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Doctoral Thesis

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Thesis for the degree of Philosophiae Doctor

Trondheim, December 2014

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There are in nature neither rewards nor punishments — there are consequences.

Robert G. Ingersoll

Preface

This work was carried out at the Industrial Ecology Programme (IndEcol) and the Department of Energy and Process Engineering (EPT) at the Norwegian University of Science and Technology (NTNU) in Trondheim, Norway, in the period from 2010-2014. The thesis has been submitted to the Faculty of Engineering Science and Technology (IVT) in partial fulfilment of the requirements for the degree of philosophiae doctor.

Kjartan Steen-Olsen

Trondheim, December 2014

Abstract

Whether inspired by motives of fair attribution of environmental responsibilities or the search for the most effective scheme for market driven emission abatements, reliable analyses of the environmental consequences of consumption is sought after by policymakers, researchers and the environmentally concerned citizen alike. A solid framework for such analyses exists in the form of input-output analysis, though such analyses have not been as widespread as their potential usefulness and pertinence might suggest. Following the ascent of environmental issues as a central item on the agenda even in top-level international policy negotiations, and facilitated by advances in modeling and computational capabilities, recent years have seen an increased focus on the development and application of comprehensive global input-output models for environmental assessments. The work presented here is an attempt to capitalize on the present suite of available global input-output databases to assess environmental pressures embodied in consumed goods and services, commonly referred to as environmental footprints of consumption, and discuss the reliability of the databases through a comparative assessment.

By extending the economic input-output models with environmental data, environmental flows, either directly or virtually embodied in products, can be tracked through the economy as it is modeled in the input-output system. In my PhD work, presented herein in the form of four resulting scientific papers, I have contributed to extending and adapting a global model with supplementary data to allow improved analyses of environmental pressures embodied in traded and consumed products, and to assessing a group of global models.

For a large-scale assessment of global flows of embodied land and water use, we combined data from the comprehensive FAO database on worldwide production and trade of agricultural and forestry products with a global multiregional input-output model. The more detailed representation of these products types, which are the ones that mostly contribute toward land and water use embodied in consumption, allowed increased accuracy in the modeling of these footprint types. Using this improved model to assess carbon, land and water footprints for each of the EU member states, we found EU average footprints per capita of 13.3 tons of CO_2 equivalents, 2.53 hectares of world-average bioproductive land, and 179 m^3 of consumed surface and ground freshwater for 2004. A further contribution analysis revealed the EU to be a net importer of all these environmental pressures embodied in traded products; however, there was also large such flows internally

among EU countries, with some countries having large net exports or imports with fellow EU countries.

In a further investigation of the potential for using global multiregional input-output models to assess environmental footprints even at the micro level, we extended one such model with data from the Norwegian consumer expenditure survey to allow a detailed assessment of the carbon footprint of Norwegian households from 1999 to 2012. We found a carbon footprint of 22.3 tons of CO_2 equivalents for the average Norwegian household in 2012, which was an increase of 26% since 1999. We put particular emphasis on documenting in a transparent manner the approach taken to harmonize these two datasets, so as to facilitate similar analyses for other countries, even by non-specialists, and to encourage further improvements to this method towards a common standard. The emphasis on a didactic approach was based on the rationale that an understanding of the extent and nature of environmental footprints of consumed products is vital in order to design efficient consumer-oriented emissions reductions strategies, and the recognition that detailed consumer expenditure surveys are already available in most countries, following a standard statistical framework.

In addition to these model extensions and associated footprint assessments, we have worked on assessing the reliability of the input-output databases themselves. Firstly, from the availability of several global multiregional input-output models arose the question as to whether they all coherently model the global economic structure. To give a first, tentative answer to this, we performed a comparative analysis of some macro indicators based on value added embodied in consumption, by first harmonizing three of the most important multiregional input-output databases currently available to a common framework. The comparison of gross value added embodied in the consumption of countries or of products showed significant differences even at the aggregate level. However, this observed disagreement was coherent to that found in the territorial accounts, suggesting that the most important contributor to model disagreement is in the value added accounts themselves.

Furthermore, we addressed the potential importance of the well-known limitation of limited product detail in input-output systems. We evaluated the accuracy of carbon footprint multipliers of individual input-output products by analyzing the sensitivity of such multipliers to the level of product detail in the model. This effect was evaluated by assessing, for four global models individually, how carbon multipliers react to aggregation of the input-output system. Throughout, the analysis showed carbon multipliers to be highly sensitive to reduced model detail, even if models are able to give reasonable overall footprint results.

Environmentally extended multiregional input-output analysis is a powerful tool that can provide important contributions to international as well as regional policy debates on a range of environmental challenges. Through recent collaborative research efforts these databases are now so detailed and extensive that comprehensive assessments can be made of international supply chains. Though input-output tables may never be perfect, the input-output community has some way to go still in terms of improving the sectoral detail and data foundation underlying the models.

Samandrag

Anten det er motivert av ynskje om rettferdig fordeling av ansvar for miljøpåverknader, eller av jakta på det mest effektive systemet for marknadsdrivne utsleppsreduksjonar, er pålitelege analysar av totale miljøkonsekvensar av forbruk noko som er etterspurt både av politikarar, forskarar og den einskilde miljømedvitne forbrukar. Eit solid rammeverk for slike analysar fins allereie i form av miljøutvida økonomiske kryssløpsanalysar, men slike analysar har tidlegare ikkje vore i so utbreidd bruk som det ibuande potensialet i metoden kanskje skulle tilseia. I kjølvatnet av at miljøutfordringar dei seinare åra har klatra til å verta sentrale tema på dagsordenen i politiske forhandlingar sjølv på høgste internasjonale nivå, har ein derimot i dei seinare år sett auka fokus på vidareutvikling og bruk av stadig meir omfattande kryssløpsmodellar, dels takka vera den valdsame utviklinga i lett tilgjengeleg datamaskinkraft som er ein føresetnad for å handtera slike store datamengder. Arbeidet som er presentert i denne avhandlinga er eit forsøk på å nytta dei ulike globale kryssløpsdatabasane som no er tilgjengelege til å analysera miljøpåverknader som er implisitt innbakt (eng. *embodied*) i forbruk av varer og tenester, ofte kalla miljøfotavtrykk, og å vurdera kor pålitelege databasane er gjennom samanliknande analysar.

Ved å utvida dei økonomiske kryssløpsmodellane med fysiske miljødata, kan flyten av slike innbakte miljøpåverknader sporast gjennom det globale økonomiske systemet slik det er representert i kryssløpsmodellane. I doktorgradsarbeidet mitt, presentert her i form av fire vitskaplege artiklar, har eg bidrege i arbeidet med å vurdera, utvida og tilpassa desse globale modellane med supplerande data for å kunna utføra betre analysar av innbakte miljøpåverknader.

I ein storskalaanalyse av dei globale handelsstraumane av innbakt forbruk av landareal og ferskvassressursar, i tillegg til dei karbonutslepp som ein oftare analysert miljøindikator, kombinerte me data frå FAO sin omfattande database over global produksjon og handel med ei mengd jord- og skogbruksprodukt med ein global kryssløpsmodell. Ved å representera desse produkttypane, som er dei viktigaste når det gjeld land- og vassforbruk, i meir detalj, kunne desse miljøpåverknadene analyserast med større presisjon. Ved å nytta denne forbetra modellen til å analysera fotavtrykk for EU-regionen, fann me gjennomsnittlege fotavtrykk per EU-borgar på 13,3 tonn CO_2-ekvivalentar, 2,53 hektar land, og 179 m^3 vassforbruk i 2004. Vidare analysar av korleis desse miljøpåverknadene var innbakt i internasjonal handel synte at EU-regionen var ein nettoimportør av alle miljøpåverknadene frå resten av verda. Det var òg store slike straumar internt i EU

mellom dei ulike medlemslanda, slik at nokre EU-land hadde stor nettoeksport eller -import av innbakte miljøpåverknader gjennom handel med resten av EU.

I ein vidare analyse av moglegheitene for å nytta globale kryssløpsmodellar for å evaluera miljøfotavtrykk jamvel på mikronivå, utvida me ein slik modell med data frå den norske forbruksundersøkjinga frå Statistisk sentralbyrå. Denne modellen vart so nytta for å rekna ut karbonfotavtrykket til norske husstandar frå 1999 til 2012. For 2012 synte analysen eit gjennomsnittleg karbonfotavtrykk på 22,3 tonn CO_2-ekvivalentar per husstand, noko som var ein auke på 26% sidan 1999. I dette arbeidet vart det lagt særleg vekt på å skildra framgangsmåten som vart nytta for å samstemma dei ulike datagruppene, for å bidra til eit felles rammeverk for slike analysar og leggja til rette for liknande studiar for andre land, og for å gjera det mogleg å både gjennomføra og forstå slike analysar òg for dei som ikkje er ekspertar på kryssløpsmodellering. Dette var tufta på argumentet om at ei forståing av omfanget og karakteren av miljøfotavtrykket av hushaldsforbruk er sentralt for å kunna utforma effektive strategiar for utsleppsreduksjon retta mot forbrukarar. Vidare er forbruksundersøkjingar ein ressurs som er tilgjengeleg etter eit internasjonalt standardisert system gjennom statistiske byrå i dei fleste land, som mogleggjer slike analysar med meir eller mindre same metode òg for andre land.

I tillegg til desse modellutvidingane og fotavtrykksanalysane som har vorte utført med dei utvida modellane, har me arbeidd med å vurdera kor pålitelege dei globale kryssløpsmodellane er for slike analysar. At det no er tilgjengeleg fleire slike globale modellar førte til spørsmålet om i kva grad dei ulike modellane skildrar ein verdsøkonomi som i det store og heile er den same. For å freista å gje eit første tentativt svar på dette vart det gjennomført ein samanliknande analyse av ein del makroindikatorar basert på verdiskaping innbakt i forbruk, ved å først harmonisera tre av dei mest sentrale globale kryssløpsmodellane i eit felles rammeverk. Samanlikninga av innbakt verdiskaping for ulike land og økonomiske sektorar synte monalege avvik sjølv på aggregert nivå. Dette avviket var likevel i stor grad i samsvar med tilsvarande avvik observert i det underliggjande datamaterialet.

Vidare undersøkte me den potensielle effekten av liten grad av produktdetaljar, ein velkjend veikskap ved kryssløpsmodellar. Presisjonen av karbonfotavtrykksmultiplikatorar vart evaluert ved å analysera kor sensitive desse multiplikatorane er for variasjonar i detaljnivået i kryssløpstabellane. Dette vart gjort ved å studera korleis multiplikatorane varierte når modellane vart aggregerte til eit lågare detaljnivå. Det gjennomgåande resultatet var at desse utslaga var til dels svært store, jamvel om modellane kan gje fornuftige analysar av fotavtrykk på makronivå.

Miljøutvida multiregional kryssløpsanalyse er eit kraftig metodeverkty som kan gje viktige bidrag til både internasjonale og regionale politiske debattar om ei mengd ulike miljøutfordringar. Som ei følgje av målretta forskarsamarbeid i dei seinare åra er no desse databasane so store og detaljerte at ein kan gjera omfattande analysar av internasjonale leveransekjeder. Like fullt har forskarsamfunnet framleis eit stykke att når det gjeld å forbetra sektordetaljnivå og datagrunnlaget som modellane er bygde på.

Acknowledgments

Several people are owed great thanks. First of all I would like to thank my supervisor, Professor Edgar Hertwich, for his continued support and guidance through the process. His wealth of knowledge and his thorough understanding and overview of the research frontier has been an invaluable help in directing my research. Furthermore, I am deeply grateful to Professor Manfred Lenzen at the University of Sydney for inviting me to a three month research stay and for supervising my work with two of the papers included in this thesis during and after my stay.

Dr. Jan Weinzettel was a de facto co-supervisor during the first year of my work when we worked closely together on compiling the OPEN:EU model from which my first paper came about. Together we spent months identifying, solving, and re-solving a seemingly endless number of challenges we encountered, ranging from methodological issues through inconsistent or incompatible data to programming difficulties. Though at times painful, this process was an invaluable learning experience for me, largely thanks to Jan's sharp mind and his patience with a novice colleague…

I also wish to thank to Dr. Richard Wood, input-output mastermind and my office-mate during four years. However busy he might be, whenever I was stuck on an input-output concept he would always take the time to find a blank (well, reused) piece of paper and a pen, start by drawing a square, and take it from there.

I would like to thank all my friends and colleagues at IndEcol, who have made these years so enjoyable, in, but also outside the IndEcol offices. I have especially enjoyed the many cabin trips together with all of you. I hope to be able to resume this tradition now that these last few months of intensive work has come to an end!

I am forever grateful to my parents for their never faltering love and support.

Finally, I thank my wife and best friend Ruth, who has been a constant support to me throughout this process; her love, care, humor and wisdom I will forever cherish.

Kjartan Steen-Olsen

Trondheim, December 2014

Contents

List of publications

This thesis is based on the four papers listed below as "primary publications", all of which I have been the lead author of, and all of which are appended to this thesis to represent the scientific contribution of my work. During my work as a PhD candidate over the past few years, I have co-authored an additional five articles published in peer-reviewed scientific journals, as well as two book chapters; all listed below as "secondary publications". Two of the secondary publications have been appended to the thesis for the benefit of the reader, as they are relevant to frame and provide background for the primary publications.

Primary publications

PAPER I

[1] Steen-Olsen, K.; Weinzettel, J.; Cranston, G.; Ercin, A. E.; Hertwich, E. G., Carbon, Land, and Water Footprint Accounts for the European Union: Consumption, Production, and Displacements through International Trade. *Environmental Science & Technology* **2012,** *46*, (20), 10883-10891.

Author contribution: Research idea and co-design, data collection, modeling, analysis, and writing

PAPER II

[2] Steen-Olsen, K.; Owen, A.; Barrett, J.; Guan, D.; Hertwich, E. G.; Lenzen, M.; Wiedmann, T., Accounting for value added embodied in trade and consumption: An intercomparison of global multiregional input-output databases. **2014,** *Submitted for publication in Economic Systems Research*.

Author contribution: Research co-design, data collection, modeling, analysis, and writing

PAPER III

[3] Steen-Olsen, K.; Owen, A.; Hertwich, E. G.; Lenzen, M., Effects of Sector Aggregation on CO2 Mutlipliers in Multiregional Input-Output Analyses. *Economic Systems Research* **2014,** *26*, (3), 284-302.

Author contribution: Research idea and design, data collection, modeling, analysis, and writing

PAPER IV

[4] Steen-Olsen, K.; Wood, R.; Hertwich, E. G., The carbon footprint of Norwegian household consumption 1999-2012. **2014,** *Submitted for publication in Journal of Industrial Ecology.*

Author contribution: Research co-design, data collection, modeling, analysis, and writing

Secondary publications

SUPPORTING PAPER AI

[5] Ewing, B.; Hawkins, T.; Wiedmann, T.; Galli, A.; Ercin, A. E.; Weinzettel, J.; Steen-Olsen, K., Integrating Ecological and Water Footprint Accounting in a Multi-Regional Input-Output Framework. *Ecological Indicators* **2012,** *23*, 1-8.

SUPPORTING PAPER AII

[6] Steen-Olsen, K.; Hertwich, E. G., Life cycle assessment as a means to identify the most effective action for sustainable consumption. In *Handbook of Research on Sustainable Consumption*, Reisch, L.; Thøgersen, J., Eds. Edward Elgar Publishing. In Press: 2014.

OTHER PUBLICATIONS

[7] Weinzettel, J.; Hertwich, E. G.; Peters, G. P.; Steen-Olsen, K.; Galli, A., Affluence drives the global displacement of land use. *Global Environmental Change* **2013,** *23*, 433-438.

[8] Owen, A.; Steen-Olsen, K.; Barrett, J.; Wiedmann, T.; Lenzen, M., A structural decomposition approach to comparing MRIO databases. *Economic Systems Research* **2014,** *26*, (3), 262-283.

[9] Stadler, K.; Steen-Olsen, K.; Wood, R., The "Rest of the World" – Estimating the Economic Structure of Missing Regions in Global MRIO Tables. *Economic Systems Research* **2014,** *26*, (3), 303-326.

[10] Weinzettel, J.; Steen-Olsen, K.; Hertwich, E. G.; Borucke, M.; Galli, A., Ecological footprint of nations: Comparison of process analysis, and standard and

hybrid multiregional input–output analysis. *Ecological Economics* **2014,** *101*, 115-126.

[11] Wood, R.; Steen-Olsen, K., Sustainability Assessment from a Global Perspective with the EXIOPOL Database. In *The Sustainability Practitioner's Guide to Multi-Regional Input-Output Analysis*, 1 ed.; Murray, J.; Lenzen, M., Eds. Common Grounds Publishing LLC: Champaign, IL, 2013; pp 163-175.

1. Introduction

The set of environmental challenges faced by humanity have in common that they relate to the way we interact with nature in order to run and maintain our civilization. The global economy is, in effect, a machine that takes inputs from nature, in the form of energy and material resources, and converts it to useful products for final consumption. Its byproducts are effluents back to nature in the form of emissions and waste. This overall process has come to be known as the 'industrial', 'social', or 'socio-economic' metabolism (Ayres and Simonis, 1994; Fischer-Kowalski and Amann, 2001; Haberl et al., 2011).

Essential for addressing environmental challenges is a systematic accounting framework. In this context the machine or system allegory is useful because these challenges occur at the intersection between the economic and the natural system. Traditionally, environmental accounting has also been performed at this point, for instance, water consumption or CO_2 emissions have been accounted at and allocated to the countries, companies, etc. in which they occurred. This approach to environmental accounting, to allocate emissions and other environmental stress to the process in which they occur, is called the production-based accounting (PBA) principle. Under this regime, emissions from a power plant are allocated to the power plant, tailpipe emissions from private vehicles are allocated to the car owners, and so on. At the macro level, the PBA principle entails a straightforward allocation of emissions to the countries in which they occur, similar to the approach taken in the Kyoto protocol.

As an alternative to this, the consumption-based accounting (CBA) approach allocates environmental interventions to the point of final consumption, based on the assumption that all the exchanges between the systems can in principle be associated with some final consumption. Fundamentally, the reason why the CBA approach has been so scarcely applied is the fact that unraveling the inner workings of the black box which is the global economy is a formidable task. The global economy consists of a massive amount of processes interlinked in a complex web of interactions and interdependencies, which makes the allocation process extremely challenging. However, brave attempts have been made, and as of quite recently, joint efforts in the scientific community aided by advances in computational power have led to the availability of a few extensive "multiregional input-output" (MRIO) models with global coverage, with the power to perform such analyses at a large scale.

This thesis sets out to explore how these databases can expand our understanding of the relationship between consumption activities and the environmental pressures directly or indirectly associated with them. It is based on four scientific papers which address various aspects of this: First, Paper I describes the process of extending one of the MRIO databases with additional environmental data to assess land and water use, and the application of the resulting model to the EU member countries. The following two papers have a methodological focus, delving more into comparing the various models: Are the various models in agreement in their description of the global economy (Paper II), and how important is the weakness of limited detail in the models, taking carbon footprint multipliers as a case study (Paper III)? Finally, in Paper IV the focus is shifted to an evaluation of the usefulness of global MRIO databases for environmental analyses at the micro (household) level, through combination with household expenditure survey data.

1.1 Background: The challenge of environmental sustainability

The gradual realization that our species has now grown to a size and a level of industrial development where the sum of all of our activities is starting to interfere with Earth's natural cycles and equilibria to an observable degree, has been troubling to come to for mankind. In a seminal piece, Kenneth Boulding aptly describes this as a fundamental transition from a "cowboy" to a "spaceman" economy (Boulding, 1966). Historically, the natural riches had seemed–for all practical purposes–limitless; Earth had been a cornucopia of resources just waiting to be reaped, and whenever a settlement had outgrown the capacity of the surrounding lands to sustain it, or all the local resources had been exhausted, there was always new land to settle beyond the horizon.

In the more recent era, we have been faced with the recognition that there are indeed finite limits to resource stocks and carrying capacity, and that we are starting to approach several of these. Thus as realized by Boulding, rather than cowboys, we are in fact spacemen aboard "Spaceship Earth", forced to make do with what we carry with us onboard. The fundamental lesson is that the global economy, though itself an open system, operates within the limits of nature, which is a closed system, apart from a steady input of solar energy. The challenge of sustainability is that our economic system is affecting the state variables of the natural system, obliging us to take into account the impacts of our collective actions on nature. The conversion from a cowboy to a spaceman economy thus entails assessing and understanding natural limits, and adjusting our society according to them.

The rapid growth of the global population and its affluence, energy use and material throughput since the industrial revolution has been accompanied by increasing demands and strains on the capacity of nature to supply mankind with resources and energy and assimilate our effluents, to the point where its ability to do so is in several respects exceeded (Vitousek et al., 1997; Crutzen, 2002; Rockström et al., 2009). The list of pressures exerted on the natural system by humanity is long and diverse. Anthropogenic **climate change** induced by the combustion of fossil fuels has been discussed for years, and consequences including snow and ice cover reduction, increased frequency of heat waves and extreme precipitation events, are now being observed (IPCC, 2013).

Most of the biosphere is dominated by man, in several regions of the world the majority of the **biological production** is appropriated by humans (Haberl et al., 2007). Currently, about half the global marine **fish stocks** are fully exploited, while another third are overexploited (FAO, 2010; Godfray et al., 2010). **Biodiversity** loss is occurring at an especially alarming rate, with species currently going extinct at a rate unprecedented since the last of the mass extinction events, which marked the end of the age of dinosaurs some 66 million years ago (Rockström et al., 2009). Human interference with the natural **nitrogen** cycle is now so significant that human inputs to the cycle exceed natural inputs, and the accumulation of reactive nitrogen in the environment carries a long range of negative consequences to the environment and human health (Galloway et al., 2003; Rockström et al., 2009). Human appropriation of **freshwater** for agricultural, industrial or household use is so extensive in several of the world's major river systems, such as the Nile and the Yellow River, that little water reaches the ocean at all (Brown, 2005; UNEP, 2006). In fact, earlier this year the Colorado River reached its natural destination in the Sea of Cortez for the first time in 16 years (Postel, 2014).

Resources such as freshwater, forests, crops and fish are renewable, yet they are still only available at certain finite levels during any given period of time, in other words they are *rate* constrained. Several other resources are non-renewable, leaving us no choice but to economize on the stocks available. Many countries have already had to deal with inevitably dwindling **oil and gas reserves** for a while, and similar scarcities of other non-renewable resources such as **phosphorus** (Cordell et al., 2009)[1] and **helium** (Nuttall et al., 2012) might become real issues in the not too distant future. In the other end of the system, there are limitations to the capacity of the Earth to absorb the discharge from society in the form of emissions and landfilling.

1.2 The need for a holistic, society-wide approach to sustainability

Some important common characteristics of the items in the (far from exhaustive) list presented above can be identified:

a) *With respect to their importance to humanity, the natural services under threat range from highly important to directly life-sustaining.* Obviously, all of society is crucially dependent on immediate access to clean drinking water. Similarly, phosphorus is an essential nutrient for plant growth and, by extension, for feeding humanity.
b) *The challenges are for the most part global in that overstepping a critical level carries repercussions for society at large.* For example, the extent of adverse impacts of climate change is global, their distribution independent of where the actual emissions occur. In terms of mitigation, this represents an additional well-known challenge known as the "tragedy of the commons" (Hardin, 1968): At the margin, the benefits of an additional unit of emissions is reaped by the emitter, whereas the environmental costs are shared by all.
c) *If unchecked until reaching some critical level, the adverse changes are essentially irreversible, or carry long recuperation times.* With the case of fossil fuels, the process is unidirectional: There is a finite amount available for consumption, with essentially zero addition to the reserve base. Though other resources replenish themselves, managing e.g. fish stocks is far from straightforward due to time lags, nonlinear system responses and critical thresholds which can lead to abrupt stock collapses (Whittaker and Likens, 1973; Moxnes, 2000).

[1] It should be noted that since the work of Cordell et al. (2009), the imminence of phosphate rock scarcity has been considerably reduced thanks to recent discoveries in Western Sahara which have quadrupled global reserve estimates (Van Kauwenberg et al., 2013).

The items above show the urgent need for collective actions by the global community. However, the challenge of achieving the necessary changes is rendered increasingly complicated by two additional characteristics:

d) *The exertion of environmental pressures is increasingly disconnected from their ultimate driving forces.* On account of the immense complexity of today's global industrial system, environmental interventions are separated in both time and space from the final demand of consumer goods and services which drives them.
e) *The challenges are in many senses interconnected, directly or indirectly, and myopic efforts to mitigate one threat alone may bear a cost in the exacerbation of another.* In the context of environmental assessments and sustainability research, the term *problem shifting* is used to describe cases where an environmental mitigation effort has reduced or even negative net environmental benefits because the problem is shifted in time, space or simply outside the scope of accounting, or because it is replaced by a different problem altogether (Finnveden et al., 2009). A recent, much discussed example has been the advocacy of and incentives for the use of biofuels for climate change mitigation, which could lead to extensive land use change and resulting pressures on food production and biodiversity (Tilman et al., 2009).

In the interest of moving society towards a truly sustainable state, it is vital that mitigation efforts are implemented in a way that minimizes problem shifting. This aim cannot be achieved unless environmental policies, and the research undertaken to inform them, take a holistic approach to the challenge of sustainability. Taking a holistic approach requires studying systems as a whole, rather than in parts. In a globalized and highly industrialized society, consumer products may contain parts manufactured in all corners of the world, and supply chains may involve long series of processing and assembly steps.

1.3 Consumption-based accounting of environmental pressures

The need for a system-wide approach to environmental protection provides the rationale for the development of systematic and reliable schemes for consumption-based accounting of environmental impacts of human activities. This is, however, not to say that the traditional production-based approach should be wholly abandoned even if perfectly accurate CBA models were available. It is not without reason that PBA remains the most common accounting approach. Fundamentally, directing mitigation efforts at the source makes sense when technological solutions, such as waste treatment or switching to alternative technologies, are readily available. This has been the approach taken to curb problems like emissions of ozone depleting substances or pollution of freshwater systems.

