple times within a time frame for different purposes, the flow of NPP can only be used once. Further, it allows one to take differences in productivity as caused by, for example, soil quality or climate, into account. The embodied HANPP approach allocates NPP from land use to biomass products ($HANPP_{harv}$), the amount of unused extraction, as well as the productivity foregone owing to land conversions ($HANPP_{luc}$). This allows one to calculate the global HANPP associated with the consumption of biomass products in a country and contrast it with the HANPP that is associated with domestic land use (Erb et al. 2009; Haberl et al. 2012, 2009; see Kastner et al., this issue).

Upstream Land Requirements: What Do the Results Mean?

In contrast to accounts for upstream energy (e.g., Bullard and Herendeen 1975; Lenzen 1998) or emissions (see Peters et al. 2009), accounting for land requirements is a young field. A small, but growing, number of global studies has been conducted. Following up on the comparison conducted by Kastner and colleagues (2014b), examples of upstream land accounts were chosen for this review that represent biophysical accounting as well as economic modeling approaches.[3] Owing to coverage by studies based on the same land-use and harvest statistics provided by the United Nations' Food and Agricultural Organization (FAO), the focus is on results for upstream requirements for cropland. In order to ensure a certain degree of comparability, neither eHANPP nor the EF were included in this review.

Biophysical Accounting Approaches

The reviewed studies based on biophysical accounting are *factor approaches*: Import and export flows (commonly in tonnes per year) are multiplied by a factor (e.g., hectare per tonne) in order to translate them into units of "embodied resource." In the case of land, the factor reflecting the (national) average land requirement per unit of product (total harvested area of product divided by total production of product) is commonly multiplied with the quantity of the product exported (e.g., Saikku et al. 2012; Würtenberger et al. 2006). Land requirements can thus be estimated based entirely on data in biophysical units. In expressing biomass trade flows in terms of their upstream land requirements, the biophysical approach allows for consideration of spatially explicit yield factors, so long as the trade data are available at a corresponding level of detail. As a prerequisite thereto, trade flows must be traced to their point of origin by correcting for re-exports: For example, to assess the land requirement of soybeans produced in Mato Grosso, shipped to Rotterdam, and imported by Austria, Brazilian, rather than Dutch, yields should be used (Kastner et al. 2011b).

The point of departure in factor approaches is that the individual traded product and total domestic land use theoretically corresponds to the summation of the land requirements of domestic production. In practice, this type of bottom-up approach faces issues of double-counting and allocation: Where one production process yields more than one product, the associated land requirements must be allocated to the co-products. This allocation can be based on the relative biophysical or economic properties of the coproducts with significant impacts on the results (see Haberl et al. 2009). For example, if both vegetable oil and cake are derived from one crop, the land requirements of that crop could be allocated to oil and cake according to their respective share in total mass, energy content, or economic value of the crop. Multiproduct production does not have to occur simultaneously: Current production may be partially based on past resource use, which must also be allocated (and depreciated). In the oil crop example, the original deforestation and conversion into cropland enabled not only the production of crops in the first, but also in all following years. The factor approaches additionally face the challenge of system boundary definition. Points of truncation must be chosen in the supply chains for each product, whereby the analysis is limited to specific time periods, sectors, and production processes (Lenzen and Dey 2000; Suh 2004; Wiedmann 2009). For example, cropland is required in the production of vegetable oil used as industrial lubricant in agricultural machinery with which wheat for export is harvested. Truncation occurs if the oil cropland is not included in the upstream land requirements of the exported wheat. Biophysical accounts of upstream land requirements commonly allocate responsibility for land use according to relative volumes of final consumption by mass or energy content: The largest share of upstream land associated with the production of an oil crop would be allocated to that country that imports the largest share of the oil crop products for final consumption.

Economic Modeling Approaches

EEIOA is widely used in accounting for upstream resource requirements. EEIOA is a top-down approach that provides a mathematical solution to the allocation and truncation issues. Input-output tables (IOTs) of monetary flows[4] per year are used to represent production and final demand in one economy (single-region input-output model) or in several economies or regions (multiregion input-output [MRIO] model). The IOTs are extended by data on biomass harvest or land requirements of each economic sector. Based on the IOTs, the Leontief (1970) inverse is calculated: a matrix of multipliers, which reflect the direct and indirect inputs from all other sectors required by one sector in order to produce one unit of output to final demand. By multiplying the Leontief inverse with the matrix of land requirements of each sector or product, the land use associated with monetary domestic and foreign final demand can be estimated (Bicknell et al. 1998). The upstream land requirements calculated by the EEIOA approach cover all direct and indirect inputs without truncation, so long as they occurred during the year under investigation.

In MRIO models, the country-level IOTs are linked by bilateral monetary trade data. By considering only imports for final demand, the EEIOA approach does not require additional correction for re-exports. The transformation of biomass harvest data into units of area is technically possible at the same level of detail as under biophysical accounting. The distribution of land use to final demand, however, occurs at the level of detail prescribed by the resolution of the IOTs. If only one sector is reported for all biomass extraction, as was the case in Austria until 2000 (Schaffartzik et al. 2014), then upstream requirements for all land-use types will be distributed to other sectors and final demand without distinction. Even in less highly aggregated IOTs, the allocation of upstream land requirements to traded products based on average prices may be unsuitable for product categories with very different unit prices (Weinzettel et al. 2014). Under the structure of the Global Trade Analysis Project (GTAP; see www.gtap.agecon.purdue.edu/), for example, the

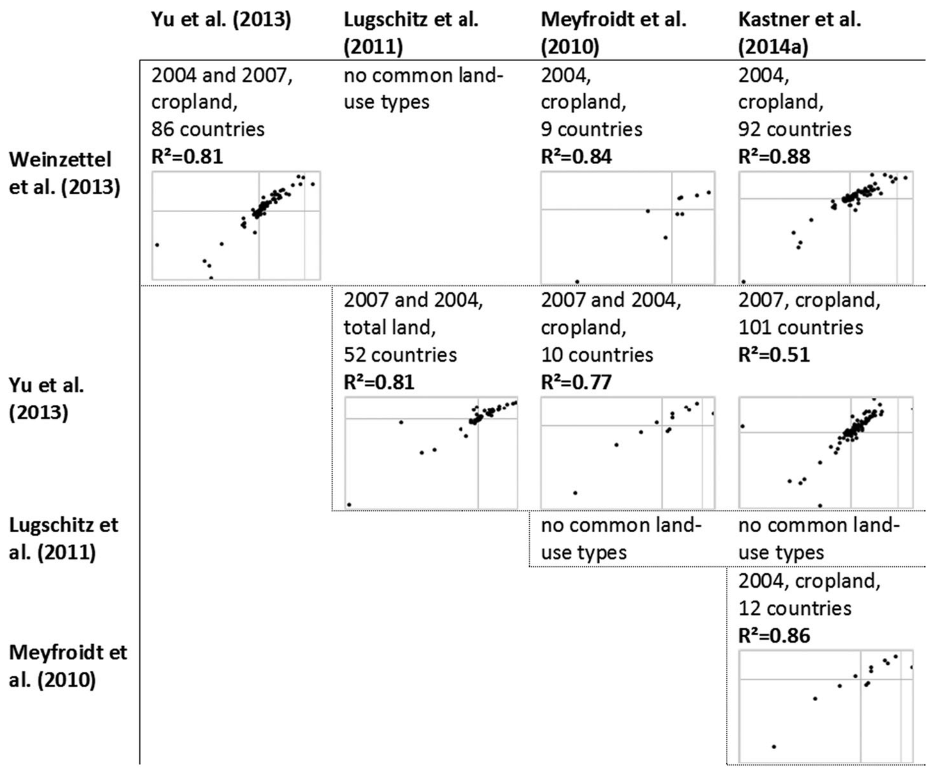

Figure 2 Overview of common characteristics (reference year, land-use type, and countries) of five studies on upstream land requirements, x-y plot of results, test for positive linear correlation (R^2) of net imports in hectares per capita.

same resource intensity per unit of monetary export would be assigned to Malaysia's exports of palm oil (705 US$/t in 2007) and cocoa butter (4,385 US$/t in 2007) because both belong to the category of vegetable fats. Factoring in the different yields for oil palm fruit and cocoa beans and the respective commodity trees of these products, the prices can roughly be translated into 2,981 US$/ha for palm oil and 2,056 US$/ha for cocoa butter. Whereas it would take only 1.4 ha of cocoa bean production to obtain the same economic value as from 1 ha of palm fruit production, almost 10 ha are required to produce the same physical amount of final product (all calculations based on data from FAO [2014]; on the issue of prices, also see Liang and Zhang [2013]). This example shows that unambiguous distribution of land requirements requires a greater degree of detail in the underlying economic data than is usually available.

The allocation by monetary value under the EEIOA approach differs fundamentally from the allocation by mass in biophysical accounting. For example, in 2010, the average price of palm oil consumed domestically in Indonesia was slightly lower (by approximately 77 US$/t) than the average price of palm oil exports (FAO 2014). Whereas the EEIOA approach would allocate the same upstream land requirement to each dollar spent on palm oil (if the IOT data were available at such a level of detail), biophysical accounting would allocate the same amount of upstream land to each t of palm oil exported. In this example, the upstream land requirements associated with Indonesian palm oil exports would be slightly lower under a biophysical than an EEIOA approach.

Finally, the EEIOA approach is highly dependent on the quality of the monetary IOTs; flows which are misrepresented in these tables will impact the results for upstream resource requirements.

Results for Upstream Land Requirements

Of the currently available EEIOA-based studies of upstream land requirements, three were selected (Lugschitz et al. 2011; Weinzettel et al. 2013; Yu et al. 2013), which coincide in at least one land-use category (cropland or total land) with one of the biophysical accounts (Kastner et al. 2014a; Meyfroidt et al. 2010). The EEIOA studies' IOTs were all constructed using GTAP data and their land-use data stemmed from the database of the United Nations' Food and Agricultural Organization (FAO). More information on the underlying data and methods is available in the Supporting Information on the Web.

As examples for the biophysical accounting approach, the global study by Kastner and colleagues (2014a) and the 12-country study by Meyfroidt and colleagues (2010) on upstream cropland requirements were used. Additionally, a number of national or regional case studies based on the factor approach are compared in the Supporting Information on the Web. As with the EEIOA-based studies, the study by Kastner

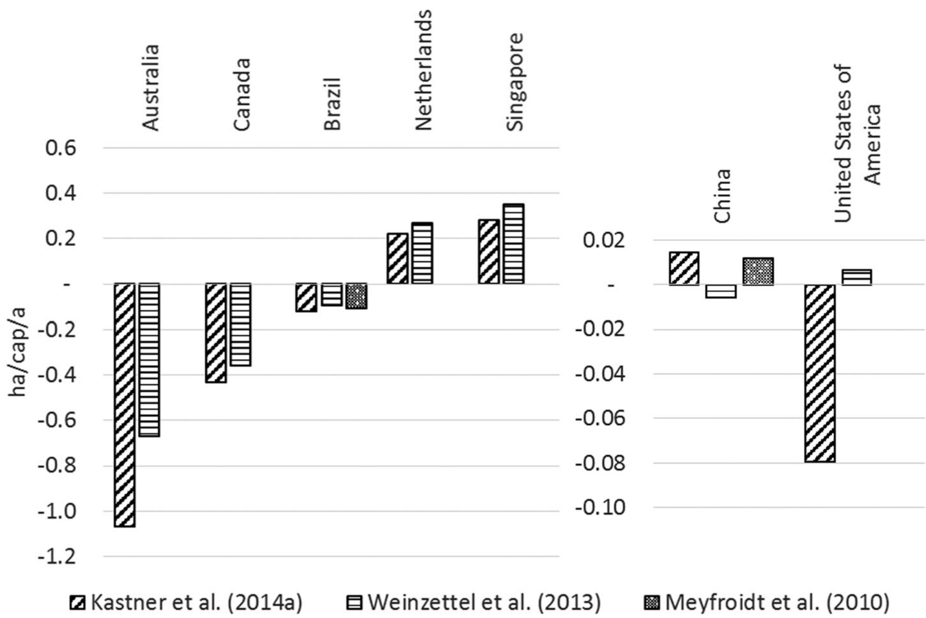

Figure 3 Net imports of upstream cropland in hectares per capita and year (ha/cap/a) in 2004 from three different studies for a selection of countries. Negative net-import values indicate that exports are greater than imports, that the country is a net exporter of upstream cropland. Please note the difference in scaling of the y-axes.

and colleagues (2014a) corrects for re-exports by tracing trade flows to their point of origin (Kastner et al. 2014a, 2014b).

Given the fundamental conceptual differences between biophysical and economic approaches and the large differences in results for China's cropland requirements described by Kastner and colleagues (2014b), a systematic divergence in the global studies' results might have been expected. This was not confirmed by a batch comparison (see figure 2). For the year 2004, a good overall fit ($R^2 = 0.88$) was found for net cropland imports as calculated by Weinzettel and colleagues (2013) using an EEIOA approach and the biophysical account of Kastner and colleagues (2014a). The initial biophysical modelling performed by Weinzettel and colleagues (2013) prior to the allocation via the IOTs partially explains this good fit of results. For 2007, the EEIOA-based results by Yu and colleagues (2013) did not match the biophysical results well ($R^2 = 0.51$). The x-y plots included in figure 2 show that the goodness of fit was strongly influenced by a small number of outliers. For the major net exporters of cropland, for example, the results as estimated by Yu and colleagues are generally slightly higher than those presented by Weinzettel and colleagues (2013), with the exception of Australia: Here, net exports are almost twice as large under the approach by Weinzettel and colleagues. In comparing the results generated by Kastner and colleagues (2014a) with those calculated by Yu and colleagues (2013), three outliers are highly visible in the x-y plot: The net cropland imports of the United Arab Emirates are more than twice as large under the approach used by Yu and colleagues (0.8 hectares per capita [ha/cap]) than under the approach of Kastner and colleagues (0.3 ha/cap). Namibia is a strong net exporter according to Yu and colleagues (–1.4 ha/cap of net imports) and a net importer in the study by Kastner and colleagues (0.1 ha/cap). The assessment of Australia differs again: –0.4 ha/cap of net imports (Yu et al. 2013) and –0.9 ha/cap (Kastner et al. 2014a). The results for the biophysical accounting approach applied by Meyfroidt and colleagues (2010) are similar to those generated for this small selection of 12 countries by all other approaches.

In the analysis presented in figure 2, high values of R^2 indicate strong positive linear correlation between the results, but not necessarily that the results themselves are identical. Whereas results are similar for many important biomass producers and consumers, they are diametrically opposed for China and the United States in some cases (see figures 3 and 4).

Both the biophysical and the EEIOA-based approach identify the large, biomass-extracting economies of Australia, Canada, and Brazil as net exporters of upstream cropland (represented by negative net imports in figures 3 and 4). The small, densely populated countries of the Netherlands and Singapore are net importers of cropland. Under the biophysical accounts (Kastner et al. 2014a; Meyfroidt et al. 2010), China is one of the most dominant global importers of upstream cropland. Owing to the large Chinese population, this translates into comparatively small per capita net imports. The EEIOA approaches (Weinzettel et al. 2013; Yu et al. 2013) characterize China as a net exporter of cropland (also see Kastner et al. 2014b). In theory, EEIOA applications are built on the assumption that the same price is paid for one unit of the same good or service throughout the economy (homogenous price assumption; Weisz and Duchin [2006]). In practice, this is often not the case. Chinese economic data reveal, for example, that "services of hotels and restaurants" are associated with high indirect land requirements (through monetary inputs from land-using sectors) and constitute a relevant export category (over one third

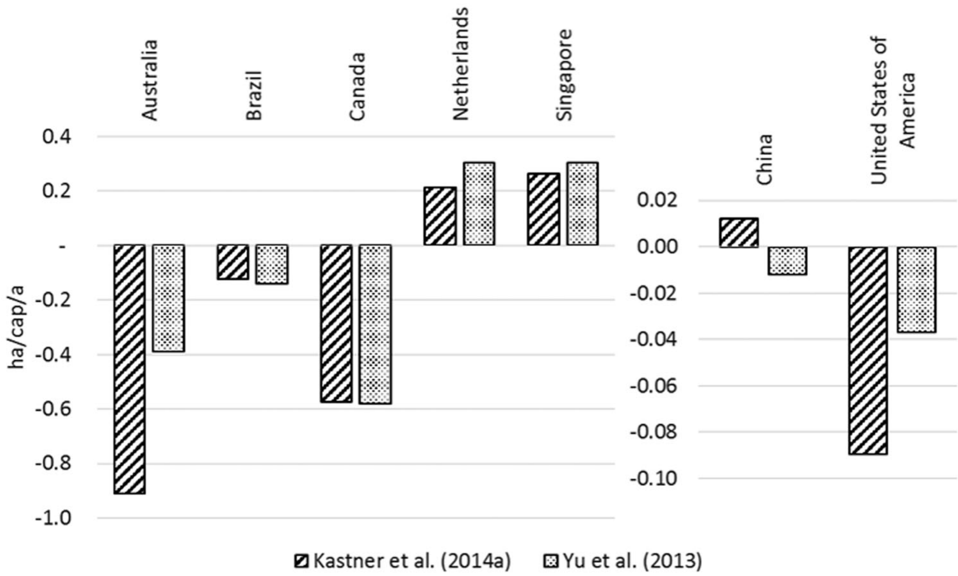

Figure 4 Net imports of upstream cropland in hectares per capita and year (ha/cap/a) in 2007 from two different studies for a selection of countries. Negative net-import values indicate that exports are greater than imports, that the country is a net exporter of upstream cropland. Please note the difference in scaling of the y-axes.

of the total monetary output generated in this sector). The latter is mainly composed of the expenditures of residents of other countries (i.e., of tourists).[5] If tourists were to pay a higher average price for a meal (in a hotel or restaurant) than a Chinese resident, then the calculated upstream land requirement associated with that meal would be higher, regardless of the amount of resources used in its production. This example illustrates a difference in allocation logic, compared to the biophysical approach, which does not consider exports of services. Though conceptual differences between biophysical and economic approaches in the accounting for indirect land requirements may have an impact in this case, the other country results suggest that it is not systematic.

The allocation of upstream land requirements by shares in biophysical consumption produces results that are comparable to those of the allocation by shares in monetary final demand (e.g., for Brazil), results that differ but reflect a comparable tendency (e.g., for Australia), and results that reflect opposing tendencies (e.g., for China). There are also cases in which the allocation according to an economic principle produces results that are comparable to biophysical allocation, but not to another application of the economic principle: The United States, another example of a large global producer and consumer of biomass, appears as a net exporter under the biophysical account and the EEIOA approach of Yu and colleagues (2013) (see figure 4) but are a net importer according to Weinzettel and colleagues (2013) (see figure 3).

Conclusions

It cannot be expected that the distribution of monetary inputs and the composition of monetary outputs in an economy correspond to the patterns of biophysical flows and land requirements (Hubacek and Giljum 2003). Considering the fundamental conceptual differences between the economic and the biophysical accounting approach, the degree to which results are comparable is remarkable. In their review of methods to quantify land-related leakage, Henders and Ostwald (2014) concluded that owing to limitations in the underlying data or assumptions required in the modeling process, all approaches are subject to fundamental uncertainties. All approaches additionally differ at the underlying conceptual level, especially in the principles by which they allocate responsibility for land use. Although results are often described using the same terminology and are directly compared, they must be interpreted as providing different types of information.

The two main groups of approaches that have been the object of this review, the economic and biophysical accounts, both produce results that are referred to as upstream land requirements or possibly as land footprints, embodied or virtual land. Even though they go by the same names, different studies have produced noticeably different results for these indicators (see Kastner et al. 2014b). Based on the assumption that China in 2007 could not simultaneously be a net exporter of 16.6 million hectares (Mha) of cropland (Yu et al. 2013) and a net importer of 16.5 Mha of cropland (Kastner et al. 2014a), uncertainties and errors in data and methods are being investigated. As with any new indicator (and with more established indicators, too, as the corrections to gross domestic product [GDP] show), these are necessary steps. Taking into account the conceptual differences between the approaches to calculating upstream land, it could also be beneficial to make slightly more precise statements about China's net imports of cropland:

a) Assuming that each economic activity in China and in all of its trade partners produces only one specific product that is sold at the same price throughout the production system and to different types of final demand, the monetary imports to final domestic demand in China in 2007 corresponded to monetary flows in the global production system to which a total of 25.9 Mha of cropland had been allocated. The monetary exports from China to final domestic demand in other countries in 2007 corresponded to monetary flows in the global production system to which a total of 42.5 Mha of cropland had been allocated.

b) After correcting biophysical trade flows for re-exports by assuming no consumer preference for domestically produced or imported goods, converting traded biomass-based products into primary crop equivalents and converting the latter into associated cropland requirements, biomass-based imports for Chinese consumption in 2007 corresponded to 22.4 Mha of cropland. Biomass-based exports from Chinese cropland in 2007 corresponded to 5.9 Mha of cropland.

In thus describing the results for upstream land requirements generated by Yu and colleagues (a) and Kastner and colleagues (b), it no longer seems that the characterization of China as a net exporter by one and a net importer by the other is necessarily a contradiction.

Accounting for upstream land requirements remains a highly important, but still insufficiently understood, research challenge, perhaps even the wildest frontier of sociometabolic research that currently exists. How upstream land requirements and thus responsibility for global land use are allocated is not only a choice of method, but also a political decision with significant impacts on the results. In translating trade flows into land requirements, product-specific differences in land productivity and land-use intensity must additionally be taken into account. By considering these underlying conceptual issues, a differentiated interpretation of upstream land requirements becomes possible.

Acknowledgments

The authors thank three anonymous reviewers for their valuable comments on an earlier version of this article. Funding by the EU-FP7 project VOLANTE (FP7-ENV-2010-265104), the European Research Council (ERC Starting Grant 263522 LUISE), and the Austrian Science Fund (FWF; project P20812-G11) is gratefully acknowledged. This study contributes to the Global Land Project (www.globallandproject.org) and to long-term socioecological research (LTSER) initiatives within LTER Europe (www.lter-europe.ceh.ac.uk/). Anke Schaffartzik is the recipient of a DOC-team fellowship of the Austrian Academy of Sciences (ÖAW).

Notes

1. In the scientific literature, upstream land requirements have also been denoted as “land footprints,” “embodied land,” and “virtual land.”
2. The year 2007 is chosen here because it is the most recent year for which the upstream land accounts presented in the following section provide results.
3. One of the economic modelling approaches contained in this review (Weinzettel et al. 2013) includes an initial biophysical accounting step, the results of which are then further allocated by means of monetary input-output tables.
4. Owing to lack of physical IOTs (Weisz and Duchin 2006), all current EEIOA approaches use monetary IOTs (Turner et al. 2007).
5. In order to ensure consistency with the system of national accounts, the system of environmental-economic accounting also applies the residence principle (also see UNSTATS 2014).

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Supporting Information

Additional Supporting Information may be found in the online version of this article at the publisher's web site:

Supporting Information S1: This supporting information provides an overview of methods to measure upstream land requirements, background information on EEIOA-based global studies, a comparison of the results of global studies, and a comparison of the results of regional and national studies.

Ecological Economics 128 (2016) 177–186

Contents lists available at ScienceDirect

Ecological Economics

journal homepage: www.elsevier.com/locate/ecolecon

Analysis

Consumption-based material flow indicators — Comparing six ways of calculating the Austrian raw material consumption providing six results

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ARTICLE INFO

Article history:
Received 24 March 2015
Received in revised form 15 January 2016
Accepted 2 March 2016
Available online xxxx

Keywords:
Raw material consumption
Material footprint
Material flow accounting
Input–output analysis
Sustainable resource use
Resource efficiency

ABSTRACT

Understanding the environmental implications of consumption and production depends on appropriate monitoring tools. Material flow accounting (MFA) is a method to monitor natural resource use by countries and has been widely used in research and policy. However, the increasing globalization requires the consideration of 'embodied' material use of traded products. The indicator raw material consumption (RMC) represents the material use – no matter where in the world it occurs – associated with domestic final demand. It provides a consumption-based perspective complementary to the MFA indicators that have a territorial focus. Several studies on RMC have been presented recently but with diverging results; hence, a better understanding of the underlying differences is needed. This article presents a comparison of Austrian RMC for the year 2007 calculated by six different approaches (3 multi-regional input–output (MRIO) and 3 hybrid life-cycle analysis-IO approaches). Five approaches result in an RMC higher than the domestic material consumption (DMC). One hybrid LCA-IO approach calculates RMC to be lower than DMC. For specific material categories, results diverge by 50% or more. Due to the policy relevance of the RMC and DMC indicators it is paramount that their robustness is enhanced, which needs both data and method harmonization.

1. Introduction

In recent years, economy-wide material flow accounts (EW-MFA in the following referred to as MFA; Eurostat, 2001a; Fischer-Kowalski et al., 2011) have been expanded towards capturing the global raw material use associated with a nation's final consumption. These consumption-based accounts complement the production-based MFA indicators and consider the upstream material requirements of traded goods in addition to the materials extracted domestically. The significance of these global upstream material flows is that they make up 40 to 400% of physical trade flows, depending on the resource and the estimation method (UNEP et al., 2015).The most prominent indicator which includes these upstream flows is raw material consumption (RMC) (Weinzettel and Kovanda, 2009; Muñoz et al., 2009; Schoer et al., 2012; Schaffartzik et al., 2014) also referred to as material footprint (Schoer et al., 2012; Tukker et al., 2014; Wiedmann et al., 2015). Such a consumption-based perspective on material use has also been called for in important policy papers (European Commission, 2011; OECD, 2011): "*This* [*the territorial MFA measure*], *however*, […] *does not deal with* […] *the potential shift of burden across countries. Because this provisional lead indicator only gives a partial picture, it should be complemented by a 'dashboard' of indicators* […] *seeking to take into account the global aspects of EU consumption.*" (European Commission, 2011, p. 20-21). For a long time, no empirical data was available to show how much consumption-based indicators differ from production-based accounts. For industrialized countries, however, the common understanding was that material-intensive production is increasingly outsourced to other countries, resulting in a reduction of domestic material use (Bringezu et al., 2003; Giljum and Eisenmenger, 2004; Muradian and Martinez-Alier, 2001). A consumption-based indicator of material use was expected to be higher than the production-based measure. In recent years, methods have been developed and results published for such a consumption-based material flow indicator (Muñoz et al., 2009; Weinzettel and Kovanda, 2009, 2011; Schoer et al., 2012; Bruckner et al., 2012; Wiebe et al., 2012; Wiedmann et al., 2015; Schaffartzik et al., 2014; Tukker et al., 2014; Giljum et al., 2015). Comparative studies aiming at evaluating differences in consumption-based accounts have been published for carbon and energy (Arto et al., 2014;

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http://dx.doi.org/10.1016/j.ecolecon.2016.03.010
0921-8009/© 2016 Published by Elsevier B.V.

Inomata and Owen, 2014; Moran and Wood, 2014; Owen, 2013, 2015; Owen et al., 2014; Steen-Olsen et al., 2014a; Weinzettel et al., 2014; Wiedmann, 2009a; Wiedmann et al., 2007), and for land (Bruckner et al., 2015; Kastner et al., 2014; Schaffartzik et al., 2015). A comparison of different RMC results was published for the EU (Schoer et al., 2013), which compares two calculation methods, and for a set of countries in a more recent report presented by the OECD, which compares results generated by multi-region input–output (MRIO) models and a hybrid LCA-IO approach (OECD, 2015). With this research, we go further and deeper by providing a consistent comparison of two methods and six datasets to calculate RMC for one specific country — Austria.

The method used to account for all materials used in a national economy is economy-wide material flow accounting (EW-MFA, in the following referred to as MFA; Eurostat, 2001b; Fischer-Kowalski et al., 2011). MFA is part of the environmental accounts (Eurostat, 2014; United Nations, 2014) which are a satellite to the system of national accounts. MFA has been implemented in the European statistical reporting (European Parliament and the Council, 2011) and is included in the United Nations System of Environmental-Economic Accounting, SEEA (United Nations, 2014).

Domestic material consumption (DMC) is the most prominent indicator in MFA and accepted as a headline indicator for resource use and resource efficiency (BM LFUW, 2012; European Commission, 2011). DMC is calculated as the balance of materials domestically extracted (DE, domestic extraction) plus imports minus exports (Eurostat, 2012; Fischer-Kowalski et al., 2011). Trade flows are accounted for with the mass they have upon crossing a national border. Usually, traded goods are at different stages of processing, and the physical mass of traded commodities differs from the mass of materials extracted to initially produce them. Economies specialized in the export of highly processed commodities may use imported primary or secondary products as material inputs into this production or can use materials which were domestically extracted and processed. If the latter type of production for export is dominant, DMC will be higher because the mass of domestically extracted raw materials is larger than the mass of imported secondary goods. The indicator DMC illustrates the domestic (in the sense of territorial) material use of a national economy comprising all material flows entering an economy (either through imports or domestic extraction activities) and remaining there (i.e. not exported). These materials may be used and turned into waste and emissions in the production process, transformed into stocks (buildings, infrastructure, or durable goods), or they may serve final consumption. Materials included in DMC become waste or emissions at the end of their use-phase so that DMC can also be interpreted as an indicator of waste potential (Weisz et al., 2006). DMC has a strong focus on the national economy and its production structure and is closely linked to national policy and legislation.