The challenge of anthropogenic greenhouse gas (GHG) emissions is representative of another class of environmental problems in the context of mitigation. GHG emissions are pervasive in society through their intimate connection with energy use; hence the consumption of a given product will have incurred emissions in most (if not all) stages of the product's supply chain. Also for climate change mitigation PBA has its merits: It is unambiguously defined and usually straightforward to quantify, characteristics that are essential towards gaining acceptance within the field of environmental policy-making. However, arguments can be made in favor of keeping complementary accounts taking the consumption perspective. Since emissions ultimately occur in response to some final demand, and there is a need to assess sustainability holistically, CBA can provide an important complementary approach to assess environmental impacts, with the potential to assign environmental responsibilities on a more just basis. CBA might also alleviate some of the practical problems experienced from PBA; strict emission regulations in some regions can lead to accumulation of emission-intensive industries in regions with more lenient regulations, the so-called 'pollution haven' hypothesis (Levinson and Taylor, 2008). In relation to the Kyoto agreement on GHG emission abatement, the term *carbon leakage* has been used to refer to the direct or indirect transfer of emission-intensive industries to countries with no binding emission reduction targets (Peters and Hertwich, 2008). Another argument in favor of accounting for impacts from the consumer perspective is that the success of mitigation efforts at the scale required to tackle a global challenge like climate change also depends on voluntary actions by private citizens. Obviously, consumers collectively carry significant potential for direct changes through their consumption patterns; perhaps even more relevant is the indirect potential of voluntary actions by a few dedicated individuals or groups, which in turn might induce policy changes to incentivize or mandate such behavior. However, without a thorough understanding of the environmental impacts embodied in various activities and consumer products, including full life-cycle effects and potential trade-offs, there is a significant risk of ineffective action and possible rebound effects (Hertwich, 2005).

1.3.1 Industrial ecology

Industrial ecology emerged as an academic field in the 1990s, based on the assertion that environmental sustainability issues should be addressed by taking a systematic, economy-wide approach, acknowledging the role of industries as central actors in environmental mitigation rather than simply as subjects of environmental regulations. By combining engineering expertise with understandings from ecology of how nature consists of a plethora of individual subsystems which draw upon one another in a finely tuned balance that maximizes the energy and material efficiency of the natural system as a whole, it was suggested that the economy should strive to mimic this behavior. This idea had been discussed theoretically by, e.g., Herman Daly two decades earlier (Daly, 1968), and the term “industrial ecology” was coined around the same time (Ministry of International Trade and Industry (MITI), 1972). The start of industrial ecology as an active academic field is usually attributed to an article by Frosch and Gallopoulos (1989) and a following colloquium on industrial ecology hosted by the American National Academy of Sciences (Jelinski et al., 1992).

Central to industrial ecology is the acknowledgment that the technosphere–i.e. the global industrial-economic system, the sum of all human activity and man-made structures–operates within the natural system. The technosphere is ultimately fueled by inputs from nature in the form of resources and energy, and also depends on nature to assimilate and regenerate the material wastes of society. Since the natural system has finite capacities to do so, the technosphere must respect its limits if it is to be sustainable. This is what industrial ecology is concerned with: The transformation of society towards a state in which the utilization of matter and energy is maximized, so that these exchanges with nature are minimized.

1.4. Input-output analysis

Input-output analysis (IOA) is an accounting framework established by Russian-American economist Wassily Leontief in the years before the Second World War (Leontief, 1936) and further developed by him and his team of researchers in the following decades, work for which he was awarded the Nobel prize in economics in 1973[2]. The fundamental element of IOA is an input-output table (IOT), which is a tabular representation of the economic transactions between the sectors in an

[2] Though the framework was new, the conceptual idea was not; in the mid-eighteenth century French economist François Quesnay had sketched a similar concept, the so-called *tableau économique*.

economy over a year. By using intersectoral transactions records to represent production functions, a model of how sectors require inputs from each other to produce their output is obtained, thereby allowing assessments of the upstream repercussions of an exogenous demand on any sector through an infinite supply chain. Today, national statistical offices routinely compile such tables, or similar tables, through the UN-based System of National Accounts (European Commission et al., 2009).

Input-output analysis is one out of a handful methods currently applied in industrial ecology. In the early years of industrial ecology, process analyses such as life cycle assessment (LCA) were especially prevalent. LCA is a bottom-up-type analysis in which factors embodied in a certain product through its life cycle are enumerated by establishing a network of industrial processes upstream and downstream and quantifying the energy and material inputs and outputs of each successive process (Finnveden et al., 2009). Similarly, material flow analysis (MFA) assesses stocks and flows of materials in an economy by taking a systemic mass balance approach (Baccini and Brunner, 2012).

Although LCA was the most important analytical tool in the early years of industrial ecology, perhaps because of its more immediate relevance to industries, the potential usefulness of IOA had been suggested from the onset (Duchin, 1992). The application of IOA to study environmental issues dates back longer still, however. In the 1960s, a dawning realization of the possible environmental damage caused by human activity[3], including long-term and indirect effects, had led several authors to propose using the already existing input-output framework to account for the environmental externalities associated with production and consumption (Daly, 1968; Ayres and Kneese, 1969; Leontief, 1970). Applications followed in the years to come, primarily concerning energy embodied in consumption (Bullard and Herendeen, 1975; Herendeen and Tanaka, 1976; Hannon et al., 1978).

More recently, input-output analysis has attracted more and more interest for its potential for analyses at the macro level. Like LCA and MFA, IOA takes a systems-based approach to analyze the flows of matter and energy in society. Mathematically, IOA is very similar to LCA[4]. The main difference lies with the approach taken to data collection and system boundaries: Whereas LCA is based on the compilation of a flow chart of the processes relevant to a product in study,

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[3] An iconic milestone is Rachel Carson's (1962) pioneering book *Silent Spring*.
[4] A formal mathematical description of the method is provided in Chapter 2.

trying to gain as much detail as possible on every stage in the supply chain (or rather: network) upstream and downstream from the demanded product, the top-down approach of IOA mitigates the issue of “cut-off errors” associated with LCA. These errors arise due to the practical requirement of defining a system border as process flow charts grow unmanageably large, meaning that impacts occurring in processes outside the boundary will be excluded from the analysis. Analyses have shown that the errors incurred through this cut-off can be as high as 50% (Lenzen and Dey, 2000). As a drawback however, IO models in general have far less detail compared to LCA systems that were composed for the purpose of the analysis of a particular product.

Due to the systemic nature of environmental challenges, there was a need for input-output frameworks able to accurately account for process networks transgressing national borders. Isard (1951) laid out the framework of multiregional input-output (MRIO) tables. In light of the large amounts of data and the extensive computations required to compile and work with such tables, and the very limited computational power available at the time, it was an impressive feat when Leontief and his team completed their world model consisting of 15 geographical regions, each with 45 economic sectors (Leontief, 1977). In the years to come, other MRIO tables would be constructed, but on account of the lingering challenges of data availability and accuracy as well as computational constraints, it is really only since a few years ago that detailed MRIO tables with global coverage have become available to researchers (Kanemoto and Murray, 2013). Using such an extensive table, Hertwich and Peters (2009) produced a first set of comprehensive carbon footprint accounts for the countries and regions of the world.

Over the last two decades or so, IOA and MRIOA have been applied to study a growing list of environmental and social externalities, including issues as diverse as greenhouse gas emissions (Hertwich and Peters, 2009; Davis and Caldeira, 2010), land and water use (Lenzen and Foran, 2001; Wilting and Vringer, 2009; Feng et al., 2011; Weinzettel et al., 2013), biodiversity (Lenzen et al., 2012), air and water quality (Kim et al., 2001; Levinson, 2010), labor (Alsamawi et al., 2014a; Simas et al., 2014) and inequality (Alsamawi et al., 2014b).

1.5 Aim of the research

In the preceding sections I have attempted to show how the current global situation of rapid growth of humankind and its impacts on the natural system on which we depend, obliges us to adopt a new way of thinking, where environmental and sustainability concerns are approached in a holistic manner. Furthermore, I have argued in favor of extending the responsibility for the environmental impacts of society to the final consumers of the outputs of the industrial system, based on the argument that this final demand is the driving force of all economic activity and ultimately for all the associated environmental impacts.

The aim of this thesis is to contribute to the understanding of the potential for IO-based assessments of environmental impacts embodied in consumption, in light of the rapid development in this field over the last few decades. Input-output databases are now becoming so extensive, both with sector and region detail and with environmental extensions, and advanced computing capabilities are now so widespread, that large scale environmental assessments are feasible at a level unattainable only a decade or two ago. The papers appended to this thesis came about as responses to sequential research questions arising from one another as I explored one path along this extensive tree or network describing the overall path towards a truly sustainable global society.

The principal research questions of the work have been:

- How can the extensive global MRIO databases available best be exploited to analyze the environmental impacts of consumption, and what lessons can we learn from such analyses?
    - How can existing multiregional input-output tables be merged with complementary data sources to provide comprehensive, detailed assessments of specific environmental pressures embodied in trade and consumption; at the (inter-)national as well as at the household level?
    - What can be said about the environmental impacts of consumption at the national and the household level, using these models?
- How reliable are current global MRIO models with respect to various environmental assessments, given their inherent limitations?
    - Are the various databases consistent in their overall representation of the global economy?
    - What are the strengths and weaknesses of such databases for environmental assessments?
    - How important are the limitations of the top-down nature of the MRIO databases towards environmental assessments focused on specific products or final consumers?
    - How can the databases be improved in this respect by drawing on additional data pertaining to the specific object of study?

The appended research papers attempt to answer these questions. The research aims of the individual papers were:

- Paper I — To develop an MRIO-based framework for assessing three fundamental environmental issues simultaneously, and use this to investigate the impacts of consumption and trade for the EU member countries
- Paper II — To assess the robustness of the underlying MRIO table for such analyses through a comparative assessment of several global multiregional input-output databases
- Paper III — To assess the accuracy with which MRIO systems can estimate environmental impacts per unit final demand at the detailed or micro level, by evaluating the sensitivity of carbon multipliers to MRIO sector detail
- Paper IV — To evaluate the potential for MRIO analysis to assess the scale and composition of the overall environmental impacts of micro-level entities such as households, by combining MRIO tables with detailed datasets such as household expenditure surveys

1.6 Structure of the thesis

The thesis is organized in four main chapters. In this introductory chapter I have established the background and rationale for my research, introduced the concepts of consumption-based accounting and input-output analysis, and stated the aims of my research. Following this, I devote Chapter 2 to a methodological description of environmental input-output analysis to the extent that it is relevant for my work. Chapter 3 contains a summary of each of the main research papers appended, including accounts of the rationale as well as descriptions and discussions of the main findings. Chapter 4 provides an overall discussion and some concluding remarks, as well as some thoughts on future research. The appendix contains full versions of the primary papers as well as two supporting papers.

2. Method: Input-Output Analysis

2.1 IOA fundamentals

An input-output table is a tabular representation of the economic activity in a specific geographic region. The IOT consists of three main components. First, the central item is the interindustrial transactions matrix $\mathbf{Z}$ $(n \times n)$, which records all sales and purchases between all the n economic sectors in the economy. This matrix describes how the various sectors require inputs from other sectors in order to produce their respective outputs for sales to other sectors, but also for final consumption. The final consumers include actors such as private households and federal governments. This final consumption is recorded in the second main component of the input-output table, the final demand matrix $\mathbf{Y}$ $(n \times d)$, which lists each of the d final demand groups' final consumption of the products from each of the n sectors. In input-output analyses the assumption is that this final demand is the driving force of the economic activity in the system as a whole.

As described above, the final demand matrix records all the sectors' sales except to other sectors. Conversely, sectors also make payments other than to other sectors. These payments are recorded in the third IOT component, the value added matrix $\mathbf{V}$ $(k \times n)$. The value added includes all non-industrial inputs to production organized in k categories such as taxes, wage payments, and profits to shareholders. In a balanced IO system, the total payments made by each industry should equal its sales, so that a vector $\mathbf{x}$ of gross industrial output by sector can be calculated from both perspectives:

$$\mathbf{Zi} + \mathbf{Yi} = \mathbf{x} = \mathbf{Z'i} + \mathbf{V'i} \qquad (1)$$

In Equation (1) and throughout, $\mathbf{i}$ represents a summation vector of ones while $\mathbf{I}$ denotes the identity matrix, both assumed to be of the appropriate dimensions[5].

In input-output analysis, this table of transactions is taken to represent production functions, in other words the total payments by a certain industry j, tallied in the jth column of $\mathbf{Z}$ and $\mathbf{V}$, represent the inputs required by j in order to produce a total of x_j units of its output. Hence, by simply dividing each of these columns by

[5] We use bold uppercase/lowercase variable names to represent matrices/vectors, respectively, while scalar variables are denoted in lowercase italics. All vectors are assumed to be column vectors by default, hence row vectors are denoted with a transposition sign ('). A circumflex (^) denotes diagonalization.

the purchasing sector's total sales, matrices $\mathbf{A}$ $(n \times n)$and $\mathbf{V}_\mathrm{c}$ $(k \times n)$of sectoral input requirements per unit of output produced are obtained:

$$\mathbf{A} = \mathbf{Z}\hat{\mathbf{x}}^{-\mathbf{1}} \quad (2)$$
$$\mathbf{V}_\mathrm{c} = \mathbf{V}\hat{\mathbf{x}}^{-\mathbf{1}} \quad (3)$$

Under the assumption that $\mathbf{A}$ gives production functions for all sectors, an element a_{ij} of $\mathbf{A}$ describes sector j's purchases of sector i's output per unit produced of its own output. Standard practice is to describe the transactions in $\mathbf{Z}$ and $\mathbf{Y}$ in monetary units (say, \$); the unit of a_{ij} is thus dollars' worth of i per dollar worth of output from sector j. This is the assumed input that sector j requires from sector i to produce a single unit of its output.

By further assuming that these coefficients are static, by extension, for sector j to produce b dollars' worth of its output, it requires inputs of ba_{ij} dollars' worth of sector i's products. For instance, a coefficient $a_{flour,bread} = 0.1$ means that in order to produce 20 dollars' worth of bread, the bread sector directly requires inputs of 2 dollars of flour from the flour sector. As explained above, these coefficients are determined as average values for the IOT reference year by (in the bread example) dividing the bread sector's total payments to the flour sector by the bread sector's gross output for that year.

By inserting Equation (2) into the fundamental material balance (1), then, we derive an expression describing total output $\mathbf{x}$ as a function of the total final demand by sector, $\mathbf{Yi} = \mathbf{y}$:

$$\mathbf{x} = \mathbf{Zi} + \mathbf{y} \quad (4)$$
$$\mathbf{x} = \mathbf{Ax} + \mathbf{y} \quad (5)$$
$$\mathbf{x} = (\mathbf{I} - \mathbf{A})^{-1}\mathbf{y} = \mathbf{Ly} \quad (6)$$

By assuming that the coefficients of $\mathbf{A}$ are fixed, i.e. the input from sector i to sector j depends only on the amount produced by sector j, Equation (6) holds true for any final demand $\mathbf{y}^*$, yielding the total output $\mathbf{x}^*$ from all sectors induced by that final demand[6].

[6] Henceforth, the asterisks are omitted, and any vector $\mathbf{x}$ is assumed to be the gross output associated with $\mathbf{y}$ which represents any final demand.

Whereas the elements of $\mathbf{A}$ quantify inputs required per unit produced, an element l_{ij} of the *Leontief inverse* $\mathbf{L} = (\mathbf{I} - \mathbf{A})^{-1}$ gives the total output of sector i induced per unit final demand of sector j's output, including all indirect production occurring upstream in the production chain. For instance, returning to the bread example above, the coefficient $l_{flour,bread}$ includes the total amount of flour production required per dollar worth of bread delivered for final consumption. For this reason, $\mathbf{A}$ is called the *direct* requirements matrix, whereas $\mathbf{L}$ is the *total* requirements matrix.

A Taylor series expansion of the Leontief inverse in Equation (6) helps shed light on how a final demand leads to additional activity upstream in a theoretically infinite supply chain of sector interdependencies:

$$\begin{aligned}\mathbf{x} = \mathbf{L}\mathbf{y} = (\mathbf{I} - \mathbf{A})^{-\mathbf{1}}\mathbf{y} &= \sum_{i=0}^{\infty} \mathbf{A}^{i}\mathbf{y} \\ &= (\mathbf{I} + \mathbf{A} + \mathbf{A}^{2} + \mathbf{A}^{3} + \cdots)\mathbf{y}\end{aligned} \tag{7}$$

The terms inside the parentheses in the written out series expansion in Equation (7) are referred to as production *tiers*. The series shows how a given final demand generates activity upstream in the supply chain. To deliver the desired output for final consumption (the "zeroth" tier, $\mathbf{y}$), there must be additional production of the direct inputs required to produce this ($\mathbf{A}\mathbf{y}$). This production in turn requires direct inputs of its own ($\mathbf{A}^{\mathbf{2}}\mathbf{y}$), and so on.

2.2 Factors embodied in consumption

In the previous paragraphs we have outlined how IOA can be used to determine the gross output by each sector following an exogenously given final demand. Following Equation (3), $\mathbf{V}_{\mathrm{c}}$ gives direct factor requirements per unit output from each sector. By assuming these to be fixed in the same way as the direct requirements coefficients in $\mathbf{A}$, the factor contents accumulated in the supply chain to produce a certain delivery for final consumption is given by:

$$\mathbf{v} = \mathbf{V}_{\mathrm{c}}\mathbf{x} = \mathbf{V}_{\mathrm{c}}\mathbf{L}\mathbf{y} \tag{8}$$

In IO terminology, this is referred to as factor contents *embodied* in consumed products.

In environmentally extended input-output analysis (EE-IOA), the IO system is appended with a matrix $\mathbf{F}$ (dimensions $s \times n$) of s environmental extensions. This lists the total direct environmental interventions by each industry over the year, e.g. tons of CO_2 emitted, m^3 of water consumed, kWh of energy used, etc. This matrix can include as many environmental extensions as desired, and each extension can be expressed in any unit desired. In EE-IOA, this matrix is treated mathematically exactly like $\mathbf{V}$. First, it is converted to coefficient form, analogously to Equation 3:

$$\mathbf{S} = \mathbf{F}\hat{\mathbf{x}}^{-1} \tag{9}$$

The total environmental factor contents of consumption are then given by:

$$\mathbf{d} = \mathbf{Sx} = \mathbf{SLy} \tag{10}$$

The Leontief inverse gives in a single matrix the direct relationship between any final demand and the resulting total output by each sector (Equation (6)), and the row vector given by summing down its columns ($\mathbf{i'L}$) gives the overall gross output resulting for a unit final demand on each sector. As such, if the jth element of $\mathbf{i'L}$ is 1.6, a final demand of one dollar placed upon sector j leads to 1.6 dollars' worth of outputs from all sectors of the economy combined. This includes the final demand itself, hence a final demand on sector j incurs upstream economic activity that leads to an additional 60% gross output.

Correspondingly, each element m_{ij} in the matrix $\mathbf{M} = \mathbf{SL}$ (dimensions $s \times n$) gives directly the total amounts of the ith environmental extension embodied in one unit final demand of commodity j. We refer to any such matrix of factors that directly relates a unit of final demand with the resulting total (direct + indirect) output, impact, factor use etc. as a *multiplier* matrix.

2.3 Multiregional input-output analysis[7]

So far we have assumed that the IO system comprises an entire (global) economy. In practice, standard IOTs are constructed by national statistics offices for their own country. In this case, imports and exports must be accounted for as well to complete the transactions accounts. The simplest solution to preserve the

[7] The derivations in this section are based on the account given by Peters & Hertwich (2004).

production balance for the region is to include exports as an additional category $\mathbf{y}^{\mathrm{ex}}$ of final demand:

$$\mathbf{x} = \mathbf{A}^{\mathrm{d}}\mathbf{x} + \mathbf{y}^{\mathrm{d}} + \mathbf{y}^{\mathrm{ex}} \tag{11}$$

With the introduction of more than one region we have included superscript d (domestic) to the original variables. $\mathbf{y}^{\mathrm{d}}$ is now the final demand on domestic industries, and $\mathbf{A}^{\mathrm{d}}$ is the domestic requirements matrix. Note that the latter is now distinctly different from the requirements or technology matrix $\mathbf{A}$, because it only includes input requirements from domestic sectors. The overall technology matrix is in fact a sum of domestic and imported inputs to production:

$$\mathbf{A} = \mathbf{A}^{\mathrm{d}} + \mathbf{A}^{\mathrm{im}} \tag{12}$$

In order to analyze factors embodied in imported goods, the import requirements matrix $\mathbf{A}^{\mathrm{im}}$ must be known, as must the environmental intensities of production of the imported goods. Unfortunately, this information is rarely available from statistical offices. Furthermore, $\mathbf{A}^{\mathrm{im}}$ will in reality be a sum of contributions from many trading partner regions. Methods of simplification exist to allow analyses to be conducted with limited data availability, such as the domestic technology assumption, where all imported goods are assumed to have been produced with identical technologies as their domestic equivalents (Lenzen et al., 2004; Peters and Hertwich, 2004). However, due to the increasing importance of international trade and the heterogeneity of regional technologies, such simplifications can be associated with large errors (Peters et al., 2004; Peters and Hertwich, 2006).

In response to an increasing degree of regional specialization and international trade of goods and services, several multiregional input-output tables (MRIOTs) have been compiled in recent years. In an MRIOT, domestic IOTs for several regions are linked together. The matrices and equations remain the same, however the dimensions are increased: Assuming that there are m regions, and that each region is modeled with n sectors, the dimensions of $\mathbf{Z}$, $\mathbf{V}$ and $\mathbf{Y}$ are increased to $(mn \times mn)$, $(k \times mn)$ and $(mn \times md)$, respectively. Written out in matrix form, Equation (4) for a multiregional system becomes:

$$
\begin{bmatrix} \mathbf{x}_1 \\ \mathbf{x}_2 \\ \vdots \\ \mathbf{x}_m \end{bmatrix} = \begin{bmatrix} \mathbf{Z}^{11} & \mathbf{Z}^{12} & \cdots & \mathbf{Z}^{1m} \\ \mathbf{Z}^{21} & \mathbf{Z}^{22} & \cdots & \mathbf{Z}^{2m} \\ \vdots & \vdots & \ddots & \vdots \\ \mathbf{Z}^{m1} & \mathbf{Z}^{m2} & \cdots & \mathbf{Z}^{mm} \end{bmatrix} \begin{bmatrix} \mathbf{i} \\ \mathbf{i} \\ \vdots \\ \mathbf{i} \end{bmatrix} + \begin{bmatrix} \mathbf{Y}^{11} & \cdots & \mathbf{Y}^{1m} \\ \mathbf{Y}^{21} & \cdots & \mathbf{Y}^{2m} \\ \vdots & \ddots & \vdots \\ \mathbf{Y}^{m1} & \cdots & \mathbf{Y}^{mm} \end{bmatrix} \tag{13}
$$

Each sub-matrix in Equation (11) has the dimensions of the corresponding matrix described in Section 2.1. Along the diagonal of the block matrix $\mathbf{Z}$, each sub-matrix $\mathbf{Z}^{rr}$ represents the domestic IOT of region r, while an off-diagonal sub-matrix $\mathbf{Z}^{rs}$ represents sales from each of the sectors in region r to each sector in region s. Similarly, in the final demand matrix $\mathbf{Y}$ an off-diagonal sub-matrix $\mathbf{Y}^{rs}$ represents direct imports of region r's products by final consumers in region s. Using this framework, the upstream effects of final demand can be analyzed consistently through the entire global economy.

The mathematical framework laid out in this chapter has been underlying the analyses conducted throughout the papers included in this thesis. For specific methods the reader is further referred to the methods sections of the individual papers.

3. Summaries of papers and discussion of main findings

This section presents a summary of the papers appended to the thesis, presenting and discussing their main findings. We present the research questions and rationale for each paper, how they arose over the course of the research period, and how they relate to the fundamental research questions of the thesis.

Paper I (Steen-Olsen et al., 2012) presents accounts of carbon, land and water footprint indicators for the EU countries, and analyze how environmental pressures embodied in consumption are displaced internationally through trade. The analysis was performed using a model based on the theoretical groundwork laid in **Supplemental Paper AI** (Ewing et al., 2012).

Paper II (Steen-Olsen et al., Submitted-a) features a comparison of three of the most important global MRIO databases currently available, with the aim of determining whether they coherently model the global economy.

Paper III (Steen-Olsen et al., 2014) is focused on the accuracy with which input-output systems estimate environmental impact multipliers for individual commodities. The analysis is based on a comparison of carbon footprint multipliers calculated from full and aggregated versions of four MRIO tables.

Paper IV (Steen-Olsen et al., Submitted-b) addresses the potential for using consumer expenditure surveys to supplement input-output tables in order to improve IO-based assessments of environmental impacts of consumption at the household level, taking the Norwegian household carbon footprint as a case study to highlight benefits, limitations and challenges for further improvements. **Supplemental Paper AII** (Steen-Olsen and Hertwich, In press) served to inform the current understanding of environmental impacts of specific household consumption activities.

Paper I: Carbon, Land, and Water Footprint Accounts for the European Union: Consumption, Production, and Displacements through International Trade

Paper I (Steen-Olsen et al., 2012) came out of an EE-MRIO model that was constructed by the authors as part of the EU FP7 project "*One Planet Economy Network: Europe*" (OPEN:EU), which aimed to develop a "footprint family" of sustainable development indicators and integrate these in a modeling framework for evidence-based policy. The OPEN:EU project was initiated as a result of the WWF's 2006 Living Planet Report (WWF, 2006), which concluded that the environmental impacts of the European economy were nearly three times the sustainable level. The OPEN:EU project was founded on the recognition that as the world's largest economy and a consumer of disproportionally large shares of the global supply of energy and resources, the EU should take the lead in the transition to a sustainable global economy.

Acknowledging that true sustainability requires attention to a multitude of challenges simultaneously, three well-developed environmental footprint indicators were identified by the project researchers; see Supporting Paper AI in Appendix E (Ewing et al., 2012). These indicators, though by no means a complete account of sustainability, were developed independently to quantify human-induced environmental pressures within three central dimensions of environmental sustainability. Slightly modified from the original set, the footprints chosen for our analysis include the carbon footprint (CF), assessing anthropogenic GHG emissions contributing to global warming; the land footprint (LF), a variant of the Ecological Footprint which quantifies human appropriation of crops, forests and animal products, represented as productivity-weighted hectares of land ("global hectares", gha)[8]; and the blue water footprint (WF_b), which quantifies human requirements for ground and surface water. As described by Ewing et al. (2012) these independent indicators were joined in a common framework based on a multiregional input-output model to analyze indirect impacts in supply chains.