As globalization intensifies, national resource productivity may change – for better or for worse – depending on the role of trade for consumption rather than on the level of consumption and a consumption-based measure of material use is urgently needed. In other words, indicators are required that assess the materials globally required to satisfy domestic final demand, and provide information on the total material extraction, no matter where it occurs, which directly and indirectly satisfies this final demand. In MFA, the information on upstream material requirements of traded goods (i.e. the materials used to produce traded goods) is provided by the raw material equivalents (RME; Eurostat, 2001a) of imports and exports and is then included in an indicator raw material consumption (RMC; Weinzettel and Kovanda, 2009; Muñoz et al., 2009; Schoer et al., 2012; Schaffartzik et al., 2014; Eurostat, 2001b), which is also termed material footprint (Schoer et al., 2012; Tukker et al., 2014; Wiedmann et al., 2015).

To date, calculations quantifying RMC use two main approaches: 1) a coefficients approach using material coefficients from life cycle inventories (LCI) to calculate upstream material requirements. In the context of MFA, this approach was initially developed by the Wuppertal Institute in the 1990s, and applied mainly for the calculation of the total material requirement (TMR) indicator (Bringezu et al., 2004; Bringezu and Bleischwitz, 2009; Dittrich et al., 2012); 2) an environmentally extended input–output analysis (EE-IOA) approach employing information on the monetary structure of production and final demand including trade to allocate direct as well as indirect upstream material requirements to final demand. EE-IOA has been applied to various resource use domains to account for upstream carbon and greenhouse gas emissions (Munksgaard and Pedersen, 2001; Peters, 2008; Hertwich and Peters, 2009; Davis et al., 2011), land requirements (Weinzettel et al., 2013; Yu et al., 2013), water (Daniels et al., 2011; Hoekstra and Chapagain, 2006; Hoekstra and Hung, 2005), a compound measure of Ecological Footprint (Ewing et al., 2012; Galli et al., 2012, 2013; Moran et al., 2013; Weinzettel et al., 2014; Wiedmann, 2009a), pressure on biodiversity (Lenzen et al., 2012b), as well as in recent years also for material flows (Muñoz et al., 2009; Weinzettel and Kovanda, 2009; Wiebe et al., 2012; Bruckner et al., 2012; Kovanda and Weinzettel, 2013; Schoer et al., 2013; Wiedmann et al., 2015; Schaffartzik et al., 2014; Tukker et al., 2014; Giljum et al., 2015).

Within EE-IOA models, the calculation of upstream requirements of imports is a challenge because it requires information on the material use, production structures, and international trade relations of all trade partners. EE-IOA-based RMC accounts use different approaches to solve this problem: Single-region IO Approaches (SRIO; Miller and Blair, 2009; Muñoz et al., 2009; Tukker et al., 2013b; Wood et al., 2009) apply the RME multipliers of domestic production derived from the IO model to all imports (commonly termed the 'domestic technology assumption', DTA). Other studies try to overcome the limitations of the DTA (which cannot accurately reflect the production structures of other countries) by combining the IO model with material coefficients based on data from LCI databases; this approach is commonly termed hybrid LCA-IO approach (see Suh, 2004) and we will stick to this term in this article. This hybrid LCA-IO approach accounts for those imported products that are not produced in the observed economy and thus are not represented in domestic IO multipliers with the help of material coefficients. Multi-regional input–output models (MRIO; Tukker and Dietzenbacher, 2013; Wiedmann, 2009b; Wiedmann et al., 2011) link monetary IO tables from many economies or regions (the number of economies varies between MRIO models with mostly a relatively large 'rest of the world' aggregate) and cover the whole world-economy. Material extraction required to produce the traded goods and services is allocated to the country of final demand via the monetary IO structure. To date, several studies have been published in which upstream material requirements were calculated. Some of them present results for single countries or regions (Muñoz et al., 2009; Weinzettel and Kovanda, 2009; Schaffartzik et al., 2014), while others calculated RMC or material footprints for a large number of countries or aggregate regions covering the whole world (Bruckner et al., 2012; OECD, 2015; Tukker et al., 2014; Wiedmann et al., 2015).

This article presents an application of six different calculations for RMC to the same country, Austria, and for the same year, 2007, and presents the range of results that can currently be obtained. The approaches cover two of the three methods discussed above: three calculations follow the hybrid LCA-IO and three the MRIO approach. The calculations were done for one single country, i.e. Austria, because hybrid LCA-IO models are only available for single nations or regions. MRIO results on the other hand are available for the whole globe covering a set of countries and regions. In order to compare both methods we limited the calculations to one country. Austria was chosen as a case-study country because the authors have a national IO model and detailed MFA data at hand for Austria (for model specifications see Schaffartzik et al., 2014). In addition

Austria's material flows have been studied extensively, making it a prime country for methodological comparisons (e.g. Eisenmenger et al., 2011; Schaffartzik et al., 2014; Wenzlik et al., 2015). Following a short description of the different approaches, the results are presented and discussed for total RMC and four aggregate material categories. Finally, we compare RMC results to DMC and assess the relevance of both indicators for policy application.

2. Methods Applied to Calculate Austrian RMC

Most RMC calculations are based on monetary input–output tables, which depict the structure of the economy as intermediate inputs among industries and as final demand (including capital investment and exports) for the output of these industries. From these input–output tables, the so-called Leontief inverse, a matrix of multipliers which reflects the inputs required directly and indirectly to produce one unit of output to final demand (Leontief, 1970), is calculated. The input–output model is extended with vectors for raw material inputs to each economic industry to calculate the material use associated with final demand. The following approaches were applied in this research:

1. *Hybrid LCA-IO approaches* use national IO tables but integrate life cycle inventory based material coefficients to provide multipliers for imported products which are not produced domestically and thus not represented adequately in the national IO structure. The calculation of the RME of imports is usually done with a single LCI dataset which does not allow for reflecting the fact that different parts of a product supply chain may pass different countries with different material efficiencies of production. For this research, we used three hybrid LCA-IO approaches:
   - in the *SEC* approach we used a model developed by Schaffartzik et al. (2014), which uses the Austrian IO table and integrates coefficients from the GEMIS database (Öko-Institut, 2009) to cover the extraction and processing of materials for metal production (iron, copper, aluminum), fertilizer production, and petroleum and gas extraction (Schaffartzik et al., 2014).
   - in the *Eurostat* approach we used RME coefficients for imports and exports provided by Eurostat (Schoer et al., 2012), which are derived from a detailed European IO model (166 × 166 industries), augmented with specific life cycle inventory based material coefficients for metal products and products from fossil fuels. These coefficients represent the average European trade structure. In our calculation, the Austrian imports and exports were multiplied with the Eurostat coefficients (at the level of single goods or products) resulting in the RME of imports and exports. Adding the RME trade balance to the Austrian domestic extraction (DE) results in the RMC.
   - in the *Eurostat-SEC* approach we combined the two approaches above. The Eurostat RME coefficients are used for imports and the resulting RME of imports are integrated in the environmental extension vector of the SEC model and then the SEC IO model was run to calculate RMC. By that, the calculation of the RME of exports was based on the specific Austrian IO structure and not on European averages as in the Eurostat approach.
2. *Multi-regional input–output (MRIO) approaches* were developed to better represent foreign production structures (Tukker and Dietzenbacher, 2013; Wiedmann, 2009b; Wiedmann et al., 2011, 2007). An MRIO framework integrates domestic IO tables for all countries (or country groups) with trade matrices linking all countries. MRIO models allow for a complete representation of global supply chains across countries, which is not the case in the aforementioned co-efficient approaches. An important attribute of MRIO models is that they are additive and closed at the global level, i.e. total global DE equals total global RMC. Tukker and Dietzenbacher (2013) identified five dominant global MR EE IO models of which four have been used in this comparison, partly compiled by teams represented by the authors of this paper:
   - The World Input–Output Database (WIOD, 2013) was developed in an FP7 European research project (Dietzenbacher et al., 2013; Timmer et al., 2012) and the publicly available version has a resolution of 35 industries and 40 countries.
   - the Global Trade Analysis Project (GTAP, 2013) in the version of GTAP v8 (Narayanan et al., 2012) is the basis for an IO model used by Giljum et al. (2015) which offers a high disaggregation for primary industries and provides data for 109 individual countries and 20 country groups.
   - EXIOBASE (2013) is a detailed MRIO model developed in two FP7 European research projects (Tukker et al., 2013a; Wood et al., 2015). EXIOBASE distinguishes 48 countries/regions, 163 industries and 200 products. The EXIOBASE 2.0 version of June 6, 2013 was used in the calculations. EXIOBASE is initially compiled as a global MR EE Supply and Use Table (SUT) and then is transformed into an IOT (Tukker et al., 2013a; Wood et al., 2015). For this analysis, a product by product IOT was used.

In addition to the above mentioned MRIOs, we will also show RMC results from the Eora model (Eora, 2014; Lenzen et al., 2012a, 2013) which integrates the national input–output data of 187 individual countries at a high level of resolution. The different national classifications and levels of sectoral aggregation are bridged with Eora-specific correspondence tables. In contrast to the MRIO models mentioned above, Eora uses different MFA data for Austria in the environmental extension. Thus, results for Eora cannot be directly compared but can serve as additional data point illustrating the range of possible results for Austria.

The Global Resource Accounting Model (GRAM; Bruckner et al., 2012), another MRIO model, was not used in this comparison, because it's sectoral aggregation is very high and thus comparable to WIOD. Additionally, GRAM is not generally accessible for free, whereas WIOD is open to the public. For these reasons we used WIOD as a representative for a highly aggregated MRIO.

The hybrid LCA-IO approaches are very different from MRIO approaches, which is expected to have a significant effect on results. Even though the approaches are very different, a comparison is necessary because both approaches are used to calculate the same indicator, i.e. RMC, and derived resource efficiency indicators. Information on the difference in results also across different calculation methods is thus needed.

More information on the different approaches and models is made available in the Supporting Information.

Table 1 summarizes the main characteristics of the approaches used (including Eora). The Eurostat-SEC approach is not listed because it combines the characteristics of the SEC and Eurostat approaches.

Mining industries are aggregated differently in IO tables: the Austrian IO table aggregates mining of oil, natural, gas, and ores and thus materials like crude oil, extracted in large amounts (1 million tons) and at low prices, and like gold, used in small amounts measured in grams per capita and at high prices, in one industry. The extraction of coal and peat and the extraction of sand and stones form the other two mining sectors, respectively. WIOD aggregates all mining of abiotic materials in one industry. The other MRIO-based approaches report fossil fuel energy carriers, metals, and non-metallic minerals in separate industries, sometimes even more than one for each material category (see Table 1). Results presented below have to be understood too in light of these differences in aggregation where higher aggregation is considered to cause less plausible results (Bouwmeester and Oosterhaven, 2013; de Koning et al., 2015; Steen-Olsen et al., 2014b).

Table 1
Main characteristics of the datasets applied in calculating the RMC of Austria.

		SEC	Eurostat	EXIOBASE	WIOD	GTAP	Eora
		Schaffartzik et al. (2013)	Schoer et al. (2012)	Tukker et al. (2013c)	Dietzenbacher et al. (2013), Timmer et al. (2012)	Giljum et al. (2015)	Lenzen et al. (2013), Lenzen et al. (2012a, 2012b), Wiedmann et al. (2015)
Approach		Hybrid LCA-IO	Hybrid mixed units LCA-IO	MRIO	MRIO	MRIO	MRIO
Regional resolution (no. of countries)		1 + DTA & LCI	1 + DTA & LCI	43 + 5 RoW	40 + RoW	109 + 20 regions	186
Sectoral resolution		59	166	163	35	57	25–510
Resolution of primary sectors for allocation of materials	Biomass	3	16[a]	17	1	12	2–40
	Fossil fuel	1	10	4		3	1–7
	Metal ores	1[b]	18	8	1	3	1–8
	Non-metallic minerals	1	5	3		1	1–8
% of non-metallic minerals directly allocated to construction sector		50%	50%	0%	50%	50%	0%
Resolution of material extension (no. of material categories)		54	12	48	12	18	35

Legend: DTA = domestic technology assumption; RoW = rest of the world.Eora is only used as a reference point from literature, however, for a better understanding we included the methodological details of Eora also in this table.
[a] 16 of 20 are non-zero.
[b] Due to data confidentiality, the 2007 Austrian IO table has an aggregated mining sector for oil, natural, gas and ores, and a sector for coal and peat.

2.1. Material Flow Data Used for the Calculation

In RME accounts, the monetary IO models are extended by material extraction data (in the hybrid LCA-IO models also with material imports) in order to calculate the RME of traded goods. Data on material extraction are part of the MFA framework, which include all materials extracted within a particular country as well as all physical imports and exports. Accounting methods and system boundaries are standardized and closely match the conventions of the system of national accounts (Eurostat, 2001a, 2012; United Nations, 2014).

In all models employed for the calculation of RMC we used exactly the same material extraction data for Austria which is sourced from Statistics Austria (Statistics Austria, 2013). The MRIO models additionally contain DE data for all other countries or regions in the world, which is derived from www.materialflows.net (SERI, 2013). With regard to trade, the three hybrid approaches used physical trade data from the Austrian MFA (Statistics Austria, 2013). The MRIO approaches rely on monetary bilateral trade data to link national or regional IO tables (SI for details).

3. Raw Material Consumption (RMC) in Austria: Results and Discussion

Austria's physical imports are higher than its exports, making the country a net-importer. The physical trade balance (PTB = imports − exports; no upstream flows are included) amounted to 4 tons per capita (t/cap) in 2007; 70% of net-imports were fossil energy carriers, 20% were metal-based products. When upstream material requirements of trade are taken into account, net-imports increased considerably in all calculation approaches except for the Eurostat approach (Fig. 1). The raw material trade balance (RTB = RME imports − RME exports = RIM − REX) is around twice as large as the PTB at 7.5 t/cap in the SEC model, 8.4 t/cap in WIOD, 9.1 t/cap in GTAP, and 9.6 t/cap in EXIOBASE. In the Eurostat approach, the RTB is only 0.1 t/cap, which is lower than the PTB. This results from negative trade balances for biomass and non-metallic minerals (see Fig. 1). The Eurostat RTB being lower than the PTB implies that Austria is, considering all upstream requirements, supplying as many resources to the world as it consumes, suggesting that Austria's imports are less material intensive than its exports.

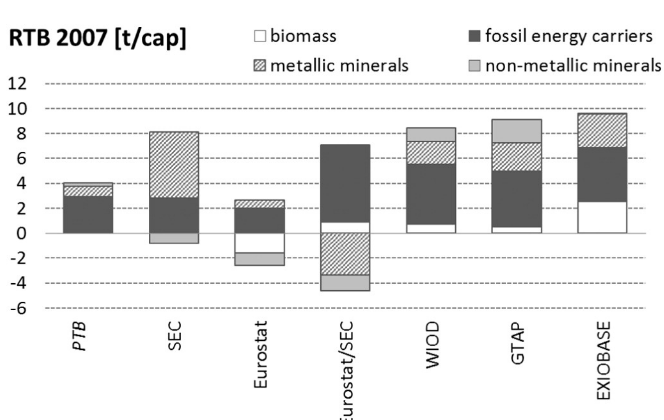

Fig. 1. Austrian raw material trade balance (RTB) in 2007 in tons per capita (t/cap).

The coverage of RME flows usually differs in hybrid IO-LCA and MRIO approaches: Just as in MFA, the RIM and REX calculated in hybrid LCA-IO include all biophysical imports and exports entering or exiting a country no matter whether these goods are destined for domestic final demand or for intermediate use. MRIO-based approaches, on the other hand, usually report as imports the raw material equivalents of domestic final demand which is not met by domestic production. Imports used in the production of goods and services for export are not included in the RIM, nor the REX of the country. In MRIO calculations, the REX of Austria only comprises domestic extraction of materials supporting production of exported goods. RIM and REX in MRIO-based approaches can therefore be expected to be lower than those from MFA or hybrid IO-LCA, which is shown in Fig. 2. The figure also shows that Austrian export flows from MRIO mostly comprise biomass and construction minerals; metals and fossil fuels are not extracted in Austria in significant amounts. In trade balances and also in RMC the different perspectives on imports and exports balance out and thus these indicators (RMC and RTB) can be compared between the different approaches.

3.1. Biomass Materials

The physical amounts of biomass materials exported and imported by Austria are similar and direct net-imports are negligible at 0.1 t/cap in 2007. In Austria, agriculture and forestry are economically important. Extensive livestock systems in mountainous regions and the relative amount of grazed biomass (in t/GDP) is slightly higher than in most other European countries. In addition, Austria imports significant amounts of semi-manufactured biomass products, especially wood-based products and high-energy

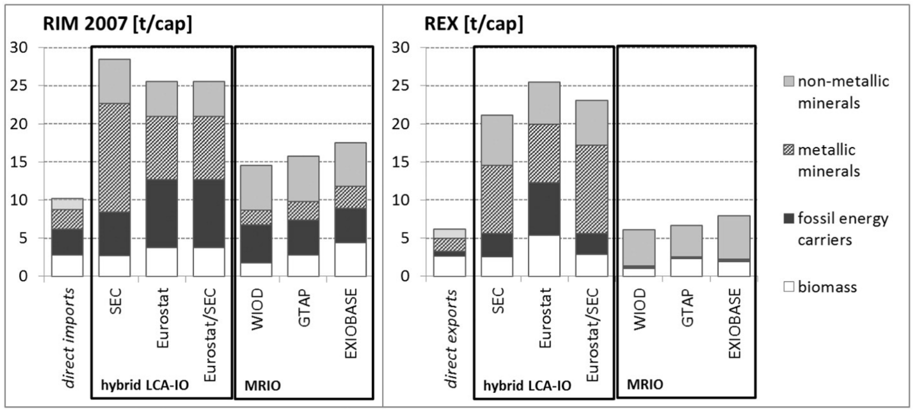

Fig. 2. Raw material equivalents of imports (RIM) and of exports (REX) for Austria in 2007 in tons per capita (t/cap).

animal feed, which are further processed in paper production and livestock systems, and then exported (Eisenmenger et al., 2011).

The MRIO approaches all result in a biomass RTB that is higher than the PTB. EXIOBASE and GTAP have the highest disaggregation of biomass producing industries and products in their models (EXIOBASE differentiates 16 industries, GTAP 12 industries). However, the sequence of RTB results from low to high does not follow the disaggregation level of biomass-processing industries or biomass products (likewise described by Steen-Olsen et al. (2014b) for CO_2 emissions): GTAP (0.5 t/cap) results in an RTB only slightly higher than PTB, whereas EXIOBASE delivers the highest biomass RTB (2.6 t/cap). The result from WIOD lies in between with 0.7 t/cap.

The Eurostat/SEC approach calculates biomass RTB of 0.9 t/cap, the SEC hybrid approach arrives at 0.1 t/cap of biomass RTB, which means upstream flows of imports and exports are of similar size. The Eurostat calculation results in a negative RTB value (−1.6 t/cap), turning Austria into a net-exporter of biomass.

The Eurostat approach result in a biomass RTBs which is very different from those calculated under the other approaches that it raises the question about whether the Austrian production structure in agriculture and forestry is possibly not well represented by average European multipliers. Higher RIM and significantly higher REX in the Eurostat than the SEC approach (see Fig. 2) may point to Austrian production structures being less input-intensive than the European average in monetary terms, translating to a lower material intensity as compared to average European production. The extensive livestock systems in Austria's mountainous regions are an example of this. The lack of disaggregation in the Austrian IO table with regard to biomass producing industries (agriculture includes livestock farming and is distinguished only from forestry and fishing as biomass-extracting industries) is another possible reason for the differences in results. In comparison to the EU average, Austria might be producing meat in extensive farming using a high amount of grazing while crop production might be less intensive than in other European countries and be associated with lower upstream requirements. Due to the high aggregation of primary sectors, however, both types of production are calculated to have the same upstream requirements per unit of output to final demand. Following our previous example, if meat with high upstream material requirements is mainly exported and crop products with relatively lower upstream material requirements meet domestic final demand, the material requirements of the former would be under- and of the latter over-estimated using average European multipliers.

A comparison between biomass RIM or REX and the respective trade flows from MFA – which is for the above mentioned reasons only possible for the hybrid IOs – reveals that the SEC model calculates RIM and REX to be lower than the respective direct imports or exports. Mathematically, this is possible: If the product of the Leontief multiplier (L) and the price (exports[$]/exports[kg]) is smaller than 1, the RME of the exports will be smaller than the exports themselves. However, practically this is impossible because directly traded biomass goods by definition are included in the RME of imports and exports and therefore RIM and REX cannot be lower than direct trade. Also Merciai and Heijungs (2014) showed that a calculation of material footprints with input–output tables can lead to a violation of the mass balance principle. Obviously, there are still issues to be solved in the application of physical flows to monetary IO models.

3.2. Fossil Energy Carriers

Austria does not extract significant amounts of fossil energy carriers but satisfies its demand through imports with a positive physical trade balance of 2.8 t/cap. All MRIO approaches calculate the RTB of fossil energy carriers to be significantly higher than the PTB and higher than the RTB from hybrid LCA-IO approaches. EXIOBASE calculates 4.3 t/cap of fossil energy carrier RTB, GTAP 4.5 t/cap, and WIOD results in 4.8 t/cap. The SEC and the two Eurostat approaches result in fossil energy carrier RTBs lower than the PTB (Eurostat: 2 t/cap, SEC: 2.7 t/cap). The Eurostat/SEC approach exceeds all other results (6.2 t/cap).

In the SEC model, fossil energy RIM are 1.7 times higher than imports; in the Eurostat approach RIM are 2.6 times larger than imports. Fossil energy REX are 5 times higher than exports under the SEC approach and 11 times higher under the Eurostat approach. The REX calculated with average European coefficients (Eurostat coefficients) results in 7 t/cap which is more than double the REX of the SEC approach (3 t/cap). The higher sectoral aggregation in the SEC approach may cause this difference. Furthermore, Austria has a higher share of hydro-power in domestic electricity production compared to other European countries and no nuclear power plants. The average European coefficients might not capture Austrian energy use structure well. In the SEC approach, upstream requirements of imports and exports are similar, and thus the RTB of fossil energy carriers of 2.7 t/cap is only slightly lower than the PTB (2.8 t/cap). The RTB in the Eurostat and the Eurostat/SEC approaches is lower than the PTB, implying that exports are more fossil fuel intensive than imports to Austrian final demand.

3.3. Metallic Minerals

Austria does not extract metal ores in significant amounts but relies on imports of metal goods. All approaches identify Austria as a net-importer of metal goods. The aggregation of Austria's low-level metal mining activities in only one sector does not represent the flows of metals and waste rock through the economy well. The hybrid IO-LCA models try to achieve more detail by using LCA coefficients as multipliers for imported metals.

The SEC approach (integrating 32 LCA coefficients for metals and metal products) results in the highest metal RTB of 5.3 t/cap. The Eurostat approach uses around 2500 LCA coefficients for metals and delivers an RTB result (0.7 t/cap) close to the direct PTB. The combined Eurostat/SEC approach results in the lowest RTB (−3.3 t/cap). The MRIO approaches deal with metals differently: WIOD, which aggregates metals, minerals, and fossil energy carriers into one sector, results in a metal RTB of 1.8 t/cap. GTAP takes a more detailed perspective and disaggregates the metal mining industry to three different industries. GTAP results in a metal RTB of 2.3 t/cap. With eight industries EXIOBASE has the most highly disaggregated IO table and calculates RTB to be 2.7 t/cap. The difference between the highest and the lowest estimated metal RTB is 8.6 t/cap.

The comparatively high RTB for metals in the SEC approach suggests that the applied LCA coefficients may lead to overestimation of the RME of imports. The application of LCA-based coefficients to the macro level is often criticized for introducing potential for double-counting and for truncation of upstream requirement chains due to system boundary definitions (Suh et al., 2010; Reap et al., 2008; Suh et al., 2004). Errors in the process of truncation occur when an analysis only follows up a finite number of supply-chain stages (e.g. due to limited resources) and then omits further upstream contributions to the life cycle of the functional unit. Such truncation errors can be severe for applications where significant impacts lie deep within the supply-chain network (see Lenzen, 2000). Double-counting occurs in cases where a functional unit is appraised that in itself is the supplier of inputs for other functional units. For example, if a power plant is analyzed and the fuel used by that power plant added to its LCA inventory, then this fuel component is double counted in applications where the power plant appears as a part of the upstream supply chain (see Lenzen, 2009). Double-counting also occurs, when one functional unit supplies more than one product (co-production). In that case problems can occur in making a choice with regard to the allocation of inputs to both or proportionally between products (based on economic value, weight, and energy content or other).

Finally, the aggregation of all mining of all metals into one industry is likely to mean that the average distribution of all the outputs of this industry (including oil and gas) is unlikely to be equally appropriate for all types of metals (Bouwmeester and Oosterhaven, 2013). A detailed discussion of this price inhomogeneity is provided by Weisz and Duchin (2006).

3.4. Non-Metallic Minerals

Non-metallic minerals cover materials such as sand, stones, and clays used for construction of buildings and transport infrastructure, which are extracted and used in bulk quantities, as well as minerals for fertilizer production or diamonds, which are used in small amounts at much higher average prices. Because of their comparatively low price per unit of mass, bulk construction minerals are hardly traded. They made up 50% of domestic extraction in the European Union at the beginning of the millennium (Weisz et al., 2006). Non-metallic minerals form part of the upstream requirements of many traded products through the use of infrastructure and buildings in the production and transport of goods and the use of construction minerals therein. This use of construction materials is reflected in hybrid IO-LCA approaches through Life cycle inventory based material coefficients but not in MRIO-based approaches; in the latter, expenditure on construction minerals corresponds to a capital investment and is therefore reported as a category of final demand. Other non-metallic minerals appear as upstream requirements in all accounting approaches. Fertilizers, for example, are an important upstream input into agricultural production.

In the case of Austria, half of the physical extraction of construction minerals is carried out by the construction sector (Eisenmenger et al., 2011; Milota et al., 2011; Schaffartzik et al., 2014). Therefore, the extraction of non-metallic minerals is allocated equally to the mining of sand and stones and the construction sector in all approaches except EXIOBASE and Eora, which follow a standard allocation of domestic extraction to primary industries (Table 1).

The non-metallic mineral RTB especially reflects the fundamental difference in how physical inputs into building- and infrastructure-stocks are accounted for under the hybrid IO-LCA and the MRIO-based approaches. Although the former approaches do account for these inputs within the production structure, assumptions have to be made in the use of life cycle inventory based material coefficients which significantly affect the results. Most significantly, choices must be made as to how construction mineral inputs into stocks are distributed both over time and for co-produced products to all of the outputs of each sector. Austria is a net-importer of non-metallic minerals with a PTB of 0.3 t/cap. The three hybrid approaches change Austria to a net-exporter, −0.8 t/cap in the SEC approach, −1 t/cap in Eurostat and −1.3 t/cap in the Eurostat/SEC approach, the latter being the lowest results for RTB. EXIOBASE results in a balanced RTB (0 t/cap). The two other MRIOs result in a positive RTB with Austria being a net-importer: WIOD results in 1.1 t/cap and GTAP 1.9 t/cap.

3.5. Raw Material Consumption

The most widely used indicator in standard MFA is domestic material consumption (DMC = domestic extraction + PTB). The Austrian DMC was 25 t/cap in 2007. By replacing direct trade flows with RME, the indicator raw material consumption (RMC) (Muñoz et al., 2009; Weinzettel and Kovanda, 2009; Schoer et al., 2012; Schaffartzik et al., 2014), also referred to as material footprint (Schoer et al., 2012; Tukker et al., 2014; Wiedmann et al., 2015), is obtained. Among the six approaches investigated here, the lowest RMC of 21 t/cap resulted from the Eurostat approach, while the highest RMC of 29.9 t/cap resulted from the EXIOBASE approach (Fig. 3). The range of RMC results (9 t/cap between the highest and the lowest RMC result) was thereby larger than any of the differences to the DMC (between −4 and +5 t/cap).

GTAP and EXIOBASE arrive at an RMC of 30 t/cap, the SEC approach estimates 28.4 t/cap, and WIOD 29.4 t/cap (see Fig. 3). In comparison, Eora, which uses a slightly lower MFA input (100 Mt instead of 170 Mt, i.e. −40%, mostly non-metallic minerals), calculates RMC to be even higher, i.e. 33 t/cap. Higher sectoral disaggregation, i.e. a larger number of sectors explicitly represented in the IO table, has been shown to enhance the interpretability of results (Miller and Blair, 2009; Lenzen, 2011; Bouwmeester and Oosterhaven, 2013; Steen-Olsen et al., 2014b). WIOD uses the smallest number of sectors, followed by GTAP and then EXIOBASE, while Eora combines national IO tables with varying sectoral resolutions (Table 1). This hierarchy of sectoral detail does not directly translate to the same sequence in RMC results, where the RMC of GTAP and EXIOBASE is lowest, WIOD ranges in the middle, and Eora results in the highest RMC (see also Steen-Olsen et al., 2014b for CO_2 emissions).

With the exception of the Eurostat result all approaches yield RMCs that are higher than DMC. But not enough information is currently available to verify whether the highest RMC estimate is more appropriate for Austria as an economy dependent on net-imports of many materials with high upstream requirements or whether an RMC only slightly higher than DMC better reflects the high export orientation of the Austrian economy in which revenues from exports account for over 50% of GDP.