Based on the theoretical framework laid out in (Ewing et al., 2012) and (OPEN:EU, 2009), the OPEN:EU model was constructed and used to analyze the environmental impacts of consumption of the EU member countries in Paper I. The model was based on an MRIO table constructed from version 7 of the Global Trade Analysis Project (GTAP) database (GTAP, 2007), which represents the global economy with 113 regions and 57 economic sectors for the reference year 2004 (Peters et al., 2011). We capitalized on the comprehensive database compiled by the Food and Agriculture Organization of the United Nations (FAO) on the production and trade of primary products of agriculture and forestry to construct a detailed table modeling these product flows through the economy. The table was estimated in part from, and set up to work as an extension to, the GTAP MRIO table. This enabled an improved representation of land and water footprints in particular, as

[8] The Ecological Footprint translates human consumption into requirements on the biological capacity of the Earth in terms of hectares of biologically productive land. The global stock of biologically productive land is modeled as belonging to either out of five different types of land, each with their own assumed degree of biological productivity.

they chiefly relate to these primary products, which are usually not represented in great detail in IO systems.

Applying the model to analyze consumption in the EU, we found per-capita carbon, land and blue water footprints for the EU as a whole of 13.3 tCO_2e, 2.53 gha and 179 m^3, respectively, while the corresponding global averages were found to be 5.7 tCO_2e, 1.22 gha and 163 m^3. For all three indicators, the EU was a net importer of factors embodied in trade from the rest of the world; these net imports constituted 23% of the CF, 23% of the LF and 38% of the WF_b of the EU.

Beyond these averages, the analysis showed large variations among the individual EU countries, and revealed that in the case of the CF and LF, some countries were actually net exporters of embodied factors. Still, for both the CF and the LF, each country's per-capita footprint was higher than the world averages in all cases but the Romanian carbon footprint. Four north-eastern EU countries–Finland, Sweden, Estonia and Latvia–had significant net exports of embodied land use, due to a combination of large forestry and fisheries industries and low population densities. The country comparison for the WF_b showed the largest variation due to climatic differences. Even though all countries were net importers of embodied blue water use, several countries, generally the transition economies in Central and Eastern Europe had WF_b/c of only a fraction of the EU and global averages (which were found to be almost on par). The very high variation was mostly due to the fact that the analysis explicitly excluded direct use of rainwater (green water in water footprint terminology); the lion's share of the water footprint is related to agriculture, and crops in many European countries are largely rainfed[9]. For the same reason, arid or semi-arid countries with large agriculture sectors such as Spain and Portugal showed very high WF_b/c.

The MRIO framework further allows an analysis of the displacements within the EU. This analysis highlighted Poland as a hub for GHG-intensive industry to satisfy consumption elsewhere in the EU, while the UK was the largest net importer of

[9] Whether or not to include green water use depends on the research question. Though blue water use is arguably more closely related to water stress issues since it is directly extracted by man from limited reservoirs, green water use is not without consequences since water consumed by the growing crops is cycled back to the atmosphere through plant transpiration and as such is made unavailable for other plants and surface water. In our case, a further reason to exclude green water has been that rainfall can be argued to be a property of the land on which it falls, contributing to its potential biological productivity; hence it is already accounted for in the land footprint.

embodied GHG emissions from fellow EU countries. As for land use, France was the largest net supplier of embodied land use within the EU, while the UK and Italy had the largest net imports. Again, the variation between countries was the largest when studying embodied blue water. Spain emerged as the undisputed exporter of embodied blue water use in Europe. At the other end of the scale, Germany and (again) the UK were the largest net importers of embodied pressure on water supply.

Overall, the compilation of the OPEN:EU model and the analysis performed here showed the potential for improved MRIO-based environmental assessments by capitalizing on supplementary data. The analysis confirmed the disproportionately high pressures exerted on the environment by consumers in affluent European countries. However, the breakdown by member state revealed significant differences with respect to per capita footprints as well as among the different footprints. Different regions specialize in the use of different factors of production to supply products for the global market based on their respective resource endowments, for instance, the Nordic countries export embodied land use through their large forestry sectors.

Paper II: Accounting for value added embodied in trade and consumption: An intercomparison of global multiregional input-output databases

The OPEN:EU model used in Paper I was built around an MRIO table constructed from the GTAP7 database. As of quite recently there is now a handful such extensive global-coverage MRIO systems available, hence the question arose as to the uncertainty associated with the MRIO part of the OPEN:EU model and the analysis based on it. An analytic calculation of uncertainty accumulation in MRIO analyses is unfeasible (Lenzen, 2000); however with the parallel emergence of a handful global MRIO databases, a cross-comparison of the overall model agreement in terms of some central indicators can serve as a valuable indicator of the credibility of MRIO-based assessments.

Paper II (Steen-Olsen et al., Submitted-a) consists of such an intercomparison of three of the most important global MRIO databases available: Eora, GTAP8 and WIOD. MRIO-based environmental assessments have usually had little or no discussion or analysis of the uncertainty of the database chosen for the analysis, instead the focus is generally on the accuracy of whatever environmental extension or additional layers were added to it. Nor was there in general any discussion of the choice of one database over any of the others. Traditionally, this had been justifiable in light of the very limited availability of such extensive global-coverage MRIO tables. In recent years, however, in response to a growing interest in assessments of emissions embodied in trade and consumption at the international level, several MRIO projects have emerged in parallel to compete with the MRIOT derived from the GTAP database, which was not compiled with IOA in mind at all[10].

We compared GTAP8 with two such MRIO tables. First, the Eora database was developed by researchers at the University of Sydney. It features a time series of MRIO tables from 1990 to 2011, with a high level of detail. Second, the World Input-Output Database (WIOD) was constructed by a consortium led by the University of Groningen, based on the OECD input-output tables. In order to compare the MRIO tables in their "purest" format, we focused on value added embodiments rather than CO_2 or any other indicator which is not intrinsically a part of the input-output system. By excluding such extensions we also avoid the added uncertainties associated with data in physical units.

To allow a comparison across the databases, they must be converted into a harmonized framework. To avoid making any changes to any of the databases apart from simple aggregation, we defined a "Common Classification" (CC) framework as a set of regions and sectors such that all the MRIO tables could be directly aggregated into it. Taking the principle of greatest common factor, the resulting CC system consisted of a set of 17 sectors and 41 regions, with the regions being especially detailed for Europe due to the European focus of WIOD.

The comparison was focused on the matrix $\mathbf{\Phi}$, which was calculated for each MRIO database. $\mathbf{\Phi}$ is a concatenation of m matrices $\widehat{\mathbf{v}_c'}\mathbf{L}\widehat{\mathbf{y}^r}$ $(mn \times mn)$, one for each region's final demand activities. This matrix simultaneously gives value added accounts from two perspectives:

1. The production perspective: The production-based accounts are found by summing up $\mathbf{\Phi}$ horizontally, across columns. This is the value added generated in each region-sector, as assumed by each MRIO database.
2. The consumption perspective: The consumption-based accounts, i.e. accounts of value added embodied in consumption, are found by summing

[10] On the GTAP website it is expressly stated that «*the GTAP Data Base is NOT a repository of Input-Output tables. (…) Users building IO tables based on this information do that under their own risk, and are assumed to understand the limitations imposed by the process of data base construction.*» https://www.gtap.agecon.purdue.edu/databases/v8/default.asp.

up $\mathbf{\Phi}$ vertically, across rows. This represents the results of the input-output analysis through the Leontief inverse, in other words the reallocation of the production-based accounts through an infinite supply chain, as outlined in Section 2.1.

On the whole, we find discrepancies between the databases that are not insignificant, when comparing macro indicators. A comparison of global value added generation in each sector according to the different models yielded a high degree of agreement for some sectors, but rather low for others; the relative standard deviations (RSD) ranged from 2% to 31%, with an output-weighted average of 8%. An analogous comparison was performed from the consumption perspective, that is, on how the models allocate the global value added to final demand on the various sectors upon application of the Leontief inverse. Again, some considerable discrepancies were observed, but interestingly the agreement and disagreement in the individual sectors were largely the same as those observed from the production perspective. In other words the process of reallocating value added from where it is generated to where the final demand is put, which entails running the production-based value added accounts through the Leontief inverse, in effect an infinite series of sector interdependencies, did not tend to amplify discrepancies in the source data. This is encouraging for MRIO compilers and analysts, as it suggests that efforts to ensure the accuracy of some key sectors or regions in the MRIO system can be sufficient to obtain acceptable uncertainty levels.

The three MRIO databases represent the same global economy described in terms of a common set of regions and sectors and as such they should in theory be the same. To further determine whether any one of them was markedly different from the others, a quantitative statistical comparison was performed, again based on the $\boldsymbol{\phi}$ matrices for the aggregated versions. This was performed using several statistical indices developed to evaluate matrix (dis-)similarity, by comparing each of the 3!=6 possible model pairings individually. As there is no single universally accepted method to evaluate for a set of matrices if any one of them is more or less similar to one another, we decided to present and apply several candidates identified in a literature review, and report how each indicator ranks each model pair as more or less similar. The result of this comparison, though tentative, singled out the Eora database as more dissimilar, whereas GTAP and WIOD were deemed closer to one another by most of the indicators.

Paper III: Effects of sectoral aggregation on CO_2 multipliers in MRIO analyses

Although forced into a common framework for the sake of comparison, the MRIO databases compared in Paper II were in fact quite different in terms of the level of detail with which they describe the global economy. Though MRIO models may be able to give fairly accurate footprint estimates at the macro level, another question remains as to their accuracy in modelling individual products. For assessments of environmental factors such as GHG emissions, factor intensities can be highly different between sectors. As such, the intercomparison led to the question of how sensitive environmental multipliers (in this case represented by CO_2) are to sector detail in EE-MRIOA.

In Paper III (Steen-Olsen et al., 2014) we investigate the three MRIO databases studied in Paper II, as well as the EXIOBASE, an MRIO table compiled by a research consortium as part of the EU FP7 project EXIOPOL. The paper describes the results of an investigation of how CO_2 multipliers change when information is removed from the background economy. Each database was aggregated to the CC classification system described previously, and subsequently carbon footprint multipliers were calculated for each CC region-sector using both the aggregate and the full versions of each database, referred to as pre- and post-aggregation, respectively. The post-aggregated multipliers were calculated by first calculating footprints from the full versions of the respective databases, then aggregating the footprints as well as the final demand to the CC level before calculating implicit multipliers as the ratio between them.

Since the four databases come with different levels of sector detail, aggregating all of them down to a common baseline entails significant loss of detail for some but only minor changes for others. The comparison of the pre- and post-aggregated multipliers for each database showed large variations between the aggregated CC multipliers across regions, and significant aggregation errors when the multipliers were calculated from the aggregated as opposed to the full models. The aggregation error was significantly larger when the original database was more detailed and thus had lost more detail in the aggregation. This finding underlines the importance of high level of detail for the carbon multiplier values for individual sectors, even if overall carbon footprint analyses for a basket of goods, such as a national carbon footprint analysis, will generally be less affected due to the variation in relative importance among multipliers.

The procedure of aggregating to the CC baseline entailed various degrees of aggregation not just between databases, but between CC sectors. A comparison of the full set of carbon multipliers for the original sectors with the multiplier of the sectors defined to represent them in the CC system revealed very large differences. Countries with the most detailed classification included in the Eora MRIO table include several hundred commodities, and the multipliers of the individual sectors aggregated into the same CC sector differ by an order of magnitude in many cases. The same was observed for the other databases, and for service sectors as well as for manufacturing and primary industries.

Paper IV: The carbon footprint of Norwegian household consumption 1999-2012

Though very well suited to analyze factor embodiments at an (inter)national level, MRIO tables are not ideally set up for analyses at the household level. Household consumption often exists as a consumption category of its own in IO systems, but IO sectors and commodities are usually not very well matched with actual household purchases and activities. The analysis in Paper III showed that the CO_2 embodiments per unit of consumption can vary very much at the product-specific level.

In Paper IV (Steen-Olsen et al., Submitted-b) we are concerned with the improvement and adaptation of MRIO tables to be better suited to inform consumers and policymakers about the household carbon footprint of consumption, how specific purchases contribute to it, and how changes in consumption can contribute to footprint reductions. The paper addresses the use of consumer expenditure surveys (CES) as a source of complementary information on household activities. Practical challenges for the adaptation of CES data to MRIO footprint analyses are discussed, and a suggested procedure is outlined taking the Norwegian CES combined with the EXIOBASE 2 MRIO database as a case to analyze the Norwegian household carbon footprint (hhCF) developments from 1999 to 2012.

The regular collection of consumer expenditure surveys by national statistics offices is a firmly established statistical tradition. A CES consists of the surveying of a large number of households drawn randomly from the population, which are instructed to record all their purchases and expenses over a given period of time. Purchases are usually classified within the COICOP system, a harmonized system of commodities defined by the United Nations statistics division. The COICOP is a highly detailed scheme with a hierarchical structure of several levels, allowing users to define a custom version suited to the consumption characteristics of the population surveyed. The version currently used by Statistics Norway (SSB) for the Norwegian CES features 183 unique commodities.

The combination of a CES dataset with an IO system entails several practical challenges that must be addressed. These challenges include harmonization in terms of product classification systems, valuation and pricing schemes, differing reference years, as well as consolidation in cases of disagreement between the datasets. By addressing these challenges the footprint of households can be analyzed; however to fully capitalize on the full extent of the CES data, the biggest challenge is to assign environmental multipliers as found in the MRIO system to individual COICOP commodities.

We apply the EXIOBASE 2 database to the Norwegian CES and present a temporal analysis of the Norwegian household carbon footprint from 1999 to 2012. A main focus of the paper is to give a didactic account of the procedure taken to combine the datasets and discuss limitations, assumptions, and remaining challenges to further improve the accuracy of such analyses, to encourage the further use of this sort of analysis also for non-experts of IOA. The results of our case study showed an average carbon footprint per household of 22.3 tCO_2e in 2012, a 26% increase since 1999. As confirmed by several previous studies, transport, housing and food were the most important consumption categories contributing towards the total carbon footprint.

4. Discussion and conclusions

Supplementing the discussion of the individual papers in Chapter 3 as well as in the papers themselves (Appendices A-D), this chapter provides a discussion of the scientific contribution of the work presented in the thesis as a whole, as well as its limitations. Finally, I offer some concluding remarks and my thoughts on future research in this area.

4.1 Scientific contribution

As the scope of this thesis is rather wide and the principal research questions of a rather general nature, no single scientific contributions of this thesis can be identified. Instead, I will here discuss the contributions of the various papers in turn, and rather take the bird's eye perspective in the conclusions in Section 4.3.

The model I co-developed under the OPEN:EU project, which led to Paper I, I believe made two significant contributions (though their credit is by no means mine alone): First, the harmonization of several independently developed and quite different environmental indicators into a common methodological framework that allows them all to be assessed simultaneously for any arbitrary final demand, with a full global MRIO model to assess indirect/upstream effects. Most water and land footprint assessments had thus far been based on process models. In Supplemental Paper AI, the idea of a "footprint family" integrated into a common modelling framework based on an MRIO framework was conceptualized, and the following work in the OPEN:EU project brought this concept from the drawing-board to a working model, which is described in Paper I. Second, the development of a parallel transactions matrix of physical flows of primary products of agriculture, designed to be used alongside the MRIO database as a regular environmental extension matrix. This allowed a fuller utilization of the wealth of data available from the FAOSTAT database, by allocating individual crops to primary users rather than to the original producer, which would in practice be one of the few agriculture or forestry sectors in the MRIO table. As to the analysis of EU footprints in Paper I; though national footprint accounts had been constructed previously, a complete set of accounts for all the individual EU countries, including three footprint types simultaneously as well as analyses of consumption and trade internally in the EU as well as with external trading partners I think was a real contribution that provided relevant and valuable information to researchers and policymakers alike.

Papers II and III were intended to contribute as cogs in the IO community's combined effort of assessing and improving the reliability of global MRIO databases to further their use in research as well as in the domain of environmental policymaking. In Paper II our aim was to provide a first take on the fundamental question of whether, or to what degree, various available MRIO databases do in fact give structurally similar descriptions of the global economy. Since the availability of several such databases is a recent luxury to MRIO analysts, this very broad research question was one that had not yet been studied in any detail. As such, we aimed to perform our comparative analysis at a very fundamental, basic level; hence the analysis is very general, with comparisons only at the macro level without any in-depth investigations of individual sectors or structural paths. For the same reason, we used value added as the embodied factor, rather than any environmental or social factor, to allow a comparison of the databases in their "purest" form. The rationale for this was to allow our work to serve as a stepping stone for further, more in-depth research without imposing any constraints on analysts. In this paper we were able to show that there are considerable differences between the various global MRIO systems even at the most aggregate sector levels; however, our results also indicated that much of the error may come from disagreements in the raw data (in this case, value added accounts) rather than originating in the model compilation stage.

The analysis in Paper III on the effects of aggregation I believe provided an important update to the classic analytical works on the subject in light of 1) the aforementioned recent availability of several comprehensive MRIO databases with global coverage, which allowed an analysis on highly relevant real-world tables rather than smaller single-regional or hypothetical systems, and 2) the more recent trend of using MRIO for environmental rather than economic assessments, for which several of the conclusions of previous studies might not hold. Based on the analysis, we were able to show that MRIO-based carbon multipliers are quite sensitive to the level of detail with which the background economy is described in the MRIO system, to the degree that the choice between any of the most important databases available today may significantly affect model results. This sensitivity is not limited to the detail of the sector of the individual multipliers themselves, but also that of the supplying sectors upstream in the supply chain.

The rationale for Paper IV was twofold. First, to culminate my research for this thesis by moving from the very large-scale (global or European) level of Papers I-III down to the national level and use the EE-MRIO framework to analyze environmental impacts embodied in consumption at the much more tangible or "real" household level. Little such work had been done focusing on Norway since the pioneering work by Herendeen (1978), and hence in my view this paper filled a knowledge gap which can hopefully serve to inform the Norwegian climate debate. Second, a special emphasis was put on providing a didactic account of the procedure taken in our analysis. Again, this was motivated by a conviction that MRIO-based environmental assessments such as this one need overall to be performed more coherently, to reduce ambiguity, but also that it may serve as a basis for further work also by researchers outside the field of input-output analysis. Furthermore, the didactic approach was specifically intended to lower the threshold of the results being used by actors outside the world of academia as well as within it.

Overall, it is my hope that the papers that constitute this thesis may help pave the way towards a more widespread use of consumption-based accounting of environmental externalities in general, at the (inter)national as well as the household level, and specifically the use of global MRIO frameworks to construct such accounts.

4.2 Limitations

The wide and general scope of the thesis carries the implication that the research questions addressed at each stage are not exhaustively explored. The specific limitations associated with each study will not be repeated here; instead I offer a few general remarks on some of the prevailing weaknesses that imply uncertainties associated with the results of MRIO-based environmental assessments.

Perhaps the most central recurring issue is that of limited product and industry detail in MRIO systems. Though probably not a critically important limitation for footprint assessments at the international level, it is certainly problematic for any meso- or micro-level assessments. Even among the most detailed MRIO tables currently available, firms and products that are in some respect entirely different will be grouped together, and the resulting hypothetical aggregate created as a weighted average of them may not be very representative of any real product. This is the case, for instance, with luxury or high-end products, which are assumed to be the same as average or low-end products of the same type, as discussed in Paper IV. Similarly, typical MRIO sectors such as “Rubber and plastic products”, “Fabricated metal products”, “Electrical machinery and apparatus n.e.c.” obviously cover wide ranges of products that are physically–and environmentally–very different.

Another drawback of many MRIO-based assessments, including those in this thesis, is that of time lags. The analysis for Paper I, though published in 2012, had to be conducted for the year 2004, as that was the reference year for the most recent version of the GTAP database. In the field of environmental footprint accounting, especially concerning greenhouse gas emissions, technologies and consumption patterns can change significantly over the course of a few years; a glance at the developments in China over the past decade underscores this point. However, the current trend in MRIO database compilation is to construct time series with a lag of only a couple of years. Even if these require some assumptions to be made, they represent a significant step forward for EE-MRIOA.

Uncertainties in IO tables and the analyses based on them have traditionally been overlooked or under-communicated. This, I admit, is a weakness also of the work presented here. Uncertainties have only been qualitatively discussed, with no real systematic sensitivity or uncertainty analyses undertaken. With the recent exception of Eora, uncertainties in MRIO tables are usually not supplied, and to estimate these through reverse engineering approaches is challenging. In the case of GTAP, for instance, the database is compiled and processed by the research consortium based on the individual country data submitted by volunteering partners, potentially after already having processed the data themselves. It is hoped that Eora will set a new standard in this respect, so as to increase the credibility of EE-MRIOA.

4.3 Conclusions and future work

Based on the findings presented in this thesis, I offer some conclusions linking back to the original research questions, and provide my take on the future of EE-MRIOA.

Global MRIO databases can offer highly useful insights as to how consumption is linked with production and environmental interventions through global supply chains. In essence, through the continued advances in MRIO table compilation, little by little the 'black box' that is the global economy is made transparent. Once this fundamental framework describing the economy is in place, it makes a powerful tool for environmental analyses through links with secondary data. In this work we have successfully applied information on both the producer and the consumer side to MRIO tables to undertake detailed assessments of specific research questions while at the same time utilizing the full ability of the MRIO system.

The analysis of consumption in the EU showed that through imports of products carrying embodied land use, water use, and carbon emissions, European consumers impose significant pressures on ecosystems in other regions. These embodied pressures imports are considerably larger than their reciprocal flows, in other words there is a net exertion of pressures on the environment from Europe to the rest of the world.

The investigations of the reliability of the MRIO databases confirmed that there are still considerable uncertainties associated with them and any analysis based on them. This is especially the case when attempting to draw conclusions at the detailed level. Nevertheless, the existence of these uncertainties should not be taken as a discouragement to conducting such analyses. Understanding the nature of the mechanisms through which our demand for goods and services affect the natural system is paramount to creating a society that is able to operate within the limits of nature. Our analyses have shown that today's generation of MRIO databases can provide important results useful for international policymaking. Though there are considerable uncertainties associated with specific case studies based on IOA, tendencies can still be identified and useful results obtained as long as the nature and scale of uncertainties is appropriately identified and discussed. Furthermore, it is through the combined effort of individual researchers performing such assessments that the field advances towards higher levels of confidence and detail.

On the methodological level, I hope that the set of global MRIO databases now available, and the current efforts going into assessing their performances comparatively, will ultimately lead to community-wide efforts to establish standards for table compilation. Environmentally extended MRIO databases carry vast potentials for environmental footprint assessments, yet they have so far not been as widely adopted outside the IO community as one might have hoped. I believe several factors contribute to this. First, the sheer dimensions of matrices and datasets can certainly be off-putting, even to researchers. Furthermore, a certain initial investment required both in terms of data management and in terms of understanding the principle of tracing supply chains through the matrix structure, even though the actual mathematical equations involved are in fact few and fairly straightforward. Second, some of the conventions and standards of input-output modelling may seem non-intuitive to anyone without any background in economics. This includes valuation schemes, where purchases are converted to basic prices which exclude direct product taxes and records trade and transport margins as separate payments, final demand components such as gross capital formation, and economic sectors are defined in a way that may be difficult to relate to. Third, the apparent[11] 'black-box'-nature of the Leontief inverse which in a single matrix captures effects accumulated through infinitely long and complex supply chains. Fourth, the lack of standards, ranging from database compilation to apparently trivial sources of confusion such as differing variable name conventions, and finally, a perceived high degree of uncertainty of IOA due in particular to sector aggregation and the use of monetary transactions to describe physical flows.

These are all possible obstacles to the widespread use of MRIO-based footprint accounting, and to consumption-based accounting of environmental impacts in general. I believe a coherent effort in the IO community to alleviate these represents a low-hanging fruit towards the goal of reaching out to policymakers and private citizens with our research.

There is still much ground to be covered before EE-MRIOA can provide accurate, detailed accounts of the overall environmental pressures associated with a certain set of consumed products. However, I would point out the practical challenges mentioned above as especially relevant. In a time where rapid action is required at every level of society to alleviate the formidable pressures exerted by humanity on nature, it is imperative that the IO community is able to reach out to policymakers and the public in general with its insights.

[11] Apparent, because expressing the inverse as a Taylor series expansion can in fact shed light on the contents of this box.

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Appendix A: Paper I

Steen-Olsen, K.; Weinzettel, J.; Cranston, G.; Ercin, A. E.; Hertwich, E. G., Carbon, Land, and Water Footprint Accounts for the European Union: Consumption, Production, and Displacements through International Trade. *Environmental Science & Technology* **2012,** *46*, (20), 10883-10891.

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Carbon, Land, and Water Footprint Accounts for the European Union: Consumption, Production, and Displacements through International Trade

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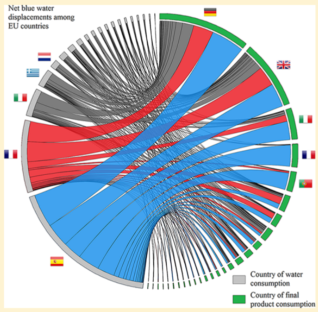
S Supporting Information

ABSTRACT: A nation's consumption of goods and services causes various environmental pressures all over the world due to international trade. We use a multiregional input−output model to assess three kinds of environmental footprints for the member states of the European Union. Footprints are indicators that take the consumer responsibility approach to account for the total direct and indirect effects of a product or consumption activity. We quantify the total environmental pressures (greenhouse gas emissions: carbon footprint; appropriation of biologically productive land and water area: land footprint; and freshwater consumption: water footprint) caused by consumption in the EU. We find that the consumption activities by an average EU citizen in 2004 led to 13.3 tCO_2e of induced greenhouse gas emissions, appropriation of 2.53 gha (hectares of land with global-average biological productivity), and consumption of 179 m^3 of blue water (ground and surface water). By comparison, the global averages were 5.7 tCO_2e, 1.23 gha, and 163 m^3 blue water, respectively. Overall, the EU displaced all three types of environmental pressures to the rest of the world, through imports of products with embodied pressures. Looking at intra-EU displacements only, the UK was the most important displacer overall, while the largest net exporters of embodied environmental pressures were Poland (greenhouse gases), France (land), and Spain (freshwater).



INTRODUCTION

Among the many environmental concerns the global community will be faced with in the 21st century, three major challenges stand out as particularly important. First, considerable efforts are currently directed toward the task of minimizing anthropogenic greenhouse gas (GHG) emissions and the potential harmful climate change effects caused by them. Furthermore, the rapid growth in population and material wealth over the previous century has led to widespread concerns about the state of two resources of vital importance to all life on earth: freshwater and biologically productive land. These three areas of concern are all highly important, and, though essentially different, they are all interconnected and mutually influencing each other, and direct efforts to alleviate one problem might well imply hidden trade-offs with others. As such, it is reasonable to suggest that these (and ideally other) environmental challenges should be assessed simultaneously when politicians and leaders are shaping policies and making investments with a sustainable future in mind.

However, assessing the environmental impacts of GHG emissions and human appropriation of land and water at the macro level is nontrivial, and several approaches exist. Based on the argument that environmental pressures are ultimately driven by consumption of goods and services, several studies and pressure indicators follow the principle of consumer responsibility and attempt to allocate full life-cycle environmental responsibilities of purchased commodities to final consumers.[1] As a way to communicate this idea to a wider audience, the "footprint" term has been adopted for various quantitative measures of environmental stress that adhere to the principle of consumer responsibility. Galli et al.[2] define a "Footprint Family" of three of the most well-recognized footprints available, to be used in assessments of the three environmental issues discussed previously. The Footprint Family includes the carbon footprint (CF), the Ecological Footprint (EF), and the water footprint (WF). The carbon footprint is a measure of total GHG emissions embodied in consumption, measured in tons of CO_2-equivalents.[3] The Ecological Footprint quantifies embodied biological resources in terms of required area of biologically productive land. The measurement unit is the global hectare (gha), which is defined as a hectare of average productivity.[4] Finally, the water footprint measures direct and indirect freshwater requirements in m^3,

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distinguishing among green water (direct rainwater consumption by plants), blue water (ground and surface water), and gray water (a measure of water pollution expressed as the water requirements to dilute emissions).[5]

Footprint indicators are commonly used for assessments at the individual or company level; however for policy making purposes it is useful to construct footprint accounts for countries or regions instead. Such national footprint accounts can provide understanding of the relative importance and nature of a country's impacts in a global perspective, and shed light on the underlying drivers of these impacts. Moreover, the footprint approach allows a quantification of impacts of domestic consumption on nature worldwide. Several attempts to construct national footprint accounts have been made: Global Footprint Network (GFN) regularly constructs national accounts of Ecological Footprints for most countries of the world, addressing both land use and CO_2 emissions;[6] the Water Footprint Network has made similar accounts for water footprints;[7] and Hertwich and Peters[3] presented carbon footprint accounts, which were later updated by Peters et al.[8,9] and are displayed online at ref 10.

Consumption-based accounting adds considerably to analysis complexity compared to territorial accounts, because products may accumulate significant embodied impacts far upstream in complex international supply chains. Multiregional input−output (MRIO) analysis is able to account for complete supply chain effects by taking a top-down approach. While MRIO analysis has been systematically applied in carbon footprint calculations, the Ecological Footprint and the water footprint were developed as bottom-up approaches based on direct land and water use of key sectors. Still, there have been several studies that have used input−output (IO) techniques for measuring EF and WF. Lenzen and Murray[11] and Wiedmann et al.[12] demonstrated the advantages of using IO for EF accounting. McDonald and Patterson[13] used MRIO to analyze regional interdependencies for New Zealand's EF, while Feng et al.[14] used a global MRIO model to calculate WF, following similar MRIO-based analyses for China.[15,16] A material footprint, quantifying the cumulative amount of material natural resources (domestic extraction used all over the world) embodied in the consumption of the EU, has been estimated by ref 17 and denoted as raw material consumption, which belongs to a group of economy-wide material flow indicators.

In this study we analyze environmental footprints of the EU27 countries, and how environmental pressures are displaced among them and to the rest of the world. We present comprehensive accounts of the three footprints, calculated using a common model framework—based on a global MRIO model—to account for supply chain effects (see the Supporting Information (SI) for detailed accounts). A drawback to input−output based EF and WF assessments has been loss of detail at the product level, because land and water use depend heavily on agricultural products, which are usually aggregated into a few bulk categories in input−output tables. The extended MRIO model used in this work partially overcomes the traditional disadvantages of low product detail within the MRIO system by including satellite accounts track the production and international trade of a range of specific primary crop and forestry products.[18] By quantifying three different footprint indicators simultaneously and under a common methodological framework, we are able to assess pressures on three different compartments of the environment in a coherent manner, allowing a fuller picture of the true environmental pressures put on the planet by consumption activities in the EU. This should help to avoid environmental pressure shifting caused by focusing on a sole type of environmental problem.

We include carbon footprints (CF), blue water footprints (WF_b), and land footprints (LF) in our analysis. The LF is equivalent to the Ecological Footprint excluding carbon uptake land, since this is directly related to CO_2 emissions already captured by the CF.[19] We also chose to focus on the blue component of the water footprint, since gray water is not a measure of water consumption in the direct physical sense but of water pollution, and green water is direct rainwater consumption, which as argued by ref 20 is a pressure that would be double counted in combination with the LF; however the interested reader can find results for the complete EF and WF indicators, as well as a discussion of the various footprint indicators, in the SI. Note that according to the "Driver, Pressure, State, Impact, Response" (DPSIR) framework used by the European Environmental Agency,[21] these are all pressure indicators. They present a single quantitative measure, which can be broken down in more detail, but they do not assess the resulting impacts.

A key interest for this analysis has been the displacement of environmental pressure through trade. We speak of a displacement when the environmental pressure occurs in another country than the country of final consumption of the product whose production is the immediate cause of the environmental pressure, following the discussion of land use studies.[22] In other words, if a pressure is displaced from country A to country B, emissions, land use, or water use occurring in country B serve the consumption in country A. Previous research on displacements of environmental pressures through international trade has indicated that Europe generally tends to have net imports of embodied pressures from other regions of the world. For the water footprint, the results of Hoekstra and Mekonnen[7] indicate that a large share of the water footprint of European countries (especially in western Europe) tends to be external compared to developing countries. Peters and Hertwich[23] showed that many EU countries are net importers of embodied CO_2 emissions, and that a significant share of this displacement was to fellow member states. Weinzettel et al.[19] found that Europe overall displaces land use to other regions of the world, especially Latin America and Asia.

In the following section, the model is described. This is followed by a section presenting the results of our analysis, while the final section provides a discussion of our main findings.

METHODS AND DATA

The analysis was carried out using a global MRIO model based on the GTAP 7 database,[24] following the method described by Peters et al.[25] The model year is 2004, as this is the reference year for the GTAP 7 database. The model tracks economic transactions among actors in the global economy, aggregated into 57 economic sectors in 113 regions, allowing the establishment of a model of sectoral interdependencies among regions through the application of the Leontief inverse.

The MRIO framework allows the tracking of environmental impacts through complex international supply chains. However, MRIO tables describe only aggregated groups of products and sectors. To provide a higher level of detail including specific crops, we created a parallel system to explicitly track the production and trade of primary products of agriculture and forestry, since this production accounts for the majority of land and water use globally.[7,26] Compared to a full disaggregation of these sectors in the MRIO model, which would require extensive new data and labor for a long list of products, our method is a compromise.

We followed the approach suggested by Ewing et al.,[27] and created an extension matrix $P^{use,x}$ with rows representing sales of each primary product (in physical units) from each region, distributed to the regions and sectors which purchase them in their primary form, in the columns. There is an additional matrix $P^{use,y}$ for direct purchases by the final demand sector. The columns thus follow the dimensions of the MRIO system, while the number of products and countries on the rows can be as detailed as desired. A more comprehensive presentation of the model can be found in the SI.

The allocation of agricultural products to intermediate and final consumers in the extension matrix allows the utilization of the extensive amounts of data available for these products.[27] We used data on production and international trade from the Food and Agriculture Organization of the United Nations (FAO) to allocate products to consuming countries.[28] We also used information on consumption by specific sectors, including the use of agricultural products such as seed and livestock feed. The remainder was allocated based on the sales structure of the corresponding sector in the MRIO model.

Our environmental extension matrices thus represent the use of primary products, produced in individual countries, by specific industries and final consumers. Using the standard input−output methodology we were then able to estimate total national requirements of specific primary products produced in specific countries. This bill of requirements was then converted to associated footprints by applying crop- and country-specific land and water use intensities (see the SI for more on these). Finally, footprints not directly related to primary products were calculated from a second set of extension matrices following traditional practice for environmentally extended MRIO, and the two contributions were added to arrive at a total. For more details and a mathematical description of the method, see the SI and ref 18.

RESULTS

The total carbon footprint of the EU in 2004 was 6.5 billion tons CO_2-equivalents ($GtCO_2e$), representing 18% of the world total (Table 1). As the EU constituted 7.6% of the global population, its average CF of 13.3 tCO_2e/p was well over twice the global average. Similarly, the EU's land footprint of 2.53 gha/p was just over twice that of the world overall. In terms of blue water the EU footprint of 179 m^3/p was only 10% above the global average.

Table 1. Total and per Capita Footprints for the EU and the Rest of the World (RoW) in 2004, Displacements within the EU, and between the EU as a Whole and the Rest of the World

	CF		LF		WF_b	
	$GtCO_2e$	tCO_2e/p	Ggha	gha/p	Gm^3	m^3/p
EU	6.5	13.3	1.23	2.53	87	179
RoW	30.0	5.1	6.58	1.11	958	162
world	36.5	5.7	7.82	1.22	1045	163
displacements						
among EU countries	0.79		0.20		9	
from EU to RoW	2.01		0.38		37	
from RoW to EU	0.50		0.10		4	

The trade analysis quantifies environmental pressures occurring in other countries to serve domestic final consumption of goods and services. When looking at the exchanges between the EU and the rest of the world, the analysis showed that the EU displaced far more pressures to the rest of the world (RoW) than the RoW displaced to the EU. EU displacements to RoW were about a factor of 4 higher than the corresponding displacements to the EU for CF and LF, and a factor of 9 higher for WF_b.

In the following paragraphs we examine the footprint results for the EU member countries, and how environmental pressures are shifted internally in the EU through trade.

Footprints for the EU27 Member States. *Carbon Footprints.* All EU countries except for Romania had CF per capita above the global average. The very high footprint of 41.6 tCO_2e/p for Luxembourg should be taken cautiously due to Luxembourg's unique economic structure with parts of the work force commuting from neighboring countries, although the high affluence level of Luxembourg would also predict high CF levels.[3] Excluding Luxembourg, the disparity among member states was still rather large, ranging from 19.8 tCO_2e/p for Belgium to only 5.6 tCO_2e/p for Romania.

The degree to which individual countries imported and exported embodied GHG emissions varied considerably, as shown in Figure 1. For the EU overall, 57% of the emissions constituting its total CF occurred in the countries of final consumption, 12% were displaced internally, and 31% occurred in countries outside the EU. Though the EU overall was a net importer of embodied emissions from the rest of the world, five individual member states (the Czech Republic, Poland, Estonia, Bulgaria, and Romania) were net exporters of embodied emissions overall.

Land Footprints. All EU countries were 35% or more above the global average land footprint of 1.23 gha/p, except for Malta, which was close to the global average (Figure 1). In the upper end of the footprint ranking we again find Luxembourg standing out, this time joined by the Nordic countries Finland, Sweden, and Denmark. Of the four, Finland was especially high at 6.8 gha/p—2.3 times above EU and 4.8 times above global averages. From the domestic pressure perspective, land use per capita was especially high in Finland, Sweden, Estonia, and Latvia. This was mostly due to large forestry and (particularly in Estonia's case) fishing industries, combined with the lowest population densities within the EU27 region which serve to exalt the level of land use per capita.

The geographic distribution of the land use that formed the total LF followed a pattern similar to that for CF. Fifty-three percent of the EU's LF was associated with domestic land use, 16% was associated with land use in other EU countries, and 31% was land used outside the EU. Eleven EU countries were net exporters of embodied land use (Estonia, Finland, Latvia, and Sweden had the highest net exports per capita), but since the more populous countries were generally importers (Poland was an exception), the overall result for the EU was a net displacement of land use to other countries. Malta and Cyprus were especially dependent on displacing land use to other countries due to their dry climate.

Blue Water Footprints. There were very large differences among individual EU countries due to different biophysical conditions and consumption patterns, with footprints ranging

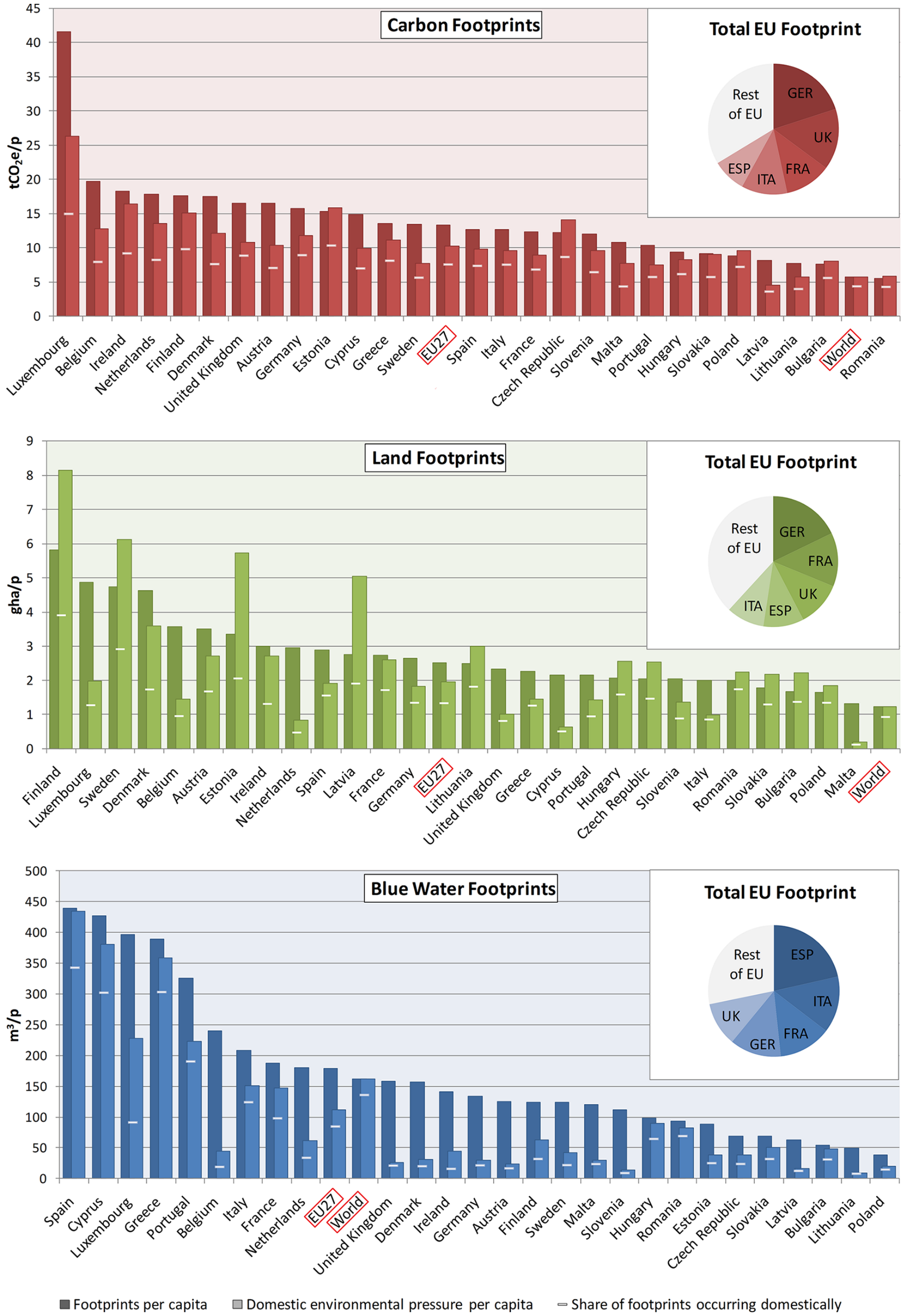

Figure 1. Carbon, land, and blue water footprints (darker columns) and the environmental pressures occurring within the borders of each country (lighter columns) per capita for the individual EU27 countries, as well as EU27 and global averages. The white markers show the part of the footprint which occurs as pressure on the domestic territory, or in other words the part of the environmental pressure on the domestic territory which was induced by domestic final demand. The pie charts show the top five contributing countries to the EU's total footprints.

from 438 m^3/p for Spain to as little as 39 m^3/p—less than a quarter of the global average—for Poland. In fact, most EU countries had WF$_b$ levels per person below the global average, but the very high footprints of Mediterranean countries, where

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Table 2. Net Displacements of Environmental Pressures by Each EU Country to the Rest of the EU[a]

	CF		LF		WF_b	
	$MtCO_2e$	tCO_2e/p	Mgha	gha/p	Mm^3	m^3/p
Austria	14.8	**1.8**	1.6	0.2	200	**24.5**
Belgium	2.1	0.2	6.9	**0.7**	238	**22.9**
Bulgaria	−7.3	−0.9	−2.7	−0.3	−28	−3.6
Cyprus	0.6	0.7	0.4	0.5	−16	−19.5
Czech Republic	−23.7	−2.3	−5.4	−0.5	23	2.2
Denmark	2.5	0.5	−0.8	−0.1	110	20.3
Estonia	−2.9	−2.2	−2.5	−1.8	3	2.0
Finland	−7.4	−1.4	−10.4	−2.0	−11	−2.1
France	**23.6**	0.4	−16.4	−0.3	−773	−12.8
Germany	9.9	0.1	10.0	0.1	**1390**	16.8
Greece	1.7	0.2	3.0	0.3	−245	−22.1
Hungary	−3.1	−0.3	−4.6	−0.5	−92	−9.1
Ireland	−7.6	−1.9	−1.8	−0.4	18	4.5
Italy	16.4	0.3	**18.7**	0.3	50	0.9
Latvia	2.0	0.8	−4.6	−2.0	11	4.8
Lithuania	0.5	0.1	−1.5	−0.4	20	5.9
Luxembourg	0.6	1.2	0.5	**1.2**	−12	−26.8
Malta	0.1	0.2	0.1	0.3	8	20.3
Netherlands	−20.9	−1.3	9.7	0.6	76	4.7
Poland	−43.3	−1.1	−9.8	−0.3	74	1.9
Portugal	4.4	0.4	1.4	0.1	235	22.5
Romania	−11.0	−0.5	−5.5	−0.3	−110	−5.1
Slovakia	−4.0	−0.7	−2.0	−0.4	−31	−5.7
Slovenia	0.6	0.3	0.5	0.2	33	16.8
Spain	−12.4	−0.3	6.5	0.2	−2346	−55.0
Sweden	15.9	**1.8**	−9.9	−1.1	97	10.7
United Kingdom	**48.2**	0.8	**18.7**	0.3	**1078**	18.1

[a]The columns of country totals all sum to zero, except for rounding. The two highest and lowest values in each column are highlighted by bold text.

Table 3. Top Five Net Pressure Displacements (ND) between EU Member States[a]

CF ($MtCO_2e$)			LF (Mgha)			WF_b (Mm^3)		
displacement from/to			displacement from/to			displacement from/to		
imported products	ND	GD	imported products	ND	GD	imported products	ND	GD
Germany/Poland	12.5	17.8	Italy/France	6.25	7.20	Germany/Spain	542	573
chem., rubber, plast. prd.		10%	wheat		22%	vegetables, fruits, nuts		58%
machinery and equipmt. nec.[b]		10%	cattle, sheep, goats, horses		20%	food products nec.		9%
motor vehicles and parts		9%	cereal grains nec.		15%	beverages and tobacco prod.		8%
Germany/Czech Republic	9.7	13.6	Germany/Poland	3.89	4.68	UK/Spain	387	405
electricity		23%	wood products		52%	vegetables, fruits, nuts		46%
machinery and equipmt. nec.		12%	food products nec.		11%	food products nec.		10%
chem., rubber, plast. prd.		11%	motor veh. and parts		4%	beverages and tobacco prod.		10%
France/Germany	8.6	23.8	UK/Sweden	3.39	3.50	France/Spain	348	637
chem., rubber, plast. prd.		18%	wood products		56%	vegetables, fruits, nuts		40%
motor vehicles and parts		12%	paper products, publishing		23%	food products nec.		15%
machinery and equipmt. nec.		11%	business services nec.		3%	animal products nec.		6%
UK/Germany	8.5	20.3	UK/France	3.09	4.26	Germany/France	327	396
motor vehicles and parts		20%	cereal grains nec.		17%	cereal grains nec.		15%
chem., rubber, plast. prd.		18%	food products nec.		15%	motor veh. and parts		11%
machinery and equipmt. nec.		10%	beverages and tobacco prod.		12%	chem., rubber, plast. prod.		10%
UK/Spain	7.6	13.6	Netherlands/Germany	3.09	4.18	UK/France	325	355
air transport		23%	fishing		32%	cereal grains nec.		24%
transport nec.		14%	cereal grains nec.		14%	motor vehicles and parts		14%
motor vehicles and parts		11%	food products nec.		10%	chem., rubber, plast. prod.		10%

[a]The gross displacement (GD) value corresponding to each displacement is also shown, as well as which products imported by the displacing country that contribute most to the total GD. [b]Not elsewhere classified.

agricultural systems are more dependent on irrigation, pulled the footprint per person for the EU27 overall above the global average. Spain alone, with only 9% of the EU population, contributed 21% of the EU's total WF_b. Finally, an interesting feature of Figure 1 is that the ten former East Bloc countries in the EU were also the ten countries with the lowest WF_b per capita.

A higher degree of pressure displacement was found for WF_b than for the other footprints: 42% of the EU's blue water footprint was water used outside the EU, while 47% was domestic water use and 10% was water use displaced between EU countries. All EU countries were net importers of embodied blue water, even the countries with very high levels of domestic blue water use turned out to have footprints that were even higher. Indeed, as seen in Table 1 the flows of embodied blue water into Europe were large compared to the outflow and also compared to the flows of embodied carbon and land use. Although the average EU WF_b per capita is about the same as the global average, this is at the same time the footprint where the EU consumption causes relatively the most displacements to the rest of the world. Both these observations are explained by the low levels of domestic blue water consumption per capita in European countries; see Tables S-1 and S-2 of the SI.

Footprints Shifted Internally among EU Countries. Though the EU overall displaced all three pressure types to the rest of the world, the detailed trade analysis showed that there were also significant shifts of pressures internally in the EU. Table 2 shows the individual member states' net imports of embodied pressures from other EU countries. In the following paragraphs, these shifts through trade are explored and results of the contribution analysis are presented to identify which products led to the largest pressure displacements. Table 3 lists the largest net pressure displacements between EU countries, and the most important traded products related to these displacements, while Table 4 shows the largest product-specific displacements between EU countries. Note that whereas Table 4 compares gross flows, Tables 2 and 3 compare net flows of embodied pressures between countries, i.e. the ranking is performed on the difference between the reciprocal displacements between each pair of countries.

Table 4. Top Five Product-Specific Gross Pressure Displacements (GD) between EU Member States, i.e. the Products Imported by One EU Country That Cause the Largest Environmental Pressure in a Fellow EU Country, and the Absolute Values of These Displacements

CF ($MtCO_2e$)		LF (Mgha)		WF_b (Mm^3)	
displacement from/to		displacement from/to		displacement from/to	
imported products	GD	imported products	GD	imported products	GD
Italy/France	4.7	Germany/Poland	2.4	Germany/Spain	331
cattle, sheep, goats, horses		wood products		vegetables, fruits, nuts	
France/Germany	4.4	UK/Sweden	2.0	France/Spain	256
chem., rubber, plastic prod.		wood products		vegetables, fruits, nuts	
UK/Germany	4.0	Italy/France	1.6	UK/Spain	185
motor vehicles and parts		wood products		vegetables, fruits, nuts	
UK/Germany	3.6	Italy/France	1.4	Italy/Spain	101
chem., rubber, plastic prod.		cattle, sheep, goats, horses		vegetables, fruits, nuts	
Germany/Netherlands	3.4	UK/Poland	1.4	Spain/France	97
petroleum, coal products		wood products		cereal grains nec.[a]	

[a]Not elsewhere classified.

Carbon Footprints. Comparing countries overall, Table 2 shows that Poland was by far the largest net exporter of embodied GHG emissions, while the United Kingdom held an even clearer position as the largest net importer. Per capita, Austria and Sweden had the highest net imports. The net result for each country consists of a sum of net exchanges with the remaining member states. Thus, for instance, France's large net import of embodied GHG emissions actually included a significant net export (2.6 $MtCO_2e$) to Italy (see SI Table S-3 for details).

The single largest net export of embodied GHG emissions between two EU countries was from Poland to Germany (12.5 $MtCO_2e$, see Table 3). Overall, Germany was an important trader of embodied CF, as apparent in Table 3. The second largest exchange was emissions embodied in German imports from the Czech Republic, a large part of which was embodied in electricity imports. The emissions embodied in the gross electricity exports from the Czech Republic to Germany amounted to 3.1 $MtCO_2e$. This was still less than the emissions embodied in the largest single product flow, "Bovine cattle, sheep and goats, horses" exports from France to Italy (4.7 $MtCO_2e$), as reported in Table 4. On the list of largest product-specific flows of embodied emissions we also find motor vehicle exports from Germany to the UK, suggesting the importance of the German automotive sector on the overall carbon footprints in European countries.

Land Footprints. In total, the main net importers of embodied LF were the United Kingdom and Italy. France had the largest embodied land use exports (absorbed land use in the terminology of ref 29). Per capita, the main net exporters of embodied LF were the northeastern cluster of Latvia, Estonia, Finland, and Sweden. The biggest net importers of embodied LF per capita were the small and densely populated Benelux (Belgium, Netherlands, Luxembourg) countries, as evident from Table 2.