Raw material equivalents are calculated to attribute global material extraction to the final demand which it ultimately satisfies. By that, RMC measures upstream material requirements, no matter where they occur, required to satisfy domestic final demand; this opens the perspective towards the global level and beyond the realm of the

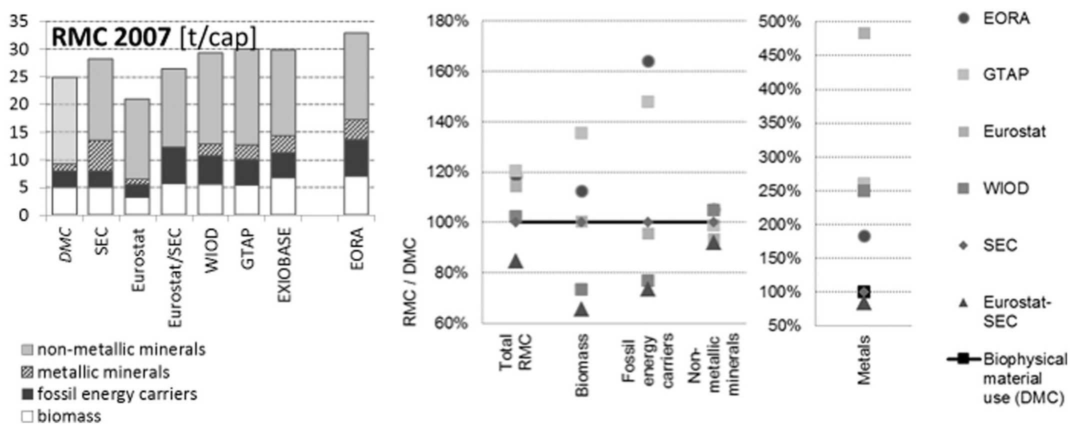

Fig. 3. Austrian raw material consumption (RMC) in 2007 in tons per capita (t/cap) (left) and Austrian RMC as a share of DMC (RMC/DMC; right).

national policy. The DMC indicator, on the other hand, represents (largely) a production perspective, accounting for all material used and transformed within national boundaries, minus physical exports.

Both DMC and RMC have been related to GDP in order to report resource efficiency, i.e. the amount of GDP generated per unit of material. With the different perspectives of DMC and RMC, both relations to GDP provide different messages, which still need to be better understood (conceptually and with regard to the underlying method). While GDP includes final domestic demand and revenues from net exports in monetary terms, DMC includes domestic extraction and physical net-imports to reflect that in trade material and money flow in opposite directions, and RMC includes all global material extraction directly and indirectly required to meet monetary domestic final demand. Both the DMC and the RMC address important but different aspects of resource use, and neither of these indicators is a perfect counterpart to GDP.

With regard to policy relevance, the two indicators can serve different purposes. DMC has a focus on the national, in the sense of territorial, processes and thus can be used for analyzing the domestic economic production structure and its efficiency in satisfying domestic final demand. National policy or legislation can directly address DMC. RMC broadens the perspective towards including global resource use issues. Material use activities outside the national economy cannot be directly addressed through national policies but only indirectly through international trade law, and to a limited extend via regulations such as safety or eco-design standards and, voluntary measures along the supply chains (for example against child labor or illegal logging). The focus on the entire supply and use chain which the RMC indicator entails caters well to policy and consumer preferences, especially but not only in the Global North, to consume more sustainably.

4. Discussion and Conclusions

In this article we presented results from the calculation of RMC for Austria for the year 2007 based on six different approaches, i.e. three datasets based on hybrid MF-IOA, and three datasets based on MRIO analysis. Results from the MRIO Eora were additionally added as further reference point from literature. These approaches represent the most widely applied and recently published methods and models to calculate RMC (or material footprint). RMC results range from 21 t/cap up to 30 t/cap or even 33 t/cap in Eora. With a variation of 9 t/cap between the lowest and the highest result for RMC (9 t/cap make up for 30–40% of RMC or DMC) the difference is higher as compared to any difference between the DMC (25 t/cap) and the RMC. This puts the results for RMC closer to DMC as one would expect for a highly industrialized country. An analysis on the level of four material categories reveals that for two material categories, i.e. biomass and non-metallic minerals, not only the level but also the sign of the trade balance changes, turning Austria from a net-importer to a net-exporter.

The calculations based on the SEC hybrid IO-LCA approach as well as the MRIOs WIOD, GTAP, and EXIOBASE show highest correspondence (only 1.6 t/cap difference between the highest and the lowest result). However, they still differ with respect to the composition along the four material categories. In the SEC approach metal ores make up for 70% of RTB, whereas in the three MRIOs 50–60% of RTB are fossil fuels. The calculations based on the Eurostat coefficients deliver results that are significantly lower than the other results, both for the RTB as well as the RMC. A comparison to Eora reveals that Eora delivers the highest estimates of RTB and RMC. The high RTB for non-metallic minerals is standing out not only in total mass compared to the other material categories but also compared to the other approaches.

Our calculations made it possible, for the first time, to directly compare RME-based indicators such as the Raw Material Consumption (also termed Material Footprint) or the Raw Material Trade Balance derived from different calculation methods. The results presented, provide a first overview of the deviation of results from different models. The rather large differences in outcome, in combination with the policy relevance of indicators like DMC and RMC indicates an urgent need for a better understanding of the sources of such differences. Due to the fact that only very recently global databases have become available to analyze environmental footprints (Tukker and Dietzenbacher, 2013), analyses why such databases lead to different results are still scarce, and mainly have been concentrating on carbon footprints (e.g. Owen, 2015). For materials, our work already points at two main reasons for differences: the use of two inherently different approaches (hybrid LCA-IO approaches versus MRIO) and the use of different databases for global, per country domestic extraction data. The latter point recently has been mitigated by the publication of a harmonized, global resource extraction database by the UNEP Resources Panel (UNEP et al., 2016). Using techniques like matrix difference statistics, structural path analysis, and structural decomposition analysis Owen (2015) found that for carbon footprints using such harmonized extension data was a factor that could reduce uncertainties by 50%. Other important factors were the structure of the domestic IO table and the fraction of a countries' final demand as percentage of global GDP. We expect that in the case of RMC, additional factors will play an important role like sector aggregation, allocation of material extraction to sectors (i.e. the composition of the material extension vector), level of aggregation of materials, product and price inhomogeneity per sector, and allocation and truncation efforts in the LCI coefficients in the hybrid LCA-IO approaches. From the results presented, no clear preference for one method (hybrid IO-LCA) or the other (MRIO) can yet be drawn.

The hybrid IO-LCA approach does not require a global MRIO database and hence its application is easier for individual countries. A drawback is however that in the LCA approach it is highly complex and data-intensive to take into account that different parts of the supply chain are located in different countries with differences in resource-efficiency of production processes. Also, by using different data sources for domestic extraction and resources embodied in imports (the latter from LCA data), if this method is applied for all countries in the world the total global material extraction embodied in final consumption will not equal the total global material extraction, which per definition should be equal. The MRIO approaches have the disadvantage, their compilation is very time consuming, but does take into account trans-national supply chains and inherently ensures that at global level material extraction equals materials embodied in final demand. Any decision about which approach to apply has to take into account very different perspectives and needs among users, i.e. robustness, transparency, easiness to compile, temporal and spatial coverage and applicability. Further research to provide a good understanding and analysis of the different approaches in particular their strengths and weaknesses as well as their particular perspective and focus is required and needs to be made transparent.

Finally, we discussed the two indicators DMC and RMC next to each other and showed their different but complementing perspectives. DMC represents a production or better territorial perspective which is also interpreted as "domestic waste potential". Among the strengths of DMC is the easiness to compile, because DMC is based on standard national statistical data; DMC can also be directly addressed through national policy and legislation. RMC on the other hand is a consumption based approach, referring to the global material use required to satisfy domestic final demand. As such, RMC can address issues of global responsibility and a fair distribution of natural resources. With the different perspectives of DMC and RMC, a relation of the two to GDP, as it is done in resource productivity (or efficiency) indicators, provide some but different messages, which still need to be better understood (conceptually and methodologically). It also needs to be emphasized that both indicators only cover one environmental pressure category – raw material use – and need to be complemented with other environmental, social and economic indicators when monitoring national sustainability performance.

Acknowledgments

The research on the Austrian RMC account presented in this article was supported by the Austrian Federal Ministry of Agriculture, Forestry, Environment and Water Management, grant number BMLFUW-UW.1.4.18/0039-I/2/2014. Writing of this paper was in part supported by the EU FP7 DESIRE project, grant agreement number 308552. Anke Schaffartzik was the recipient of a DOC-team-fellowship of the Austrian Academy of Sciences (ÖAW) at the Institute of Social Ecology.

Appendix A. Supplementary data

Specifications of the calculation models, material flow accounting indicators and data, and detailed RME results presented in data tables. Supplementary data associated with this article can be found in the online version, at http://dx.doi.org/10.1016/j.ecolecon.2016.03.010.

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nature climate change

ARTICLES

PUBLISHED ONLINE: 19 DECEMBER 2016 | DOI: 10.1038/NCLIMATE3165

Unequal household carbon footprints in China

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Households' carbon footprints are unequally distributed among the rich and poor due to differences in the scale and patterns of consumption. We present distributional focused carbon footprints for Chinese households and use a carbon-footprint-Gini coefficient to quantify inequalities. We find that in 2012 the urban very rich, comprising 5% of population, induced 19% of the total carbon footprint from household consumption in China, with 6.4 tCO_2/cap. The average Chinese household footprint remains comparatively low (1.7 tCO_2/cap), while those of the rural population and urban poor, comprising 58% of population, are 0.5–1.6 tCO_2/cap. Between 2007 and 2012 the total footprint from households increased by 19%, with 75% of the increase due to growing consumption of the urban middle class and the rich. This suggests that a transformation of Chinese lifestyles away from the current trajectory of carbon-intensive consumption patterns requires policy interventions to improve living standards and encourage sustainable consumption.

The growing climate crisis[1] shows that becoming wealthy, while enabling a clean-up of the local environment[2], drives economic activity and subsequently carbon emissions, often in distant places[1,3,4]. The concept of a carbon footprint is increasingly used in the public debate on responsibility and mitigation of climate change to describe the direct and indirect carbon emissions of consumption along the international supply chain[5–8]. To achieve absolute reductions of emissions fairly, proposals grounded in climate justice have been put forward to target high-emitting individuals across all countries[9–11] while ensuring minimum levels required for a human development[11]. In 2013 a growing global upper class of top 10% consuming households already contributed 40–51% of global emissions from fossil fuels and other sources with their footprints, a third of them in emerging economies such as China[10]. At the same time the global poor (lower 50% of global income distribution) are driving 10–13% of global greenhouse gas emissions[10]. Improved methods[12], as employed in this study, provide important information for policymakers to explicitly consider the interactions and trade-offs between measures targeting inequality, poverty, climate mitigation, and towards sustainable lifestyles for the emerging middle class and rich households.

China, which recently announced a stronger focus on bolstering domestic consumption over its current export orientation, is steadily moving towards carbon- and resource-intensive consumer lifestyles, tracking the way of high-income countries[1,5,8,13,14]. The sheer scale of the Chinese economy means that the future global climate is strongly determined there[1,14,15]. Since the 1980s, a rapid reduction of the proportion of people living below the poverty line of 1.9 US dollar (2011 purchasing power parities) income per day has been achieved, from 88% in 1981 to 11% in 2014[16]. At the same time income inequality grew substantially to a Gini Index of 0.55 in 2010, leading to a stop of official reporting on the Gini coefficient for incomes[17], an established indicator on income distributions. A clear urban–rural divide of energy consumption can be observed in China, where rural households often use traditional and locally polluting energy carriers, such as straw, wood or coal, while electricity and natural gas is slowly penetrating these areas[18,19]. In urban areas, modern energy carriers such as electricity, natural gas and LPG are dominant, and mobility is the main driver of direct household energy use[18,19]. Annually, approximately 20 million people move from rural to urban areas, and future population growth is projected to be concentrated in cities, which entails large new infrastructure and housing requirements[5]. Especially in urban areas, a sizeable middle class and a small segment of households with high incomes has emerged[5,8,20], while large swaths of rural China and migrant workers coming to cities still largely remain in poverty[21]. Increasing consumption in urbanizing China has been identified as an important driver of household carbon footprints over the past 20 years, due to the growing urban population and incomes, while decreasing carbon intensity of the Chinese economy only weakly dampens these trends[5,8,22]. These growing disparities in incomes and carbon footprints are driven by government investment policies favouring coastal and urban areas[17]. But in a globally carbon-constrained future with the urgent need for absolute reductions of annual emissions[1,23], relying on economic growth to lift all boats while also decreasing inequality and improving human development can become very challenging. Clearly, decarbonizing the energy system via production-focused efficiency measures and energy-pricing reforms is essential[3,24,25]. But developing carbon-free lifestyles beyond the current trajectory of increasing carbon footprints while becoming wealthy will require more substantial debates on the limits of green consumerism and the potential towards sustainable consumption[11,20,26–28].

Herein we present the unequal distribution of carbon footprints between Chinese households along national and international supply chains for 13 income groups (5 rural and 8 urban). We quantify inequality between urban–rural and 13 income groups with a carbon-footprint-Gini coefficient (CF-Gini), for the latest available years (2012 and 2007). Gini coefficients are used to quantify inequality[10,17,29] and were applied to production-based territorial emissions[29], cumulative historical territorial emissions[30], interregional assessments of household footprints[7], urban Chinese carbon footprints[20], and estimates of household carbon footprints

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NATURE CLIMATE CHANGE DOI: 10.1038/NCLIMATE3165

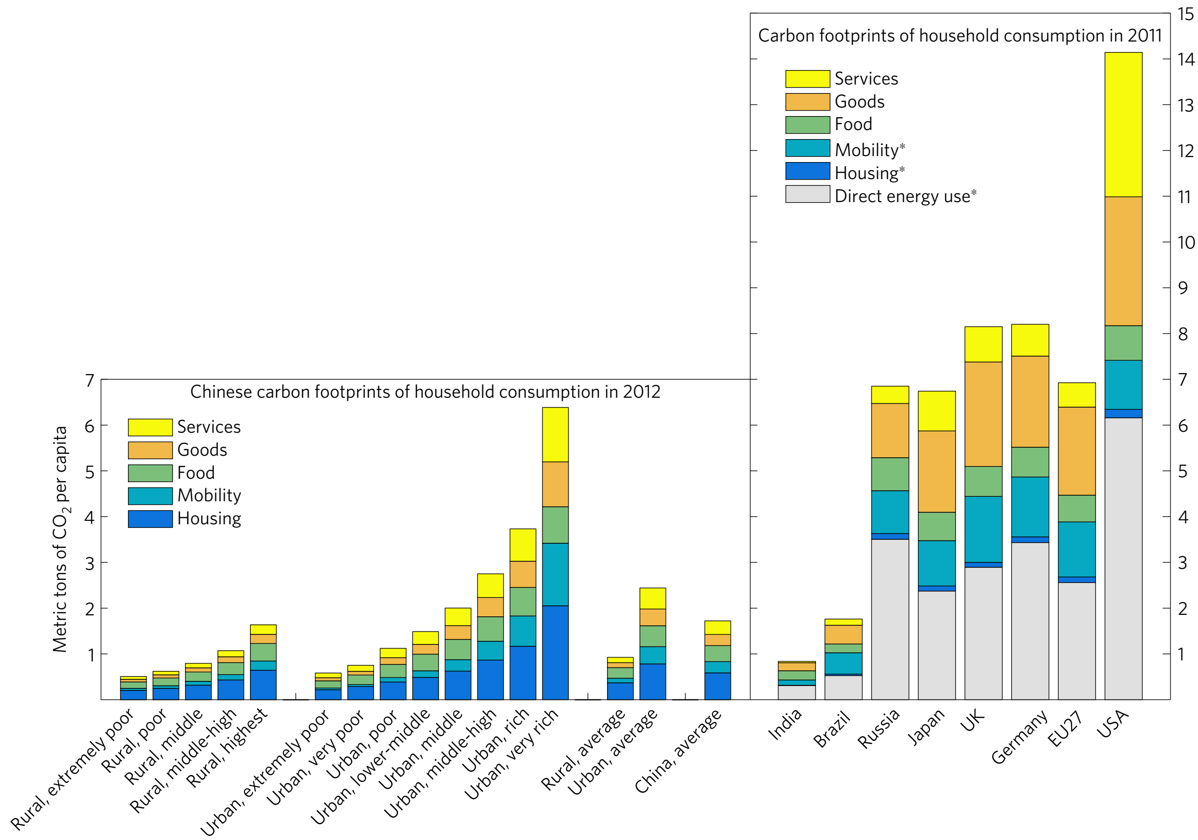

Figure 1 | Carbon footprints of Chinese and international household consumption in 2012 and 2011, respectively, from fossil fuels and cement production. *Due to data constraints the emissions from direct energy use of international households could not be allocated to the respective indirect emissions from mobility and housing. For Chinese households mobility and housing includes direct and indirect emissions.

across countries[10,12]. For income inequality and carbon footprints an inverse relationship was found[31–36]. We utilize a detailed Chinese Input-Output Table with the latest, substantially revised Chinese emissions statistics[37,38] and a Multi-Regional Input-Output Model (GTAP 9 database) for all other countries. Emissions data sets cover carbon emissions from fossil fuels and cement production.

From a production-based territorial perspective, Chinese carbon emissions are 6.7 tons of per capita in 2013[37]. However, from a consumption-based perspective, the majority of Chinese emissions are related to capital investments (48%) and exports (20%) as main drivers[3,22], while households induce only 17% of the national footprint in 2012. We find that the average Chinese household footprint is only 1.7 tCO_2/cap in 2012, more than double the Indian average (0.9 tCO_2/cap), similar to the Brazilian average (1.5 tCO_2/cap), but one quarter of that in the EU27 (6.7 tCO_2/cap) and one sixth of that in the USA (10.4 tCO_2/cap) (Fig. 1 and Table 1). However, due to high income inequality in China[17], 5.3% of the Chinese population, the very rich urban dwellers, have carbon footprints of consumption at 6.4 tCO_2/cap, nearly four times of the average Chinese. The three richest urban groups, 21% of Chinese population, induce 48% of the total Chinese household carbon footprint. At the same time rural China and the urban poor, 58% of the population, induce only 31% of the total household footprint, all below the national average of 1.7 tCO_2/cap. The total household carbon footprint of 1,354 million Chinese is estimated at 2,332 Million tons of CO_2. In comparison, the total footprint of 1,247 million Indians is only half (1,152 $MtCO_2$), while 500 million Europeans, 37% the population size of China, have 1.4 times the total footprint (EU27: 3,347 $MtCO_2$) and 312 million US-Americans, 23% the population of China, also have 1.4 times the total carbon footprint (3,262 $MtCO_2$) (Table 1).

Urban residents, 53% of the Chinese population, induce 75% of the national household carbon footprint in 2012. Their average per capita footprint is 2.4 tCO_2 (Table 1). The top 5.3% very rich urban Chinese spend 7,237 US$ per year and have a per capita footprint of 6.4 tCO_2—which is very similar to the national averages of OECD countries, that is, Japan (6.6 tCO_2 with 27,692 US$), Russia (5.9 tCO_2 and 7,585 US$), the EU27 average (6.7 tCO_2 and 21,082 US$) and Germany (7.6 tCO_2 with 20,374 US$) (Fig. 1 and Table 1). This richest urban group comprises approximately 71 million people, or 5.3% of the entire Chinese population, inducing 19% of the total household carbon footprint in 2012 (Table 1). The total footprint of that richest group amounts to 455 $MtCO_2$, 1.6 times the entire Brazilian household footprint (290 $MtCO_2$). The second group, the urban rich, 5.3% of total Chinese population, spends approximately 4,298 US$ per capita and has an average footprint of 3.7 tCO_2. The urban middle class, divided into three income groups, spends 1,725–3,159 US$ and has a per capita footprint of 1.5–2.8 tCO_2. In total, the two urban rich groups and middle class together induce 69% of the national Chinese household carbon footprint. At the same time the urban poor, divided into three groups, totalling 10.5% of Chinese population, spend only 650–1,270 US$ and have footprints of 0.6–1.1 tCO_2/cap. This means their carbon footprints are below the Chinese (1.7 tCO_2) and Brazilian average (1.5 tCO_2) but similar to the Indian average (0.9 tCO_2), and in the same range as Chinese rural households. The extremely poor in urban areas have a footprint of only 0.5 tCO_2/cap.

Consumption of Chinese rural households, 47% of the population, induces 25% of the national household carbon footprint in 2012 (Table 1). The average rural carbon footprint is 0.9 tCO_2/cap—one-quarter of the urban average. Further decomposing the rural population into five income groups yields footprints of

Table 1 | Household carbon footprints from fossil fuels combustion and cement production, population size and final demand across income groups in China for 2012 and 2007 and for international comparisons for 2011.

	Population (million people, in 2012)	Household expenditure per capita (2011/12 US$ MER)	CF elasticity of consumption (2012)	CF cap (tCO_2, in 2012)	Total CF ($MtCO_2$, in 2012)	Shares in total CF (%; in 2012)	Total CF 2007 ($MtCO_2$)
China, total	1,354 (100%)	1,908	1.00	1.7	2,332	100	1,954
Urban China, total	712 (53%)	2,803	0.97	2.4	1,738	75	1,429
Very rich	71 (5.3%)	7,237	0.98	6.4	455	19	374
Rich	71 (5.3%)	4,298	0.97	3.7	266	11	220
Middle-high	142 (10.5%)	3,159	0.97	2.8	392	17	322
Middle	142 (10.5%)	2,334	0.95	2.0	285	12	236
Lower-middle	142 (10.5%)	1,725	0.96	1.5	212	9	175
Poor	71 (5.3%)	1,270	0.98	1.1	80	3	65
Very poor	36 (2.6%)	838	1.00	0.8	27	1	21
Extremely poor	36 (2.6%)	650	1.00	0.6	21	1	17
Rural China, total	642 (47%)	916	1.12	0.9	594	25	525
Highest	128 (9.5%)	1,611	1.13	1.6	210	9	185
Middle-high	128 (9.5%)	1,054	1.13	1.1	138	6	117
Middle	128 (9.5%)	785	1.13	0.8	102	4	92
Poor	128 (9.5%)	625	1.10	0.6	80	3	73
Extremely poor	128 (9.5%)	506	1.11	0.5	65	3	58
India	1,247	939		0.9	1,152		
Brazil	193	7,707		1.5	290		
Russia	143	6,585		5.9	845		
Japan	127	27,692		6.6	843		
UK	63	26,479		5.7	361		
Germany	80	26,169		6.4	511		
EU27	500	21,082		6.7	3,347		
USA	312	34,853		10.4	3,262		

Carbon footprint (CF) elasticities were calculated using the basic income elasticity approach, where the relative change of each income groups CF/cap from the average CF/cap is divided by the relative change of each income groups expenditure/cap from the average exp/cap in 2012 (for details see Methods and Supplementary Information; US$ at 2011/2012 market exchange rates (MER)) (all numbers were rounded).

0.5–1.6 tCO_2/cap. Even the richest rural households, 9.5% of the Chinese population, spend only 1,611 US$ per capita and have a footprint of 1.6 tCO_2/cap, which is similar to the urban lower-middle class (1.5 tCO_2/cap). The rural middle and middle-high classes have footprints of 0.8–1.1 tCO_2/cap, spending 785–1,054 US$ per capita. The two poorest rural groups, 19% of the entire Chinese population, have footprints of 0.5–0.6 tCO_2, which together is only 6% of the total national household carbon footprint, and less than the Indian average footprint.

Between 2007 and 2012, the total Chinese household carbon footprint increased by 19% or 378 $MtCO_2$, with 82% of these increases due urban consumption (Table 1). The urban 'very rich', 5.3% of population, took 21% of the total increase, almost the increase for all of rural China, with 47% of the population (80 versus 70 $MtCO_2$). Per capita footprints in urban areas increased on average by 2% from a relatively higher level, while those in rural areas increased by 28%. The poor in urban and rural areas, together 29.5% of population, increased their footprint by 16%, but induced only 10% of the increase in total household footprint. The two richest urban and one rural richest groups together, 20% of population, increased their footprints by 20%, thereby taking 40% of the total increase. The urban middle class induced 41% of the total increase.

Interestingly, for the carbon-footprint elasticities of consumption we find slightly elastic relationships for the middle class and richer urban income groups (0.97–0.98), while for the urban poor and rural groups we find proportional to inelastic relationships (1–1.13) (Table 1). For the urban very rich, a 1% increase in expenditure would lead to +0.98% in carbon footprints, while +1% increase of expenditures of the rural poor would lead to +1.11%. This means that coming out of poverty is relatively carbon-intensive, due to low-quality commercial energy such as coal, first purchases of appliances and so forth. Richer households tend to use growing incomes for higher-quality commercial energy (electricity, LPG, natural gas), and especially more goods and services, which are relatively less carbon-intensive. When comparing elasticities for 2007 and 2012, interestingly rural households become slightly more carbon-intensive, while for urban households the carbon-footprint elasticity decreased (see Supplementary Information). These patterns replicate across countries[7,8,19,39], where generally with rising affluence the marginal carbon intensity of consumption decreases, but larger overall expenditure still means higher total carbon footprints than less affluent households.

When looking at the contribution of each income group's consumption pattern to the total Chinese footprint, it becomes evident that the urban rich and middle class are driving the categories mobility, goods, and services, while footprints from food and housing are less unequal (Fig. 2). For example, 78% of the total footprint of mobility, 74% for goods and 75% for services is due to the urban middle class and the two rich groups, although these income groups constitute only 42% of the population. At the same time the urban and rural poor together, which also amount to 29.5% population, induce only 7% of the mobility-related emissions and 10% of the total Chinese carbon footprints from goods as well as from services.

Finally, we quantify inequality between carbon footprints of Chinese income groups using Lorenz curves and carbon-footprint-Gini Indices. In a Lorenz curve the cumulative share of population is plotted against their cumulative footprints, where the Gini Index then quantifies the area under that curve. We find that CF-Gini

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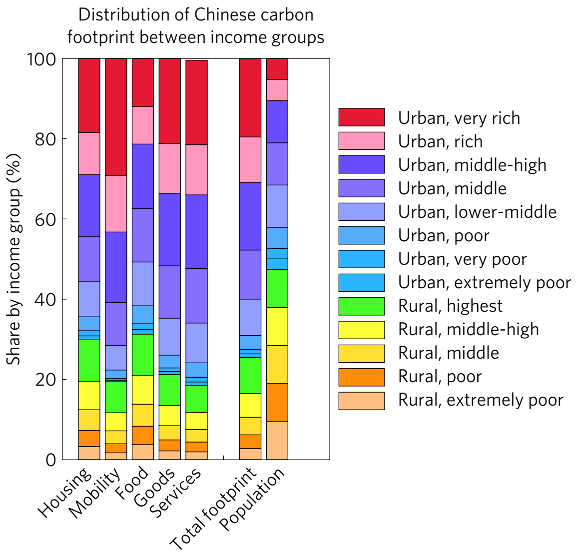

Figure 2 | Relative distribution of household carbon footprints from fossil fuels and cement, income and population size among 13 income groups in 2012.

indices for total and goods footprints are similarly unequally distributed as household expenditure in 2012 (around 0.4) (Fig. 3a). We find higher inequality for carbon footprints of services (0.5) and mobility (0.6), while those for food and housing (0.3) are more equally distributed among the Chinese population. Between 2007 and 2012, national inequality decreased slightly across all categories (Fig. 3b), except for rural food and housing-related carbon footprints, which is also the major contributor to increasing per capita footprints in rural areas (Fig. 3). While urban inequality did not change significantly (Fig. 3b,c), rural inequality increased (Fig. 3b,d).

To encourage economic growth, China's government has enacted policies focusing on increased domestic consumption as a substitute for its declining growth in investment and exports, while also announcing an absolute emissions peak for 2030. Recently, Chinese emissions growth did slow down[37], largely driven by a stabilization of coal use[13,38]. Substantial policy efforts in carbon taxation, feed-in tariffs for renewables, and accelerated deployment of renewables and nuclear have been modelled to achieve this stabilization of Chinese emissions at modest costs until 2030[25], while most of these current Chinese climate policies consider regional inequality only by using differentiated goals between provinces. However, at the same time it is clear that stabilizing the climate at 1.5–2 °C will require unprecedented absolute global reductions of emissions over the next two to three decades[1,15].