All EU countries were net exporters of embodied land use to some EU countries, while having net imports from others. For example, the UK's large net import from the rest of the EU also contained a significant net export of embodied land use to Spain; 0.4 Mgha (see SI Table S-4). Malta stands out with net embodied land use imports from all EU countries except Greece.

By far the largest net shift of embodied land use between EU countries, shown in Table 3, was Italy's displacements to France. Important contributing Italian imports were wheat, barley, and other grains, but also livestock. Wood products were another major carrier of embodied land use in the EU trade market. Regarding the largest product-specific gross land use displacements in the EU shown in Table 4, German imports of wood products led to land use in Poland of 2.4 Mgha, while Swedish land use due to British imports of wood products constituted the second most important flow (2.0 Mgha).

Blue Water Footprints. The most striking net displacements of pressures within the EU were related to blue water consumption. Blue water embodied in Spanish exports dominated completely, with only French exports coming remotely close. The displacement of blue water use to Spain mainly came from consumption in Germany and the United Kingdom; see Table 3. Though the ranking of countries is shifted in the per capita domain in Table 2, Spain still massively dominates with net per capita exports of embodied blue water more than twice those of Luxembourg at second place. The situation is largely explained by the fact that Spain is an important producer of several rather water intensive crops, while at the same time being highly dependent on irrigation due to a semiarid climate.

In the analysis of net blue water exchanges we find that Lithuania had net imports of embodied blue water from all other EU countries, the only such case in our results. Another interesting result is that Portugal's net embodied blue water imports from Spain (277 Mm^3) were considerably larger than its total net imports (235 Mm^3). In fact, despite being a net importer overall, Portugal was a net exporter of embodied blue water to most other EU countries.

Imports of embodied blue water from Spain and France to other EU countries dominate the lists of top country-specific and product-specific displacements of embodied blue water in Tables 3 and 4. In particular, the net German displacement to Spain stands out at 542 Mm^3. The corresponding gross displacement was only slightly larger, and the analysis showed that about 21% of this was related to German imports of oranges, tangerines, and similar fruits. These fruits were also important components in the blue water consumption displaced from the United Kingdom and France to Spain. Another large displacement featured Spain as the importing part: Spanish imports of cereal grains led to 97 Mm^3 of blue water consumption in France, 91 Mm^3 of which were associated with imports of maize (see Table 4).

DISCUSSION

All consumption draws on the finite resources of the planet; however, this fact has become less visible to consumers in industrialized regions such as Europe, as products have become increasingly processed, and consumers are located further from production sites. An inhabitant in a modern city indirectly requires large amounts of land, fresh water, and greenhouse gas emissions to sustain her consumption activities, but she may never see any of these effects first-hand. Through international trade, pressures on the natural system are often located far away from end consumers, and people in urban areas can easily have higher total embodied resource consumption than people living in areas where those resources are abundant.

Input−output analysis has provided important insights into the environmental impacts of consumption,[1,30] which served as the basis for policy such as the EU's Roadmap to a Resource Efficient Europe.[31] The findings of the consumption-based analysis for Europe emphasize the importance of shelter, mobility, and nutrition. Analysis of trade has often focused on aggregate results, addressing questions of responsibility with regard to, primarily, climate change.[9,32] The aggregate results, however, come about through a multitude of interactions in a complex web of global supply chains. A better understanding of cross-country relationships and of the importance of product flows can offer important insights to policy and provide some understanding of economic interests.[33] Analyses can be organized according to product groups, regions,[34] countries,[35] or bilateral trade relationships.[36]

Our analysis addresses the relationships within the EU member states in more detail. On the global level, the accounting for emissions embodied in trade increases the already high carbon footprints of Europe, Japan, South Korea, and the United States.[3,9] The correct accounting for land use displacement increases the land footprint of the same economies which already extract more domestic biomass than less affluent ones.[19] Intra-European trade, however, seems to run in the other direction: heavy polluters or resource users having high *net* emissions embodied in export. Poland and the Czech Republic use a lot of coal for both electricity production and heavy industry, and some of this coal is burnt for producing exports, similar to the role China has in the world economy. Poland and the Nordic countries have a large forestry sector, so that they become net exporters of land due to the export of these products while also using a lot of these products at home. Spain and France have large and productive agricultural sectors, so they export a lot of agricultural products to the UK, Germany, and other European countries. While food production in temperate countries relies mostly on rain-fed agriculture, Spain relies heavily on irrigation, something that can be seen both within its own consumption but also in its net-trade position within the EU.

How can it be that within a global context, we find that high footprint countries have a position as net importer (or displacer), while within Europe, they tend to have a position of a net exporter? To understand the total net position, we need to understand that there is a trade-off between the scale of consumption and the efficiency of production. Affluent countries tend to have a high efficiency, i.e., a lot of value created per unit of resource use or pollution. They also have a high level of consumption. As their exports are lower resource intensity than their imports, they become net importers. Within Europe, however, the differences in consumption are much smaller and less important. Resource intensities differ because of structural and natural differences, not so much due to differences in economic efficiency. Resource-rich countries specialize in resource extraction and processing, whether it is heavy industry (Poland and Czech Republic for coal) or forestry (Finland, Sweden, Poland, Estonia for land) and agriculture (Spain, France for land). The situation for blue water is different. While any biomass production requires water, irrigation is most heavily used in regions where rainfall during the growing season is insufficient to sustain intensive agriculture. In these regions, agriculture easily dominates other water uses.

In its Renewed Sustainable Development Strategy for the EU, the European Council calls for a set of indicators for sustainable development at the "appropriate level of detail" to ensure sufficient coverage of the complex environmental challenges facing society.[37,38] The heterogeneity displayed by our results supports the notion that sustainability analyses need to simultaneously assess more than one dimension of the environmental sustainability challenge. Even within a single economic region like the EU, footprint profiles vary considerably between countries. However, more research is still needed to improve models and increase our understanding of how environmental pressures are displaced globally. First, macro level input−output models like ours generally carry significant uncertainties, and second, defining indicators to represent complex environmental issues inevitably involves some subjective choices. This is especially true for land and water use accounting. For instance, although the EF is widely used, another well recognized approach to the same environmental challenge instead attempts to measure the human appropriation of net primary productivity (HANPP)—see refs 39 and 40 for discussions of these two approaches. The issue of water use accounting is perhaps even more debated, referring to the water types distinguished in the introduction. Furthermore, the environmental impacts of land and water use very much depend on where and when the water and land use take place. The reader is referred to the SI for a further discussion of these issues.

Finally, although the focus of this study has been the displacements of environmental pressures through trade, it should be pointed out that such displacements are not problematic in themselves, but may in fact carry environmental benefits.[41] Different regions have different comparative advantages in terms of production technologies and natural endowments, and international trade can serve to optimize the global society's overall use of natural resources.[42,43] The potential negative impacts depend on local conditions; hence European displacements of forest land use to Finland, where forest stocks are large and increasing,[44] should represent less of a concern than displacements of blue water consumption to Spain, where water resources are under pressure.[45]

ASSOCIATED CONTENT

Supporting Information

Further explanation of the extended MRIO model and the footprint indicators; discussion of validity and limitations; complete lists of regions and sectors included in the MRIO model, and countries and primary products included in the physical system; extensive footprint results including detailed displacements between all 27 EU countries; results for the complete EF and WF indicators. This information is available free of charge via the Internet at http://pubs.acs.org. A version of the model can be accessed at www.eureapa.net.

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Notes

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Supporting Information:

Carbon, Land, and Water Footprint Accounts for the European Union: Consumption, Production, and Displacements through International Trade.

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Supplementary Materials and Method

Environmentally Extended Multiregional Input-Output Models

In order to keep things simple, the methodological explanation in the following paragraphs is based on a single-region input-output system; however the principle is the same in multiregional models. Input-output (IO) models are made up of matrices describing transactions between actors within an economy. Rows represent product groups while columns represent the industry, government, or household sectors which consume them. Transactions are generally accounted for in monetary terms; however some IO tables based on mass or energy transactions have been constructed.

An environmentally extended IO model constitutes a complete inventory of all economic transactions and selected environmental interventions of individual sectors within a specified region during a period of time, most commonly for a country on an annual basis. An environmentally extended IO model is generally made up of four matrices: the intermediate transactions matrix (Z), the final demand matrix (Y), the value added matrix (W), and the environmental extensions matrix (F), which can also include direct environmental interventions by households (F^{hh}) if applicable. In symmetric IO tables, an economy is modeled as consisting of n industries (we will assume the IO table is industry-by-industry, but they can also be product-by-product), d categories of final consumers, w types of production factors, and f types of environmental interventions. Z (n-by-n) is a square matrix of intermediate transactions where rows represent sales from each of the n industries included in the system, while columns represent each industry's purchases, so that an element z_{ij} gives industry j's total purchases from industry i. Each column of Y (n-by-d) contains the purchases made by a specific group of final consumers, such as households and government, from each industry. Y also contains columns for tracking changes in stocks, changes in inventories, capital investments, and exports. Entries in Y describe purchases by consumers which do not produce output that re-enters the economy.

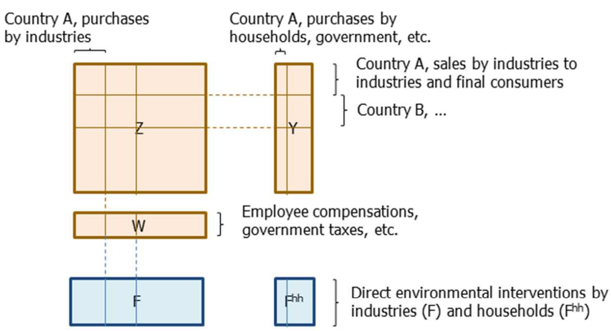

Figure S-1. Generic multiregional input-output tables with environmental extensions.

The rows of W (w-by-n) represent labor payments, taxes, subsidies, and operating surplus and the columns represent industries or product groups. For environmental analysis, the W matrix is rarely used. The F (f-by-n) matrix represents environmental interventions of each economic sector. It has one row for each included kind of intervention, such as CO_2 emissions, energy use and so on, and one column for each industry, such that its columns correspond to the columns of Z. In addition, there might be an additional matrix F^{hh}, representing direct environmental interventions by final consumers, e.g. CO_2 emissions from gas stoves in households.

Using these matrices a model can be constructed which allows the calculation of the total economic transactions and environmental interventions occurring along all supply chains associated with the production of a basket of products and services:

The total output (x) from all the industries in the economy over the defined time period can be calculated using Z and y, a column vector of total final demand, equal to the row sum of Y:

$$x = Zi + y$$

where i is a column vector of ones (for the summation of rows across columns of the matrix). Next we define a direct requirements matrix (A):

$$A = Z\hat{x}^{-1}$$

Each element (a_{ij}) of A represents the purchases of product/service (*i*) required by industry/service sector (*j*) to produce one unit of its output. Substituting into the previous eq. (1) we obtain the following:

$$x = Ax + y$$

Solving for x yields

$$x = (I - A)^{-1}y$$

where I is the identity matrix. Note that this equation holds not only for the original x and y but through the Leontief inverse $L = (I - A)^{-1}$ the total supply chain output (x*) associated with an arbitrary demand vector (y*) can be calculated.

A normalized environmental extension matrix (F) can be defined that gives environmental interventions by sector per unit output, by dividing total annual emissions etc. by total production, to arrive at a matrix with e.g. kg CO_2 emitted per dollars' worth of aluminum produced by the aluminum industry:

$$F = F_r\hat{x}^{-1}$$

The F matrix can be used to calculate total environmental interventions associated with an arbitrary final demand of products (y*):

$$E^* = F(I - A)^{-1}y^*$$

Where E* is a vector of total environmental interventions resulting from the whole production phase of the arbitrary demand vector y*.

In order to accurately represent trade flows and the economic structure involved in the production of imported products, an IO model combining several national-level IO tables through the use of international trade data is required. Such an international multiregional input-output (MRIO) table depicts interdependencies between domestic and foreign sectors with different production technology, resource use and pollution intensities and is regarded as a methodologically sound approach for the enumeration of environmental impacts from consumption. Using MRIO instead of a single region IO table does not change anything of the general concept of IOA, with the exception of international trade. Therefore, all exports are not part of final demand in MRIO model, but are allocated to the users in other regions. Exports to industries abroad are included in Z, while only when exports are used for final consumption in the receiving economy they are part of the final demand matrix. In an MRIO with *m* regions, the dimensions of the matrices would increase with the factor *m* where applicable (see Fig. 1).

Model Construction

The environmentally extended multiregional input-output (MRIO) system constructed and used for the present analysis is based on the Global Trade Analysis Project Database Version 7 (GTAP 7) [1]. The GTAP 7 database models the total global economy in 2004 as 113 geographical regions, composed of 94 individual countries and 19 aggregate regions. GTAP is based on datasets provided by a worldwide network of national dataset providers as well as the UN Commodities Trade Database. The process of constructing a multiregional input-output database from the GTAP 7 database is described in [2]. The construction of the extensions that was used for the present analysis is thoroughly documented in [3]. The following overview is based on similar descriptions in [3] and [4].

Both water and land use is largely determined by agricultural and forestry production. Primary crop and forestry products refer to non-processed products that are directly harvested. In this article, the term *primary product* always refers to such (biological) products. Ecological and Water footprint accounts currently available have high product level detail on such products. This kind of detail is normally not available in IO systems, a shortcoming that could lead to serious aggregation errors. Therefore, the environmental extension matrix should be based on these primary products distinguishing their country of origin with the same level of detail as is used for standard footprint accounting and which is different from the MRIO system. Therefore we distinguish two systems: the monetary (MRIO) system and the physical (footprint – environmental extension) system. These systems differ regarding detail in primary crop and forestry products classification and country aggregation. Furthermore, the two systems track trade flows in different units, monetary (Euros) and physical (tons or m^3), respectively. The monetary system follows MRIO classification, while the physical system follows the classification required for footprint calculations, in this particular case the FAOSTAT classification system.

The information about the origin and type of primary product has to be kept in order to calculate the footprints in a proper manner. Two types of information regarding the use of specific primary products by MRIO sectors and regions can be available. The first one comprises production and international trade of primary products; the second comprises information on direct use of some primary products by specific MRIO sectors, for example the use of products as seed by their producing industry. It is usually not possible to distinguish the country of origin for each particular primary product which is consumed by a specific sector within the consuming country, but the overall composition of supplying countries for each primary product is well distinguished. Since detailed information on the use of all primary products by all individual sectors of the MRIO model is generally not available, the allocation of the rest of primary products to individual sectors within MRIO regions can be done using the appropriate monetary flows within the MRIO model (the monetary flow of the respective product group of the respective region). This is generally done by the Leontief inverse in the standard approach as well, but using the same patterns for all products of one product group. The advantage of this approach is the distinction of the consuming region for individual primary products and utilizing specific data on the use of some primary products such as feed and seed by MRIO sectors. For example, if more primary products are aggregated in one MRIO product group and only one primary product is traded internationally, this detail will be kept by this approach. The distinction in the use of the rest of primary products within the same MRIO product group for intermediate consumption and final demand will not be addressed since its distribution within the same region is based on the monetary flows only.

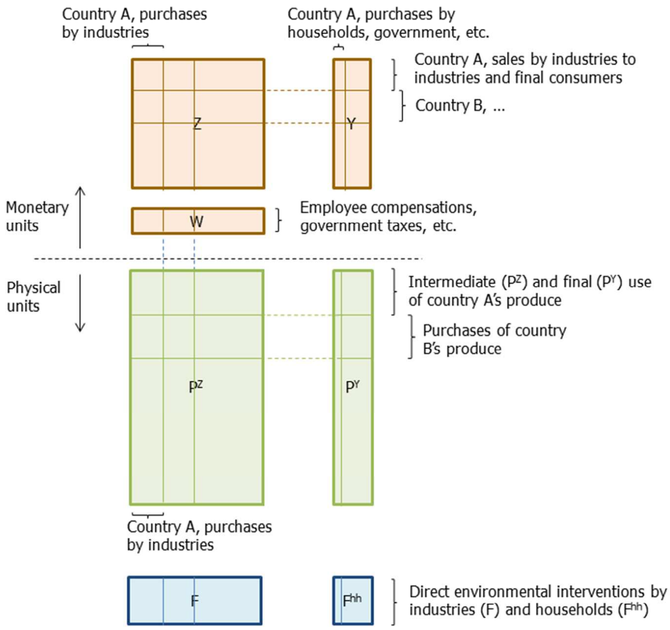

Figure S-2. Sketch of the constructed EE-MRIO model including the additional physical system (green)

Constructed according to the preceding description, the P^Z and P^Y matrices track 179 primary products from producing country to the sector and region that first uses them. Hence they both have 179 (primary products) times 238 (countries) rows. P^Z has columns corresponding to the columns of Z, while P^Y has columns corresponding to the columns of Y, see Fig. S-2. The units are metric tons for agricultural products, and m^3 for forestry products. Upon construction, these matrices were treated as regular environmental extensions, where the environmental intervention is the amounts used of individual primary products. Following a column down through the matrices Z, P^Z, and F in Fig. S-2, one can infer the purchases made by that particular sector from all sectors (Z), its total use of primary products of all 179 types and produced in all 238 countries (P^Z), and its total CO_2 emissions etc. (F). Using a set of coefficients that convert each primary product produced in each country to a corresponding set of land and water footprints, one copy of P^Z and P^Y can be created from each footprint type. Due to the extensive matrix sizes and the computational capabilities required, these matrices were constructed and subsequently aggregated across primary products. Note that this does not change any results since the matrices, once constructed, are static.

When the footprints are implemented this way into the F matrix, it is necessary to account separately for direct footprints (E_{DIR}) of primary products included in the final demand (y) and all indirect footprints (E_{IND}) of all products included in the final demand (y) using the following equations:

$$E_{DIR} = F^{hh} y$$

$E_{IND} = F(I - A)^{-1}y$

Land footprint coefficients

The land footprint describes the equivalent land and ocean area utilized by humans to derive usable biomass products, i.e. products of economic interest to people [5, 6]. This land and ocean area is weighted according to its current productivity by converting it into an equivalent area of global average productivity, measured in units of global hectares [7]. The land footprint distinguishes five different land types, namely: cropland, forest land, pasture, built up land and marine area, each with a specific, world-average productivity.

The direct land footprint (LF_D) is calculated as:

$LF_D = A_{LN} \cdot YF_{LN} \cdot EQF_L$

Where A_{LN} is the area of land type L used in country N, YF_{LN} is a country and land type specific yield factor, which converts the area A_{LN} in country N into world average area of the respective land type and EQF_L is a land type specific equivalence factor, which converts the former result into an area with a global average productivity. For each individual primary biomass product, A_{LN} is calculated as P_{LN}/Y_{LN}, where P_{LN} is the physical amount of product harvested and Y_{LN} is the country specific yield for the land type L producing that product.

The yield factor is derived as:

$$YF_{LN} = \frac{\sum_{i \in U} A_{LWi}}{\sum_{i \in U} A_{LNi}}$$

Where i is an index over all primary biomass products (set U) harvested from the land type L in country N, A_{LWi} is the area associated to each primary biomass product using world average yields and A_{LNi} is the area associated to each primary biomass product i in the studied country.

Country specific yields, production volumes and international trade data are retrieved from FAOSTAT database [8], yield factors and equivalence factors are retrieved from database of Global Footprint Network [7].

Water footprint coefficients

Water Footprint estimations (green, blue and grey) of primary crops are taken from the study by Mekonnen and Hoekstra [9]. The green, blue and grey Water Footprints of primary crops are estimated in a spatially-explicit way. Calculations are done by taking a high-resolution approach, estimating the Water Footprint of the crops at a 5 by 5 arc minute grid.

The green and blue Water Footprint of a crop (WFcrop, m3/ton) is calculated as the green or blue component in crop water use (CWUi, m3/ha) divided by the crop yield (Y, ton/ha) where i indicates the component of Water Footprint, green and blue.

$$WF_{proc,i} = \frac{CWU_i}{Y}$$

The green and blue components of crop water use (CWU, m3/ha) are calculated by accumulation of daily evapotranspiration (ET, mm/day) over the complete growing period:

$$CWU_i = 10 \times \sum_{d=1}^{lgp} ET_{i,d}$$

Where ETi,d represents evapotranspiration by type, i, either green or blue and by day, d. The factor 10 is used to convert mm into m3/ha. The summation is done over the period from the day of planting, d=1, for the entire length of growing period (lgp) until harvest.

The grey water footprint of a primary crop (WFcrop,grey, m3/ton) is calculated as the chemical application rate per hectare (AR, kg/ha) times the leaching rate (α) divided by the maximum acceptable minus the natural concentration for the pollutant considered (cmax – cnat, kg/m3) and the crop yield (Y, ton/ha).

$$WF_{proc,grey} = \frac{(\alpha \times AR)/(c_{max} - c_{min})}{Y}$$

Grey water footprints are measured based on the (human-induced) loads that enter into freshwater bodies, not on the basis of the loads that can finally be measured in the river or groundwater flow at some downstream point. Since water quality evolves over time and in the course of the water flow as a result of natural processes, the load of a certain chemical at a downstream point can be distinctly different from the sum of the loads that once entered the stream (upstream). The choice to measure the grey water footprint at the point where pollutants enter the ground- or surface water system has the advantage that it is relatively simple – because one does not need to model the processes that change water quality along the river – and safe – because water quality may improve along the flow of a river by decay processes, but it is unclear why one should take improved water quality downstream as an indicator instead of measuring the immediate impact of a load at the point where it enters the system. While the grey water footprint indicator thus does not account for natural processes that may improve water quality along the water flow, it does also not account for processes that consider the combined effect of pollutants, which may sometimes be greater than what one may expect on the basis of the concentrations of chemicals when considered separately. In the end, the grey water footprint strongly depends on ambient water quality standards (maximum acceptable concentrations), which is reasonable given the fact that such standards are set based on the best available knowledge about the possible harmful effects of chemicals including their possible interaction with other chemicals.

Uncertainties, limitations, and subjectivity

Following the "Driver, Pressure, State, Impact, Response" (DPSIR) framework, the assessment of sustainability can be subdivided into more manageable tasks of analyzing drivers (e.g., population increase), pressure (e.g., GHG emissions), environmental state (e.g., atmospheric GHG concentration, mean global temperature), impacts (e.g., more frequent severe weather events), and responses (e.g., emission taxes, energy efficiency programs). In this study we quantify pressures, thus avoiding the difficult following task of assessing the overall consequences on the environment. Especially for land and water use, the impacts are mostly local, and would depend fundamentally on specific knowledge about the local conditions. For instance, the blue water footprints calculated here is a sum of water consumption in all parts of the world, without considering the water availability at the point of extraction. Still, even though all the blue water use is aggregated in the WF_b indicator, the model keeps the detailed information. This facilitates impact assessments based on the footprint accounts.

Even the pressure accounting however, is not straightforward. The matters of how to directly quantify pressures on the climate, and on biological and freshwater resources, are not definitively settled, and our method involves some weighting and subjective choices. For instance, the carbon footprint (CF) is perhaps the least disputed among the three indicators, but even here we weight emissions of CO_2, CH_4, N_2O, and various fluorinated greenhouse gases (F gases), into tons of CO_2-equivalents, using weights that depend on the time horizon chosen. Furthermore, we do not include biogenic CO_2 emissions, though Cherubini et al. [10] point out that these too have a forcing effect during their atmospheric lifetime. The Ecological Footprint (EF), since its introduction two decades ago, has become popular as a sustainability indicator. However, it has also been criticized for being too simplistic, see [11]. The land footprint (LF) used here is defined as a subset of the EF, where we exclude energy land (carbon uptake land). The energy land is not land that is used as such; it is a certain area of forest land required to be left unused to store carbon to counteract CO_2 accumulation in the atmosphere due to anthropogenic emissions. This area is directly derived from CO_2 emission accounts; however, since these are already counted in the CF in this analysis, we did not include this part of the EF. Regarding water use, it is immediately obvious that some water use can hardly be said to have any harmful effects at all, if the extraction rate is modest and water is abundant, while extensive water consumption in water stressed regions is another story. The water footprint (WF) [12] does not account for this, but counts all water equally, which has spurred some debate [13-17].

There are several sources of uncertainty in our analysis. There are some inherent uncertainties to MRIO analyses; this relates to the aggregation of products and industries required when keeping complete records of national economies, and to the fundamental assumption that monetary transaction record can be used to represent physical flows. Moreover, there is also the question of the validity of the underlying data. The GTAP database is based on voluntary data submissions from a network of partners, and the quality of the submitted data is not always certain [2]. The physical extension matrices we constructed for this model were based on data from the FAOSTAT database, which suffers from similar data quality challenges. Finally, a limitation to our analysis is the vintage of the datasets. The model year is 2004, since this is the reference year for the GTAP 7 database. However, the GTAP 8 database has recently been released, with 2004/2007 as reference years [18].

Supplementary Data

Environmental extensions

Environmental extensions describing the land use by economic sectors and final consumption were developed following an approach proposed by Ewing et al. [19]. Actual land use was associated to the harvested primary biomass products, which were allocated to economic sectors of their first use. This approach allowed for the utilization of detailed data on international trade of specific primary crop and forestry products and their use by economic sectors based on information from the FAOSTAT database [8]. Therefore, primary crop and forestry products were treated using the high level of detail included in FAO statistics, while the input-output model with considerably broader product categories was employed to address the trade and consumption of products produced from these primary biomass products and also for primary fish products. In a second step all the primary biomass products were converted into equivalent area using country specific conversion factors. The equivalent area associated to primary biomass products used by individual economic sectors is then allocated to final consumers using the standard input-output equation:

$$L = F \cdot (I - A)^{-1} \cdot Y$$

Supplementary Results

Supplementary Figures

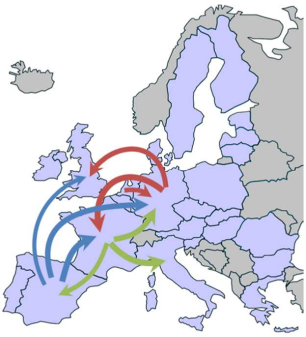

Figure S-3 Top three gross displacements of environmental pressures between EU member states, with arrows pointing in the direction of product flows, i.e. opposite of displacement. Red arrows show CF, green LF, and blue WF_b. For each footprint the arrow thicknesses indicate relative magnitudes.

Supplementary Tables

Description of Tables All descriptions are referring to the tables in the Excel workbook 'Supporting_Information.xls'.

Table S-1 in the worksheet 'Overall_FP' shows the total footprints for each EU country, as well as the EU and world totals. Table S-2 shows the results from the production or territorial perspective, meaning e.g. for WFb - how much blue water is consumed within the borders of each country. The footprint and territorial results are thus the same for the world overall. Overall land footprint results were previously published in ref 1. Population estimates were taken from the GTAP 7 database[2].

Tables S-3 through S-7 show displaced footprints. In the top 28x28 table in each sheet, an element (i,j) shows the total environmental pressures occurring in region i and allocated to final consumption in region j due to international trade. The rather small values on the diagonals (i,i) represent pressures occurring in region i which go into production chains abroad before coming back to the home region for final consumption. An example can be wheat grown in Norway and exported to Sweden as flour where it is used to produce bread which is in turn imported back to Norway for final consumption.

Each value in this top table is then broken down on the top 5 contributing products below. As such there are 28*28 small 5x3-tables, where the first column shows the product consumed, the second shows the footprint attributed, and the third converts this value to a percentage of the total shown in the big table above. Be aware that the breakdown is on products purchased in region j that lead to environmental pressures in region i, hence the products need not directly represent imports between these regions.

Consider this example: In the 'CF_trade' sheet, we see that purchases of "Motor vehicles and parts" by final consumers in Austria led to 47 ktCO2e of emissions in Belgium. This could be caused by Austrians directly importing cars or car parts from Belgium, or it could be that they purchase cars from another country or domestically. For instance we can imagine that part of this came from Austrians buying German cars, and that the German car manufacturer used electricity that was produced by Belgian coal power plants. The only thing we know directly from the table is that the purchases, somewhere in the supply chain, led to these emissions in Belgium.

In both the top tables, as well as in the bottom system of 28*28 small tables, the top 5 values are highlighted in each sheet (excluding exchanges with the aggregated "Rest of the world" region).

Note that the land footprint (LF) is a subset of the Ecological Footprint (EF), and the blue water footprint (WFb) is a subset of the water footprint (WF). The results are for the year 2004.

Tables S-8 and S-9 show the sectors and regions included in the MRIO model, based on the GTAP 7 database.

Tables S-10 and S-11 show the countries and products included in the physical extension matrices.

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Appendix B: Paper II

Steen-Olsen, K.; Owen, A.; Barrett, J.; Guan, D.; Hertwich, E. G.; Lenzen, M.; Wiedmann, T., Intercomparison of global multiregional input-output databases. *Submitted for publication in Economic Systems Research*.

Accounting for value added embodied in trade and consumption: An intercomparison of global multiregional input-output databases

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Abstract

Global multiregional input-output (MRIO) tables constitute detailed accounts of economic activities, worldwide. Global trade models based on MRIO tables are being used to calculate important economic and environmental indicators such as value added in trade or the carbon footprint of nations. Such applications are highly relevant in international trade and climate policy negotiations, and consequently MRIO model results are being scrutinized for their accuracy and reproducibility. We investigate the uncertainty of results from three major

MRIO databases by comparing underlying economic data and territorial and consumption-based results across databases. Although global value added accounts were similar across databases, we find some significant differences at the level of individual regions and sectors. Model disagreement was relatively stable from the territorial to the consumption perspective. Pairwise matrix comparison statistics indicate that the GTAP and WIOD MRIO tables were overall more similar to each other than either was to the Eora database.

Keywords

MRIO tables; MRIO comparison; Eora; GTAP; WIOD

1. Introduction

Considerable progress has been made in recent years in calculating the extent to which various production factors and social or environmental externalities are embodied in international trade flows. As part of the globalization process, the production of goods and services is increasingly separated from their consumption, with supply chains often spanning multiple countries. In most cases, global multiregional input-output (MRIO) models are used to create macro-level accounts of factor use from the consumption perspective, as these models are capable of tracking the monetary flows of goods and services between nations as well as the production factors embodied in these flows. Example applications include value added in trade (Bridgman, 2012; Johnson and Noguera, 2012; Foster-McGregor and Stehrer, 2013; Kelly and La Cava, 2013; Michel, 2013; Auer and Mehrotra, 2014; Baldwin and Robert-Nicoud, 2014; Suder et al., 2014) or the environmental (Hertwich and Peters, 2009; Galli et al., 2012; Steen-Olsen et al., 2012; Weinzettel et al., 2013; Wiedmann et al., 2013; Hoekstra and Wiedmann, 2014; Tukker et al., 2014) or social (Alsamawi et al., 2014; Simas et al., 2014a; Simas et al., 2014b) footprints of nations.

A number of global MRIO databases have been developed recently (Wiedmann, 2009; Wiedmann et al., 2011; Peters et al., 2011; Lenzen et al., 2012; Dietzenbacher et al., 2013; Tukker et al., 2013), see a summary in (Murray and Lenzen, 2013). However, with the availability of several alternative databases, the issue of potentially conflicting estimates of such accounts for a nation or a sector needs to be addressed. To what extent are the results produced by MRIO tables reliable and reproducible considering that each table may provide different results? This question is at the heart of potential policy applications seeking to address questions of differing value added or resource-intensity of traded products and responsibility for environmental or social impacts embodied in consumed goods and services.

Quantification and management of uncertainty becomes increasingly important as the number of policy relevant applications of MRIO modeling rises (Wiedmann and Barrett, 2013). The work presented in this paper aims to identify and quantify the differences between global MRIO tables and their implications for policy formulation.

In a recent study, Peters et al. (2012) address this question with carbon footprint assessments in mind. They consider the effect that different *emissions* datasets have on carbon footprint results using a single MRIO system, in this case the GTAP table, but the authors did not investigate to what extent different MRIO databases model the global economy in a structurally similar way, in terms of output and consumption levels, trade patterns, and sector interdependencies.

A recent special issue of Economic Systems Research (Vol. 26, Issue 3) presents a number of papers comparing MRIO databases and analytical results derived from their use (Inomata and Owen, 2014). Moran and Wood (2014) compare carbon footprint accounts calculated with four different databases. With harmonized emissions datasets they find national carbon footprint accounts to vary by up to 10% across models for most major economies, though significantly more for many other countries. Owen et al. (2014) apply structural decomposition across databases to compare national carbon footprints and find that for a majority of regions accounts from the GTAP and WIOD databases are more consonant than those calculated from the Eora database, and attribute a significant part of this to differences in the Leontief inverse. In a comparison of carbon footprint accounts for GTAP and WIOD MRIO databases using structural decomposition analysis, Arto et al. (2014) find that most of the overall differences can be attributed to differences in a few key regions and sectors.

In this paper we present a comparison of the Eora (Lenzen et al., 2012; Lenzen et al., 2013), GTAP[1] (Andrew and Peters, 2013) and WIOD (Dietzenbacher et al., 2013) global MRIO tables for the year 2007. We focus on the degree to which the different databases represent the global economy in a way that can be said to be structurally similar, by comparing value added accounts from a production-based and a consumption-based perspective, and from a regional and a sectoral perspective. Though we foresee our results will be particularly interesting for environmental assessments, we have chosen to conduct the analysis based on value added rather than any environmental factors so as to be able to study the MRIO tables in their "purest" form, to avoid the added uncertainties arising from including extension matrices that are not an intrinsic part of the MRIO system, especially since these represent physical rather than economic quantities. This also makes our results more relevant in terms of the many other potential applications of MRIO, such as analyses of energy, labor, material use or land use, for which the relative importance of sectors may be very different than they are for carbon emissions. Finally, we discuss sources of uncertainty and offer recommendations for the future use and development of MRIO systems.

In the following section, the basics of multiregional input-output analysis are presented along with the MRIO databases selected for this analysis, and the approach taken to compare them. In Section 3, the main results are presented, while Section 4 concludes.

2. Methods and data

2.1. Multiregional Input-Output Analysis

Input-output analysis yields economic output as the solution to a set of linear equations that describe both inter-industrial and inter-regional trade relationships. The IO identity describes output ($\mathbf{x}$) as a function of final demand ($\mathbf{y}$):

[1] We use version 8 of GTAP (Narayanan et al., 2012). Note that the original GTAP database does not include an MRIO table; however one can readily be constructed from it (Peters et al., 2011).

$$\mathbf{x} = \mathbf{L}\mathbf{y} \tag{1}$$

In Equation 1, each element l_{ij} of the total requirements matrix $\mathbf{L}$ shows the total (direct and indirect) output of region-sector i instigated per unit of final demand on region-sector j. $\mathbf{L}$ is a matrix of dimension $mn \times mn$ where m is the number of regions and n the number of sectors represented in the MRIO table. For the present analysis we pre-multiply Equation 1 by a coefficient vector $\mathbf{v}_c'$ $(1 \times mn)$ of primary inputs required per unit of production, forming

$$\mathbf{v}' = \mathbf{v}_c'\mathbf{x} = \mathbf{v}_c'\mathbf{L}\mathbf{y} \tag{2}$$

Equation 2 gives total valued added $\mathbf{v}'$ by sector, expressed as a function of final demand. In MRIO tables, $\mathbf{y}$, rather than a single column vector of total final demand, can be represented as a matrix $\mathbf{Y}$ of width m, each column vector $\mathbf{y}^{r}$ representing the final demand of a specific region r. Such a representation allows the total value added resulting from the final demand of a single region to be determined. If both vectors $\mathbf{v}_c'$ and $\mathbf{y}^{r}$ are diagonalized, $\widehat{\mathbf{v}_c'}\mathbf{L}\widehat{\mathbf{y}^{r}}$ is a matrix of the same dimensions as $\mathbf{L}$ which expresses the value added in region-sector i as a result of the consumption of region-product j. By repeating this procedure for each region, m such matrices can be produced, analyzing each region's final demand separately.

Consider for instance the vector $\mathbf{y}^{AUS}$ of Australian final demand. Assume the kth item of $\mathbf{y}^{AUS}$ represents Australian final demand of food products from New Zealand. In this case, the kth column of $\widehat{\mathbf{v}_c'}\mathbf{L}\widehat{\mathbf{y}^{AUS}}$ shows the contributions of value added by each production sector by region to the total Australian demand of New Zealand food products.

For the purpose of this analysis we aggregated each $\widehat{\mathbf{v}_c'}\mathbf{L}\widehat{\mathbf{y}^{r}}$ matrix horizontally by product type to give m such matrices of dimension $mn \times n$, and concatenated these to a new matrix $\mathbf{\Phi}(mn \times mn)$. Rather than a column representing e.g. Australian final demand of New

Zealand food products, a column in **ϕ** represents Australia's *total* final demand of food products, broken down by VA-contributing regions and sectors. In the **ϕ** matrix calculated from each MRIO table, then, we now have the global gross value added broken down by producing region and sector (as rows), and demanding region and demanded commodity (as columns). These matrices will be used for the MRIO table intercomparison in the following.

2.2. Data sources

The most recent audits of the main global MRIO initiatives (Peters et al., 2011; Wiedmann et al., 2011; Dietzenbacher and Tukker, 2013) describe six systems of which three (WIOD, Eora and EXIOBASE) were released in 2012. The other major MRIO systems available are GTAP, AIIOT, and various systems using OECD tables[2]. The present study was performed for the reference year 2007, for which there is data available for three MRIO tables; the GTAP8, Eora, and WIOD databases. Eora is available in both a homogeneous 26-sector classification and a heterogeneous system where different countries are represented by different sector classifications depending on data availability. We analyze both Eora systems, treating them as two separate tables. The outcomes calculated by the four MRIO systems will vary because each table has been constructed slightly differently. Differences between the tables can be categorized into three broad areas:

- Source data – e.g. source of national input-output tables, bilateral trade data and environmental accounts
- System structure – e.g. numbers of sectors and regions, use of supply and use tables (SUTs) or symmetric input-output tables (SIOTs)

[2] WIOD: www.wiod.org; Eora: www.worldmrio.com; EXIOBASE: www.exiobase.eu; GTAP: www.gtap.agecon.purdue.edu; AIIOT: http://www.ide.go.jp/English/Research/Topics/Eco/Io/index.html; OECD: http://www.oecd.org/trade/input-outputtables.htm

- System construction – e.g. the method by which the compilers have dealt with missing data and what techniques were used to ensure system balance

Table 1 summarizes the main differences in the tables considered.

Table 1. Characteristics of the assessed databases.

Eora (Lenzen et al., 2013)		
Source data	**National IO tables**	74 IO tables from national statistical offices Other countries' data taken from the UN National Accounts Main Aggregates Database
Source data	**Bilateral trade data**	Trade in goods from UN Comtrade database Trade in services from UN Service trade database
Source data	**Environmental accounts**	EDGAR IEA
Source data	**Value added data**	National IO tables UN National Accounts Main Aggregates Database UN National Accounts Official Data
System structure	**Region detail**	186 countries
System structure	**Sector detail**	Varies by country; ranges from 26 to 511 sectors
System structure	**Structure of IO tables**	Heterogeneous table structure. Mix of SUT and SIOTs. SIOTs can be industry-by-industry or product-by-product
System construction	**Harmonization of sectors**	Uses original classification from national accounts
System construction	**Harmonization of prices and currency**	Converts national currencies into current US$ using exchanges rates from IMF
System construction	**Off-diagonal trade data calculations, balancing and constraints**	Large-scale KRAS optimisation of an intial MRIO estimate with various constraints
Eora26 (Lenzen et al., 2013)		
Source data	**National IO tables**	As Eora
Source data	**Bilateral trade data**	As Eora
Source data	**Environmental accounts**	As Eora
Source data	**Value added data**	As Eora
Syst	**Region detail**	As Eora

	Sector detail	26 homogeneous sectors
	Structure of IO tables	As Eora
System construction	**Harmonization of sectors**	Uses concordance matrices to aggregate Eora to 26 sectors
	Harmonization of prices and currency	As Eora
	Off-diagonal trade data calculations, balancing and constraints	As Eora
GTAP (Andrew and Peters, 2013)		
Source data	**National IO tables**	Tables submitted by GTAP consortium members
	Bilateral trade data	Trade in goods from UN Comtrade database. Trade in services from UN Servicetrade
	Environmental accounts	CO_2 derived from IEA energy data.
	Value added data	Tables submitted by GTAP consortium members
System structure	**Region detail**	129 regions
	Sector detail	57 homogeneous product-by-product sector tables
	Structure of IO tables	Homogenous SIOT table structure
System construction	**Harmonization of sectors**	To disaggregate a country's non-agricultural sectors, the structure from other IO tables within regional groupings is used. For agricultural sectors data from the FAO is employed
	Harmonization of prices and currency	IO tables scaled to US$ using GDP data from the World Bank
	Off-diagonal trade data calculations, balancing and constraints	Uses 'entropy-theoretic methods' to harmonize dataset. Constraints include consumption data from the World Bank, energy data from IEA), Bilateral trade data from UN's COMTRADE database.
WIOD (Dietzenbacher et al., 2013)		
Source data	**National IO tables**	SUTs from National Accounts.
	Bilateral trade data	Trade in goods from UN Comtrade database. Trade in services from UN, Eurostat and OECD
	Environmental accounts	Emissions from NAMEA

	Value added data	SUTs from National Accounts.
System structur	**Region detail**	40 countries and a rest of the world region
	Sector detail	35 homogeneous industry-by-industry sector tables
	Structure of IO tables	Homogenous SIOT table structure
System construction	**Harmonization of sectors**	Developed concordance tables between national classifications and the 35 sectors used in WIOD.
	Harmonization of prices and currency	Supply table (from SUT) in basic prices. Use table in purchases prices. Transform the Use table to basic prices. Convert all data to current US$ using exchange rate from IMF
	Off diagonal trade data calculations, balancing and constraints	International SUTs merged to a 'World SUT' then transformed to a WIOT using the fixed product sales structure assumption. Missing data – use additional info and reallocate negative entries

The comparison in this study was performed on the tables in basic prices as far as possible. The GTAP database is valued in what the developers call 'market prices', which are similar to the basic prices used in standard input-output systems (Peters et al., 2011). WIOD and Eora are both available in basic prices.

2.3. Aggregation

Prior to data comparison between MRIO tables, we need to arrange the structure of each table to be the same in terms of the global economic regions and sectors within each region. To this end, a common classification (CC) system of regions and sectors was adapted, into which all the MRIO tables could be aggregated. The CC was defined based on the principle of greatest common factor, so that the harmonization would be strictly an aggregation exercise for all tables. A concordance matrix was constructed for each of the MRIO tables, allowing them to be aggregated to the CC structure for comparison. Table 1 lists the original dimensions of the three MRIO tables used in the study. Since the full Eora table has a heterogeneous sector classification system where regions may have different numbers of sectors, the CC was built according to the least detailed of these classifications – regions with information on just 26 sectors. The CC resulted in a system with 40 countries as well as a bulk 'Rest of the World' (RoW) region, each with 17 sectors, for a total of 697 region-sectors. For details of the aggregation to the CC for the various tables, please refer to Tables A1 and A2 in the appendix. Results for each MRIO table were calculated using the table at its full level of detail, and then post-aggregated to the common classification to avoid the well-published issues of pre-aggregation (Lenzen, 2011; Andrew et al., 2009).

2.4. Matrix comparison methods

The $\mathbf{\Phi}$ matrices alone contain significant amounts of information. They directly provide detailed value added accounts from both the production-based perspective (through their rows) and the consumption-based perspectives (through columns). Hence, a quantitative evaluation of the four $\mathbf{\phi}$ matrices may provide an indication of overall table similarity. There is no single statistical test that can be used to determine the accuracy with which a matrix corresponds to another (Butterfield and Mules, 1980), and it is suggested that to obtain an indication of similarity between the result matrices calculated by the four MRIO systems a suite of matrix comparison statistics is used (Harrigan et al., 1980; Knudsen and Fotheringham, 1986; Günlük-Şenesen and Bates, 1988; Gallego and Lenzen, 2005). The convention in matrix similarity tests is to compare elements from a matrix of superior data c_{sup} with elements from a matrix of preliminary estimates c_{act} (Gallego and Lenzen, 2005). We adopt this notation when describing the comparison equations below, but note that in this study there is no MRIO system assumed to produce superior results over another. This means that the similarity tests used must be commutative and calculate the same result regardless of which MRIO system is chosen as c_{sup} or c_{act}. The Chi-squared statistic is an example of a comparison test which calculates a different results if the variables are interchanged, and as a result it was excluded from this study.

After surveying the literature and excluding statistics that were non-commutative or directly correlated to other statistics, the following six were selected to estimate matrix similarity:

1. The mean absolute deviation (MAD) (MABS in (Harrigan et al., 1980))

$$MAD = \frac{1}{m \times n}\sum_{i=1}^{m}\sum_{j=1}^{n}\left|c_{act,i,j} - c_{sup,i,j}\right|$$

2. The mean squared deviation (MSD)

$$MSD = \frac{1}{m \times n}\sum_{i=1}^{m}\sum_{j=1}^{n}\left(c_{act,i,j} - c_{sup,i,j}\right)^2$$

3. The Isard-Romanoff similarity index (DSIM)

$$DSIM = 1 - SIM = \frac{1}{m\ \times n}\sum_{i=1}^{m}\sum_{j=1}^{n}\frac{\left|c_{act,ij} - c_{sup,ij}\right|}{\left|c_{act,ij}\right| + \left|c_{sup,ij}\right|}$$

4 R-squared (RSQ)

$$RSQ = \left[\frac{\sum_{i=1}^{m}\sum_{j=1}^{n}(C_{act,ij}-\bar{C}_{act})(C_{sup,ij}-\bar{C}_{sup})}{\left\{\sum_{i=1}^{m}\sum_{j=1}^{n}(C_{act,ij}-\bar{C}_{act})^2 \cdot \sum_{i=1}^{m}\sum_{j=1}^{n}(C_{sup,ij}-\bar{C}_{sup})^2 \cdot\right\}^{1/2}}\right]^2$$

5. The absolute psi statistic (ABSPSI)

$$ABSPSI = \sum_{i=1}^{m}\sum_{j=1}^{n}p_{ij}\left|ln\left(p_{ij}/s_{ij}\right)\right| + \sum_{i=1}^{m}\sum_{j=1}^{n}q_{ij}\left|ln\left(q_{ij}/s_{ij}\right)\right|$$

Where

$$p_{ij} = c_{act,ij}\Big/\sum_{i=1}^{m}\sum_{j=1}^{n}c_{act,i,j}$$

$$q_{ij} = c_{sup,ij}\Big/\sum_{i=1}^{m}\sum_{j=1}^{n}c_{sup,i,j}$$

$$s_{ij} = \left(p_{ij} + q_{ij}\right)/2$$

6. The absolute entropy distance (AED)

$$AED = |H_p - H_q|$$

where

$$H_p = \sum_{i=1}^{m} \sum_{j=1}^{n} p_{ij} \, ln p_{ij}$$

$$H_q = \sum_{i=1}^{m} \sum_{j=1}^{n} q_{ij} \, ln q_{ij}$$

and p_{ij}, q_{ij} are defined as above.

Each matrix comparison statistic takes a different approach to measure similarity. The first three measures can be described as ‘distance measures’. The MAD is based on the absolute distance between each element in the two matrices, independent of the magnitude of the cell values. This means that the MAD puts larger weight on the relative accuracy of larger elements. The MSD calculates squares of differences; hence large table elements will count relatively more towards the overall distance evaluation. This further emphasizes the effect of differences between cells containing large values. DSIM takes a different approach and considers the proportional difference whether the cells contain large or small elements. The next measure, RSQ, calculates how well the sets of values in each matrix correlate, and can be called a ‘goodness of fit’ measure. An RSQ value of zero indicates no correlation between the two matrices, whereas a value of one indicates perfect correlation. The final two measures are ‘information-based statistics’ (Knudsen and Fotheringham, 1986). Information theory is concerned with the quantification of information and ABSPSI and AED are extensions of the the information gain statistic developed by Kullback and Leibler (1951). Information-based statistics compare the probability distributions of the result matrices. We use the ABSPSI information gain statistic as opposed to others because it is commutative (Knudsen and Fotheringham, 1986). The AED is the absolute value of the difference in the entropies of the two result matrices. In information theory, entropy refers to the amount of information needed to specify the full system.

As a screening test for table dissimilarities to potentially indicate directions of future research, we apply this set of indicators to each of the six possible pairings between the databases, to see if any table pair is consistently ranked as more or less similar across statistics.

3 Results

We assess value added accounts from two perspectives. Through the Leontief inverse, input-output tables can be used to allocate all production and the associated value added generation, dollar for dollar, to a corresponding final consumption activity that is assumed to be its ultimate driving force. Value added accounting, as accounting of other factors of production, can thus be performed with allocation either to producers or consumers. We refer to these accounting perspectives as production-based accounting (PBA) and consumption-based accounting (CBA), respectively. PBA and CBA are both relevant, but quite different approaches to issues such as international emissions accounting. The two perspectives are interestingly different also for the purpose of MRIO table comparison: Whereas MRIO agreement from the production-based perspective depends only on MRIO basic data, consumption-based accounts are model results obtained by tracking final demand effects through an infinite number of tiers in the supply chain as represented by the model's technology matrix. Model agreement from this perspective thus also depends on the global economic structures implicitly assumed by each MRIO table through the Leontief inverse.

An initial comparison of the value added accounts accompanying each database was performed at the most aggregated level, represented by the global gross value added (GVA). While the value added data in WIOD comes from the supply and use tables from the MRIO regions' national accounts, Eora uses value added accounts from national IO tables supplied with data from the UN National Accounts Main Aggregates database (Owen et al., 2014).

GTAP relies on data submitted by consortium members, using GDP data from the World Bank for adjustments (Aguiar and Dimaranan, 2008).

The comparison showed more or less equal GVA values of 52.7 and 52.8 trillion USD for Eora/Eora26 and WIOD, respectively, while this value according to GTAP was 53.6 trillion USD or about 1.6% higher than the other two. The relative standard deviation (RSD) of 0.9% was deemed a high degree of coherence. In the following, we are interested in the economic structures assumed by each table rather than in absolute value added accounts. For this reason, we henceforth report value added accounts as shares of the global GVA rather than in absolute values unless otherwise noted.

3.1 Comparison of global value added accounts by sector

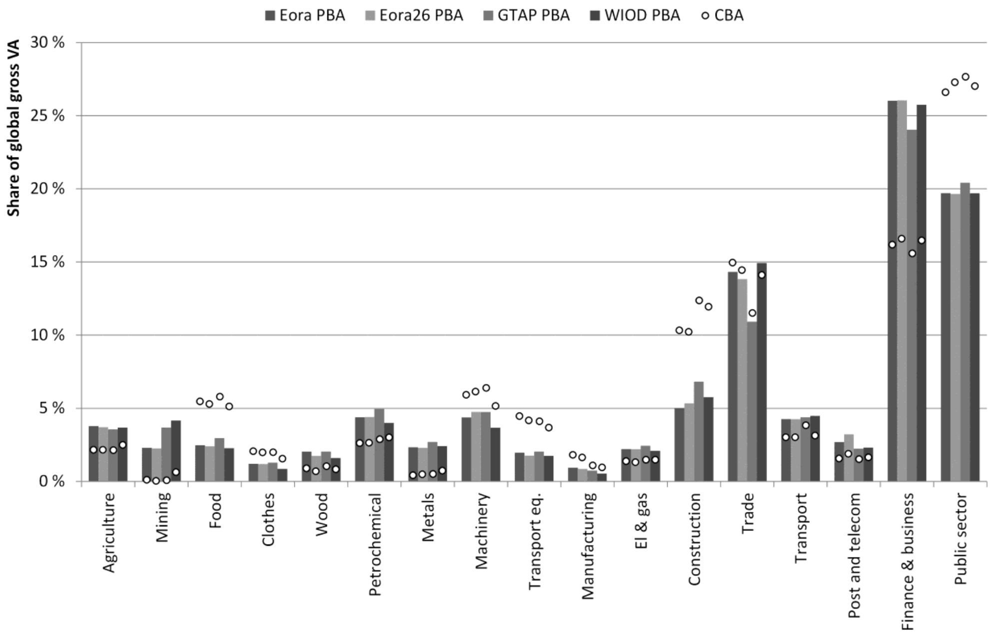
For the sectoral comparison, all data and results are aggregated across regions, such that sectors are global and all VA is distributed between the 17 CC sectors. Figure 1 shows the sectoral breakdown from both accounting perspectives as calculated by each MRIO database.