The slight decreases in expenditure inequality between Chinese households, mostly due to a small catch-up of rural households, is triggered by governmental subsidies to rural households' general purchase and income tax free policies. But our findings suggest that coming out of poverty is fairly carbon-intensive due to a larger carbon-footprint elasticity of consumption of the poorer income groups, strongly driven by their dirtier direct energy mix. However, much more problematic are the growing carbon footprints of the urban middle class and the rich, which together induce 69% of

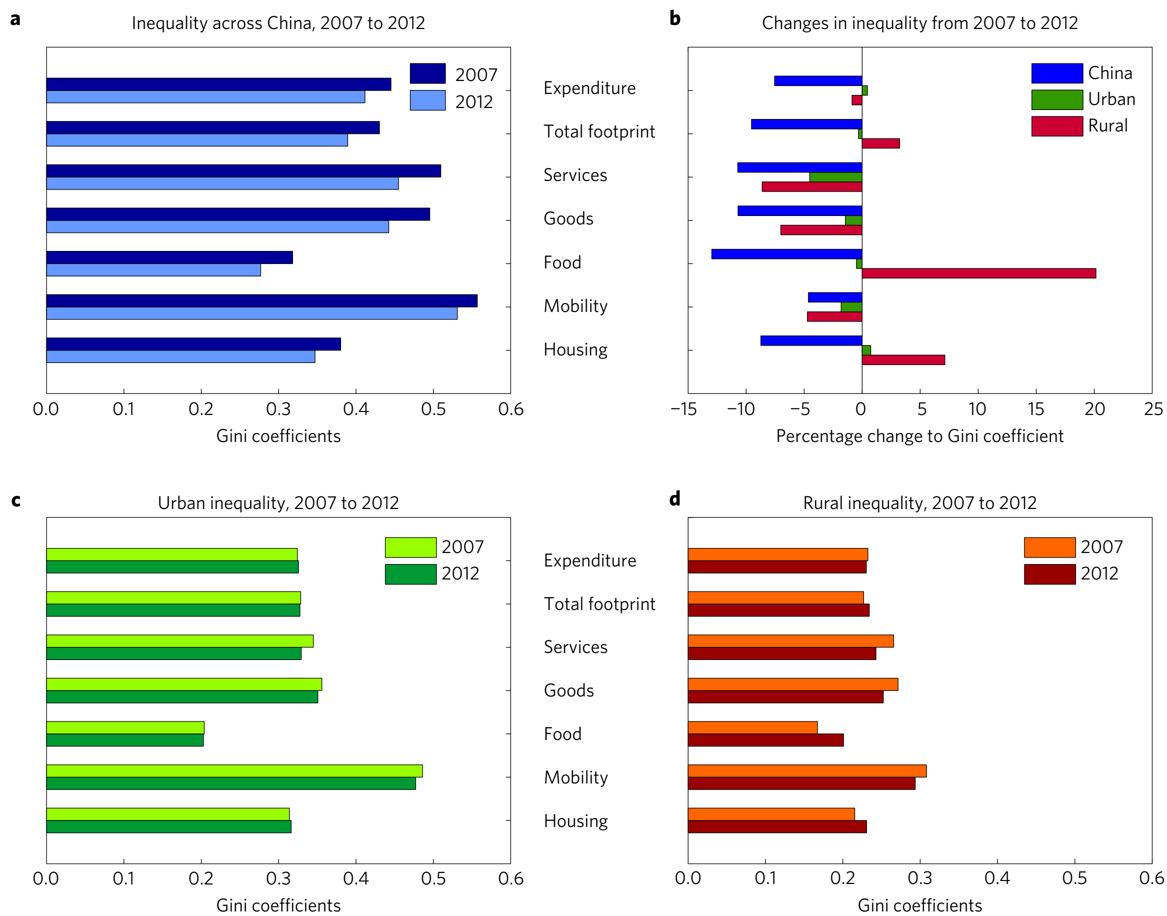

Figure 3 | Quantifying inequality. Carbon-footprint-Gini coefficients for 13 Chinese income groups in 2012 and 2007, for carbon emissions from fossil fuels and cement production.

the total Chinese household footprint and rapidly westernize their lifestyles. It has been suggested that income redistribution in urban China could reduce aggregate carbon footprints while improving living standards and income inequality[20]. From the results in Table 1 we can estimate that simply redistributing expenditure to achieve equality at 1,762$/cap, which is −8% lower than the current average expenditure, would result in a −1% decrease of total household footprints, due to differential CF elasticities. Therefore, social and redistributive policies need to be understood as interacting with climate and energy policy, as well as with efforts towards enabling sustainable lifestyles for all[17,20,31,36].

While the Chinese government is making efforts to build regional inequalities into climate policies from production efficiency and technology level approaches (for example, rich coastal versus poorer inland areas), this study reveals that there are substantial inequalities within these regions and along income groups. The CF-Gini could be useful for developing sustainable consumption programmes for those income groups which dominate the footprints of certain consumption areas, or for guiding policy design in achieving poverty alleviation while reducing emissions and increasing energy efficiency. Direct emissions from heating with coal or natural gas at present amount to 11% of the total footprint and 21% of the rural footprint. Some practical policies are designed to alleviate poverty and reduce emission at the same time. For example, Beijing municipality government set up a subsidization plan to implement a 'coal replacement by clean energy programme' for every rural household in 400 villages surrounding Beijing. By end of 2017, appropriately 4 million tonnes of coal consumption for residential usage will be saved, which is equivalent to 7 million tonnes of CO_2 emissions and 210 thousand tonnes of SO_2. The emission reduction effort is the same as three years aggregated emission discharge by 66 thousand taxis in Beijing.

Usually, shifting consumer choices is seen as yielding substantial climate mitigation benefits, for example eating less (red) meat and more vegetarian diets, less to no fossil fuelled mobility, energy-efficient dwellings, and purchasing high(er)-quality long-lived goods[40,41]. Tapping these potentials requires substantial policy guidelines, careful policy designs and matching infrastructures. At present, direct mobility emissions from fuels make up only 3% of the total household footprint, most of it by the rich and urban middle class. But following 'on the road' American culture, there are increasingly demands for cheap 4 × 4 fleets by the Chinese middle class. Domestic car manufacturers are upgrading production lines to fulfil such demands. Beijing and Shanghai have implemented tailored policies to limit absolute gasoline fleets and encourage electric vehicle (EV) purchases with heavy subsidies. However, such policies ignore China's coal-dominant energy mix. China's gasoline vehicle replacement programme with EVs is not effective at present. In fact, evidence shows that the CO_2 emissions reduction in the petroleum sector is offset by the increase in CO_2 emissions in the electricity sector[42]. The EV programme can be effective only with significant changes in the Chinese energy mix towards renewables. Therefore, green consumerism alone (even with policy guidelines) cannot drive the entire production system towards sustainability, and more systemic approaches are necessary to achieve sustainable consumption and production[11,26,28]. More sustainable urban forms and spatial planning have been identified as important long-term factors towards facilitating low-carbon lifestyles, especially in growing cities which are currently expanding their infrastructures[5,6,43,44].

Overall the required long-term transformations towards a net-zero carbon society should be included into a national discourse about the currently dominant mode of ecological modernization, green growth and conspicuous consumer lifestyles[28]. The carbon-intensive lifestyles of the wealthy are being emulated, and serve as role models, while investments in infrastructure and cities are made. Based on the CF elasticities (Table 1), a hypothetical scenario of an expenditure catch-up of all Chinese households to the average urban rich expenditure pattern (that is, mobility by cars and planes and living with an average 90 m^2 per household) can be estimated, resulting in a tripling of the total Chinese household carbon footprint. A catch-up only to the average urban middle class would translate into a 58% increase of the total footprint. But in a carbon-constrained post-Paris COP21 future, high wellbeing and human development needs to be achieved while rapidly reducing total emissions[1,13]. Reducing inequalities but preventing emission-intensive lifestyle westernization in populous developing countries can be a step forward to contribute global climate change mitigation. Cost-effectively using limited public and private funding for these societal goals will be crucial. Some countries have already achieved a high level of human development (HDI of >0.8) with an average carbon footprint of 1 ton per capita[45–47], highlighting that pathways to livable and potentially more sustainable societies exist.

Methods

Methods and any associated references are available in the online version of the paper.

Received 29 April 2016; accepted 27 October 2016; published online 19 December 2016

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Acknowledgements

This work was supported by National Key R&D Program of China (2016YFA0602603 and 2016YFA0602604), the National Natural Science Foundation of China (41629501, 71521002, 41461118, 41501605), Austrian National Science funded project 'MISO—modelling the global metabolic transition' (P27590), the UK Economic and Social Research Council funded project 'Dynamics of Green Growth in European and Chinese Cities' (ES/L016028/1), the UK Natural Environment Research Council funded project 'Integrated assessment of the emission-health-socioeconomics nexus and air pollution mitigation solutions and interventions in Beijing' (NE/N00714X/1). Many thanks go to J. Minx, F. Krausmann and J. K. Steinberger for their feedback on the manuscript, to G. Peters for support with the global emissions data set for the GTAP-MRIO and to L. Yu for his feedback on the concordances to bridge the GTAP-MRIO and the Chinese national IOTs classifications.

Author contributions

D.W. and D.G. designed the research, performed calculations and discussed the results. D.W. wrote the paper. D.G., Z.L., J.M., N.Z. and Y.-M.W. collected data and contributed to writing the paper.

Additional information

Supplementary information is available in the online version of the paper. Reprints and permissions information is available online at www.nature.com/reprints. Correspondence and requests for materials should be addressed to D.W., D.G. or Y.-M.W.

Competing financial interests

The authors declare no competing financial interests.

Methods

We quantify the household carbon footprints of 13 Chinese income groups (5 rural and 8 urban) for 2012 and 2007, using a detailed Chinese Environmentally Extended Input-Output model (IOT)[48] and a global Multi-Regional Input-Output Model derived from the GTAP database (MRIO). The Chinese IOT has 135 sectors in producers' prices[49], and is extended by China's sectoral CO_2 emissions from fossil fuel combustion and cement production, each corrected for the latest published estimates on coal carbon contents and energy use, which significantly altered the previously available official data[38,50]. Household consumption patterns for eight urban and five rural income groups are derived by disaggregating the urban and rural household final demand vectors in the Chinese IOTs with data on the respective consumption structures from the China Urban Lifestyle and Price Yearbooks[50]. These list average incomes and consumption patterns per income group, discerning 8 major classes of expenditure items and 58 sector specific items, which were mapped to the 135 sectors of the Chinese IOTs. In particular, we convert all 58 consumption categories into percentage to total per capita consumption; we then produce a concordance matrix to map the 58 sector-specific items with 135 IOT sectors; finally, we disaggregate the urban household vector in IOT into 8 income groups. We repeat the same process and utilize income-grouped household expenditure data provided from Chinese Rural Statistics to disaggregate the rural household average consumption into 5 income groups.

All international upstream emissions were calculated via an MRIO derived from the GTAP database for 2011 (140 countries × 57 sectors) and 2007 (129 countries × 57 sectors)[51]. International emissions of fossil fuels and cement production by sector are derived from the GTAP emissions database and corrected with the latest revised Chinese emissions statistics[38,50]. This is important because households directly consume imports and the Chinese economy requires imported intermediate inputs to produce domestic final output, which constitutes an important international inter-industry feedback[52]. The limitations of this study are first, that non-CO_2 greenhouse gas emissions are not included due to lack of available sector level data for China. Second, Input-Output Analysis generally considers only the annual emissions (flows) within the same year, which means that cumulative emissions (stocks) required to build existing infrastructure and buildings are not accounted for—which can be seen as an issue especially for housing-related footprints, which are driven at present by electricity, natural gas and household appliances. More generally, this means that the consumption of capital is currently not endogenized in our model, as this is an ongoing issue for IO analysis in general (see Supplementary Information for a longer discussion on the methodological limitations). The concordances between the MRIO and the Chinese IOT are derived from the GTAP documentation (see Supplementary Information).

Carbon-footprint results for 135 Chinese and 57 international sectors were aggregated to five major categories of consumption: housing, mobility, food, goods and services (Supplementary Information). Emissions from direct household energy use of coal, natural gas and electricity are allocated to the category housing, while oil emissions are allocated to mobility[53]. Chinese national emission data is available online at China Emission Accounts and Datasets - CEADs (http://www.ceads.net). Due to data constraints in the GTAP-MRIO, direct energy use for non-Chinese households cannot be completely allocated, and thus was kept separate (see Fig. 1 and Supplementary Information).

The consumption-based carbon-footprint-Gini coefficient presented herein is based on the well-known Gini coefficient, which is derived from Lorenz curves, initially proposed by Lorenz in 1907 and widely used to measure inequality[7,10,17,29,30]. The original Lorenz curve plots population shares against income shares, where the area below that curve is defined as Gini coefficient, ranging from 0 to 1. A straight 45° line in the Lorenz curves would indicate perfect equality; similarly, a Gini coefficient of 0 indicates perfect equality, and 1 indicates perfect inequality. In this paper, we present a consumption-based carbon-footprint-Gini index across 13 income groups and their carbon footprints. Let us define the following variables for group n: C_i^n is the carbon footprint of group n for product i, Pop^n is the population size of income group n, and $p^n = (\mathrm{Pop}^n / \sum_m \mathrm{Pop}^n)$ is the population share of group n. $C^n = (C_i^n / \sum_m C_i^n)$ is then the share in total household carbon footprint of group n for product i. Define the area between the actual allocation curve and perfect equal allocation curve as X, the area below the actual allocation curve as Y. Then the Gini index is defined as $X/(X+Y)$. Supplementary Table 3 provides the population and carbon footprints of each group in 2007 and 2012.

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7. Full research articles for objective two: Integrating biophysical structures into ew-MEFA, to monitor progress towards a more sustainable circular economy

RESEARCH AND ANALYSIS

Maintenance and Expansion

Modeling Material Stocks and Flows for Residential Buildings and Transportation Networks in the EU25

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Keywords:

construction and demolition waste
dynamic stocks and flows modeling
industrial ecology
material flow analysis (MFA)
recycling
societal metabolism

Supporting information is available on the *JIE* Web site

Summary

Material stocks are an important part of the social metabolism. Owing to long service lifetimes of stocks, they not only shape resource flows during construction, but also during use, maintenance, and at the end of their useful lifetime. This makes them an important topic for sustainable development.

In this work, a model of stocks and flows for nonmetallic minerals in residential buildings, roads, and railways in the EU25, from 2004 to 2009 is presented. The changing material composition of the stock is modeled using a typology of 72 residential buildings, four road and two railway types, throughout the EU25. This allows for estimating the amounts of materials in in-use stocks of residential buildings and transportation networks, as well as input and output flows. We compare the magnitude of material demands for expansion versus those for maintenance of existing stock. Then, recycling potentials are quantitatively explored by comparing the magnitude of estimated input, waste, and recycling flows from 2004 to 2009 and in a business-as-usual scenario for 2020. Thereby, we assess the potential impacts of the European Waste Framework Directive, which strives for a significant increase in recycling.

We find that in the EU25, consisting of highly industrialized countries, a large share of material inputs are directed at maintaining existing stocks. Proper management of existing transportation networks and residential buildings is therefore crucial for the future size of flows of nonmetallic minerals.

Introduction

Ongoing efforts in the European Union (EU) to further improve its environmental performance and move toward a sustainable development path have led to the formulation of the Waste Framework Directive (WFD), which was put into national laws by 2010 (EPC 2008). Among pushes toward improved waste prevention and comprehensive national waste management plans, the directive also mandates quantitative goals for increased recycling as an important step toward a "recycling society"—especially construction and demolition waste, which is, after carbon emissions, the second largest waste stream of the European economies, has been targeted with a compulsory recycling rate of at least 70% of weight by 2020 (EPC 2008).

These steps have been taken while there are already large accumulated in-use stocks of buildings and infrastructure throughout Europe. The central issue of stocks is that they usually have service lifetimes of at least years to decades, making them

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DOI: 10.1111/jiec.12216
Editor Managing Review: Seiji Hashimoto

Volume 00, Number 0

"suitable for depicting the influences a system's history has on its present—and hence for analysing temporal developments" (Faber et al. 2005, 155). Generally, stocks and the physical services they provide are closely linked to resource use and emissions, starting during extraction and processing of materials, during stock construction, while utilizing the services and operating the stock, for maintenance, and, finally, at the end of the service lifetime when demolished and landfilled or recycled (Pauliuk and Müller 2014). Material flow accounts for Europe consistently show that nonmetallic minerals make up 30% to 40% of domestic material consumption (Eurostat 2012a; Steger and Bleischwitz 2011; Steinberger et al. 2010; Weisz et al. 2006), all of which are accumulated as stocks. Aging buildings already require significant amounts of resource use for maintenance and refurbishment (Deilmann et al. 2009), especially when taking into account that their energy efficiency needs to be improved rapidly for effective climate-change mitigation (Nemry et al. 2010; Meijer et al. 2009). But, at the same time, building stocks are still being expanded, thereby possibly running counter toward these efficiency efforts (Pauliuk et al. 2013a; Sandberg and Brattebø 2012). Linked with this expansion, especially of cities, is also the ongoing growth of already substantial transportation networks crisscrossing Europe (Steger and Bleischwitz 2011), inducing large maintenance requirements in materials, energy, and financial terms. At the same time, stocks can also be seen as a future secondary resource for urban or technospheric mining (Krook and Baas 2013; Brunner, 2011; Hashimoto et al. 2007). Therefore, systematic knowledge about the dynamics of in-use stocks, their lifetimes, as well as expansion and maintenance requirements and subsequent end-of-life (EOL) waste flows and potentials for recycling are important stepping stones for a materially efficient (Allwood et al. 2011) recycling society and the potential transformation toward sustainability (Pauliuk and Müller 2014).

In this work, we utilize a dynamic material stocks and flows model for residential buildings and the road/railway network of the EU25 for 2004–2009 to investigate the relationships between these stocks and materials inputs and outputs of nonmetallic minerals. The changing material composition and size of stocks is modeled by utilizing detailed age/type matrices for 72 residential building types, four road and two railway types, combined with empirical growth and demolition rates as well as maintenance rates. In an otherwise business-as-usual scenario until 2020, some of the effects of strongly increased recycling mandated by the European WFD are explored.

Literature Review and Scope Definition

The majority of existing literature on in-use stocks and flows has thus far focused on metals (Müller et al. 2010; Graedel et al. 2013; Müller et al. 2014), with only a smaller proportion of studies investigating nonmetallic minerals, mostly from an industrial ecology and waste management perspective. Kapur and colleagues (2008, 2009) modeled the in-use stocks of cement in the United States, using a top-down dynamic model, thereby demonstrating that per capita stocks doubled over the past 50 years, and whereas in-use stock increase is slowing down since the 1970s, they are still growing by approximately 2% to 3% per annum. Further, more than 80% of all cement utilized during the last century is still in use. Future maintenance and replacement requirements, especially for roads, are expected to be substantial because of stock aging (Kapur et al. 2009). Japanese studies further highlighted that, additionally to concrete, large amounts of sand, gravel, crushed stones, and aggregates are also stocked, with an overall in-use stock increase of nonmetallic minerals by a factor of 3 between 1970 and 2000 (Hashimoto et al. 2009). Often, these materials are used in foundations and other more permanent infrastructure, which makes them unlikely to be recoverable, thereby substantially reducing the future potential recycling amounts (Hashimoto et al. 2009). Further, in Japan, stocked materials in various infrastructure, such as disaster prevention structures, harbors and airports, water and sewerage networks, and structures of the primary sector are each approximately the same amount as those in residential buildings and roads (Hashimoto et al. 2007). Studies on China highlighted the rapid accumulation of in-use stocks of residential buildings (Huang et al. 2013), strong spatial inequality among urban and rural building stocks (Hu et al. 2010), as well as between Western hinterlands and coastal regions (Han and Xiang 2013) and explored potentials for dematerialization and a "circular economy," also including the roads network (Wen and Li 2010; Shi et al. 2012; Guo et al. 2014).

A recent study on the cement cycle of Ireland, which also only recently turned into a booming economy, showed that EOL waste only amounts to 1% of concrete use because the majority of stocks have only been accumulated in the last 20 years (Woodward and Duffy 2011). One of the issues visible in this study is that the data quality on waste flows and collection rates is mixed, giving rise to substantial imbalances between EOL flows, recycling estimates, and reported waste (Woodward and Duffy 2011). Dynamic top-down studies for Norway and the Netherlands, which are long-standing wealthy economies, suggest that in-use stocks of residential buildings are substantial and can be expected to keep growing slowly, with a probable saturation in the mid-twenty-first century, alongside strongly increasing EOL waste flows (Bergsdal et al. 2007; Müller 2006; Sartori et al. 2008) and subsequent challenges for waste management (Bohne et al. 2008). Overall, residential buildings in most of Europe are a mixture of different age and building types: 30% to 40% of buildings were built in the period 1945–1970 and another 20% to 40% between 1971 and 1990 (Meijer et al. 2009; Nemry et al. 2010). Demolition rates have been ranging from 0.05% to 0.2% in the period 1980–2005 (Thomsen and van der Flier 2009), resulting in a rapid accumulation of stocks across Europe. Maintenance, refurbishments, and replacements of these buildings to adapt to current and future standards, also in regard to climate-change mitigation, have large implications financially as well as for materials and energy use and emissions (Nemry et al. 2010; Pauliuk et al. 2013a; Sandberg and Brattebø 2012; Thomsen and van der Flier 2009).

Additionally, road networks constitute a significantly sized stock of nonmetallic minerals, especially in Japan with its stricter building standards owing to frequent natural disasters (Hashimoto et al. 2007; Kapur et al. 2008; Schiller 2007; Tanikawa and Hashimoto 2009). For European roads and railways, data exist on kilometers (km) of network, but not on detailed age/type distributions, hindering the application of a dynamic cohort-based modeling approach. But as a result of short service lifetimes, compared to buildings, material turnover can be expected to be much higher (Schiller 2007; Steger et al. 2011; Tanikawa and Hashimoto 2009).

In this study, the following research questions are posed, also with the aim of evaluating the European level data situation for future policy recommendations:

1. What is the size of in-use stocks of nonmetallic minerals in residential buildings and the road/railway network of the EU25? What are the magnitudes of directly related flows of nonmetallic minerals into and out of these stocks resulting from maintenance and expansion? How do they compare to economy-wide material use?
2. What are the dynamics of these stocks and flows over time and how could increased recycling under the WFD until 2020 affect these flows?

One of the major constraints for this study is one of data availability on stock characteristics, material composition and age distributions, as well as demolition and maintenance rates. Although it is clear that stocks in various infrastructure as well as public and commercial buildings are substantial, European-level data sets are hardly available. Retrospective studies further face the problem of scale—although it is possible to exactly identify, for example, the material content of a single house or a road, doing so for a whole country is impossible. Different approaches exist to deal with this, for example, with representative building typologies (Nemry et al. 2008, 2010), average material intensities per floor space and building period (Bergsdal et al. 2007; Müller 2006; Schiller 2007), coupling of spatial databases of settlement structures, building standards and age distributions (Tanikawa and Hashimoto 2009), linking consumption of materials with service-lifetimes and waste factors (Cochran and Townsend 2010), and combining input-output methods with data on floor space, building regulations, and construction activity (Hashimoto et al. 2007).

Methodology: Definitions, Data, and the Model

We apply a dynamic bottom-up approach, utilizing time series of extents (E) of each stock type (s), either in number of buildings or km of road and rail, by country (c) and year (t), as well as material intensities (M), in metric tons, for each stock type (s) and material (m) (see supporting information available on the Journal's website for details of compilation; tables 1 and 2). These material intensities are grouped in "concrete and asphalt" as well as "other construction minerals" (bricks, stones, tiles, sand and gravel, and aggregates). Combined, these two groups constitute 96% of domestic material consumption (DMC) of nonmetallic minerals in the EU25 (Eurostat 2012b). The remaining 4% are "salt, fertilizer, and other products."

Using the MATLAB programming language, the following equation is used to estimate annual material stocks (MS) of nonmetallic minerals as sum over the multiplication of extents (E) by respective material intensities for each stock type (S), country (c), and year (t) (equation 1).

$$MS_{c,m,t} = \sum_{s} E_{S,c,t} * M_{S,c,m} \tag{1}$$

For residential buildings and their material compositions, a typology of 72 "typical" buildings, developed by Nemry and colleagues (2008, 2010), representing 80% of residential buildings in the EU25 for the year 2003, is used (table 2; see the Supporting Information on the Web; Nemry et al. [2008]). Time series on annually finished and demolished dwellings and total dwelling stocks are compiled from Eurostat, European Housing Statistics reports ,and other national sources for 2003–2009 (see the Supporting Information on the Web for annual data and sources; table 2 for an overview). A recent survey, also using various sources for the 2000s and highlighting large data gaps showed that, in EU27 plus Norway and Switzerland, residential buildings constitute 75% of total European floor space, whereas the remaining square meters are commercial (15%), educational (4%), sport (1%), hospitals (2%), and of "other" (3%) nature (BPIE 2011). For these very heterogenous building types, no data on material composition are available. Data on roads and railways are compiled from Eurostat, UNECE, and the European Road Foundation, and material intensities for four road and two railway types were taken from the literature (see the Supporting Information on the Web; table 1). Because of very mixed data quality, various data points had to be either linearily interpolated or taken from the EU-wide average (see the Supporting Information on the Web for detailed documentation). No similar data on other infrastructure, such as bridges, tunnels, dams, ports, sewers, and so on, were available.

Based on these time series, we also constructed the business-as-usual scenario for 2020, using the average national growth and demolition rates from 2003 to 09 (table 2) as well as the lifetimes for each stock type that are subject to maintenance (see the Supporting Information on the Web; table 1). For this scenario, the banking and subsequent public debt crisis of 2007–2008 is also taken into account: For countries that experienced stronger-than-average 2% growth of their dwelling stock between 2003 and 2009 (Spain, France, and Ireland), the average of the remaining EU25 countries (0.78%) was used for the projection from 2010 onward.

Expansion of the stock of buildings and transportation networks and subsequent material inputs (*MI_expansion*) are defined by net increase of stock extent and are calculated in two steps: First, the annual net increase of stock extent[1] (*E_add*) of stock type (s), in country (c) between year (t) and t+1, is calculated from equation (2) for $E\_change > 0$, whereas the net decrease of extent (*E_decl*) uses equation (2), for $E\_change < 0$.

$$E\_change_{s,c,t+1} = E_{s,c,t+1} - E_{s,c,t} \tag{2}$$

Table 1 Material intensities and maintenance cycles for residential building types, roads and railways[a]

Zone 1: Northern Europe									
	Single family houses (11 types)			*Multi-family houses (11 types)*			*High-rise buildings (3 types)*		
Number of buildings, EU25 (2003)	21,353,400			1,629,200			234,200		
Material intensities [metric tons per building]									
	min	***avg***	***max***	***min***	***avg***	***max***	***min***	***avg***	***max***
Structurally used									
Concrete	0	119	216	0	1,493	3,565	3,176	3,627	4,530
Other construction minerals	94	178	425	0	681	2,542	0	280	840
Nonstructural use									
Other construction minerals	25	42	65	154	313	549	424	778	1,355
Renovation cycles (Nemry et al. 2008) [years per building]									
Nonstructural use									
Other construction minerals	25	26	30	25	26	28	27	28	28
Zone 2: Central Europe									
	Single-family houses (11 types)			*Multifamily houses (11 types)*			*High-rise buildings (3 types)*		
Number of buildings, EU25 (2003)	35,084,100			2,243,320			129,440		
Material intensities [metric tons per building]									
	min	***avg***	***max***	***min***	***avg***	***max***	***min***	***avg***	***max***
Structurally used									
Concrete	0	119	184	0	1,257	3,402	2,312	2,936	4,185
Other construction minerals	0	143	384	0	732	2,547	0	456	1,368
Non-structural use									
Other construction minerals	32	43	54	227	269	404	11	351	567
Renovation cycles (Nemry et al. 2008) [years per building]									
Non-structural use									
Other construction minerals	24	26	28	26	26	27	26	26	26
Zone 3: Southern Europe									
	Single-family houses (9 types)			*Multifamily houses (10 types)*			*High-rise buildings (3 types)*		
Number of buildings, EU25 (2003)	2,412,500			296,600			3,100		
Material intensities (metric tons per building)									
	min	***avg***	***max***	***min***	***avg***	***max***	***min***	***avg***	***max***
Structurally used									
Concrete	0	142	211	0	1,198	3,402	2,312	2,936	4,185
Other construction minerals	0	89	384	0	400	2,547	0	912	1,368
Non-structural use									
Other construction minerals	32	40	54	150	254	404	476	537	567
Renovation cycles (Nemry et al. 2008) [years per building]									
Non-structural use									
Other construction minerals	24	26	29	25	26	27	26	26	26

(Continued)

Table 1 Continued

	EU road and railway network					
	Motorways	*State roads*	*Provincial*	*Communal*	*Single track*	*Double tracks*
km in EU25 (2003)	58,310	383,944	1,380,274	3,336,382	125,132	74,764
Material intensities (metric tons per km)						
Asphalt, concrete	14,341	3,489	2,932	1,393	308	616
Other construction minerals	25,864	8,011	6,280	4,208	3,419	6,837
Infrastructure lifetimes (sources in Supporting Information on the Web) (years per km)						
Asphalt, concrete	17	20	20	20	47	47
Other construction minerals	60	60	60	60	60	60

[a]Structural use covers, for example, load-bearing walls and foundations; non-structural use includes roofs, bricks in non-load bearing walls, tiles, plaster, etc; see the supporting information on the Web for details and compilation procedures.

Min = minimum; Avg = average; Max = maximum; EU = European Union; km = kilometers.

This additional stock extent (*E_add*) is multiplied by the respective material intensity (*M*) for stype type (*s*), country (*c*), and material (*m*) (equation 3) to arrive at the material inputs due to expansion.

$$MI\_expansion_{c,m,t} = \sum_{s} E\_add_{s,c,t} * M_{S,c,m} \quad (3)$$

Maintenance, on the other hand, is defined as material inputs required to keep the stock extent and the services it provides constant, despite stock aging and EOL flows. Maintenance therefore includes two material inputs: first, ongoing renovation cycles of nonstructural components of residential buildings, such as roofs, tiles, nonload carrying walls (table 1), as well as the renewal of worn down layers of a roads; second, also replacement constructions of stocks that were demolished because they were at the end of their service lives are included, for example, rebuilding of railway sections or roads, as well as buildings. The lifetimes for each building type from Nemry and colleagues (2008) are used to assign probabilities of demolition (i.e., the older a building, the higher the probability). For this short period of estimations and because of lack of data on the change of composition of residential buildings, we assume that each stock type (i.e., single-family house and communal road) is replaced by a similar type, but of up-to-date construction standards. For roads and railways, we assume, based on the literature, that maintenance, in the form of renewal of the asphalt layers, happens every 17 to 20 years (depending on road type; table 1; see the Supporting Information on the Web), and for railways, the aggregates and concrete sleepers need replacement every 47 to 60 years (table 1; see the Supporting Information on the Web).

Maintenance material inputs (*MI_maintenance*) are therefore calculated in two steps: First, all roads, rails, and buildings (E) that are at the end of service life (*E_EoL*) and need replacement are calculated using country (*c*) and time (*t*) specific empirical demolition rates (*Demol*) for entire buildings and the reciprocal of lifetimes as an approximation of replacement requirements for roads and railways (*Demol*). Renovation cycles for nonstructural building components as well as upper road layers and railways sections are calculated by the reciprocal of the lifetimes (*LT*) from table 1 (equation 4):

$$E\_EoL_{s,c,m,t} = E_{S,c,t} * Demol_{S,c,t} + E_{S,c,t} * \frac{1}{LT_{S,c,m}} \quad (4)$$

The maintenance material inputs (*MI_maintenance*) are then the sum over the multiplication of all EOL stocks (*E_EoL*) by the specific material intensity (*M*) of the new infrastructure/building type replacing it,[2] as well as the material intensity (*M*) for the stock components that were renovated (Table 1 and supporting information available on the Journal's website) (equation 5).

$$MI\_maintenance_{c,m,t} = \sum_{S} E\_EoL_{s,c,t} * M_{s,c,m} \quad (5)$$

The material outputs from stock (MO) are estimated by the EOL stocks *(E_EoL)* from equation (4) and the stocks coming out of use as a result of stock decline (E_decl), taken from equation (2). The sum over both stocks multiplied by the material

Table 2 Overview on major parameters of stock extent and change[a]

	Composition of dwelling stocks and share of residential buildings not covered (Nemry et al. 2008, 18)				*Housing growth and demolition rates*			*Average change of roads network 2000–2009*				*Average change of railways network 2000–2009*		
				Not	*Growth*		*Demolition*							*Recycling rate*
	Single family	Multifamily	High-rise	represented by dwelling types	Avg 2003–09	2010+	Avg 2003–09	Motorways	State roads	Provincial	Communal	Single tracks	Double tracks	of C&D waste
Austria	41%	46%	1%	12%	1.2%	1.2%	0.54%	0.4%	–0.2%	0.1%	0.0%	0.6%	1.4%	60%
Belgium	63%	20%	2%	15%	0.7%	0.7%	0.15%	0.4%	0.1%	0.1%	0.6%	–0.3%	0.6%	68%
Cyprus	50%	20%	0%	30%	4.0%	0.8%	0.03%	1.6%	0.7%	0.8%	2.5%	0.0%	0.0%	1%
Czech Republic	28%	30%	18%	24%	0.7%	0.7%	0.04%	2.7%	0.1%	0.0%	0.9%	0.3%	–0.1%	23%
Denmark	40%	33%	6%	21%	0.8%	0.8%	0.15%	2.7%	–2.0%	–0.3%	0.2%	–0.8%	0.7%	94%
Estonia	27%	32%	25%	16%	0.6%	0.6%	0.15%	2.2%	0.7%	1.0%	1.0%	1.4%	0.6%	92%
Finland	38%	47%	0%	15%	1.1%	1.1%	0.15%	4.9%	0.0%	0.0%	1.8%	0.0%	0.2%	26%
France	40%	28%	10%	22%	1.6%	1.6%	0.07%	2.2%	–5.0%	0.4%	0.9%	–0.7%%	0.2%	62%
Germany	41%	42%	4%	13%	0.5%	0.5%	0.12%	1.0%	–0.3%	0.1%	0.2%	–1.8%	0.3%	86%
Greece	44%	31%	0%	25%	0.8%	0.8%	0.15%	0.6%	–2.3%	0.0%	0.0%	0.3%	1.4%	5%
Hungary	42%	20%	14%	24%	0.8%	0.8%	0.11%	3.3%	0.1%	–0.1%	0.0%	0.3%	–0.1%	16%
Ireland	70%	4%	0%	26%	2.6%	0.8%	0.74%	2.3%	0.3%	0.7%	0.3%	–1.6%	–0.4%	80%
Italy%	34%	39%	12%	15%	0.8%	0.8%	0.15%	0.2%	0.5%	0.9%	0.7%	–0.6%	1.6%	0%
Latvia	24%	65%	0%	11%	0.8%	0.8%	0.10%	0.0%	–0.1%	0.1%	1.0%	–0.6%	0.2%	46%
Lithuania	31%	56%	0%	13%	0.2%	0.2%	0.01%	0.4%	0.1%	2.4%	3.5%	–0.4%	–1.3%	60%
Luxembourg	42%	17%	8%	33%	0.8%	0.8%	0.15%	0.4%	0.0%	0.0%	0.0%	0.0%	0.0%	46%
Malta	50%	30%	0%	20%	0.8%	0.8%	0.15%	0.0%	0.0%	4.3%	0.0%	0.0%	0.0%	0%
Netherlands	50%	28%	5%	17%	0.3%	0.3%	0.29%	1.3%	–0.3%	–0.1%	0.5%	–0.6%	0.6%	98%
Poland	35%	36%	18%	11%	0.5%	0.5%	0.04%	1.5%	0.3%	–0.1%	0.5%	–1.3%	–0.1%	28%
Portugal	44%	16%	14%	26%	1.0%	1.0%	0.15%	2.8%	2.1%	–0.8%	0.3%	–0.6%	2.2%	5%
Slovakia	43%	23%	16%	18%	0.3%	0.3%	0.02%	2.4%	0.9%	–0.4%	–0.2%	–0.1%	–0.1%	0%
Slovenia	47%	23%	8%	22%	0.7%	0.7%	0.04%	4.5%	1.7%	0.4%	0.3%	0.3%	0.0%	53%
Spain	26%	27%	22%	25%	8.5%	0.8%	0.20%	5.3%	–0.9%	–0.3%	0.1%	–0.2%	2.5%	14%
Sweden	40%	45%	0%	15%	0.4%	0.4%	0.03%	3.0%	0.4%	0.0%	0.4%	–0.2%	1.7%	0%
UK	53%	18%	1%	28%	0.9%	0.9%	0.08%	0.8%	–1.3%	1.0%	0.2%	0.6%	0.6%	65%
Average EU25	42%	31%	7%	20%	1.1%	0.8%	0.15%	1.9%	–0.2%	0.4%	0.6%	–0.2%	0.5%	47%

[a]See the Supporting Information on the Web for annual data as used in the modeling and details of compilation and detailed sources.
C&D = construction and demolition; Avg = average.

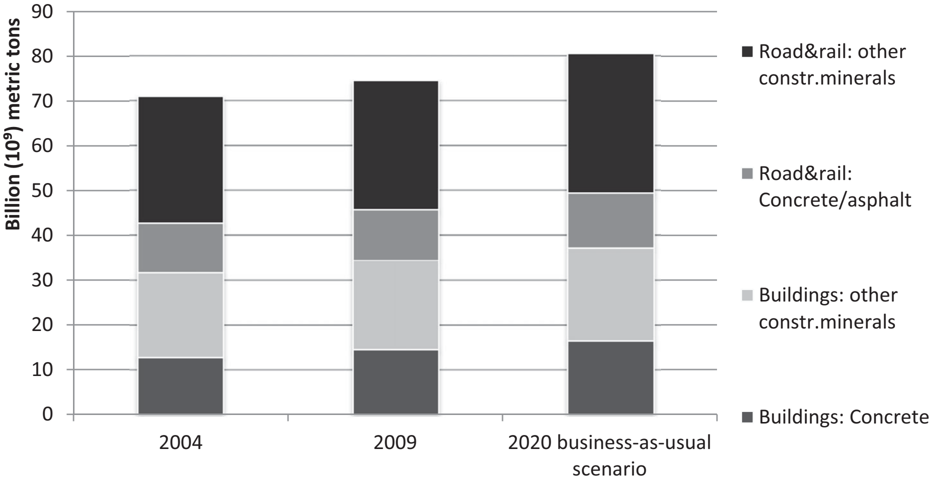

Figure 1 Stocks of nonmetallic minerals in residential buildings, roads, and railways in the EU25 (own calculations).

intensity (M) of these specific EOL stocks then yields the total material output from stocks by country (c), material (m), and each year (t) (equation 6).

$$MO_{c,m,t} = \sum_{S} \left(E\_EoL_{s,c,t} + E\_decl_{s,c,t}\right) * M_{s,c,m} \quad (6)$$

The amounts of recycled materials (MR) are calculated using country (c) specific recycling rates (*Recy_rate*) (equation 7). The remainder of the material output from stocks is then counted as construction and demolition waste, again not considering stock hibernation.

$$MR_{c,m,t} = MO_{c,m,t} * Recy\_rate_c \quad (7)$$

These recycling rates were sourced from a recent metastudy for the European Commission (EC), which also highlighted the difficult data situation in regard to EOL waste flows and recycling estimates (Monier et al. 2011; table 2).

Results on Material Stocks and Flows for the EU25

In-use stocks of nonmetallic minerals in roads are estimated at 39 billion tons, in railways at 1 billion tons, and for residential buildings at 35 billion tons in 2009 (figure 1). On average, per capita stocks therefore amount to 128 tons of nonmetallic minerals in roads, 72 tons in residential buildings, and 3 tons in railways.

Between 2004 and 2009, stocks were expanded, with an average annual net increase of 160 million tons stocked in roads, 3 million tons in railways, and 542 million tons in residential buildings (figure 1). For the business-as-usual scenario until 2020, the increase of material stocks slows down to 0.7% per annum, which is a result of the reduction of housing growth rates in Spain, Ireland, and France (table 2; *Methodology* section). Still, total estimated stock increases to 81 billion tons of nonmetallic minerals in 2020 in that scenario.

Modeled inputs into stocks amount to 1,989 million tons, of which annual maintenance of the roads network makes up the majority, with on average of 47% or 930 million tons per annum. Maintenance inputs into buildings amount to 204 million tons, which are 10% of total estimated inputs, on average, between 2004 and 2009 (figure 2a,b). Replacement construction only takes up a small share of total inputs— on average, annually 24 million tons of concrete inputs and 15 million tons of other nonmetallic minerals. Expansion of the housing stock used 28% or 548 million tons of inputs annually, on average, whereas the remaining 15% or 307 million tons are related to the expansion of roads and, to a minor share, railways. Further, the modeled annual buildings expansion flows decreased slightly by approximately 3% between 2004 and 2009, reflecting the effects of the banking/public debt crisis and subsequent economic recession from 2007 onward.

Overall, modeled outputs from stocks amount to, on average, 1,178 million tons, of which construction and demolitions waste from roads makes up the largest fraction, with, on average, 49% or 628 million tons annually (figure 2c,d). Of that, outputs from railway stocks make up only 23 million tons, or 2%, on average. Buildings demolition waste amounts to 110 million tons or 8% of total output from stocks, on average. Overall, recycled construction minerals amount to approximately 550 million tons, of which 81%, on average, results from maintenance and demolitions of the roads and railway network, whereas the remainder stems from buildings stocks.

The mass of estimated material flows into and from stocks are all in the range of only 0.8% to 1.6% of stocks (figure 3). The major share of net additions to stocks are the result of the expansion of the residential building stock (0.7%). In maintenance inputs, mostly roads and only lightly railways make up the majority (1.3%). For the outputs from stocks, materials from roads make up most of the recycling as well as waste.

Bottom-up Flows and Economy-wide Material Consumption in 2009

Bottom-up estimates of inputs into stocks can be compared with economy-wide consumption figures. Inputs flows into stocks (above) are modeled at 1,908 million tons in 2009,

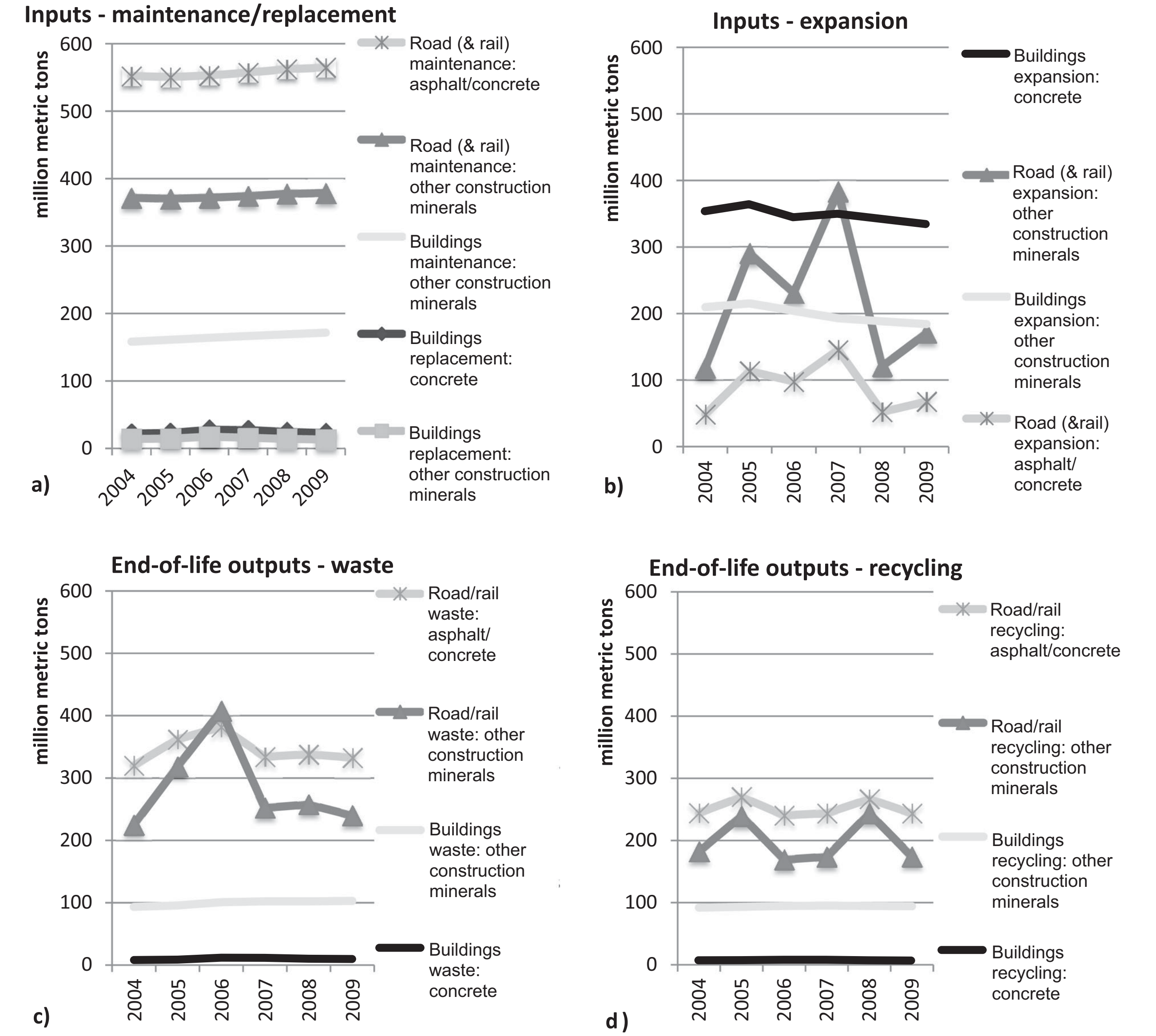

Figure 2 Material inputs and outputs from stocks. Strong fluctuations in road- and railway-related flows, are to some extent, the result of mixed data quality (own calculations; see the Supporting Information on the Web).

which would account for 61% of overall DMC of construction minerals in the EU25 (figure 4). Over the whole time period 2004–2009, it is 56%, on average. But not all of those inputs are virgin materials—one has to consider that the recycling flows are, to some extent, already replacing those. This means that if all the estimated recycled materials are used to replace inputs into stocks and are not used for other purposes not covered in this study, still 1,389 million tons of virgin materials, or 44% of the DMC, is required for stock maintenance and expansion.

These results raise the question of where the "remaining" 39% to 56% of DMC of construction minerals used annually are destined. Uncertainties in model parameters (lifetimes, material intensities, demolition, and recycling rates) definitely play a role here. Additionally, these estimates do not cover all societal material stocks, for example, bridges, ports, airports, tunnels, underground networks, and commercial and public buildings are not included.

Increased Recycling Resulting from the European Waste Framework Directive in an Otherwise Business-as-Usual Scenario for 2020

The European WFD (2008/98/EC) states, in Article 11, that "by 2020, the preparing for re-use, recycling and other material recovery, including backfilling operations using waste to substitute other materials, of non-hazardous construction and demolition waste [. . .] shall be increased to a minimum of 70% by weight," from the European average of 46% in 2004–2009

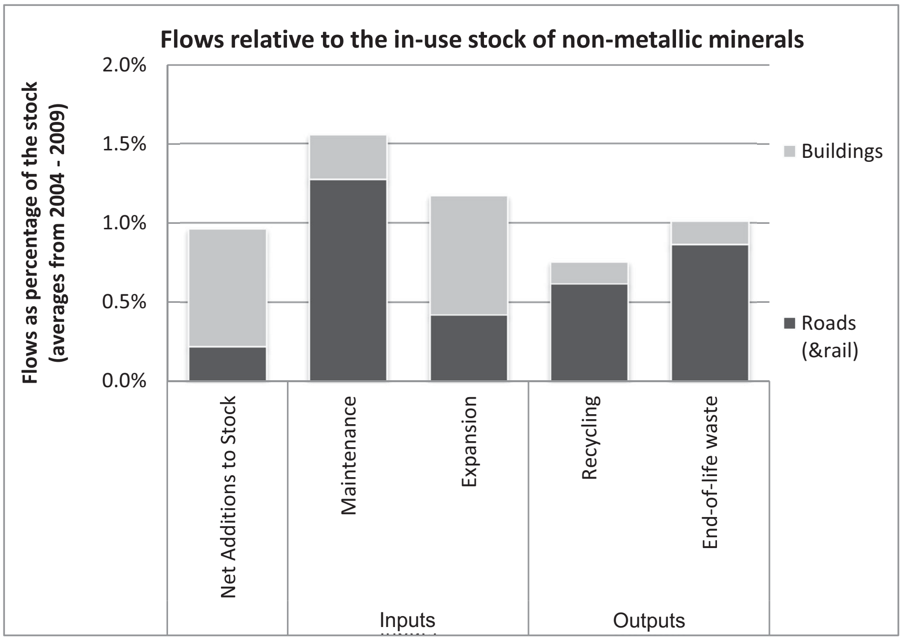

Figure 3 Modeled material flows and the quantitative relationship to the stock for 2004–2009.

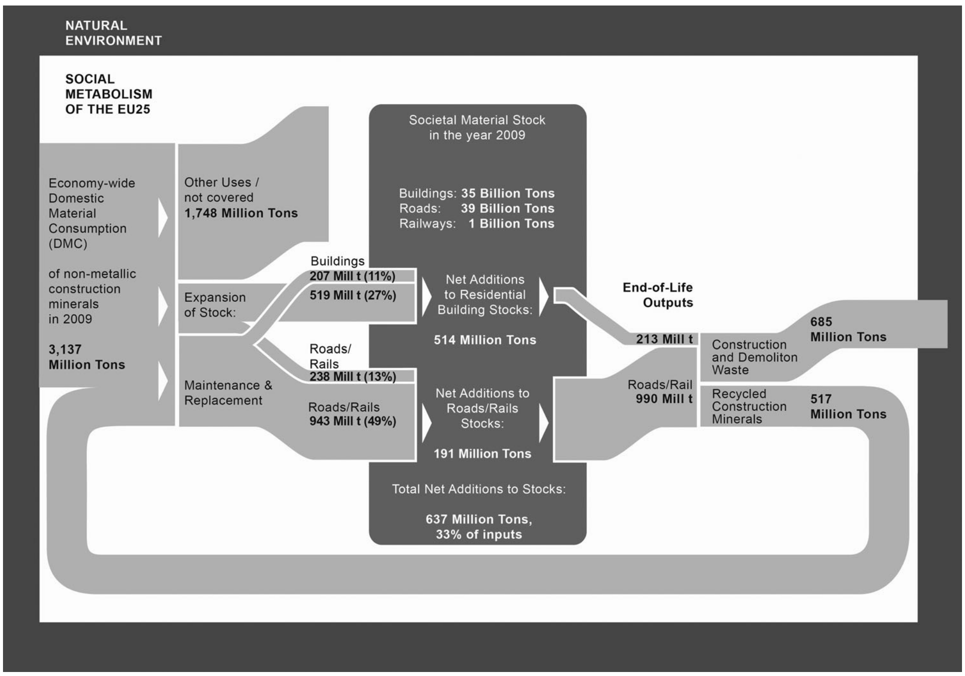

Figure 4 Economy-wide consumption of nonmetallic minerals versus estimated inputs and outputs from stocks of residential buidlings, roads, and railways in 2009 (own calculations; DMC of non-metallic minerals without "other products, salt and fertilizer"; Eurostat [2012a]). DMC = domestic material consumption.

(Monier et al. 2011; Mudgal et al. 2011). Using the data and model described above (tables 1 and 2; *Methodology* section; Supporting Information on the Web), a business-as-usual scenario is modeled, in which only recycling rates are increased according to the EU WFD targets, whereas all other factors, such as demolition and growth rates[3] as well as renovation and maintenance rates, are held constant (a so-called ceteris paribus assumption).

In such a business-as-usual scenario, annual inputs of non-metallic minerals into residential buildings and the road/railway network in 2020 are estimated at 1,829 million tons, which is a decrease of –4% compared to 2009 (figure 5). This decrease stems from reductions in expansion-related inputs (–22% in 2020 compared to 2009), the majority of which is a result of reduced housing expansion in the scenario (–46% reduction of housing expansion material inputs). Maintenance inputs

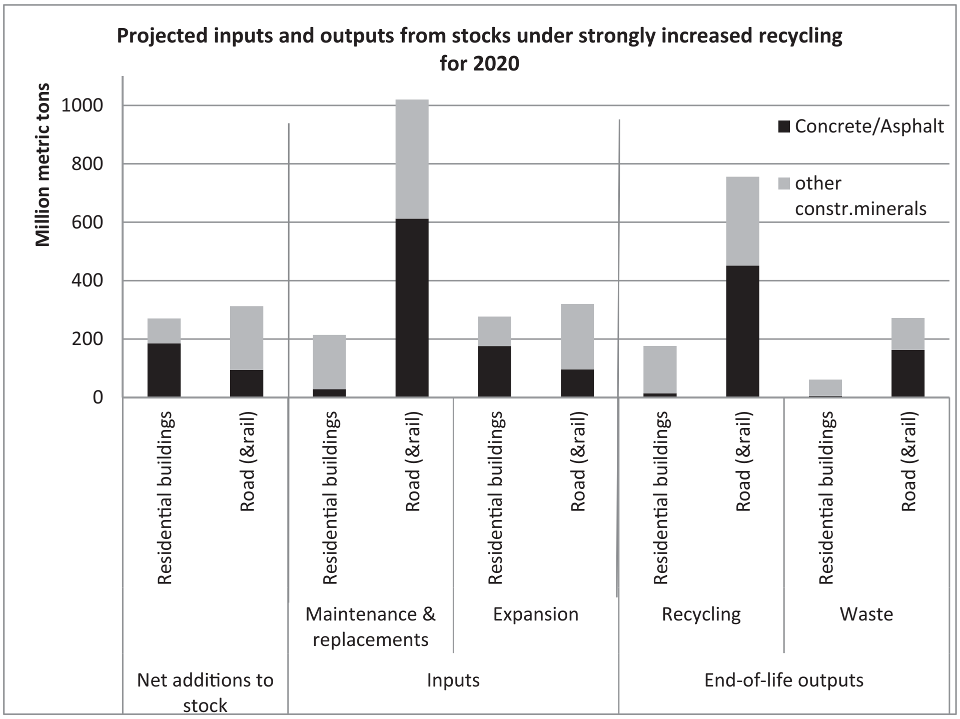

Figure 5 Projected results for the business-as-usual scenario exploring the effects of increased recycling under the Waste Framework Directive's goals in 2020.

increase in the scenario (8% from 2009) and make up the majority of the modeled inputs in 2020 (55%), where all the concrete and 8% of other construction minerals go into replacement construction. Modeled outputs from stocks amount to 1,264 million tons, an increase of 5% compared to 2009. Increased recycling rates in 2020 result in 932 million tons of recycled materials, an increase of 70% over 2009. This means that if the WFD is fully implemented until 2020, 51% of virgin inputs into the stocks of residential buildings, roads, and railways could be sourced from recycled materials, given the otherwise business-as-usual scenario of ongoing stock expansion (figure 5). If all recycled materials are used for maintenance alone, 75% of these flows could be covered, materials quality considerations put aside.

Verification of Results and Discussion of Limitations and Uncertainties

This study focuses on the European level and relies on a variety of European data sources, assumptions, and simplifications, all of which are outlined in the *Methodology* section and discussed in more detail in the Supporting Information on the Web. Major problems were low data quality on extent and change of dwelling stocks, which are collected at irregular intervals and published in the "Housing Statistics Reports." For roads, data gaps and statistical breaks are common (see the Supporting Information on the Web). Research and policy recommendations would greatly benefit from improved data gathering and harmonization.

The few existing estimates of construction minerals stocks in selected EU countries have arrived at 130 to 325 tons per capita, lower estimates covering only residential buildings and upper estimates including all infrastructure, buildings, and subsurface constructions (Bergsdal et al. 2007; Daxbeck et al. 2009; Müller 2006; Rubli et al. 2005). Estimates in this study cover residential buildings and roads/rails and arrive at 203 tons per capita in 2009 for the EU average, of which housing accounts for 36% (72 tons). This comparatively low value for building stocks can be partly attributed to an underestimation resulting from incomplete primary data: The typology of 72 buildings that was used represents only 80% of the EU residential dwelling stock (Nemry et al. 2008, table 2).

Further, stocks in road infrastructure also seem underestimated, where a study for Germany arrived at 6.2 to 8.2 billion (10^9) tons of stocks in the road system for 2009 (Steger 2012), whereas this study finds 5.5 billion tons in 2009, which is 12% to 32% lower. Germany-specific material intensities, which are slightly higher than those used in this study, are the cause. Generally, the uncertainties for road and rail estimates are higher than for buildings because of the lack of information on the age composition of roads, the need for more specific material intensities over time and by region, as well as the overall low data quality for communal and provincial roads, which actually constitute the majority of the network (see the Supporting Information on the Web for Details).

In regard to stock growth, a recent study presented direct estimates of the economy-wide "net additions to stock" for the

Czech Republic, arriving at 58 million tons of net additions of construction minerals in 2002 (Kovanda et al. 2007). Estimates presented herein for 2005, although focused on the EU25 aggregate, amount to only 5 million tons for the Czech Republic. Although a direct comparison of the two values is complicated by the fact that the results refer to different years and that additions to stocks can vary considerably from year to year, it is clear that this study does underestimate net additions as a result of the different scope. Several factors contribute here: (1) This study only covers 76% of residential buildings of the Czech Republic (table 2), 92) roads and railways are estimated without any supporting infrastructure (bridges, tunnels, train stations, and so on), and (3) this study does not cover various other infrastructure and public or commercial buildings, whereas Kovanda and colleagues (2007) use an economy-wide approach.

Recycling estimates of construction and demolition waste presented in this study agree well with a recent report to the European Environmental Agency (Tojo and Fischer 2011), which arrived at 554 million tons in 2007 for the EU27 versus the 488 million tons from this study for the EU25. As a waste estimate, this model yielded 685 million tons in 2007, which is 24% less than the 896 million tons reported by Tojo and Fischer (2011). Monier and colleagues (2011, 15) further conclude, in a review of the literature, that 310 to 700 million tons of construction and demolition waste for the EU27 is a plausible range. Eurostat recently included a first estimate of construction and demolition waste for the EU25 for 2010 in their database, which is 330 million tons (Eurostat 2013). Uncertainties for lifetimes and maintenance rates, material intensities, and demolition and recycling rates have to be mentioned here as issues. Further, the explicit calculation of road maintenance as flows, which involves in situ recycling, which usually only counts as waste if moved from the site in other studies, does affect the results. Additionally, "hibernating" stocks no longer in use, but not turned into a waste flow, as well as the ratio of below- and above-ground stocks can play an important role in explaining these differences, given that they strongly influence the amounts of EOL stocks actually turning into waste flows (Hashimoto et al. 2009). Overall, the results of this study for waste and recycling are higher than the Eurostat numbers, but well within the ranges discussed by other European studies.

Summarizing, the results obtained from our model are lower than what more detailed country-level studies find, although the overall size of comparable stocks is quite similar and waste and recycling estimates are in a very plausible range; the reasons for the underestimation in this study can be attributed to incomplete data at the EU scale, generally conservative assumptions, and no explicit treatment of stock hibernation, for which no data exist. Inclusion of commercial and public buildings, as well as other infrastructure, such as underground networks, sewers, ports, dams, and so on, would also definitely refine the results and substantially increase total stock estimates, as has been demonstrated for specific countries (Rubli et al. 2005; Hashimoto et al. 2007, 2009; Steger et al. 2011). An economy-wide approach therefore would constitute an important further step.

Outlook and Remaining Research Gaps

The majority of residential buildings in the EU25 have been constructed between 1945 and 1990 and their service lives are expected to end during the mid-twenty-first century, requiring strongly increasing replacement construction, recycling, and waste treatment (Bergsdal et al. 2008; Müller 2006). Based on past housing survival, these projections assume 60 to 120 years of service life. But empirical demolition rates for buildings are generally very low, with a EU25 average of 0.15% (table 2, 0.01–0.5%; Thomsen and van der Flier [2009]). Implicitly, this would be a lifetime for the entire stock of 667 years (inverse of demolition rate), which is not a realistic figure, but shows that this projected strong increase of demolitions is not yet happening. Because demolitions are usually driven by other factors than actual technical EOL (Thomsen and van der Flier 2011), lifetime extension and increased focus on renovations and refurbishments are an important policy option toward more sustainable and efficient resource use. In this way, the material input as well as the waste and recycling side could be tackled while also making the building stock more energy efficient (Pauliuk et al. 2013a). Additionally, also roads, but probably also other infrastructure, have to be included in considerations toward more sustainable resource use, given that they constitute a large part of the material stock and also induce large material flows (Hashimoto et al. 2009; Schiller 2007; Steger 2012).