The columns in Figure 1 show the overall distribution of VA generation according to the data underlying each table. Though the global GVA was similar across databases, there are some differences in how they each attribute its generation to the various sectors of the economy. Across sectors the RSD ranged from 2-31%, with an output-weighted average of 8%. The Mining sector stands out (RSD 31%), with global GVA generation according to Eora/Eora26 42% lower than the average of GTAP/WIOD. In absolute terms, in addition to the Mining sector the GVA generation differs especially for the sectors ‘Trade’ and ‘Finance & business’. GVA according to GTAP in these tertiary sectors is significantly lower than the others, despite the global GVA being higher.

The results of the Leontief reallocation to consumed products are shown for each table as the markers accompanying each column in Figure 1. The reallocation causes significant changes for some sectors, whereas others are less affected. The finance and business sector becomes less important towards the global total, while the public sector increases to be the most important sector from the CBA perspective. Again, these overall trends are the same across tables. A visual comparison of the relative position of the markers compared to the height of the columns suggests that the pattern of change is largely the same across tables; i.e. if table A gives a somewhat larger VA estimate in the PBA than table B, it is generally also larger to more or less the same degree in the CBA. The RSD calculation corroborates this notion. Though the RSD now ranges from 2-131 % the weighted average RSD is similar as in the PBA at 7%.

This result is somewhat surprising. The conversion from PBA to CBA involves shifting value added data through the production system described by the Leontief inverse, which includes relationships with all other sectors through an infinite series of intersectoral dependencies. The change is significant; some sectors, especially resource extraction and similar primary sectors, here represented by mining and quarrying, all but vanish during the reallocation from PBA to CBA because their outputs are almost exclusively intermediate goods. The public administration sector becomes relatively more important because it delivers its output mostly to final demand, and because it draws heavily upon other sectors. Still, the analysis showed that this reallocation was performed in the tables in such a way that the overall table spread was in fact slightly reduced. Peters et al. (2012) investigated carbon emissions using a previous version of GTAP with five different carbon emission inventories and made a similar observation; they explain this by noting that the data for large regions, which constitute a significant part of the consumption based accounts of other regions through international trade, are more in agreement than data for small regions.

Figure 1. Comparison of production- and consumption-based accounts (PBA, columns; and CBA, markers) for the 17 CC sectors, aggregated to the global level, as calculated by the four MRIO databases.



RSD_P	2 %	31 %	12 %	17 %	12 %	9 %	7 %	12 %	8 %	22 %	6 %	14 %	13 %	3 %	17 %	4 %	2 %
RSD_C	8 %	131 %	5 %	12 %	16 %	7 %	26 %	9 %	8 %	30 %	6 %	10 %	11 %	12 %	10 %	3 %	2 %

Note: The agreement between tables from both accounting perspectives are shown as relative standard deviations in the bottom of the figure.

3.2 Comparison of global value-added accounts by region

An important area of application for global MRIO tables is within international climate and environmental policy making, where consumption-based accounts can serve to illustrate how countries and regions depend on and influence each other, directly as well as indirectly through trade in products embodying environmental pressures. The political acceptance of consumption-based accounts to inform this debate hinges on their credible construction via MRIO analysis. The construction of such accounts depends on detailed information on international trade; however, information on the distribution of imported goods between different sectors is usually lacking, and must be estimated by MRIO compilers. This disaggregation may affect how the table allocates economic activities in the various countries to consumption in other countries.

In the previous section the tables were post-aggregated across regions; in this section an analogous aggregation is performed across sectors to rather allow a comparison of table agreement from a regional perspective. Although the deviation of the global GVA was in the order of 1% from the average across databases, a comparison of some macro-indicators by region as shown in Table 2 reveals considerably larger differences.

Table 2. GVA generated, total; GVA embodied in consumption, total; GVA embodied in consumption, generated abroad; GVA generated, embodied in consumption abroad, for each CC region.

	GVA (PBA)			GVA (CBA)			GVA (imported)			GVA (exported)		
Reg	Avg.	Max diff.	High/low database	Avg.	Max diff.	High/low database	Avg.	Max diff.	High/low database	Avg.	Max diff.	High/low database
USA	14 073	3 %	e/G	13 856	5 %	E/G	2 104	20 %	E/W	1 078	19 %	W/E
RoW	6 916	4 %	e/W	6 927	12 %	E/G	1 672	34 %	W/G	1 704	41 %	G/e
JPN	4 266	2 %	W/E	4 281	3 %	G/E	570	16 %	E/W	641	16 %	E/G
CHN	3 390	7 %	W/G	3 254	3 %	W/e	549	16 %	G/e	910	15 %	W/G
DEU	3 111	7 %	e/W	3 147	15 %	G/E	595	30 %	G/e	1 048	31 %	E/G
GBR	2 502	12 %	G/E	2 617	8 %	G/e	682	23 %	E/W	491	22 %	W/e
FRA	2 363	6 %	G/E	2 450	11 %	G/E	490	12 %	G/W	465	31 %	E/W
ITA	1 957	5 %	G/W	1 990	7 %	G/e	425	10 %	G/e	437	33 %	E/W
CAN	1 385	12 %	e/W	1 349	5 %	e/W	283	9 %	W/e	362	45 %	E/G
ESP	1 312	9 %	G/E	1 374	8 %	G/e	345	7 %	G/e	230	18 %	G/W
BRA	1 165	21 %	G/E	1 271	17 %	G/e	134	7 %	E/G	138	33 %	G/e

IND	1 101	15 %	G/E	1 173	21 %	G/e	158	56 %	G/e	160	29 %	W/e
RUS	1 094	15 %	G/E	1 162	17 %	G/e	156	63 %	G/e	253	49 %	W/e
MEX	1 005	4 %	e/G	989	5 %	W/G	157	22 %	W/e	199	16 %	G/W
KOR	924	35 %	G/E	1 050	36 %	G/e	239	6 %	E/W	254	2 %	e/E
AUS	838	9 %	W/G	848	10 %	W/G	178	23 %	E/G	152	10 %	W/e
NLD	714	7 %	G/W	743	23 %	G/E	187	7 %	W/e	265	45 %	E/G
TUR	576	22 %	G/E	635	15 %	G/e	131	11 %	G/e	78	62 %	G/e
IDN	443	13 %	e/G	421	11 %	E/G	91	14 %	E/W	112	27 %	e/W
BEL	436	14 %	e/W	433	50 %	G/E	116	91 %	G/e	196	62 %	E/W
SWE	421	6 %	G/W	429	11 %	E/W	122	16 %	E/W	140	3 %	G/e
POL	376	11 %	G/E	397	11 %	G/W	115	15 %	G/e	93	44 %	W/e
TWN	351	40 %	G/E	397	55 %	W/E	59	80 %	W/e	140	36 %	G/e
AUT	342	3 %	G/W	348	5 %	G/W	114	13 %	E/W	122	9 %	E/e
DNK	272	7 %	G/W	282	12 %	G/E	76	30 %	G/e	90	20 %	G/e
GRC	270	17 %	G/E	289	6 %	G/e	96	21 %	E/W	33	106 %	G/e
IRL	228	17 %	G/E	247	8 %	e/G	79	13 %	G/e	115	75 %	G/e

FIN	221	7 %	G/W	231	17 %	G/e	52	27 %	G/e	72	19 %	E/W
PRT	195	22 %	G/E	212	8 %	G/e	77	25 %	E/W	39	48 %	G/e
CZE	168	13 %	e/W	168	6 %	G/W	60	9 %	G/e	71	27 %	E/W
ROU	152	15 %	G/E	162	15 %	G/e	47	23 %	W/e	30	33 %	W/e
HUN	123	10 %	G/W	130	5 %	G/W	44	22 %	G/e	48	27 %	G/e
SVK	77	21 %	G/W	82	39 %	E/W	38	33 %	E/W	27	38 %	G/e
SVN	44	10 %	G/W	46	12 %	G/W	17	17 %	G/e	16	13 %	E/W
LUX	42	65 %	G/E	49	60 %	G/W	30	101 %	E/W	21	167 %	W/e
BGR	39	26 %	e/W	42	18 %	G/W	17	30 %	G/e	12	20 %	G/W
LTU	34	21 %	G/E	37	24 %	G/E	14	37 %	G/e	10	51 %	W/e
LVA	26	17 %	G/E	28	27 %	G/e	11	45 %	G/e	6	22 %	W/e
EST	19	15 %	G/E	21	22 %	G/e	8	39 %	G/e	7	22 %	G/e
CYP	19	13 %	G/E	20	5 %	E/W	9	25 %	G/W	4	302 %	G/e
MLT	6	14 %	G/E	7	11 %	G/W	4	61 %	G/e	3	155 %	G/e

Notes: Regions shown ranked by GVA generated. For each measure, columns show average across databases (trillion USD), disagreement of the highest estimate relative to the lowest, and which databases give the highest/lowest estimate (E: Eora; e: Eora26; G: GTAP; W: WIOD). Averages calculated excluding Eora26 to avoid giving undue weight to the Eora database. For GVA (PBA) Eora/Eora26 are practically identical.

Across regions, the difference in gross value added generation from the lowest to the highest estimate is 13% on the median; in the consumption-based accounts it is 11%. Though there is a weak tendency of larger deviations for the smaller countries, there are significant differences also among the larger economies in the PBA or the CBA accounts. In many of these cases, the largest disagreement is between GTAP and Eora, with GTAP generally giving the higher estimates. Although the slightly higher global GVA in GTAP relative to Eora obviously amplifies this effect to some degree, rescaling the results in Table 2 so that global GVA is the same across databases (not shown) only leads to minor changes and does not change this picture. Overall, the level of error is stable from the production to the consumption perspective; however for individual regions it is sometimes significantly increased (e.g. Belgium) or decreased (e.g. Portugal).

In the right half of Table 2 the same comparison across databases is performed on the analysis of value added embodied in traded products for each region, specifically the amount of value added embodied in consumption that is generated in a different region, and the reciprocal assessment of the amount of domestic value added generation attributable to consumption abroad. The differences observed through this comparison are generally larger, with medians of 22% and 31% for the imports and exports comparison, respectively. There are also a few regions for which this estimated value differs by a factor of two or more between the most deviating results.

3.3 Calculations of matrix similarity

In an attempt to get a more complete, quantitative evaluation of the table agreement, a series of matrix comparison statistics were calculated for each of the six possible pairs of tables. The comparisons were performed on the value added multiplier matrices $\boldsymbol{\phi}$, and Table 3 shows how each statistic ranks the pairs in terms of similarity. Not surprisingly, Eora-Eora26 is overall the most similar pair combination. The entropy-based AED statistic is seen to disagree with the other indicators throughout, interestingly deeming both Eora and Eora26 to be more similar to GTAP than to each other. Using the mean rank from the six indicators as the criteria, second to Eora-Eora26 in similarity is Eora26-WIOD, followed by GTAP-WIOD. Conversely, Eora-GTAP is the most dissimilar table pair by all indicators except the AED.

Table 3. Ranks of similarity statistics for the 6 matrix pairings. 1 = most similar pair, 6 = least similar pair.

	Eora26	**GTAP**	**WIOD**
Eora	Rank MAD = 1 Rank MSD = 4 Rank DSIM = 1 Rank RSQ = 3 Rank ABSPSI = 1 Rank AED = 3 **Mean Rank = 2.17** **St Dev Rank = 1.33**	Rank MAD = 6 Rank MSD = 6 Rank DSIM = 6 Rank RSQ = 6 Rank ABSPSI = 6 Rank AED = 1 **Mean Rank = 5.17** **St Dev Rank = 2.04**	Rank MAD = 5 Rank MSD = 5 Rank DSIM = 4 Rank RSQ = 5 Rank ABSPSI = 5 Rank AED = 6 **Mean Rank = 5.00** **St Dev Rank = 0.63**
Eora26		Rank MAD = 4 Rank MSD = 3 Rank DSIM = 5 Rank RSQ = 4 Rank ABSPSI = 4 Rank AED = 2 **Mean Rank = 3.67** **St Dev Rank = 1.03**	Rank MAD = 3 Rank MSD = 1 Rank DSIM = 2 Rank RSQ = 1 Rank ABSPSI = 3 Rank AED = 4 **Mean Rank = 2.33** **St Dev Rank = 1.21**
GTAP			Rank MAD = 2 Rank MSD = 2 Rank DSIM = 3 Rank RSQ = 2 Rank ABSPSI = 2 Rank AED = 5 **Mean Rank = 2.67** **St Dev Rank = 1.21**

4 Discussion and conclusions

The question of to what degree global MRIO databases are similar is highly important. Currently, such tables are not compiled by national or international statistical offices, but by a few research groups collecting various datasets from various sources and using various methods to harmonize, disaggregate and balance their tables. Since the national data that go into MRIO tables will ultimately be supplied by many different sources all over the world, and there is no suggested standard for constructing MRIO tables from them, there are potentially significant, yet mostly not quantified uncertainties. Nevertheless, these tables are increasingly being used by researchers to perform analyses that serve to inform national and international policies, at present notably within the fields of climate and environment. The differences observed in the comparison of macro-level indicators at the sector and region level were non-negligible in most and considerable in many cases. These errors may stem from several factors; taking the MRIO database descriptions in Table 1 as a point of departure, we offer some suggestions below.

First of all, the level of detail is quite different between the databases. Several countries in Eora, generally those that play larger roles in the global economy, are represented by several hundred economic sectors, whereas in WIOD the same countries are represented by only 35 sectors. For analyses of individual sectors or regions, lack of detail may result in large aggregation errors, especially for environmental MRIO analyses. However, for analyses at the macroeconomic level similar to the one conducted here, the effect of these errors is less pronounced, as supported by the agreement found here between Eora and Eora26, and by Steen-Olsen et al. (2014).

The method by which compilers populate the off-diagonal trade blocks in the MRIO transactions matrix can explain some of the discrepancies. International trade data are not available by importing sector; hence the bulk of the elements in the MRIO transactions table must be estimated. This challenge is met through somewhat different means in the databases assessed. Still, the importance of these uncertainties is likely limited, since whatever the method applied by the compilers will still be constrained by import/export totals at the national level. Again, analyses at a more detailed level might be affected more.

Valuation schemes represent a continuous challenge for input-output compilers and analysts, which may have to deal with data in basic, producers' or purchasers' prices, or variants of these as in the case of GTAP discussed previously. Data are typically available in the valuation most useful to the data supplier, and records might include or exclude various types of taxes and margins. The general IO preference is to use basic prices. IO analysis is not the primary focus of the GTAP consortium, which does not strictly adhere to IO conventions. For instance, the 'market price' valuation used in GTAP differs from the basic price in that it includes domestic margins as inputs to production (Peters et al., 2011). The effect of this deviation in the GTAP database from the true basic price as used in Eora and WIOD we have not analyzed, but judging from the findings here (cf. Table 2) it may be the case that this effect is more important than previously assumed.

The question of how to measure how similar two or more MRIO databases are is not trivial. In our study we have focused on similarity of the model results as a practical proxy. In addition to this, several statistical measures of matrix similarity were identified, and applied to the overall multiplier matrices $\mathbf{\Phi}$, to determine if any two were consistently deemed more dissimilar to each other. This was found to be the case for Eora and GTAP, although these results should be considered to be indicative only.

It is clear that an analysis of value added embodied in consumption based on a model where 15% of the global value added generation occurs in the CC "Trade" sector may, depending on the research question, conclude quite differently than if the model only puts this at 11%. The same is true for analyses of any environmental or social indicators. On the other hand, the overall stability in differences observed in going from the production to the consumption perspective is interesting. Consumption-based account modeling is the essence of input-output analysis, and results depend on the structure of the full multiregional table, the elements of which must for a large part be estimated by compilers due to limited data availability. If it were confirmed that the uncertainty in consumption-based accounts is mainly the result of errors in the extension matrices, focus could be shifted to this rather than the more daunting task of adjusting individual entries in the full transactions matrix; however recent work by Owen et al. (2014) attribute a considerable share of uncertainties also to the Leontief inverse.

When extending the MRIO databases with physical-environmental tables to analyze environmental impacts from a consumption-based perspective, another possibly significant source of error is introduced with the linking of physical to monetary flows. The potential data errors will generally be larger in these datasets than for a value added table, as the latter is constructed in parallel with, and constrained by, the other MRIO tables whereas emission and similar table extensions have no such inherent relationship with the economic sectors they are allocated to. For this reason the value ranges can be much larger in environmental extension matrices than in value added tables – most industries will have value added generation over sectoral output ratio of 25-75%, whereas emissions are not constrained in this manner.

This paper is intended as a first step towards a more fundamental understanding of the reliability of global MRIO tables and how it can be improved. In the attempt to construct a basis for further research toward this end, we have focused on general comparisons at the macro level. It is to be expected, however, that comparisons on a more detailed level, such as on results of advanced contribution analyses, will give larger differences than observed here. We suggest further investigations along this vein as an area of future work, focusing on identifying regions and sectors that contribute especially to overall model uncertainty (see

Arto et al. (2014) for some recent work in this direction) and determining their underlying causes. We believe the matrix distance measures presented here can prove useful for future comparative studies that delve more into subsystems of the global economy (i.e. of the databases), investigating database representations of individual regions or economic sectors.

Analyses of the importance of different or ambiguous valuations in input-output databases should be undertaken, especially considering the popularity of the GTAP database for MRIO assessments. Investigations into the differences between GTAP and Eora should pay particular notice to this aspect. Furthermore, based on the results found here and by Peters et al. (2012) suggesting that underlying data quality is of particular importance towards errors in analyses of social or environmental extensions, further uncertainty assessments focusing on various relevant extensions should be performed. For MRIO compilers, the present study suggests that efforts towards improving database quality are best spent working with national statistical agencies and data suppliers to improve data quality, but also to acquire a better, preferably quantitative, understanding of the uncertainties accompanying individual datasets.

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Appendix C: Paper III

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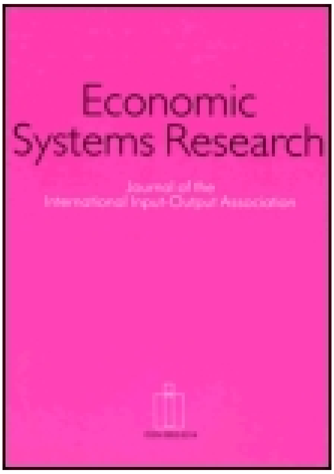

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EFFECTS OF SECTOR AGGREGATION ON CO_2 MULTIPLIERS IN MULTIREGIONAL INPUT-OUTPUT ANALYSES

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The past few years have seen the emergence of several global multiregional input–output (MRIO) databases. Due to the cost and complexity of developing such extensive tables, industry sectors are generally represented at a rather aggregate level. Currently, one of the most important applications of input–output analysis is environmental assessments, for which highly aggregate sectors may not be sufficient to yield accurate results. We experiment with four of the most important global MRIO systems available, analyzing the sensitivity of a set of aggregate CO_2 multipliers to aggregations in the MRIO tables used to calculate them. Across databases, we find (a) significant sensitivity to background system detail and (b) that sub-sectors contained within the same aggregate MRIO sector may exhibit highly different carbon multipliers. We conclude that the additional information provided by the extra sector detail may warrant the additional costs of compilation, due to the heterogeneous nature of economic sectors in terms of their environmental characteristics.

Keywords: MRIO databases; Aggregation; CO_2 multipliers

1. INTRODUCTION

In the pursuit of effective policies and strategies to lessen the environmental burdens of our society, a key element is the accounting scheme chosen to keep track of environmental interventions. One central accounting decision to be made is whether emissions should be tallied at the point where they occur, or at the point of final consumption of the goods or services being produced. The two schemes may be referred to as accounting from the production or the consumption perspective, respectively. Production-based accounting has the advantage of being unambiguous and rather straightforwardly measured, and it is fundamentally the same principle which is used in environmental regulations such as the US Clean Air Act and the Kyoto Protocol, where regions are made responsible for the emissions occurring within their borders.[1] Consumption-based accounting

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[1] The territorial approach, which is adopted in the Kyoto Protocol, is similar but slightly less comprehensive than the production approach, because emissions from international shipping and aviation are not allocated to any country.

(CBA) is the principle of attributing responsibilities of environmental pressures to the point of final consumption rather than to the processes where the pressures occur. For example, the CO_2 emissions from a steel mill are allocated to the final consumers of the products requiring the steel either directly (i.e. the final product contains steel) or indirectly (i.e. the final product required inputs of steel somewhere in its supply chain). In the terminology of CBA, consumption activities are said to *embody* a certain amount of environmental pressures, accumulated through the supply chain. The CBA principle rests on the assumption that any activity in the global economic system, and hence all emissions, occurs with the ultimate goal to deliver some product or service for final consumption.

CBA is of current interest within the realm of environmental policy-making: firstly, it facilitates the design of demand-side policies, by identifying the consumption activities that matter more or less for a given environmental issue. Secondly, it is being put forward by some as a more equitable principle for designing international climate and emissions agreements, which would also help to avoid the leakage effects experienced in the Kyoto Protocol (Peters and Hertwich, 2008; Chen and Chen, 2011; Peters et al., 2011b; Aichele and Felbermayr, 2012; Kanemoto et al., 2014).

Environmentally extended input–output analysis (EEIOA) is the prevailing method for large-scale assessments of environmental pressures embodied in consumption. Input–output analysis (IOA) is an analytical framework describing the interdependencies between the sectors of an economy, developed in the 1930s by Wassily Leontief, building on Quesnay's *Tableau économique* (Leontief, 1936). It allows the calculation of output *multipliers* or estimates of the total production output by each sector of the economy required as a result of a final demand of one unit of any sector's output. By extending the economic transactions tables of a standard input–output (IO) system with accounts of emissions or other environmental indicators, emissions multipliers rather than just economic output multipliers can be calculated by the same principles.

However, the top-down nature of IO tables implies practical limitations in terms of sector detail, which will also apply to environmental assessments based on them. To be able to track all transactions in the economy, IO table compilers aggregate small firms into broader economic sectors. The characteristics of the sectors thus represent weighted averages of the characteristics of the firms aggregated within them. For IO-based environmental assessments, such aggregations could be highly important, depending on the environmental indicator being analyzed. Sectors in IO tables are generally defined on economic rather than environmental bases, and they can represent firms with completely different environmental characteristics. Consider, for instance, a hypothetical assessment of CO_2 emissions embodied in brass instruments. In a typical low-detail IO system, the copper and zinc that make up the brass would be aggregated in a 'non-ferrous metals' sector, dominated by the far more CO_2-intensive aluminium industry, thus leading to artificially high estimates of emissions embodied in the instruments.

The recognition that detailed multiregional input–output (MRIO) tables are required for environmental assessments in an increasingly globalized and diverse economy has led to the recent development of a handful of such databases by various research groups, see Wiedmann et al. (2011) for an overview. In this paper we assess four of these, calculating CO_2 multipliers in each of the full databases, as well as after aggregating all four to a defined common region and sector classification system, with the aim of studying the effects of levels of sector detail on such multipliers.

The general problem of aggregation in input–output tables has been discussed extensively, see Kymn (1990) for an overview. Several authors have assessed empirically how IO coefficients and multipliers vary with different levels of aggregation.

- Based on a 1960 IO table for Philadelphia, Karaska (1968) studied how sector aggregation affected total-material coefficients for approximately 1,000 firms in the manufacturing industries, and found that aggregation even to the most detailed Standard Industrial Classification (SIC) level (four-digit, 126 industries) resulted in an average coefficient of variation (CV) of 31%.[2] Further aggregation led to an even higher variation; the average CV was 37% after aggregation to the three-digit SIC level (96 industries), and 45% for the two-digit level (19 industries). 'Assembly'-oriented sectors displayed more variation than primary sectors more dependent on a few, large inputs. A ranking of sectors according to CV values proved quite stable independent of the level of aggregation.
- Kymn (1977) studied possibilities for aggregation of the American 1963 IO table for energy forecasts, and found a significant scope for aggregation at low accuracy costs with a careful selection of sectors for aggregation.
- Katz and Burford (1981) compared the output multipliers for the 367-sector 1967 US IO table to an aggregated version with 81 sectors, and found high levels of variation among the original multipliers compared to the multipliers of their aggregate sectors.
- Bullard and Sebald (1988) applied Monte Carlo simulations to study error propagation in the 1967 US IO table, and found that when assessments were based on linear combinations of IO coefficients, errors canceled each other to the degree that overall errors were within acceptable limits – irrespective of the level of aggregation.
- Miller and Shao (1990) examined the sensitivity of output multipliers of the 1977 US MRIO table to regional as well as sector aggregation, and found scope for significant regional aggregation, while the sensitivity to sector aggregation was somewhat higher.
- Wyckoff and Roop (1994) found that carbon embodied in imports to several European countries from the USA was reduced by about 30% when calculated with a 6-sector aggregated version of the original 33-sector table.
- Lenzen et al. (2004) investigated how Denmark's CO_2 accounts changed in a five-region MRIO table when aggregating from an average of 118 to only 10 sectors per region, and found significant errors.
- Su et al. (2010) studied the effect of sector aggregation in calculations of CO_2 emissions embodied in exports for the case of China. They used the 2002 Chinese IO tables at four levels of sector aggregation, and found that a level of around 40 sectors was sufficient to capture the majority of the embodied emissions.
- Lenzen (2011) showed that disaggregation of IO data is preferable to aggregation of environmental extension data, even if based on only a few data points.
- Bouwmeester and Oosterhaven (2013) studied carbon and water footprints using the EXIOBASE MRIO database, and quantified effects of sector and region aggregation. Their findings largely agree with Su et al. (2010) in the number of sectors required; however, this varied strongly across countries.

[2] The coefficient of variation is the same as relative standard deviation, i.e. standard deviation divided by the mean.