At this stage of research, no explicit recycling loops or cascading uses are included, rather a comparison of the magnitudes of material flows is discussed. As a next step, one would need to consider first the temporal and spatial distribution of stocks and flows resulting from the severe economic limitations on transporting and storing large quantities of recycled construction and demolition flows. Second, specific material qualities and their quantities would need to be included to gain an understanding of the actual feasibility for recycling and usability. This goes beyond this study and current data availability and is a topic for further research.

Further, an inverse relationship between buildings density and material stocked in roads and other infrastructure has been reported (Schiller 2007; Pauliuk et al. 2013b), indicating the importance of spatial planning and reducing urban sprawl. Careful densification of settlements is also being advocated for climate mitigation strategies in order to reduce the energy requirements of transportation (Newman and Kenworthy 1999; Wiedenhofer et al. 2013 and references herein). In combination, this could lead to reduced traffic loads, prolonging the lifetimes of road infrastructure and decreasing their large maintenance requirements and monetary burden on state budgets. This indicates that substantial cobenefits toward more sustainable material use and climate strategies might exist, which have to be understood more clearly (Deilmann 2009; Weisz and Steinberger 2010; Allwood et al. 2011).

Conclusions

We presented a dynamic model of material stocks in residential buildings and the road/railway network and the material flows going into their maintenance, as well as net expansion, for the EU25. Results are on the conservative lower side, compared to other studies, with per capita stocks estimates for nonmetallic minerals at 128 tons in roads, 72 tons in residential buildings, and 3 tons in railways in 2009 for the EU25.

Interestingly, maintenance-related material inputs into the stocks covered in this study amount to 34% to 58% of domestic material consumption of nonmetallic minerals in the EU25 in 2009, depending on how recycling is handled. The majority of these flows are the result of maintenance of roads, then for the renovation of buildings. Additional net expansion of stocks amounts to approximately another 28% of DMC of nonmetallic minerals, of which the majority is used for additional residential buildings. Overall, the results indicate a significant commitment of annual resource use for maintaining existing stocks.

In 2020, strongly increased recycling in an otherwise business-as-usual scenario has been estimated to only cover 51% of material input flows into residential buildings and roads/rails resulting from ongoing stock expansion. But, if all recycled materials are used for stock maintenance alone, 75% of these flows could be covered. This scenario is based on trend extrapolations from 2003 to 2009 and an increase of recycling rates for construction and demolition waste from the European average of 47% during that time toward the WFD's goal of a minimum of 70% in 2020.

Based on the results presented in this article and in line with other studies (Hashimoto et al. 2007; Müller 2006; Shi et al. 2012), the following insights emerge: The size of stocks as well as their service lifetimes are the two most important factors driving material use necessary for renewal and maintenance. This, in turn, means that a reduction of material use would be most easily achieved through a stabilization of existing stocks ("steady stocks") and an effort to prolong lifetimes of standing infrastructure and buildings. Preliminary results for European roads suggest that these stocks are a major driver of resource use, and their maintenance and net expansion need to be considered critically.

Finally, serious efforts toward more sustainable patterns of society-nature interactions need much more systematic considerations on how to use the substantial material stocks already in use in industrialized countries more efficiently and much longer. The majority of these structures have been accumulated during the rapid acceleration of the fossil-fuel–based system of high resource throughput (Fischer-Kowalski 2007; Pauliuk and Müller 2014), when sustainability was not an issue. Moving toward resource-saving longer lifetimes, improved maintenance, and renovation practices, additionally to an overall stabilization or sometimes even shrinking of material stocks and flows, therefore amounts to a critical paradigm shift (Allwood et al. 2011; Boulding 1966).

Acknowledgments

Many thanks go to Prof. Fridolin Krausmann for his ongoing support and encouragement and to the anonymous reviewers for their detailed and constructive comments, which greatly helped to improve the manuscript. The authors also thank the student research assistants Selina Schall, Monika Riegler, Christian Dorninger, and Sebastian Pawell for their thorough data work. We also thank Klaus Ringelhann for his support with the design of Figure 4. The beginnings of this study were conducted under the framework contract agreement ENV.G4/FRA/2007/0067; additional support has been granted by the Austrian National Science funded project "GLOMETRA–the global metabolic transition" (P 21012) and the EU FP7–funded project NEUJOBS (266833). Any remaining errors or mistakes are the sole responsibility of the authors.

Notes

1. Only in few countries and specific years was a decline of infrastructure extent found and then only mostly for single-track railway lines and minor roads. These are deducted from the time series E and turn into end-of-life waste.
2. End-of-life buildings are replaced by a building type of current standards, that is, a single-family house from 1960 is replaced by a single-family house type of current building standards and designs.
3. Housing growth rates for Spain, Ireland, and France have been reduced to the remaining EU average for 2010 onward, to take the economic recession into account (see *Methodology* section).

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Supporting Information

Additional Supporting Information may be found in the online version of this article at the publisher's web site:

Supporting Information S1: This supporting information contains the procedures and data sources used to compile the data sets on the residential buildings stock and road/railway network of the EU25 states.

RESEARCH AND ANALYSIS

How Circular is the Global Economy?

An Assessment of Material Flows, Waste Production, and Recycling in the European Union and the World in 2005

Willi Haas, Fridolin Krausmann, Dominik Wiedenhofer, and Markus Heinz

Keywords:

circular economy
energy transition
industrial ecology
material flow accounting
recycling
sustainable resource use

Supporting information is available on the *JIE* Web site

Summary

It is increasingly recognized that the growing metabolism of society is approaching limitations both with respect to sources for resource inputs and sinks for waste and emission outflows. The circular economy (CE) is a simple, but convincing, strategy, which aims at reducing both input of virgin materials and output of wastes by closing economic and ecological loops of resource flows. This article applies a sociometabolic approach to assess the circularity of global material flows. All societal material flows globally and in the European Union (EU-27) are traced from extraction to disposal and presented for main material groups for 2005. Our estimate shows that while globally roughly 4 gigatonnes per year (Gt/yr) of waste materials are recycled, this flow is of moderate size compared to 62 Gt/yr of processed materials and outputs of 41 Gt/yr. The low degree of circularity has two main reasons: First, 44% of processed materials are used to provide energy and are thus not available for recycling. Second, socioeconomic stocks are still growing at a high rate with net additions to stocks of 17 Gt/yr. Despite having considerably higher end-of-life recycling rates in the EU, the overall degree of circularity is low for similar reasons. Our results indicate that strategies targeting the output side (end of pipe) are limited given present proportions of flows, whereas a shift to renewable energy, a significant reduction of societal stock growth, and decisive eco-design are required to advance toward a CE.

Introduction

While resource use globally is growing at high rates and has even accelerated in the last decade (Schaffartzik et al. 2014), it is becoming evident that the scale of humanity's metabolism is unsustainable and must be reduced. The material and energy resources required to extend the current metabolic pattern of the industrial countries to the rest of the world are most likely not available, nor are the capacities of global ecosystems sufficient to absorb the outflows of industrial metabolism (UNEP 2011a; WBGU 2011). In this context, the notion of a circular economy (CE), in which material flows are made up either of biological materials, which after discard are available for ecological cycles, or of materials designed to circulate within the socioeconomic system (SES) with reuse and technical recycling as a key strategy (GEO5 2012), has gained momentum. In the debate about pathways toward a more sustainable industrial metabolism, the CE appears to be a promising strategy to meet the environmental and economic challenges of the early twenty-first century and define targets of sustainable resource use (Allwood et al. 2010; Chen and Graedel 2012; Ellen MacArthur Foundation 2013; Hislop and Hill 2011; Mathews and Tan 2011; Moriguchi 2007; Preston 2012). The CE is promoted by many governments and international organizations and is considered instrumental in the mitigation of greenhouse gas emissions (e.g., EC 2012; PRC 2008; METI 1991).

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DOI: 10.1111/jiec.12244 Editor managing review: Heinz Schandl

Volume 00, Number 0

In response to signs of resource depletion and sharp increases in both prices and related volatilities of raw material supply, promoters of the CE further argue that increasing the circularity of the physical economy is indispensable for maintaining future resource security (e.g., Hislop and Hill 2011).

A critical examination of the literature on the CE reveals a lack of precise definitions and criteria for assessing measures to improve the circularity of the economy. In this article, we refer to a simple definition used, for example, in the United Nations (UN) GEO5 report, which states that, in a CE, material flows are either made up of biological nutrients designed to re-enter the biosphere, or materials designed to circulate within the economy (reuse and recycling) (GEO5 2012).

Assessing the circularity of an economy based on these criteria, however, warrants caution[1]: In particular, the inclusion of all biomass as a "circular" material flow seems problematic and implies that biomass is produced in a renewable way and that all waste flows and emissions effectively re-enter ecological cycles. When the production of biomass is associated with net carbon emissions, loss of soil nutrients, or the depletion of nonrenewable water resources, as is often the case, biomass cannot be regarded as a circular flow proper. In practical terms, however, it is difficult to assess which share of the global biomass production meets the criteria required for a CE.

In principle, circularity can be advanced by different strategies. Alongside closing loops through recycling and reuse, a shift from fossil to renewable energy sources and translating efficiency gains into a reduction of the overall level of resource consumption is required. Recycling is, in practice, still the most widespread strategy employed to achieve a CE. For some materials, recycling is already very advanced (e.g., metals, paper, and glass) while for others, such as construction and demolition, waste considerable efforts are made to increase recycling rates (Graedel et al. 2011; Mugdal et al. 2011). But not in all cases does recycling lead to an effective reduction of material use: Energy requirements for recycling can be high, the lower quality of secondary material can lead to increased virgin material demand, or secondary materials may not be used to substitute virgin materials, but may instead drive the production of new low-price products (Moriguchi 2007). Thus, considering the wide variety of different CE strategies for different material flows and their interdependencies, it becomes increasingly important to establish frameworks on how to assess not only specific measures and improvements, but also their overall contributions both to closing material loops within the economy and making use of ecological material cycles.

The assessment presented in this article is an attempt to frame and substantiate the discussion by applying a systemic and sociometabolic perspective to assess the current level of circularity of the global economy. We define and quantify a set of key indicators to characterize the circularity of national economies and apply it to the global economy and the European Union (EU-27).

In the next section, we lay out the conceptual foundations of the material flow model we are using to analyze material flows and briefly describe the database and the assumptions we made. This is followed by a presentation of the empirical results of the circularity of global economy and the EU-27 in 2005. Based on these results, we then discuss, for each of the four main material groups, the current state of circularity and the potentials and limitations for further improvement and draw some general conclusions for further progress toward a CE.

Methodological Approach

Figure 1 shows a simple model of economy-wide material flows and depicts the different flows and processes that were quantified in this study to assess the circularity of the economy. The model we use is based on the conceptual framework and the system boundaries applied in economy-wide material flow accounting (MFA) (Eurostat 2012). It defines the flow of materials from extraction and import, by processing, immediate consumption, or temporary accumulation in material stocks to recycling or final treatment before all materials finally leave the SES as waste and emissions.

Flows were estimated for the global economy and the EU-27 for the year 2005. Material flows were calculated at a detailed level of 47 material groups following the Eurostat classification of MFA (Eurostat 2012). Rather than assessing circularity for specific materials or substances, this study aims at a comprehensive picture, taking all materials into account. Results are therefore presented at the level of main material groups: biomass, fossil fuels (FFs), metals, waste rock, and industrial and construction minerals. Table 1 provides an overview of the literature and the sources used to derive the different coefficients to estimate flows or formulate assumptions.

Inputs into the economic systems comprise domestically extracted materials and imports. A fraction of inputs is exported. We define domestically processed materials (PMs) as the sum of apparent domestic consumption of materials (DMCs; extraction plus imports minus exports) and recycled materials. Data on domestic extraction, imports, and exports were derived from a global economy-wide material flow database (Schaffartzik et al. 2014). From materials processed, we distinguish three pathways of material flows of high relevance for the CE: energetic use; waste rock; and material use.

Energetic use comprises all materials that are used for energy production. This includes the combustion of energy-rich materials, such as wood, coal, oil, or gas, to provide technical energy[2] and applies to the largest fraction of all fossil materials (except for a small share used in material applications, such as plastics or bitumen) and a comparatively small fraction of biomass (e.g., fuel wood and biofuels[3]). We also consider agricultural biomass used to feed humans or livestock to provide metabolic energy in the catabolic processes in the human body and livestock as energetic use.[4] All fossil and biomass materials used to provide energy are converted into gaseous emissions (mainly carbon dioxide [CO_2]) and other residues (combustion residues and excreta) and become domestic processed output (DPO; see below). None of these residues can be recycled within the economy in the sense that they can be used again for the original purpose. To a limited degree, cascade utilization is possible, for example, when dung is used as fuel or to produce biogas or when ash is used in chemical processes. In the MFA system, excreta or biowaste used as fertilizer is not considered as recycling within the SES in MFA, but as an output that (potentially) enters ecological material cycles within the biosphere.

Waste rock (from metal ore processing) is a flow of considerable size, which goes straight from processed materials to DPO. MFA reports metal extraction in terms of gross ore and metal content. While the extracted metal is further used within the economic process, waste rock and tailings are discarded. This flow is a major waste flow, which, with few exceptions, does not qualify for recycling.[5]

Material use comprises of all other materials, that is, all metals and nonmetallic minerals and the fractions of biomass and fossil energy carriers not used for energy generation. Material input data from the detailed global material flow database were allocated to energetic or material use according to their material properties. For material flows where the resolution of the global material flow database did not allow for this distinction to be made, we used additional data from production statistics, for example, FAO (2013) for wood products and Plastics Europe (2012) and IEA (2013) for petroleum products.

The material use fraction is further split into two pathways based on average product lifetime: We distinguish between materials that are used within 1 year (throughput materials) and materials that remain in the SES for a longer period of time, that is, they add to stocks of artefacts (stock-building materials). Throughput materials become end-of-life (EOL) waste within a year, and the largest part of this fraction is potentially available for recycling after use. Typically, these are consumer goods, such as packaging, newspapers, batteries, plastic bags, and so on. In contrast to these consumables, by far the largest amount of materials is used to build up and maintain long-life stocks of buildings, infrastructures, and other long-life goods, which remain in the socioeconomic system as in-use stocks for more than a year. This flow is denoted as "addition to stocks" and is not immediately available for recycling, but remains in use for a period of 1 year to several decades until it is discarded and becomes EOL waste. Based on a literature survey and data from production statistics (e.g., for plastics and paper), we made material-specific assumptions to estimate the stock building fraction of a material (stocking rate), for example, for construction wood, paper, plastics, iron, aluminium, and other metals (see table 1).

Annual Discard of Stock Building Materials

Several studies indicate that economies still increase their physical stocks (Hashimoto et al. 2007; Pauliuk and Müller 2014; Wiedenhofer et al. 2015; Fishman et al. 2014), while, at the same time, a considerable amount of stocks that reach their EOL time each year are discarded or demolished. To estimate the annual amount of discarded stocks, we used data from stocks and flow literature that is available for specific materials, such as iron or construction minerals, on the global and/or regional level. For materials where this type of information was not available (e.g., wood, plastics, and tin), we applied a simple so-called delayed model, which states that outflow from a stock at a given time t equals the inflow from year t minus the average lifetime of the stocks in years (Voet et al. 2002):

$$\text{Outflow}\,(t) = \text{Inflow}\,(t - \text{life time})$$

We estimated lifetimes based on literature and used the corresponding historic inflow data from the global material flow database (Schaffartzik et al. 2014).

End-of-Life Waste: We assume that all discarded stocks become EOL waste at the end of their lifetime. We do not distinguish between in-use stocks and hibernating stocks, that is, stocks that are not demolished, but remain in place unused (Hashimoto et al. 2009; Wallsten et al. 2013). The amount of EOL waste equals the amount of materials potentially available for recycling, reuse, or downcycling.

Recycling is defined as any recovery operation by which EOL waste is reprocessed into products, materials, or substances that can serve the original or comparable purposes (EP&C 2008). We estimate the amount of recycled materials on the basis of statistical data and recycling rates published in the scientific literature (see table 1). In this context, downcycling also plays an important role, which can be defined as the reprocessing of EOL waste into products of inferior quality, compared to the primary material, for example, concrete being crushed into aggregate. In practical terms, data on recycling flows often do not allow us to distinguish between re- and downcycling. We assume that, in particular, the recycling flow of construction minerals includes a considerable amount of downcycling. Case studies suggest that construction and demolition waste is often used in applications with reduced quality demands such as backfilling. Given that there is a lack of data, downcycling is subsumed under recycling in this study. We therefore overestimate the recycling flow proper.

DPO comprises all wastes and emissions that leave the SES. In order to be able to close the material balance, we do not account for DPO in their actual form as suggested by MFA guidelines (e.g., as CO_2) (Eurostat 2012), but, for reasons of simplicity, we exclude changes in mass flows resulting from oxidation or changes in moisture content.[6]

To assess the circularity of an economy based on the material flows shown in figure 1, we propose a set of key indicators:

a) Material size: PMs (gigatonnes [Gt] and tonnes per capita [t/cap])
b) Stock growth: Net addition to stocks as share of PMs (%)
c) Degree of circularity within the economy: recycling as share of PMs (%)
d) Biodegradable flows: biomass as share of PMs (%)
e) Throughput: DPO as share of PMs (%)

It is further important to note that an assessment of the CE needs to take the issue of spatial and temporal scales into account. It is not straightforward over which period of time and at what spatial scale circularity should be optimized, but this is

Table 1 Sources for data and assumptions used to calculate material flows shown in figure 1 by main material groups

Main material group	*Domestic extraction*	*Trade flows*	*Allocation to material or energetic use*	*Additions to stocks*	*Demolition and discard of stocks*	*Recycling*
Biomass			Primary crops and crop residues: assumptions based on FAO commodity balances and Krausmann and colleagues (2008)	According to use	Estimates based on assumption of lifetime delay model, see Van der Voet and colleagues (2002)	n.a.
			Wood: FAO (2013)	According to use	Estimates based on own assumptions re. lifetimes	FAO (2013)
Fossil fuel carriers			Crude petroleum: Plastics Europe (2012)	According to use	Estimates based on own assumptions regarding lifetimes	Estimates based on Plastics Europe
			Natural gas: Wood and Cowie (2004)			
Metals (content)	All: Schaffartzik and colleagues (2014)		n.a.	Iron: Wang and colleagues (2007)		
			n.a.	Aluminium: Cullen and Allwood (2013); Bertram and colleagues (2009)		
			n.a.	Other metals: Allwood and colleagues (2010)	Wang and colleagues (2007)	Hislop and Hill (2011); UNEP (2011b)
Waste rock			n.a.	n.a.	n.a.	n.a.
Industrial minerals			n.a.	Authors' assumptions	Authors' assumptions	Authors' assumptions
Construction minerals			n.a.	Estimates based on Wiedenhofer and colleagues (2015); Fishman and colleagues (2014); Kapur and colleagues (2009)		Monier and colleagues (2011); Mugdal and colleagues (2011)

Note: n.a. = not applicable

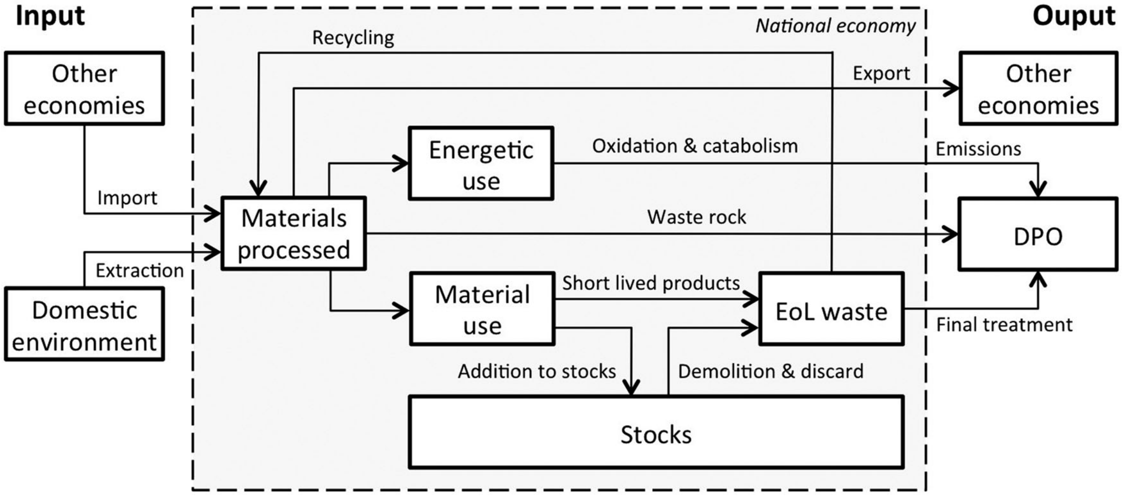

Figure 1 General model of economy-wide material flows from resource inputs imports and extraction to outputs of wastes and emissions and exports. All flows shown in the model have been quantified to assess the key characteristics of the circular economy. EoL waste = end-of-life waste; DPO = domestic processed output.

rarely discussed. We have chosen to assess circularity for a specific year (2005) and at a global scale. The observation period of 1 year has been chosen for practical reasons (MFA system boundaries and data availability), but it allows to capture the interplay of long-living stocks and annual flows and their impact on circularity only to a limited extent. The global scale chosen in this article provides a very comprehensive picture, but, ultimately, a multiscale perspective is required. It is important to observe and improve the CE at various levels, and the objectives for the CE may differ for different materials at different scales.

Robustness of the Estimate

The estimate of the different material flows entails considerable uncertainties. For some material groups, such as many metals, fossil energy carriers, and biomass, a broad knowledge of the material system and solid data exist. For some flows and some materials, the data situation is less satisfying and the level of uncertainty is considerable, in particular for recycling rates and flows of construction minerals. In a review, Monier and colleagues (2011 15) conclude, for example, that the available data and estimates of construction and demolition waste for the EU-27 vary by a factor 2. To estimate the different material flows, we used the best available information based on a broad literature survey. In general, we used assumptions that rather overestimate the degree of circularity of the economy. This refers, in particular, to the assumed rates for discard and recycling, which are at the upper limit. Further, the inclusion of all biomass as a circular material flow, regardless of the way this biomass is produced or how biomass wastes are discarded, overestimates the actual degree of circularity.

Although the level of uncertainty for specific materials may indeed be considerable, we assume that, for the overall aim of the article, which is to provide a rough, but comprehensive, assessment of the global economies circularity at the level of main material groups, the reliability of the data and our estimates is sufficient.

Current State of the Global Economy's Circularity

Based on a quantification of the different material flows shown in figure 1, we can make a rough assessment of the degree of circularity of the global economy at the turn of the twenty-first century. Figure 2 presents the size of the material flows in the year 2005 for the global economy and the EU-27 in the form of a Sankey diagram. In 2005, 58 gigatonnes per year (Gt/yr) of extracted raw materials entered the global economy. Together with 4 Gt/yr of recycled material, this added up to a total of 62 Gt/yr of processed materials (see table 2). Forty-four percent of all processed materials (28 Gt/yr) were used to provide energy through combustion or catabolic processes in humans and livestock and were converted into gaseous emissions or solid wastes leaving the SES as DPO. Another 6% of the processed material left the SES as waste rock or tailings from ore processing. This leaves 30 Gt/yr having entered the production process for material use. Of these, 4 Gt/yr were used in goods with a lifetime shorter than 1 year and 26 Gt/yr (or 43% of all processed materials) were added to stocks of buildings, infrastructures, and other goods with a lifetime longer than a year. This large flow of additions to stocks was accompanied by 9 Gt/yr of discarded stocks, which results in a total of 17 Gt/yr of net additions to stocks in 2005. According to our estimate, the total EOL waste flow from material use sums up to 13 Gt/yr. This amount of materials, which corresponds to one fifth of all material inputs, was potentially available for recycling and reuse in 2005. We estimate that roughly one third of this waste flow (4 Gt/yr) was actually recycled or downcycled, and the

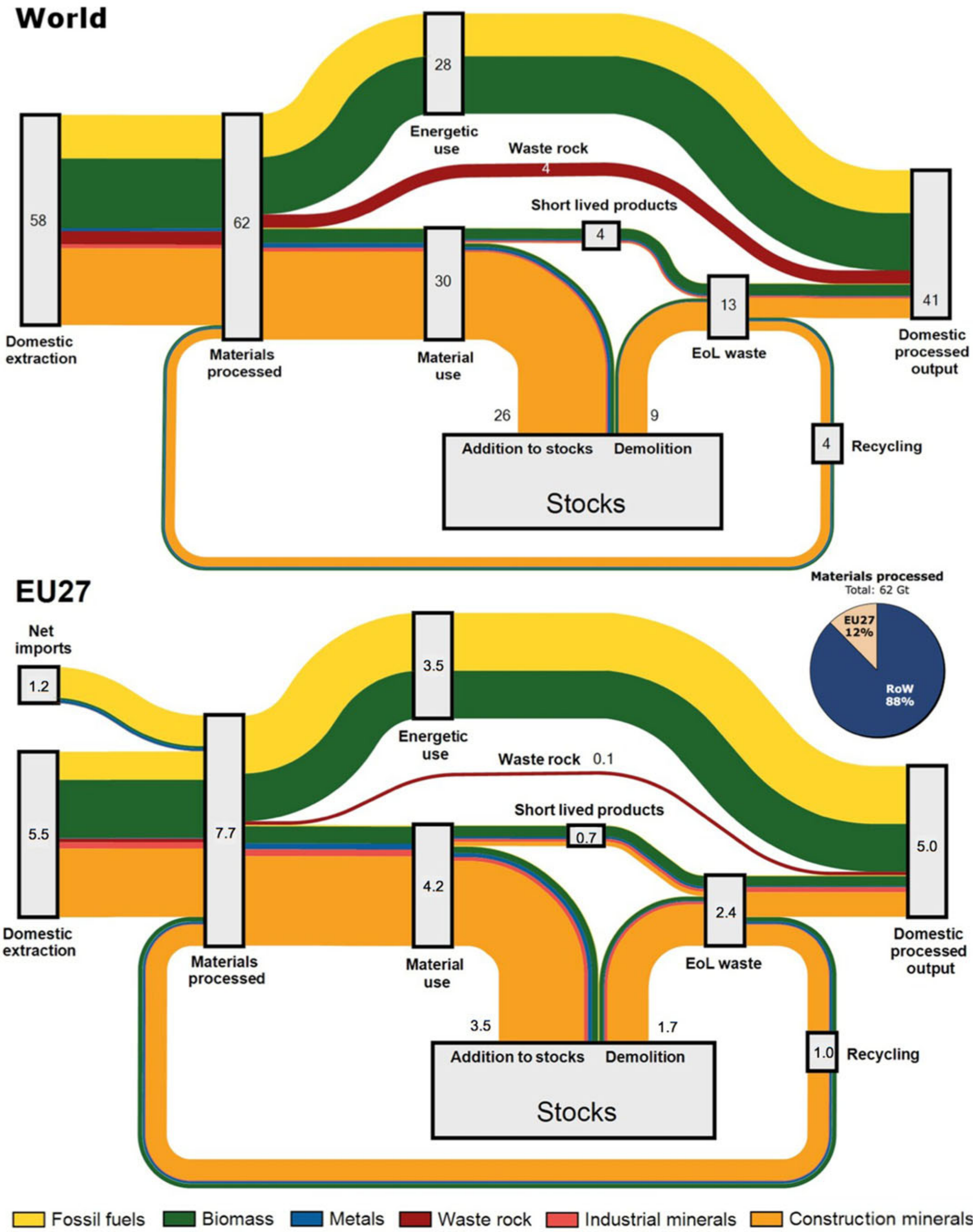

Figure 2 Sankey diagram of material flows through the global economy (world) and the EU-27 in 2005. Numbers show the size of flows in Gt/yr. For a definition of flows, see the article text. EU = European Union; EoL waste = end-of-life waste; Gt/yr = gigatonnes per year; RoW = rest of the world.

remainder was disposed to the environment directly or after treatment in waste plants and left the SES as gaseous, liquid, or solid outputs. A considerable fraction of this flow may also have remained in place as unused (hibernating) stocks (Hashimoto et al. 2009; Pauliuk et al. 2013; Wallsten et al. 2013). When related to the total material input (processed materials), the aggregate recycling rate shrinks to 6%.

From such a system-wide metabolic perspective, the degree of circularity of the global economy measured as the share of actually recycled materials in total processed materials appears to be very low, at 6%. The vast majority of all processed materials (66%) left the global economy as wastes and emissions and a large fraction (27%) were net additions to stocks of buildings, infrastructures, and other long-life goods. These materials become available for recycling only after longer periods of time, often after decades. Materials used for energy provision dominate the inputs (44% of all processed materials). This large material flow does not qualify as recycling proper within the economy at all. However, if we follow the common definition of the CE, biomass is considered a cyclical flow owing to the fact that all

Table 2 Circular economy indicators for the world and the EU-27 in 2005 (see figure 1 and the article text for definitions; per capita values are shown in figure S2 in the supporting information on the Web). Indicators above the horizontal dividing line are the proposed set of key indicators. The indicator below the line signifies a potential and others provide more detailed information

Indicator	*Unit*	*World*	*EU-27*
PM	Gt	61.9	7.7
	t/cap	9.6	15.8
Net addition to stocks as share of PM	%	28%	22%
Recycling within the economy as share of PM	%	6%	13%
Biomass as share of PM	%	32%	28%
Domestic processed output as share of PM	%	66%	66%
Flows either biodegradable or recycled in economy as share of PM	%	37%	38%
Fossil energy carriers as share of PM	%	19%	26%
Material for energetic use as share of PM	%	44%	46%
Material for material use as share of PM	%	50%	54%
Waste rock as share of PM	%	6%	1.5%
Short-lived products as share of PM	%	7%	9%
EOL waste as share of PM	%	21%	31%
Recycling as share of EOL waste (overall recycling rate)	%	28%	41%

Note: PM = processed material; EOL = end of life; Gt = gigatonnes; t/cap = tones per capita; EU = European Union.

biomass waste products re-enter the biosphere and are available for ecological cycles (CO_2, plant nutrients, and manure) and new biomass production. Hence, combining economy-internal technical cycles and economy-external ecological cycles by including all biomass yields a level of circularity of 37% globally. Considering that global biomass production is associated with deforestation, net CO_2 emissions, and soil degradation or that a considerable fraction of plant nutrients is lost to global sinks (Cordell et al. 2009; Rosegrant et al. 2009; Vermeulen et al. 2012), the actual degree of circularity of biomass is much lower. Thus, the overall level of circularity of 37% rather stands for a maximum current level and considerably overestimates the circularity of the global economy.

The EU-27 is among the regions taking the lead with respect to policies of sustainable development and sustainable resource use, but is also a major consumer of resources and producer of emissions. In 2005, the EU-27 accounted for 7.5% of the global population, but used 12.4% of the globally extracted materials. The highly industrialized region had approximately 30% of global gross domestic product (GDP) which, in capita terms, was in average US$28,600 in 2005 (in constant 2005 prices; UN 2014), approximately 200% above the global average. Average material use per capita amounted to 15.8 t/cap/yr and was 64% above the global average. The EU-27 is further a net importer of materials, which amount to roughly 20% of its DMC (Schaffartzik et al. 2014). The high import rate also indicates that a considerable amount of waste production associated with European consumption may occur elsewhere in the world (Wiedmann et al. 2013; Bruckner et al. 2012) for example, the comparatively small flow of waste rock is owing to the high import of processed metals). Figure 2 shows the size of the different material flows in the EU-27. Of the total amount of processed materials of 7.7 Gt/yr, roughly 54% went into material use, of which additions to stocks accounted for 80%. In the EU, a larger share of stocks reached EOL, compared to the global average, and the flow of discarded stocks amounted to 50% of additions (compared of 33% globally). Nevertheless, per capita net additions to stocks in the EU were, at 3.4 t/cap/yr, still much higher than the global average of 2.7 t/cap/yr (see figure S2 in the supporting information available on the Journal's website). Recycling in the EU is advanced. A total of 2.0 t/cap/yr of materials were recycled in the EU in 2005, which corresponds to 41% of EOL waste, compared to a global average of 0.6 t/cap/yr or 28% of EOL (figure S2 in the supporting information on the Web). The aggregate recycling rate (recycled material as share of processed material) was, at 12.6%, roughly twice as high as the global average. But, in spite of a higher recycling rate, DPO is large and amounted to 10.4 t/cap/yr or 66% of processed materials, as compared to 6.3 t/cap/yr in the global average. Including all biomass flows as circular flows results in a degree of circularity of 39%. But, also for the EU-27, biomass production cannot be regarded as fully circular, as discussed above. Whereas the overall degree of circularity of the EU economy is surprisingly similar as the global value, owing to the fact that the lower share of biomass in the EU's metabolism is balanced by higher recycling rates, also the size of the flows needs to be taken into account: The flows that are in a material loop within the economy or that are biodegradable, as the definition of the CE demands, amount to 6.8 t/cap/yr in the EU-27 and 3.5 t/cap in the global economy. But also the noncircular flows are much larger in the EU-27, at 6.4 t/cap/yr, as compared to 3.4 t/cap/yr globally, which indicates the significance of downscaling the overall size of social metabolism, in particular, in industrial countries in addition to advancing the degree of circularity.

Challenges for a Global Circular Economy

In 2005, the global economy processed 62 Gt/yr of materials. Twenty-eight percent of these materials were net additions to stocks of built structures and long-life goods, indicating that global in-use material stocks are growing at a high rate. At the same time, the degree of circularity measured as the share of recycled material in total processed materials was very low, at only 6%. The EU-27, a group of highly industrialized countries with relatively progressive environmental policies, processed 7.7 Gt of materials in 2005. Twenty-two percent of these flows are net additions to stocks, indicating that relative stock growth in the EU was lower than the global average. The estimate of recycling flows amounts to 13% of processed material. Whereas the degree of circularity within the economy in the EU is twice as high as the global average, the renewable biomass flows are, at 28% of the processed materials, relatively lower than the global average of 32%. Thus, the metabolism of the EU countries is also characterized by material throughput, and the distance to closed material loops appears to be surprisingly high. In this section, we discuss some of the factors responsible for the low degree of global circularity as well as the potentials and limitations of different options for furthering advance circularity by the four main material groups.

Fossil Energy Carriers

Of the 12 Gt of fossil energy carriers extracted globally in 2005, roughly 98% were used to produce energy. The energy contained in fossil energy carriers is released by combustion and in a highly irreversible manner. With the exception of plastics and a few other material applications, recycling is not an option for the group of fossil materials. For this reason, the share of recycled fossil materials in all processed fossil materials was only 0.26% (EU-27: 0.38%) and lower than for any other material group except for waste rock (see table S1 in the supporting information on the Web, circularity within the economy). Recycling potentials are limited to the small fraction of fossil materials used as raw material. Owing to source and sink problems related to FFs, a transition toward a new energy system will be required, with effects on the circularity of the economy. Whereas some of the energy solutions discussed might conserve the present linearity of the energy system, others have the potential to significantly improve circularity: Carbon capture and storage is one example that contributes to conserving or even reinforcing the economy's linearity. This technology increases the input for material and energy required by fossil-powered plants per unit of energy output and therefore reduces the efficiency of energy production (Herzog 2011). In contrast, a rising share of energy generated by solar, wind, geothermal, and tidal power plants in the total energy mix could improve circularity. These technologies are less material intensive in terms of material input per unit of energy output than the fossil energy system and thus can reduce both inputs and outputs of materials (Raugei et al. 2012). If we assume that 50% of the fossil energy carriers used in 2005 globally were to be substituted by solar, wind, and geothermal power generation,[7] according to our calculations this would reduce the size of processed materials by 10% and DPO by 15%.

Recycling is an option for part of the 2% of all fossil energy carriers that are used globally as material, mainly in the production of plastic, bitumen, and lubricants. Important recycling pathways exist for plastic and bitumen (see asphalt under nonmetallic minerals). Global recycling rates for plastic are estimated at 17% (22% in the EU) (Plastics Europe 2012), but these rates overestimate proper recycling given that, in most cases, plastic is, in fact, downcycled to replace products of lower quality (e.g., food packaging to plastic bags or flower pots) (Mugdal et al. 2011). For present recycling, the variety of different synthetic materials is a major barrier for increased material recycling. Reducing the consumption of plastics seems to be a more promising option, in particular in packaging, where 40%, and in building and construction, where 21% of all plastics are used. Concerning material properties for both uses, an almost complete substitution by biogenic materials, which are degradable in ecological cycles, is technically feasible. However, the land requirements for some substitutes are large and pose limits for actual substitution (Dornburg et al. 2003; Lauk et al. 2012).

In addition to recycling, the cascadic use of fly ash and slag, which accrue as waste product in the combustion of coal and wastes, in the production of concrete can reduce material flows and contribute to circularity. Though there are no reliable data for the current use of fly ash in cement production, experts argue that a shift to concrete mixtures containing more than 50% fly ash by mass of the cementitious material can reduce the water and energy demand of production as well as improve the workability and durability of concrete (Wang 2004). Such strategies, however, also perpetuate the use of FF carriers.

Biomass

Global biomass extraction amounts to 19 Gt/yr and the degree of circularity for this material group within the economy is low, at only 3% (7% in the EU-27). Almost 80% of all biomass is used energetically in the form of food, feed, and fuel. Similar to fossil energy carriers, for this fraction of biomass, recycling within the economic system is not feasible. However, if biomass is produced sustainably, that is, without damaging soil or water resources and without depleting ecological carbon stocks (Jordan et al. 2007), it can be considered renewable and the emitted CO_2 as well as waste flows such as excreta can largely be recycled into new primary biomass within ecological cycles. These processes can be supported by human activity, for example, when nutrient-rich excreta of humans and livestock or ash are used to fertilize agricultural ecosystems. This not only helps to close loops of essential plant nutrients, but it also contributes to a reduction of the input of industrial fertilizer based on nonrenewable mineral resources and further increases the circularity of the economy.

Additionally, there seem to be large potentials to reduce the amount of biomass inputs required to produce sufficient food for the global population. Reducing food wastes is one possible strategy, given that approximately 20% to 30% of all food is wasted along the way from harvest to consumption (Gustavsson et al. 2011). A second, even more powerful pathway involves changing dietary patterns toward a lower share of animal products, which could drastically[8] reduce the material intensity of food supply (Wirsenius 2003; Krausmann et al. 2008). Cascade utilization of by-products, residues, and excreta also has a high potential to improve overall resource efficiency (Ma et al. 2010).

Roughly one fifth of all biomass is used as raw material; wood accounts for the largest fraction of this flow: Approximately 12% of biomass (approximately 4% of globally processed materials) is wood used for construction, for other durable wood products such as furniture and for paper production. In Europe, approximately 44% of the materially used wood was recovered; of this, 64% were recycled or downcycled, 2% were reused, and 34% were used for energy generation in 2005 (Merl et al. 2007). Seventeen percent of the wood is used for paper production. Paper has a long recycling tradition with current recycling rates of 40% to 50%, both globally and in the EU-27. Whereas collection of waste paper and subsequent recycling or alternative uses have almost reached their limits, there is great potential for improvement in the prevention of paper flows, in particular, where use is inefficient (e.g., newspapers, unsolicited bulk mail, and office paper use) (Roberts 2007).

Metals

Ores account for approximately 4.5 Gt/yr or 8% of global material extraction. The actual metal content of these ores is only approximately 0.8 Gt; the reminder are tailings and processing slags of little further use. Of the pure metal, approximately two thirds are added to stocks. For many "base metals" (e.g., copper, zinc, and so on), EOL recycling rates are slightly above 50%, and only for two metals they are significantly higher: iron, with a recycling rate of approximately 90%, and lead (Graedel et al. 2011; UNEP 2011b). Lead is an exception owing to the fact that the biggest share of lead is used for just one product group: vehicle batteries, of which approximately 90% to 95% are collected and recycled. On the other end of the spectrum, there is a wide range of metals and metalloids with recycling rates below 1% (e.g., lithium and thallium). Whereas aggregate EOL recycling rates of metals are high both in the EU (76%) and globally (71%), the high flow of net additions to stock for metals keeps the degree of circularity for this material group much lower, at 40% and 36%, respectively.

There are promising strategies to make more efficient use of metals such as increasing lifetimes, more-intense uses, repair and resale, product upgrades, modularity and remanufacturing, component reuse, and using less material to provide the same service (Allwood et al. 2011). Although these strategies seem to have great potential, quantitative assessments are difficult to make and are largely lacking (Mugdal et al. 2011).

In terms of recycling, metals can theoretically be recycled infinitely. However, there are significant challenges to metal recycling (Reck and Graedel 2012; Graedel et al. 2011): At the beginning of the twenty-first century, humanity is using almost the entire spectrum of available metals. Many of these metals are used in very small quantities (nanomaterial technologies and microelectronics), in complex alloys, or in composite materials, and individual products can contain dozens of different metals. All of these factors decrease the recyclability of metal products, because the separation of metals becomes more demanding and costly and pure recycled metals are increasingly difficult to obtain. This is aggravated by the fact that recycling technologies (shredding, crushing, or magnetic sorting) are often crude and far less advanced than production technologies.

In general, metal recycling contributes not only to a reduction in the demand for virgin ores, but also has a positive effect on energy requirements. The processing steps from ore extraction to pure metal entail moving and processing huge quantities of raw material and consume large amounts of energy, both of which can be reduced through recycling. Metals are approximately 5% of the total EOL waste streams. If a product design that favors recycling is applied and if economic incentives are in place, there is a high potential to close material loops for metals to a high degree, provided that net additions to stocks are also reduced. Additionally, this could substantially reduce carbon emissions related to steel production, which amounted to 25% of global industrial carbon emissions in 2006 (Allwood et al. 2011). Increasing the recycling rate for steel from 71% to 91% would, for example, reduce the overall global sum of extracted materials by 1.3% (equals the reduction of pure metal and waste rock extraction as well as associated fossil energy carriers use) and DPO by 1.7%, compared to the present situation.

Nonmetallic Minerals

Nonmetallic minerals are the largest fraction of global material extraction and their consumption is growing at very high rates (Krausmann et al. 2009). Of the 22 Gt extracted in 2005, bulk minerals, such as sand, gravel, stone, or clay, account for roughly 95% and are subsumed under the category of construction minerals. According to our calculations, global EOL recycling rates for this material group are 33% globally and 46% in the EU-27. Similar to metals, net additions to stock are very high for nonmetallic minerals, and the overall degree of circularity is much lower, at only 11% and 23%, respectively (see table S1 in the supporting information on the Web). Proper recycling flows are even lower than that, owing to the fact that recycling statistics for construction minerals include large amounts of downcycled materials (e.g., construction and demolishing waste used as backfilling material). For asphalt (a mixture of gravel and bitumen) in situ recycling is already quite high, but quantitative assessments at the global level or for world regions are lacking. The National Asphalt Pavement Association (NAPA) assumes asphalt pavement recycling rates of over 99% for the United States (NAPA 2013). For industrialized countries in general, we assume a range from 80% to 90% (see also US DOT 1993).

Key strategies for reducing material inputs and improving circularity of this group are to stabilize or even reduce the size of stocks and extend the service lifetime of existing structures. Additionally, further closing loops for construction minerals is possible, but requires recycling-friendly design of buildings and infrastructures and regional flow management to keep transport distances short. While, in principle, nearly all types of construction materials can be recycled, recycling is not always the most sustainable option for this material group. Negative environmental and resource effects for some materials are considerable (e.g., cement recycling), and also transport intensity is a limiting factor (Blengini and Garbarino 2010). Chong and Hermreck (2010), for example, point out that saturation of local markets for recycled construction materials can become a critical factor, given that an increase in the distance between project sites and recycling facilities might counteract the benefits of recycling. The study concludes that further increases in recycling activities depend on the existence of a market for recycled materials, regional recycling capacities, total energy used to recycle, and the knowledge of the workers and designers of options for using recycled materials in construction projects. Another limitation concerns underground stocks of built structures. These are large stocks, but difficult to access, and the costs of recycling are high (Tanikawa and Hashimoto 2009). Often, underground stocks are simply abandoned and remain in the ground as so-called hibernating stocks.

The small fraction of nonmetallic minerals used for other applications than construction is a very heterogeneous group. For some of these materials (e.g., salt), recycling potentials are very low; but examples of materials with a long tradition of recycling and high recycling rates (such as glass) are also in this group. Nonmetallic mineral inputs for the production of glass account for less than 0.5%[9] of global extraction. Recycling rates in industrialized countries range from 40% to 70%. Glass can be remelted and used in new glass products without loss of physical property or quality.[10] However, according to the priorities of the CE, reuse would be more favorable than recycling. Another example is phosphate, which currently moves mainly in a linear direction from mines to distant locations for crop production, processing, and consumption. There is a high potential for improving phosphorus use efficiency, and as a result of phosphorus scarcity it will need to be recovered from waste streams from human and animal excreta to food and crop wastes (Cordell et al. 2011; Schröder et al. 2011).

Conclusions

The sociometabolic approach shows that, currently, only 6% of all materials processed by the global economy are recycled and contribute to closing the loop. If all biomass is considered a circular flow regardless of production conditions, the degree of circularity increases to 37%. The rates for the EU-27 are only slightly above the global averages. This indicates that both the global economy and that of the EU-27 are still far away from a CE. Against the background of an average growth rate in global material consumption of approximately 3.6% in the last decade (1950–2010) (Schaffartzik at al. 2014), the CE is not in sight at present. Several lessons can be learned from our systemic assessment, from a metabolic perspective, for policies aiming at the implementation of a CE.

Recycling is one of several important elements of a CE; yet, although it has the potential to increase circularity for some materials, circularity cannot be achieved on the basis of recycling alone. We identify two structural barriers for improving the circularity of the economy through recycling: A very large fraction of the materials we use still accumulates as in-use stocks. While a certain trend of stock stabilization in industrial countries can be observed, globally stocks are growing at high rates and might continue to do so. As long as additions to stocks grow at such high rates,[11] even high EOL recycling rates will make a limited contribution to overall circularity. A second barrier is the large amount of materials used for energy generation. For these materials, and, in particular, for fossil energy carriers, closing the loop is not possible and a high share of these materials keeps the degree of circularity low. Whereas sustainably produced biomass that is recycled within the biosphere can be an important component of a CE, reducing the consumption of fossil energy carriers is necessary to further raise the degree of circularity of the economy. The energy transition from fossil to renewable energy resources is therefore an important prerequisite for moving toward circularity. Reducing barriers for recycling materials used as raw materials is another important cornerstone. Although EOL recycling rates for some materials are already high, considerable improvements seem possible. This requires the consistent eco-friendly design of products (including buildings and infrastructures) that increases lifetimes, provides the same service with less material requirement, and facilitates repair and resale, product upgrades, modularity and remanufacturing, component reuse, and, finally, also EOL recycling. Achieving a reversal of the trend of global growth in resource consumption into a dynamic of reduction, or at least a steady-state physical economy, remains the greatest challenge of all.

Acknowledgments

The authors gratefully acknowledge funding from the European Commission for the project NEUJOBS (EU-FP7 project 266833) and the Austrian Science Fund (FWF; Project Glometra, P21012 G11 and project MISO P27590).

Notes

1. Further, it must be noted that circularity should not be regarded as equating to ecological sustainability: Also, the use of materials that run in cycles can have negative impacts on ecosystems and biodiversity.
2. Material used for nuclear fission in power plants is not considered in our assessment.
3. Fuel wood and biofuels account for roughly 10% of all globally processed biomass (FAO 2013; Krausmann et al. 2008; Goldemberg et al. 2014).
4. While food and feed not only provide nutritional energy for humans and livestock, but are required to building up body mass (i.e., stocks), the fraction of food/feed that accumulates in body mass is very small. On the basis of population growth, we estimate that global net additions to population stock correspond to less than 0.1% of total food supply (per year). We therefore neglect the "material use" component of food in our assessment and consider all food and feed as "energetic use."

5. Waste rock may be used, for example, as backfilling material. Owing to lack of data, this flow has not been considered in this assessment. Waste rock may also eventually become a resource again, if rising metal prices and technological development make the exploitation of remaining metal content feasible. Given that waste rock becomes DPO in the MFA system, this would be considered as extraction and not as recycling.
6. Processing and consumption change the moisture content of biomass and combustion adds atmospheric oxygen to fuels. To close the mass balance between material inputs and outputs, economy-wide MFA considers water flows resulting from changing moisture content and oxygen inputs resulting from combustion as so-called balancing items. For reasons of simplicity, we do not consider balancing items in this assessment. This means that changes in the mass of flows resulting from oxygen uptake or changes in moisture content are not taken into account.
7. Such an energy scenario is discussed and considered feasible, for example, by Jacobson and Delucchi (2011). In our calculations, we neglected the fact that also renewable energy technologies require inputs of mineral materials, for example, for infrastructure, turbines, or dams and the implications of these material flows for circularity.
8. Approximately 60% of all harvested biomass is used to feed livestock, which converts plant biomass into meat, milk, and other livestock products at a low efficiency (Krausmann et al. 2008). A change in dietary patterns toward a lower share of animal products and within animal products toward meat from monogastrics, which have a much higher feed-use efficiency than ruminants, would significantly improve the biomass efficiency of the food system (see Herrero et al. 2013; Wirsenius et al. 2010).
9. According to the World Silica Sand Market report (Freedonia Group 2012), extraction will increase to 278 million metric tons in 2016, compared to approximately 175 million tons for 2004.
10. Colored glass cannot be turned into clear glass products, but can be recycled into other colored glass products.
11. At the global level, additions to stocks in the material category of construction minerals grew by 4% annually during the period 1990–2005 (respectively by 0.8% in the EU-27).

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Supporting Information

Additional Supporting Information may be found in the online version of this article at the publisher's web site:

Supporting Information S1: This supporting information provides further material flow calculations with regard to the main material categories (table S1), a breakdown of processed materials into main material groups (figure S1), and a comparison of main material flows (figure S2) for the world and the EU-27.

Global socioeconomic material stocks rise 23-fold over the 20th century and require half of annual resource use

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Edited by William C. Clark, Harvard University, Cambridge, MA, and approved January 5, 2017 (received for review August 26, 2016)

Human-made material stocks accumulating in buildings, infrastructure, and machinery play a crucial but underappreciated role in shaping the use of material and energy resources. Building, maintaining, and in particular operating in-use stocks of materials require raw materials and energy. Material stocks create long-term path-dependencies because of their longevity. Fostering a transition toward environmentally sustainable patterns of resource use requires a more complete understanding of stock-flow relations. Here we show that about half of all materials extracted globally by humans each year are used to build up or renew in-use stocks of materials. Based on a dynamic stock-flow model, we analyze stocks, inflows, and outflows of all materials and their relation to economic growth, energy use, and CO_2 emissions from 1900 to 2010. Over this period, global material stocks increased 23-fold, reaching 792 Pg (±5%) in 2010. Despite efforts to improve recycling rates, continuous stock growth precludes closing material loops; recycling still only contributes 12% of inflows to stocks. Stocks are likely to continue to grow, driven by large infrastructure and building requirements in emerging economies. A convergence of material stocks at the level of industrial countries would lead to a fourfold increase in global stocks, and CO_2 emissions exceeding climate change goals. Reducing expected future increases of material and energy demand and greenhouse gas emissions will require decoupling of services from the stocks and flows of materials through, for example, more intensive utilization of existing stocks, longer service lifetimes, and more efficient design.

material flow accounting | socioeconomic metabolism | circular economy | carbon emission intensity | manufactured capital

The growing extraction of natural resources, and the waste and emissions resulting from their use, are directly or indirectly responsible for humanity approaching or even surpassing critical planetary boundaries (1). Both decoupling of resource use from economic development and absolute reductions in the use of certain materials and energy sources are imperative for sustainable development (2). The demand for materials and energy is to a large extent driven by constructing, maintaining, and operating in-use stocks of materials (hereafter "material stocks"), or what economists call manufactured capital (buildings, infrastructure, artifacts). These stocks transform material and energy flows into services, such as shelter or mobility (3, 4). The significance of long-lived stocks of infrastructure and buildings for determining and potentially reducing future material and energy use and greenhouse gas emissions is increasingly recognized (5, 6). This study investigates the dynamics of global stocks and flows of materials by using and expanding a material flow accounting (MFA) approach. MFA is used in industrial ecology to study the biophysical domain of society, comprising in-use stocks and the processes and flows that maintain and operate these stocks, from the extraction of primary materials to the disposal of waste and emissions (7, 8). MFA research has shown that during the 20th century global material consumption grew by an order of magnitude. It was estimated to range between 70 and 76 Pg/yr in 2010 (2, 9, 10). Primary materials are used for two main purposes (11). Currently around half of all materials extracted are used dissipatively and provide energy in a broad sense (Fig. 1*A*). This includes fossil energy carriers used for thermal energy conversion and also biomass, which are both used as fuel and constitute the primary energy source (and building-blocks) for the biological metabolism of humans and livestock. These materials are converted to carbon emissions and other waste and pollution soon after extraction. The other half of global resource extraction is used to build up more or less long-lived material stocks. This is the case for metals and nonmetallic minerals, and a minor fraction of biomass (e.g., timber) and fossil fuels used in in the chemical industry (e.g., for asphalt and plastics). These durable materials are extracted, processed, and used to construct and maintain buildings, transport and communication infrastructure, machinery, and consumer goods. The materials accumulate in socioeconomic systems and remain in use from several years up to decades and sometimes centuries.

These stocks are the material basis of wealth (3, 8). They provide services, such as shelter, mobility, and communication, and constitute the physical infrastructure for production and consumption (3, 4). Material stocks link basic services to flows of materials and energy and hence are a main determinant of material flows (4, 8). Large amounts of materials and energy are required in industry and construction to build, maintain, and

Significance

A large part of all primary materials extracted globally accumulates in stocks of manufactured capital, including in buildings, infrastructure, machinery, and equipment. These in-use stocks of materials provide important services for society and the economy and drive long-term demand for materials and energy. Configuration and quantity of stocks determine future waste flows and recycling potential and are key to closing material loops and reducing waste and emissions in a circular economy. A better understanding of in-use material stocks and their dynamics is essential for sustainable development. We present a comprehensive estimate of global in-use material stocks and of related material flows, including a full assessment of uncertainties for the 20th century as we analyze changes in stock-flow relations.

Author contributions: F.K., D.W., H.T., and H.H. designed research; F.K., D.W., C.L., W.H., T.F., and A.M. performed research; F.K., D.W., C.L., W.H., T.F., and A.M. analyzed data; and F.K., D.W., W.H., T.F., H.S., and H.H. wrote the paper.

The authors declare no conflict of interest.

This article is a PNAS Direct Submission.

Freely available online through the PNAS open access option.

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This article contains supporting information online at www.pnas.org/lookup/suppl/doi:10.1073/pnas.1613773114/-/DCSupplemental.

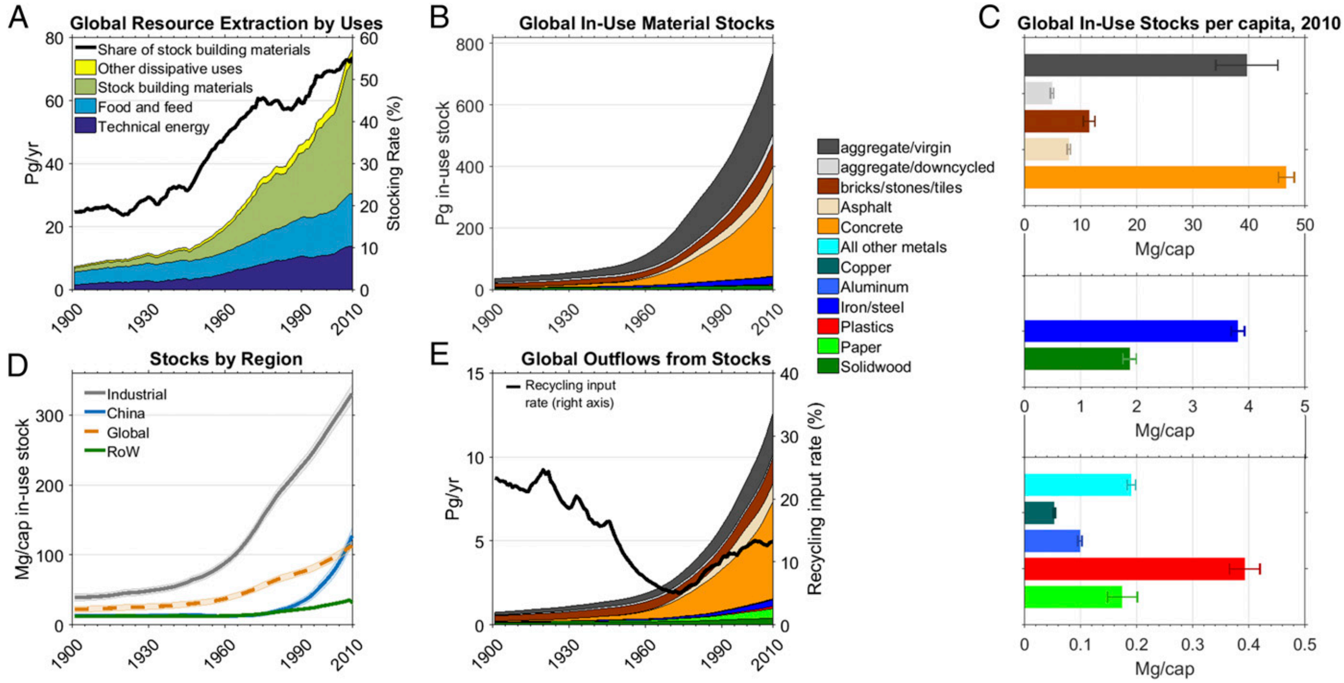

Fig. 1. Development of global material stocks and flows from 1900 to 2010. (*A*) Annual global extraction of materials by use and share of stock-building materials in total extraction (right axis). (*B*) Development of global in-use stocks of materials by 12 main material groups. (*C*) Global material stocks in 2010 including uncertainty ranges (note that the scales in *C* differ by a factor of 10). (*D*) Development of total stock per capita in the group of industrial countries, China, and the rest of the world (RoW). (*E*) Global end-of-life outputs from discarded stocks and recycling input rate (i.e., share of recycled and down-cycled end-of-life outputs from stocks in total inputs to stocks). Note that *B*, *C*, and *E* share the same legend.

refurbish stocks. Once they are in place, stocks require energy to provide services. Energy is needed to heat and light buildings, to operate railways and fuel cars, to run machinery, and power information and communication technologies. However, stocks not only affect demand for material and energy inputs, they also codetermine the amount of solid waste produced and the availability of materials for recycling in terms of quantity, quality, and time. Stock-flow dynamics and their role in shaping patterns of material and energy flows are thus key to understanding and changing resource flows and closing material loops. Circularity of resource flows in economic activities, contributing to improved resource efficiency and underpinning human well-being at much lower resource requirements, is at the core of sustainable resource-use strategies and policies instituted in Japan, China, and the European Union (12–14). Although current policy mechanisms mainly focus on products and specific industries, the importance of long-lived in-use stocks and the dynamics of material use in terms of their accumulation, maintenance, and use has not yet been adequately included in policy (11, 15, 16). Achieving absolute decoupling of resource use and emissions from economic development and a transition to sustainable resource use requires systemic knowledge about the interactions of in-use stocks and resource use at different spatial and temporal scales (4, 8, 17).

In recent years, scientific and policy interest in stocks has been growing, but studies conducted so far have been confined to specific elements, such as metals and other narrowly defined (groups of) substances or products (18–23). Comprehensive MFA-based analyses of the long-term development of material stocks accumulated in built infrastructure and durable goods, their composition and relation to demand for primary materials, and waste production are still rare (24–26) and have never been attempted at the global scale. Here we present a global estimate of the development and composition of total economy-wide material stocks, end-of-life wastes, and recycling flows by material types during the 20th century. Our estimates are fully consistent with the system boundaries and principles of economy-wide MFA (7). We expand the MFA approach by combining it with a mass-balanced top-down stock/flow model (*SI Appendix*, Fig. S1). By systematically linking all major stocks and flows of materials, we provide an important step toward a comprehensive understanding and modeling of the long-term dynamics of the global sociometabolic system (8, 27). We quantify the mass of all materials stored in buildings, infrastructure, and durable goods, distinguishing 11 types of stock-building materials from 1900 to 2010 (Dataset S1). We use Monte Carlo simulation to propagate errors for all parameters throughout the modeling. Uncertainties of results are shown as ±3 SD or 99.7% over 10^3 Monte Carlo simulations (*SI Appendix*). This research quantifies the development of stocks and also flow indicators, including material inputs to stocks and net additions to stocks, waste output, and recycled materials. We analyze stocks, inflows, and outflows and their relation to economic growth, energy use, and CO_2 emissions, and discuss implications for future growth of stocks and emissions. We focus on global results, but for aggregate stocks also show results for industrial countries, China (a major driver of global resource use over the past two decades), and the rest of the world (*SI Appendix*, Table S3).

Results

Global Stock Growth in the 20th Century. From 1900 to 2010, global material extraction grew 10-fold from 7 Pg/yr (1 Pg = 10^{15}g = 1 Gt) to 78 Pg/yr in 2010 (*SI Appendix*). Separating dissipative uses from materials used to build up stocks, we find that the share of stock-building materials in total extraction rose from 18 to 55% (Fig. 1*A*). Deducting processing losses (e.g., tailings from metal production or CO_2 from burning limestone), we calculate the actual amount of primary material inputs used to build up or renew in-use stocks of buildings, infrastructure, machinery, and durable goods. Primary inputs to stocks increased from 1 Pg/yr in 1900 to 36 Pg/yr in 2010. In the latter year, the largest of these primary material inflows (79% or 28.6 Pg/yr) was sand and gravel used in concrete and asphalt or in foundations and base course layers. Metals are often used in combination with construction minerals, for example in reinforced concrete. Primary materials amounted to 1.2 Pg/yr, more than biomass in the form of timber and paper as well as fossil fuels used as feedstock for plastics and bitumen, which amounted to 0.9 Pg/yr and 0.3 Pg/yr, respectively. A total of 4.8 Pg/yr of re- or down-cycled secondary materials added to the inflow of primary materials. The growth in the inflow of stock-building materials reflects a century of urbanization and industrialization in Europe, the United States, and other high-income countries. Buildings, transport and communication networks, supply and discharge systems, vehicle fleets, and industrial machinery were established, which constitute the material basis of modern society. In all world regions, in particular in the industrial countries, manufactured capital expanded greatly (3). Fig. 1*B* shows that material stocks grew from 35 Pg (± 18%) to 792 Pg (± 5%): that is, at an average annual growth rate of 2.9% between 1900 and 2010. Growth was fastest in the three decades after World War II at 4.0% per annum. This was the period of postwar economic boom. Large investment went into the reconstruction of war damage in Europe and Japan, and fast economic growth and urbanization in the industrial world were accompanied by the rapid expansion of material stocks. Nevertheless, most global stocks are comparatively young, which is because of the continuously high level of yearly inputs to stocks in the industrial world and more recent acceleration of stock growth in emerging economies. Globally, almost two-thirds of all materials used to build and renew material stocks between 1900 and 2010 were added to stocks since 1980. As a result, 82% of all in-use stocks are aged 30 y or younger (*SI Appendix*, Fig. S9).

Over the whole period, global material stocks grew 23-fold. They grew at a similar pace to global gross domestic product (GDP), which increased by a factor of 27, but much faster than global annual material extraction (factor of 10). Population only quadrupled and average per capita stocks surged from 22 Mg/cap (± 18%) in 1900 to 115 Mg/cap (± 5%) in 2010 (1 Mg = 10^6g = 1 t) (Fig. 1*D*). The differences are large between high-income industrial countries and the developing world. Average material stocks in industrial countries amounted to 335 Mg/cap (± 4%) in 2010, a value commensurate with the range of per capita stock estimates reported in the literature for specific industrial countries (25, 28). China has been catching up rapidly since the 1990s. Between 1990 and 2010 it increased its per capita stocks from 35 Mg/cap to 136 Mg/cap (± 8%): that is, to the level the industrial countries reached in the early 1970s. Average per capita stocks in the rest of the world are slowly on the rise as well but had reached only 38 Mg/cap (± 2%) in 2010, barely above the global average in 1900. Large growth in material stocks is expected for the second wave of urbanization in Asia, Africa, and Latin America (29). The composition of material stocks has also changed. Sand and gravel constitute a large but declining share of total stocks throughout the 20th century, from 47% in 1900 to 38% in 2010. In 1900 bricks and wood were dominant materials (44%), whereas in 2010 concrete, asphalt, and metals made up 50% of the total stock. The mass of metal accumulated in built structures and machinery increased from 0.6 to 4.1 Mg/cap, whereas biomass declined from 2.6 to 2.0 Mg/cap (Fig. 1*C*). Overall 25.6 Pg of iron, 0.7 Pg of copper, 0.4 Pg of aluminum, and 13.8 Pg of biomass were stored in in-use material stocks in 2010, of which two-thirds were located in industrial countries.

We compared our results with available estimates from studies that have quantified stocks of substances or single materials, including steel (23), aluminum (21), copper (30), carbon in timber and plastics (20), and concrete (6). As shown in *SI Appendix*, Figs. S5–S8, we find good agreement between levels and trends of material stocks despite considerable differences in the methods applied to estimate these stocks, corroborating the results from our comprehensive but less-detailed modeling approach. The uncertainty analysis reveals that despite uncertainty ranges of up to ±15–60% (3 SDs) for annual material inputs and recycling rates and ±15–30% for mean lifetimes, global stock uncertainty in 2010 was moderate at ±5%, but ranged from 6–15% for the 11 materials (Fig. 1*C*). This result shows that in our modeling approach, stochastic issues with specific data points are a relatively minor source of overall variation. Because of the tendency toward the mean (central limit theorem) in the resulting normal distributions, long lifetimes and detailed cohorts for each material smooth out large inflow variations. To check for systematic errors in the factor lifetime distributions, we also conducted a sensitivity analysis. We tested for –30% and +30% as well as +50% on the mean "best guess" lifetimes, yielding stock estimates −10% lower and +6% and +9% higher, respectively, than the mean estimate of 792 Pg in 2010. This finding suggests that lifetimes can have a substantial and nonlinear effect on overall uncertainty, but the error is still fairly small. Furthermore, we tested the influence of large-scale destructive events, such as World War II, and found little long-term impact on overall stock development. Economists have estimated that in World War II the value of destroyed physical capital ranged from 5 to 15% in European countries and from 25 to 30% in Japan and the Union of Soviet Socialist Republics (31). The assumption that 15% of existing physical stock of manufactured capital in the industrial group was destroyed by the end of the war results in a large waste flow (5 Pg). However, the impact on stock development remains small and in 2010 stocks in the industrial region were only 0.3% lower than without World War II stock destruction.

SUSTAINABILITY SCIENCE

Waste, Recycling, and Closing the Loops. Although the 20th century was a period of material accumulation, aging buildings, infrastructure, and durable products increasingly reached the end of their lifetimes and were discarded, resulting in growing end-of-life waste from stocks. We find that outflows from stocks increased from 0.8 Pg/yr in 1900 to 14.5 Pg/yr (± 7%) in 2010, half of which was concrete (Fig. 1*E*). The share of biomass was 7%; metals amounted to 5% and plastics to 1%. Not all outflows turned into waste, but a considerable fraction of the materials from discarded stocks was recycled into secondary material inputs, up from 0.3 Pg/yr in 1900 to 4.8 Pg/yr (± 25%) in 2010. The majority of the recycling flow comprises nonmetallic minerals (88% in 2010), which are mainly down-cycled as base materials for backfilling during new construction. Metals, biomass, and plastics together account for 12% of total end-of-life recycling. Because global stocks are growing, the contribution of recycling to closing material loops remained lower than the promising potential suggested by end-of-life recycling rates. For nonmetallic minerals we estimate that 37% of all end-of-life outflows from stocks are recycled, but because of the larger inputs into stocks this yields a recycling input rate (the share of recycled or down-cycled materials in the total inflow of primary and secondary materials into stocks) of only 11% (Fig. 1*E*). Metal recycling is relatively advanced and industry has taken significant steps in terms of scrap reuse and recycling (32). We find that 77% of end-of-life outputs of metals are recycled, but the share of secondary materials in total metal inputs to stock is only 27%. For biomass materials, the end-of-life recycling rate and the recycling input rate are similar at around 20%. We assume that with the onset of mass production and consumption, increasing abundance of primary raw materials and falling resource prices end-of-life recycling initially declined, until the rise of environmentalism in the 1970s drove recycling rates upward again (33, 34). Over time, aggregate recycling rates therefore follow a U-shaped trend. End-of-life recycling declined from 36% in 1900 to a trough in the 1970s of around 18% and then improved again, reaching 33% in 2010. We find that the aggregate recycling input rate (Fig. 1*E*) followed a similar trajectory, declining from 23% in 1900 to only 5% in 1970. Because of more and more stocks becoming obsolete, and increasingly effective recycling regulations and capacity in many countries, the recycling input rate recovered to 12% in 2010. However, as material stocks are growing and inputs to stocks exceed end-of-life outflows by a factor of 4, primary materials remain the main material input for building up and refurbishing in-use stocks, even if recycling rates are improving and more secondary resources become available.

Our model also yields estimates of global solid waste production from discarded stocks, an area for which data are notoriously poor. A recent study estimated the global waste flow, largely based on municipal waste statistics, to amount to 2.1 Pg/yr (35). This excludes organic waste, which is not covered in our study. We arrive at a much larger solid waste flow of 9.7 Pg/yr (± 14%) for 2010. The difference occurs because the estimate by Hoornweg et al. (35) only comprises a small fraction of the large mass flows of construction and demolition waste. Additionally, a considerable portion of our waste estimate may also be "hibernating" stocks (36). These involve abandoned infrastructure or buildings that are left in place and therefore do not appear in waste statistics. Overall, our calculations show that between 1900 and 2010 a total of 293 Pg of discarded stocks turned into solid waste, including 11 Pg of metals. These materials have been deposited in controlled or uncontrolled landfills or remain in place as unused structures, potentially polluting the environment, but they also constitute anthropogenic resource deposits for potential recovery in the future. Waste formation from stocks will continue to increase. The size and age distribution of global stocks (*SI Appendix*, Fig. S9) indicates that large material stocks currently in use may reach the end of their service lifetimes in coming decades. Assuming unchanged lifetime distributions, we estimate that by 2030 35% of the material stock in use in 2010 will be discarded, yielding 274 Pg of end-of-life outflows, about the same amount that accrued in the previous 110 y. These materials may become secondary resources and contribute to closing material loops, or they have to be disposed of. To ensure material outflows can be recycled and turned into valuable resources, it is vital to have better knowledge about where and when which types of material outflows from stocks become available (28).

Stock Productivity and Decoupling. Previous studies (9, 37) of global material flows found a long-term trend of relative decoupling of global material use from economic development during the 20th century, where economic output grew faster than annual material consumption. This resulted in a considerable and continuous improvement in material use productivity (Fig. 2*A*). Our results enable us to go beyond this observation, which ignores the role of material stocks as a production factor, to analyze economic activity trends per unit of physical capital. Our results show no significant long-term improvement in material stock productivity (Fig. 2*A*). The added value produced per unit of stocked material fluctuated between \$56/Mg in 1900 to a peak of \$75/Mg in the early 1970s. Since then, global stock productivity has been declining, reaching \$67/Mg in 2010. This decline hints toward a tight coupling of economic development and the growth of material stocks of manufactured capital. Infrastructure and capital goods are prerequisites to production and economic growth which, in turn, triggers private investments into housing, vehicles, and consumer goods. The lack of continuous improvements in average global stock productivity suggests that economic development, in particular in emerging economies, is likely to be connected to further stock growth. This finding shows that the question of whether stock growth drives economic growth or economic growth drives stock growth (and under which conditions) is complex, and poses challenges for designing policies that aim to decouple economic growth from stock growth.

Energy and Emission Intensity of Material Stocks. Practically all of the technical energy, and consequently also fossil fuel-related CO_2 emissions, are tightly linked to material stocks in one way or another. Energy is required in mining, manufacturing, and

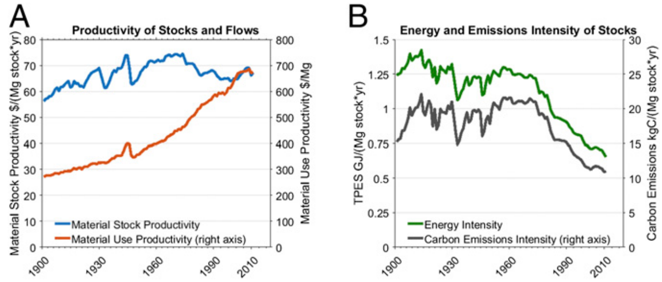

Fig. 2. Development of global stocks in relation to GDP, energy use, and CO_2 emissions 1900–2010. (*A*) Global stock productivity (GDP/material stock) and material use productivity (GDP/annual material consumption, right axis). (*B*) Energy and carbon emission intensity of material stocks. Total primary energy supply (TPES) and CO_2 emissions from fossil fuel use per megagram of material stock. Material use (domestic material consumption) is in megagrams (9), GDP in constant international dollars of 1990 (45), CO_2 emissions in kilograms of C (46), and TPES in gigajoules (9).

construction, indeed in all processes involved in building up and renewing the built environment and artifacts (6). Once stocks are in use, even larger amounts of energy are needed to heat, cool, and light buildings, keep transport moving, and power electrical appliances, among many other uses. Comparing stock size with the long-term development of energy use and CO_2 emissions from fossil fuel use, we observe moderate decoupling (Fig. 2*B*). Primary energy use per unit of stock has declined by 53% since World War I to 0.7 $GJ \times Mg^{-1} \times yr^{-1}$. Decoupling accelerated in the 1970s and since then primary energy inputs per unit of stock have declined at 1.6% per annum. The trajectory we find for CO_2 emissions from fossil fuel consumption resembles that of primary energy use (Fig. 2*B*). The aggregate CO_2 emission intensity of stocks began to improve in the 1970s and has declined by 48% since then. In 2010 an average of 11 kg C were emitted per megagram of material stock. Aggregate emission intensity can be separated into the amount of CO_2 emitted because of building and renewing material stocks and the amount of CO_2 emitted for the provision of services from stocks. Data on sectoral energy use from the International Energy Agency (38) indicate that in 2010 one-quarter of available final energy was used to manufacture stocks (industrial energy use) and three-quarters to provide services from stock (energy use in, for example, households, transport, service sectors). Assuming that this relation also roughly holds for fossil fuel-related CO_2 emissions and adding CO_2 emissions from cement production, we calculate an emission intensity for building and renewing stocks of 62 kg C per megagram of material inputs to stock and an average emission intensity of providing services from stocks of 8 kg C per megagram of in-use stock per year. This finding demonstrates that despite considerable efficiency gains, stocks and stock growth are important determinants of energy use and CO_2 emissions.

Future Stock Development and Its Impact on Flows. A major issue for sustainable development and for downsizing material and energy throughput is how the size of stocks will develop in the future. Chen and Graedel (3) have shown that the temporal evolution of a broad range of in-use stocks of manufactured capital in the United States follows a logistic function. Stocks rapidly accumulate and then saturate after a slow take-off and fast growth, often followed by another wave of capital accumulation of a new type of stock. A classic example is the expansion and saturation of transport networks, shifting from canals to railroads, roads, and airports. Such a trajectory has been projected for per capita in-use stocks of iron (23, 39, 40), as well as for aggregate material stocks in some countries (24, 25). Our global results show no sign of saturation yet; stocks continue to grow at high rates, including in wealthy and industrial countries where they have already reached a high level of 335 Mg/cap (± 4%) (Fig. 1*D*). However, in industrial countries inflows of stock-building materials have stabilized. In some countries they have even begun to decline in recent years (24, 25, 28). Our results indicate that net additions to stocks have ceased growing in the industrial group (Fig. 3*B*). If the decline in net additions to stock observed in the last years continues, this may eventually result in a saturation of stocks in the industrial world in coming decades (24, 26). This would increase the potential for closed loops and absolute reductions in primary material extraction.

At the global scale, a stabilization of material stocks, and hence of primary material requirements to build up new stocks, still seems distant if past trends continue. Industrial countries currently possess about two-thirds of all material in-use stocks, and China is rapidly catching up. Since 1990 China's share of global stocks has more than doubled from 10 to 22%, and its net additions to stock have surpassed those of the industrial countries (Fig. 3). Per capita stocks, however, are still only 41% of the level of industrial countries (Fig. 1*B*). The rest of the world was inhabited by 62% of the global population in 2010, but owned only 18% of global stocks. The average per capita stock is just 11% of the industrial level. If industrial countries and their level of stocks serve as a benchmark for other regions, this may put huge pressure on material and energy demand and contribute large additional CO_2 emissions in coming decades (6). A simple scenario calculation (*SI Appendix*) illustrates this claim. Assuming a global convergence of per capita material stocks at the industrial level by 2050 and a world population of 9.7 billion implies a fourfold increase in global material stocks to 3,137 Pg. This number would require more than a doubling of global annual net additions to stock to 59 Pg/yr, up from 26 Pg/yr in 2010 (*SI Appendix*, Table S5). We further assume that historic trends of improvements in emission intensity will continue and lead to a reduction in the emission intensity of building and renewing stocks and of stock use by 52% by 2050 (*SI Appendix*). A fourfold increase in global stocks would then still result in cumulative carbon emissions of 542 Pg C between 2010 and 2050 (*SI Appendix*, Fig. S4*B*). Of these emissions, 72% result from providing services from stocks; the remainder is from building up (18%) and renewing stocks (10%). This amount exceeds the remaining emission budget to stay within 2 °C with a 50% or higher chance, which ranges from 234 to 417 Pg C (41). Even in the highly unlikely case that full decarbonization of the energy system could be achieved by 2050, cumulative C emissions would still amount to 303 Pg C. This finding underlines that a convergence of material stocks at the high level of industrial countries is not compatible with the global climate change mitigation target agreed in Paris in 2015.

This scenario calls for rigorous decoupling of in-use stocks from material and energy throughput and service provision (4). Making services from material stocks more energy efficient and increasing the reuse and recycling of discarded stocks is one strategy. However, stock decoupling also requires a reduction, or at least stabilization, in the size of material stocks without reducing the services provided by stocks. More intensive use of stocks, extensions of service lifetimes, material substitutions, and light weighting can contribute to this goal (4, 42). Such decoupling of stocks from services and wealth would have a large impact on the global socioeconomic metabolism. Let us assume that the level of quantitative stock development the industrial world had achieved by the 1970s, after two decades of postwar growth and massive increases in wealth and quality of life, is sufficient to provide wealth. Taking into account technological improvement and efficiency gains this amount of material stock should provide more and better services today than in 1970. Global convergence at the 1970 level of industrial per capita stock of 132 Mg/cap by 2050 would result in a comparatively moderate expansion of global stocks by 61% to 1,274 Pg and a reduction in net additions to stock from 26 Pg/yr currently to an average of just 12 Pg/yr (*SI Appendix*, Table S5). It would, however, also imply a considerable reduction in the mass of material stocks in the industrial world: that is, a shrinking of infrastructure and buildings, with the side effect of making large amounts of material available for recycling. Such a contraction and convergence scenario would induce cumulative carbon emissions of 302 Pg C if historic improvements in emissions intensities were to continue and 188 Pg C if full decarbonization could be achieved by 2050 (*SI Appendix*, Fig. S4*B*). In contrast to the catch-up scenario outlined above, such a contraction and convergence pathway could be compatible with 2 °C climate goals and contribute to dematerialization.

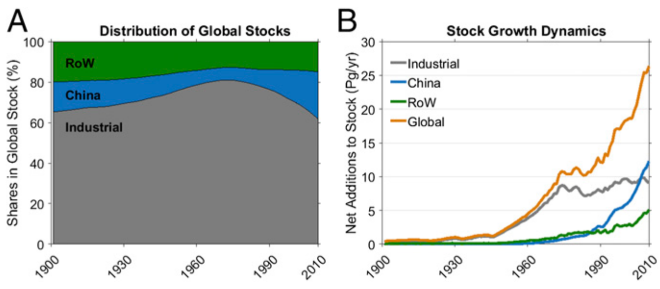

Fig. 3. Dynamics of stocks and flows in the industrial countries, China, and the rest of the world (RoW). (*A*) Distribution of global stocks across country groups. (*B*) Annual net additions to stock.

Conclusions

The 20th century has often been characterized by the emergence of a throwaway society (43). Paradoxically, it would be better described as a century of massive stockpiling. A considerable proportion of all primary materials used globally has accumulated in growing material stocks in the built environment in cities and rural areas. These link flows of materials and energy to the provision of services used by the economy and by households. In-use stock of materials has now reached 792 Pg (± 5%) and is growing in unison with GDP. Saturation, or significant decoupling of stock growth from economic development, is not in sight. Rather, growth is likely to continue, as differences in stock size between industrial and emerging economies are large, and development needs in the global South and climate change mitigation and adaptation will require revamping existing spatial structures and developing new infrastructures and settlements (6, 29, 41, 44). A global convergence to the current level of in-use stocks in industrial countries, however, would drive a massive increase in material and energy demand and greenhouse gas emissions, and undermine sustainable development and climate goals.

The sociometabolic macroperspective on stock-flow relations we provide here shows that the global economy is still far from steady state or a circular economy (11). This would essentially require a stabilization of material stocks (and a shrinking in some regions) to reduce yearly throughput. However, as long as inputs to stocks are growing and inflows to stocks are a multiple of outflows, significant improvements in closing material loops cannot be achieved, even if end-of-life recycling rates were to improve drastically. Current research and political strategies concerned with circular economy focus on closing loops at the industry or product level (14). Our results underpin the need to take the dynamics of stocks of buildings and infrastructure into account. This is where a large and still growing part of all extracted materials accumulate and after retirement eventually become available as secondary resources. With their long service lifetimes, stocks shape the dynamics of technological change and contribute to lock-in and path-dependency with respect to material-, energy-, and carbon-intensive technologies and settlement patterns. The

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stocks constitute a long legacy in driving material and energy flows and corresponding wastes and emissions. Our research indicates that decoupling global resource use from economic development, as called for in a recent United Nations Environment Program report (2), foremost requires decoupling of services from stocks and stocks from flows. This can be achieved through, for example, more intensive use of existing stocks, longer service lifetimes, and more efficient design. To reach a steady state of the physical economy, material stocks clearly deserve more attention in socioeconomic metabolism and sustainability research. To develop strategies toward a circular economy and reductions of material and energy use, improved knowledge about stock-flow dynamics, the role of stocks in connecting human well-being and resource use, and the spatial patterns of stock distribution is required.

Materials and Methods

The Material Input Stock and Output model is a top-down, input-driven, and mass-balanced dynamic stock model (22, 25, 30). It covers material inputs, stock accumulation, end-of-life outflows, and recycling for the time period 1900–2010. Drawing on a comprehensive global MFA database (9, 10) and additional sources (*SI Appendix*), annual global material use of steel, aluminum, copper, an aggregate of other metals and industrial minerals, concrete, asphalt, bricks, primary and down-cycled aggregates, paper, solid-wood products, and plastics were estimated. The annual gross additions to stock of each material group were handled as explicit cohorts and tracked throughout the entire time period, similar to a population or vintage-stock model. We covered all in-use manufactured capital, such as buildings, infrastructure, machinery, and durable goods with a lifetime longer than 1 y. Normal distributed lifetime functions were used to estimate stock dynamics and annual end-of-life outputs from stocks. Based on an extensive literature review, model parameters for lifetimes and recycling rates for all material/stock types were compiled (*SI Appendix*). A detailed description of the model, the data used and assumptions, the uncertainty analysis and the calculated scenarios, as well as numerical results are provided in *SI Appendix*.

ACKNOWLEDGMENTS. We thank Karin Hosking at Commonwealth Scientific and Industrial Research Organization for editing the manuscript. This work was supported by the Austrian Science Fund (FWF) (Project MISO P27590) and by the Environment Research and Technology Development Fund (Contract 1-1402) of the Ministry of the Environment, Japan.

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8. Curriculum vitae – Dominik WIEDENHOFER

Academic career, positions held to date

Name	**Mag.rer.nat. WIEDENHOFER Dominik, Bakk.techn.**
	HIGHER EDUCATION
2012/02 - 2017/05	**Doctoral thesis (to be defended in 2017/10)**: "Opening the black box of economy-wide material and energy flows: from accounting to stock modeling and input-output analysis". (supervision: Prof. Fridolin Krausmann).
2010/10 - 2011/07	**Master thesis:** 'Spatial and Socio-economic Drivers of Direct and Indirect Household Energy Consumption in Australia' (supervision: Julia Steinberger, PhD and Prof. Manfred Lenzen)
2008/11 - 2009/03	**Master program** *Human- and Social Ecology* at the **Alpen-Adria University** Klagenfurt, Vienna, Graz; **Graduation (MSc) with honors**
2008/02 - 2008/08	**Stipend from EU/Australia Exchange** for Sustainable Development; **Flinders University**, Adelaide, **Australia**.
2004/10 - 2008/08	**Bachelor program** *Environmental and Bio-Resource Management* at the University of Natural Resources and Applied Life Sciences Vienna; **Graduation (BSc)**
	APPOINTMENTS / POSITIONS HELD TO DATE
Since 2011/09	**Researcher and Lecturer** at the Institute of Social Ecology, **Alpen-Adria University Klagenfurt, Vienna, Graz, Austria.**
2013/06 – 2013/09	**Visiting Scientist** at the **Nagoya University, Japan**, Department of Civil Engineering, Research Lab for Environmental Systems Analysis and Planning. (with Prof. Hiroki Tanikawa)
2012/02 – 2012/02	**Visiting Scientist** at the **University of Leeds**, **United Kingdom**, Earth and Science Department, Sustainability Research Institute (SRI). (with Julia Steinberger, PhD)
2010/03 – 2010/05	**Visiting Scientist** at **University of Sydney, Australia,** Department of Physics**,** Integrated Sustainability Analysis (ISA). (with Prof. Manfred Lenzen)
2009 – 2011	**Researcher** at EcoPolicy Lab, Neue Energien 2020, FFG Project 822065
2006 & 2007	**Internships** at Austrian Alpine Association, Taricaya Research and Conservation Peru
2006 - 2009	**Teaching assistant** at the **University of Natural Resources and Life Sciences (BOKU), Austria**, Dept. of Socio-Economics (5 courses and lectures)

Peer review and organizing activities, and/or memberships in academic organisations

2017	**Reviewer for the conference & co-organizer of special session** at the 12th ISIE & ISSST joint-conference, Chicago, University of Illinois, USA
2016	**Chair of the Gordon Research Seminar on Industrial Ecology**, Vermont, USA.
2016	**Reviewer** for the joint 5th ISIE Asia-Pacific and 12th socioeconomic metabolism section conference, Nagoya University, Japan
2015	**Reviewer** European Society for Ecological Economics (ESEE) conference, University of Leeds, UK
2012 – ongoing	**Member** of International Society for Industrial Ecology (ISIE) and International Input-Output Association (IIOA)
Reviewing activities:	Journal of Industrial Ecology; Environmental Modelling & Software; Environmental Science & Technology; Resources, Conservation and Recycling; Ecological Economics; Regional Environmental Change; Sustainability; Environmental Pollution; Resources; publons.com/author/938894/dominik-wiedenhofer

CONFERENCES, BOOK CHAPTERS & CONFERENCE PROCEEDINGS (selected):

More than 40 oral contributions at national/international scientific conferences as first author or co-author and 9 book chapters.

Invited lecture at the Young Researchers Section of the Gordon Research Conference on Industrial Ecology 2014, Tuscany, Italy. "Modelling global economy-wide material stocks in the course of the socio-metabolic transition, 1900 - 2009." Dominik Wiedenhofer, Christian Lauk, Willi Haas, Hiroki Tanikawa, Tomer Fishman, Fridolin Krausmann.

Further Infos: www.orcid.org/0000-0001-7418-3477 or www.researchgate.net/profile/Dominik_Wiedenhofer