In this paper, we capitalize on the recent availability of a suite of MRIO databases with global coverage to study how the level of sector detail may affect multipliers used for environmental assessments. Though the aggregation issue has been studied before, it has mostly been treated theoretically, or based on experiments with hypothetical or small-scale tables. Furthermore, most of the older works in the above list have dealt with purely economic assessments, and the rest have generally been focused on total footprints rather than multipliers. Many of the studies found that sector aggregation may not be a big issue; however, there is reason to believe that this might not be the case for environmentally extended IOAs. There are two main reasons for this. Firstly, while there are technological limits to the variation in economic inputs and value-added coefficients between industries, emission intensities can easily differ by orders of magnitude from one industry to the next. Secondly, no matter the aggregation level of an input–output system, at some level firms will be grouped together, typically based on similarity of outputs and processes. However, processes that appear otherwise similar may in fact be very different if studying non-monetary factors such as labor or environmental interventions. For example, if studying lead pollution, it would be advantageous to have aviation fuel separate from automotive fuel.

We experiment with four of the largest global-coverage MRIO databases available, studying how CO_2 multipliers change when tables are aggregated. Compilers of these databases have chosen different levels of detail. While a highly detailed database may intuitively seem desirable, there would be clear advantages with more aggregated MRIO databases if they can be justified in terms of model accuracy. A more aggregate table can save compilers as well as analysts both time and money, and in practice, users may not always desire tables that are too detailed as the model results can be hard to interpret. It should also be noted here that the construction of a highly disaggregated IO table may be complicated by the fact that many firms have diverse product ranges that are not easily distinguished. If IO compilers allocate such firms to the sector that most closely resemble what is considered their main output, the more disaggregated system can in fact lead to a worse representation of the actual processes for such firms. However, IO systems constructed from supply and use tables (SUT) do not suffer from this problem as sectors are allowed multiple outputs.

We chose to conduct the analysis at the multiplier rather than the footprint level. This allows a better understanding of the MRIO databases' sensitivity to aggregation, since results are independent of final demand volumes. A multiplier analysis is also more useful for researchers interested in using MRIO for assessments of specific products or product groups. Readers more interested in footprints at the national level should hence bear in mind that the product multipliers are not equally important toward these totals.

In the following section, the basics of IOA are explained along with the steps taken to calculate the sets of multipliers compared. Section 3 contains the results of the analysis, while Section 4 provides a discussion of the main findings. Section 5 concludes.

2. METHODS AND DATA

2.1. Input–Output Analysis

IOA is an analytical framework using records of economic transactions between the sectors of an economy to analyze interdependencies among them. In its simplest form, an economy is described as a set of *n* economic sectors, and the gross sales between them during the course of a year are recorded in an $n \times n$ transactions matrix $\mathbf{Z}$, in which an element z_{ij} represents sector j's total purchases from sector i, usually in monetary terms. The transactions matrix describes how the sectors depend on each other's products in order to produce their own. In IO terminology, this inter-industrial consumption is called *intermediate* consumption, based on the assumption that all this activity takes place to enable the industrial system to ultimately deliver products to *final* consumers. Sales to final consumers are reported in an $n \times d$ final demand matrix $\mathbf{Y}$, where d is the number of specific final consumption categories detailed. The columns of $\mathbf{Z}$ and $\mathbf{Y}$ together contain all sales by each sector, such that a vector $\mathbf{x}$ of gross outputs can be obtained by summing across them:

$$\mathbf{x} = \mathbf{Zi} + \mathbf{Yi} = \mathbf{Zi} + \mathbf{y}, \tag{1}$$

where $\mathbf{i}$ is a summation column vector of ones of appropriate length.

Just as $\mathbf{Y}$ consists of all sales to purchasers outside the industrial system, an IO system will also contain a value-added matrix $\mathbf{V}$ that contains each sector's payments other than purchases of the products of industry sectors. This matrix contains all non-industrial payments, such as wages, taxes, and profit. In a balanced IO system, the total payments of each sector equal its total sales, so that $\mathbf{x}$ can be obtained by summation down columns of $\mathbf{Z}$ and $\mathbf{V}$ as well:

$$\mathbf{x}' = \mathbf{i}'\mathbf{Z} + \mathbf{i}'\mathbf{V}. \tag{2}$$

In IOA, the assumption is that the observed flows represent requirements for production, so that sector j's total payments, reported in column j of $\mathbf{Z}$ and $\mathbf{V}$, represent its specific requirements in order to deliver its total output, recorded as the jth element of $\mathbf{x}$. Thus, $\mathbf{Z}$ and $\mathbf{V}$ can be normalized by the vector of total outputs to coefficient ('per-unit-output') forms:

$$\mathbf{A} = \mathbf{Z}\hat{\mathbf{x}}^{-1}, \tag{3}$$

$$\mathbf{V_c} = \mathbf{V}\hat{\mathbf{x}}^{-1} \tag{4}$$

where the circumflex ˆ represents diagonalization of a vector and subscript c denotes coefficient form. $\mathbf{A}$ is called the direct requirements matrix, because its columns represent a sector's direct input requirements from every other sector in order to produce one unit of its output.

By insertion of Equation 3, Equation 1 can now be rearranged to

$$\mathbf{x} = (\mathbf{I} - \mathbf{A})^{-1}\mathbf{y} = \mathbf{Ly}. \tag{5}$$

Equation 4 can similarly be expressed as a function of the final demand:

$$\mathbf{V} = \mathbf{V_c}\widehat{\mathbf{Ly}}. \tag{6}$$

The matrix $\mathbf{L}$, referred to as the Leontief inverse, gives the total (direct + indirect) output by each sector required per unit of output delivered for final consumption.

The equations above outline the most basic form of an IO system. In EEIOA, a matrix $\mathbf{F}$ containing accounts of one or more environmental extensions is appended to the tables. It is treated analytically like $\mathbf{V}$; however, it can be specified in any unit desirable and there is no particular balancing requirement. The databases assessed here, in addition to

TABLE 1. Overview of the MRIO databases compared.

Database name	Eora	EXIOBASE	GTAP 8	WIOD
Reference year(s)	1990–2011	2000	2004, 2007	1995–2011
Number of regions	187	44	129	41
…of which were estimated	113	1	20	1
Number of sectors	26–511	129	57	35
Transaction matrix dimension	14,760[a]	5,676	7,353	1,435
Currency	US$	€	US$	US$

[a]Note that whereas the dimension is equal to the number of regions times the number of sectors for the other databases, the matrix is larger for Eora because of the occurrence of SUTs. If only counting the number of commodities (which is greater than or equal to the number of industries) available for each region with SUTs, the system dimension of Eora is a little over 10,000, or an average of 54 sectors per region. In practice, many of the smaller countries have the minimum of 26 sectors, while larger countries may have several hundred commodities.

being environmentally extended, are *multiregional*, meaning they represent an economy consisting of several regions, each with their own set of economic sectors, all interacting with each other. The analytical framework is the same as for a single region table; however, the multiregional tables provide much better representation of internationally traded goods.

2.2. The Set of MRIO Databases Analyzed

The MRIO databases studied in this analysis include the Eora, EXIOBASE, Global Trade Analysis Project (GTAP8), and World Input–Output (WIOD) databases. Out of these, Eora is the most detailed overall. It also differs from the others in that its sector detail varies among regions, from a minimum of 26 sectors up to more than 500 for the UK. Like Eora, GTAP features a high level of regional detail, specifying 129 countries and aggregate regions. The EXIOBASE and WIOD databases both have a European focus in terms of regional detail. The EXIOBASE is particularly detailed in terms of sectors, featuring a total of 129 sectors for each of its regions (Table 1). Our analysis was performed for the year 2007 for those databases available with several reference years, since this was the most recent year modeled by all these. EXIOBASE was the exception, since it was only available with 2000 as the reference year. This means that care should be taken when comparing EXIOBASE results to the other databases; however, the focus of this study is not to (directly) compare databases against each other, but rather against aggregated versions of themselves.

2.2.1. Eora

Eora (Lenzen et al., 2012; 2013) is a time series of highly detailed MRIO tables compiled by the Centre for Integrated Sustainability Analysis at the University of Sydney. Eora contains annual tables for the years 1990–2011, and the researchers aim to keep as close to the original data of each country as possible, allowing a combination of symmetric input–output tables (SIOT) and SUT in their database, as well as allowing each region to keep its original sector classification. Eora explicitly describes 187 countries. One hundred and thirteen of these were estimated by the researchers using proxies for the initial estimate, and constrained by measured raw data from the UN (UNSD, 2011) in the reconciliation process.

2.2.2. EXIOBASE

The EXIOBASE MRIO database (Tukker et al., 2013) was the outcome of the EU-funded EXIOPOL project. It features 129 sectors, with special focus on environmentally relevant sectors such as agriculture, energy, and materials. The geographic focus is on the EU; all EU countries at the time of the database as well as the EU's major trading partners are explicitly described, making up a total of $27 + 16$ countries, while the rest of the world (RoW) is described as a single lump region. A drawback of EXIOBASE is that there is no time series available and the reference year is 2000. EXIOBASE is currently undergoing updates, and the new release will have 2007 as reference year, provide more sector detail and a regional disaggregation of the RoW.

2.2.3. Global Trade Analysis Project

The GTAP, based at Purdue University, has been compiling global trade databases since 1993. GTAP does not publish an MRIO directly; however, one can readily be constructed from the tables published (Peters et al., 2011a), and it has been used for several environmental assessments (Wiedmann, 2009). GTAP has a high level of regional detail and an intermediate level of sector detail, with a focus on agriculture. The present analysis was performed with version 8 of GTAP, which features 129 regions and 57 sectors (Narayanan et al., 2012).

2.2.4. World Input–Output Database

The WIOD (Dietzenbacher et al., 2013) was constructed by a European research consortium led by the University of Groningen. Like Eora, it features a continuous time series of MRIO tables, from 1995 to 2011. Its regional focus, like that of EXIOBASE, is on Europe and its main trading partners. It was mainly constructed with economic analyses in mind, but also includes some environmental extensions. It is the least detailed of the four databases assessed here.

2.3. The Common Classification System

A 'common classification' (CC) system, comprising a set of 41 regions and 17 sectors, was adopted for the analysis. The CC was defined so that each of the four MRIO databases could be converted to this classification through a process of straightforward aggregation, taking the principle of the greatest common factor to define regions and sectors. This scheme allowed a total of 40 individual countries which were explicitly modeled in all MRIOs. In addition, the CC system includes a bulk RoW region for completeness. The CC has a better regional detail for Europe than for other continents, reflecting the regional bias of the EXIOBASE and WIOD databases.

Our choice of using the greatest common factor aggregation principle means that each sector in the CC generally corresponds one-to-one to an identical sector in at least one of the databases. This is then the constraining database for this sector in terms of our objective of maximizing the number of sectors in the CC system without requiring any disaggregation. The existence of such direct links between the actual MRIO tables and our aggregate CC versions is an important point to make for our analysis, because it implies that each CC sector is usually an actual sector in at least one database, and as such it is relevant to discuss the effects of aggregation to this sector level. As Table 2 shows, the existence of each

TABLE 2. The sectors of the common classification system.

#	Code	Sector name	Aggregated sectors			
			Eora26[a]	EXIOBASE	GTAP	WIOD
1	AGRF	Agriculture, forestry, hunting, and fisheries	1–2	1–17	1–14	1
2	MINQ	Mining and quarrying	3	18–32	15–18	2
3	FOOD	Food products, beverages, and tobacco	4	33–44	19–26	3
4	CLTH	Textiles, leather, and wearing apparel	5	45–47	27–29	4–5
5	WOOD	Wood, paper, and publishing	6	48–50	30–31	6–7
6	PETC	Petroleum, chemical, and non-metal mineral products	7	51–65	32–34	8–11
7	METP	Metal and metal products	8	66–73	35–37	12
8	ELMA	Electrical equipment and machinery	9	74–78	40–41	13–14
9	TREQ	Transport equipment	10	79–80	38–39	15
10	MANF	Manufacturing and recycling	11–12	81–83	42	16
11	ELGW	Electricity, gas and water	13	84–94	43–45	17
12	CNST	Construction	14	95	46	18
13	TRAD	Trade	15–18	96–100	47	19–22
14	TRNS	Transport	19	101–107	48–50	23–26
15	POST	Post and telecommunications	20	108	51	27
16	BSNS	Financial intermediation and business activities	21	109–116	52–54, 57	28–30
17	PAEH	Public administration, education, health, recreational, and other services	22–26	117–129	55–56	31–35

[a]Eora's own 26-sector CC system listed here for reference, as the full Eora database has a variable sector count depending on the region (Figure A1). The correspondence between each Eora region's sectors to the 26-sector classification is always many-to-one.

CC sector in at least one MRIO database holds true for all CC sectors except sector 17, ‘Public administration, education, health, recreational and other services’, where different overlapping sector definitions implied that no single sector in any database could readily be used as a CC sector to which a set of other sectors could be aggregated in the other databases. Generally, WIOD is the constraining database; although several Eora regions only include 26 sectors, those countries included in the CC system are generally more detailed in Eora. For an overview of the level of detail available in the Eora database for each of the sectors and regions defined in the CC system, the reader is referred to Figures A1 and A2 of the appendix.

2.4. Aggregation

For each database, the following set of matrices were used to calculate the CO_2 multipliers for comparison:

$\mathbf{Z}_0(mn \times mn)$, the multiregional inter-industrial transactions matrix; $\mathbf{Y}_0(mn \times md)$, the final demand matrix including direct imports; and $\mathbf{F}_0(k \times mn)$, the environmental extensions matrix.

Here, m is the number of regions, n is the number of sectors,[3] d is the number of final demand categories, and k is the number of extensions. We use μ and ν to denote the number of regions (41) and sectors (17), respectively, in the aggregate classification. Furthermore, we use the subscript 0 to refer to the full databases and 1 to refer to the aggregate versions. In the present study, $k = d = 1$, however, expanding the analysis for more stressors or more final demand categories is straightforward.

A set of binary concordance matrices was constructed in order to create aggregate (CC) versions of each MRIO database. For each database, the concordance matrix $\mathbf{C}_{01}(mn \times \mu\nu)$ was constructed from two smaller concordance matrices; one mapping region $(\mathbf{C}^{\mathbf{r}}_{01}, m \times \mu)$ and another mapping sector $(\mathbf{C}^{\mathbf{s}}_{01}, n \times \nu)$ according to the following algorithm:

(1) Create a copy of the regional concordance matrix $\mathbf{C}^{\mathbf{r}}_{01}$.
(2) Expand each element $\mathbf{C}^{\mathbf{r}}_{01}(i,j)$ to an $n \times \nu$ matrix:
  (a) If $\mathbf{C}^{\mathbf{r}}_{01}(i,j) = 1$, insert $\mathbf{C}^{\mathbf{s}}_{01}(n \times \nu)$.
  (b) If $\mathbf{C}^{\mathbf{r}}_{01}(i,j) = 0$, insert all-zero matrix $(n \times \nu)$.

For Eora, the concordance matrix had to be slightly modified to accommodate the mixed SIOT/SUT structure. The only aggregate region in the CC is the RoW region, which in the case of Eora was constructed from 147 individual countries. Since this group included countries of both the SIOT and the SUT structure, two RoW regions were constructed for Eora.

Aggregate versions of the databases were constructed from the full versions as follows:

$$\mathbf{Z}_1 = \mathbf{C}'_{01}\mathbf{Z}_0\mathbf{C}_{01}, \tag{7}$$

$$\mathbf{Y}_1 = \mathbf{C}'_{01}\mathbf{Y}_0\mathbf{C}^{\mathbf{r}}_{01}, \tag{8}$$

$$\mathbf{F}_1 = \mathbf{F}_0\mathbf{C}_{01}. \tag{9}$$

2.5. Multipliers

The comparison was performed at the multiplier level, addressing CO_2 emissions from all economic sectors. Three sets of CO_2 multipliers were calculated for each MRIO database:

$\mathbf{M}_0(1 \times mn)$, multipliers as calculated using the full tables; $\mathbf{M}_1(1 \times \mu\nu)$, multipliers as calculated using the aggregate tables; and $\mathbf{M}_{01}(1 \times \mu\nu)$, multipliers as calculated from aggregating the results of the full tables.

[3] Note that in the special case of the Eora database, n is variable. Furthermore, the matrix dimensions are larger for Eora because Eora contains supply and use tables instead of input-output tables for some regions.

For each database, the set of multipliers was calculated from the original matrices following a similar procedure as Lenzen (2001; 2011):

$$\mathbf{M}_0 = \mathbf{F}_0\widehat{\mathbf{x}_0}^{-1}(\mathbf{I}_0 - \mathbf{Z}_0\widehat{\mathbf{x}_0}^{-1})^{-1}, \tag{10}$$

$$\mathbf{M}_1 = \mathbf{F}_1\widehat{\mathbf{x}_1}^{-1}(\mathbf{I}_1 - \mathbf{Z}_1\widehat{\mathbf{x}_1}^{-1})^{-1}, \tag{11}$$

$$\mathbf{M}_{01} = (\widehat{\mathbf{Y}_1\mathbf{i}})^{-1}\mathbf{C}'_{01}\hat{\mathbf{M}}_0\mathbf{Y}_0\mathbf{i}. \tag{12}$$

In the equations above, $\mathbf{I}$ is an identity matrix of appropriate dimension. Note that Equation 12 requires $k = 1$, however for larger k the procedure can simply be repeated for each environmental extension separately.

To study the effect of aggregation on each MRIO database individually, the difference between 'pre-aggregated' multipliers $\mathbf{M}_1$ and 'post-aggregated' multipliers $\mathbf{M}_{01}$ was taken as a measure of aggregation error. We define δ^{rs} as the relative aggregation error in the multiplier for region r and sector s in the CC system:

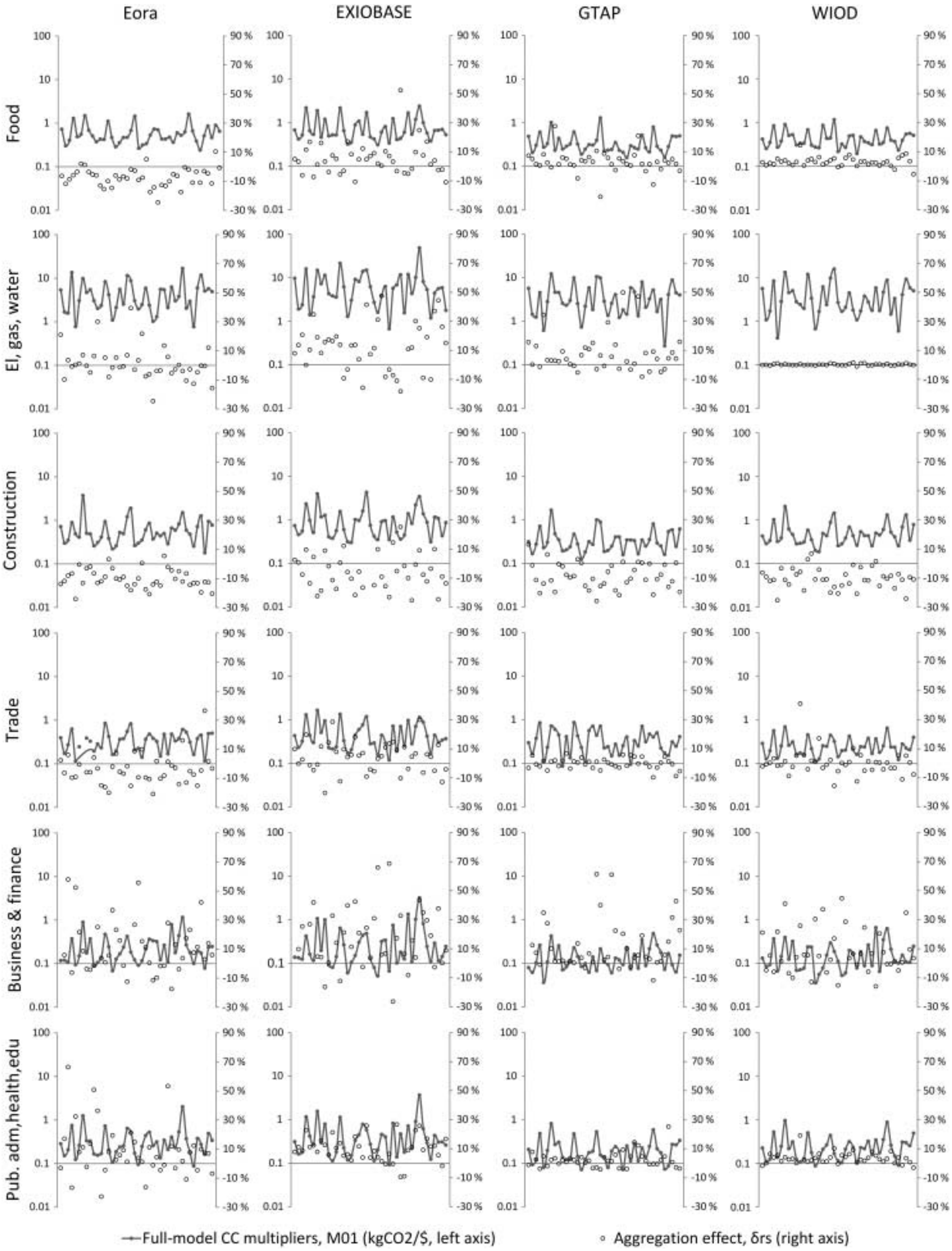
$$\delta^{rs} = \frac{\mathbf{M}_1^{\mathbf{rs}} - \mathbf{M}_{01}^{\mathbf{rs}}}{\mathbf{M}_{01}^{\mathbf{rs}}}. \tag{13}$$

3. RESULTS

CO_2 multipliers for the 17 CC commodities can be calculated directly from the aggregated IO systems ($\mathbf{M}_1$), or by utilizing the information available in the full tables ($\mathbf{M}_{01}$). Figure 1 shows, for six of the larger sectors in terms of gross output, how these differ in the various tables. Each plot contains one such pair for each of the 41 regions, showing $\mathbf{M}_1$ in terms of relative change δ^{rs}. Those multiplier values that were calculated using the aggregate MRIOs directly generally deviate substantially from their full-table counterparts. On the whole, the more detail available in the full table compared to the CC system, the larger the effects of aggregation, as expected. The differences observed in Figure 1 are generally the smallest for WIOD, which was not aggregated much in the experiment, while the additional sector detail available in the full versions of Eora and EXIOBASE influences the CC multipliers considerably. However, the aggregation effect is not necessarily manifested (only) in the sectors that were aggregated the most. The 'Construction' sector is an interesting example, as it was not aggregated at all in either table (save for a few of the Eora regions), yet across all databases, the CO_2 multiplier of the Construction sector appears significantly affected by the overall aggregation process. This reflects the fact that a sector's CO_2 multiplier also includes emissions occurring in other sectors that supply it. On the other hand, the multiplier of the 'Electricity, gas, and water' sector in WIOD, also not aggregated in our experiment, is hardly affected at all. This is explained by the very high degree of own-sector emissions in this multiplier: for the case of Australia, the share is 97%. Conversely, this share is only 30% for the Australian Construction sector in WIOD. These values are representative across regions. For the databases other than WIOD, the spread is large because the 'Electricity, gas, and water' CC sector is disaggregated into several sectors in the respective full tables.

The aggregation effects in the Construction sector is an example of another interesting result: for all four databases, the aggregate versions give multiplier estimates for the Construction sector that are quite consistently too low compared to those found from the full tables, as evident from Figure 1. A closer inspection of the multipliers reveals why:

FIGURE 1. CO_2 multipliers and their change after aggregating databases.



Notes: The figure shows CO_2 multipliers (kg/\$) for six CC sectors, as calculated by the four MRIOs. Each plot shows multipliers for each region from the post-aggregation exercise (left axes), each accompanied by a marker showing the multiplier's change when using the aggregate MRIO instead (right axes).

the Construction sector requires significant inputs of CO_2-intensive cement. However, in the aggregation, the cement sector is aggregated with several larger but less CO_2-intensive sectors into the CC sector called 'Petroleum, chemical and non-metal mineral products' (PETC). Several similar instances of near consistent effects across regions for the same database were found, though sometimes in opposite directions; note for instance the Food sector multiplier, where the aggregation generally led to reductions in Eora and increases in WIOD.

Table 3 contains a quantitative overview of the trends suggested in Figure 1, showing how and to what degree CO_2 multipliers of various sectors tend to be over- or underestimated when tables are aggregated. The first two columns for each database show the median values of $\mathbf{M}_{01}$ and $|\delta|$ across regions (RoW multipliers excluded). For all the CC sectors and across all four aggregation experiments, the aggregation led to multipliers being overestimated for some regions and underestimated for others. Most sector/table combinations tended one way or the other, however. Interestingly, the effect of aggregating the various tables, each with different levels of detail, down to the 17×41 CC dimension was not the same. The aggregation of WIOD, with 1,435 region-sectors, a level of detail not much higher than the CC, mostly manifested itself in multipliers as overestimations. In fact, multipliers for 14 of the 17 CC sectors were mostly overestimated, 11 of them overwhelmingly so (overestimated for more than two-thirds of the CC regions). The effects were mixed for EXIOBASE (5,676 region-sectors) and GTAP (7,353 region-sectors), while for Eora ($>$10,000 region-sectors), the effects were quite the opposite: multipliers for 15 of the 17 CC sectors were mostly underestimated, 11 of which overwhelmingly. Overall, the effect of aggregation was an overestimation of CO_2 multipliers for 72% of the region-sectors in the WIOD experiment, compared to 71% being underestimated for Eora. Furthermore, some of the sectors for which the aggregated version of Eora most consistently underestimated the multipliers (FOOD, ELMA, TREQ, and MANF) were also among the sectors most consistently *over*-estimated in the aggregated WIOD.

The trend of underestimation of the Construction sector multiplier, suggested earlier, is confirmed in Table 3. This is an important point pertaining to product footprinting – aggregation errors need not manifest themselves only in the sectors that were actually aggregated. Hence for the Construction sector, it is the aggregation of sectors that deliver its inputs that cause the error. The Construction sector is one out of only three sectors for which the median relative change carries the same sign for all four databases.

The relative multiplier changes in Table 3 show which sector multipliers were mostly affected by the aggregation, and indicate the degree of error. Overall, sector detail mattered significantly for the calculation of aggregated multipliers in our analysis. When the multipliers are calculated with highly detailed databases such as Eora and EXIOBASE, their values change significantly from what was estimated using the aggregate tables; in the EXIOBASE case, the median change for most CC sectors is larger than 10%. Note that since these are median errors, several individual region-sectors changed significantly more, as seen in Figure 1.

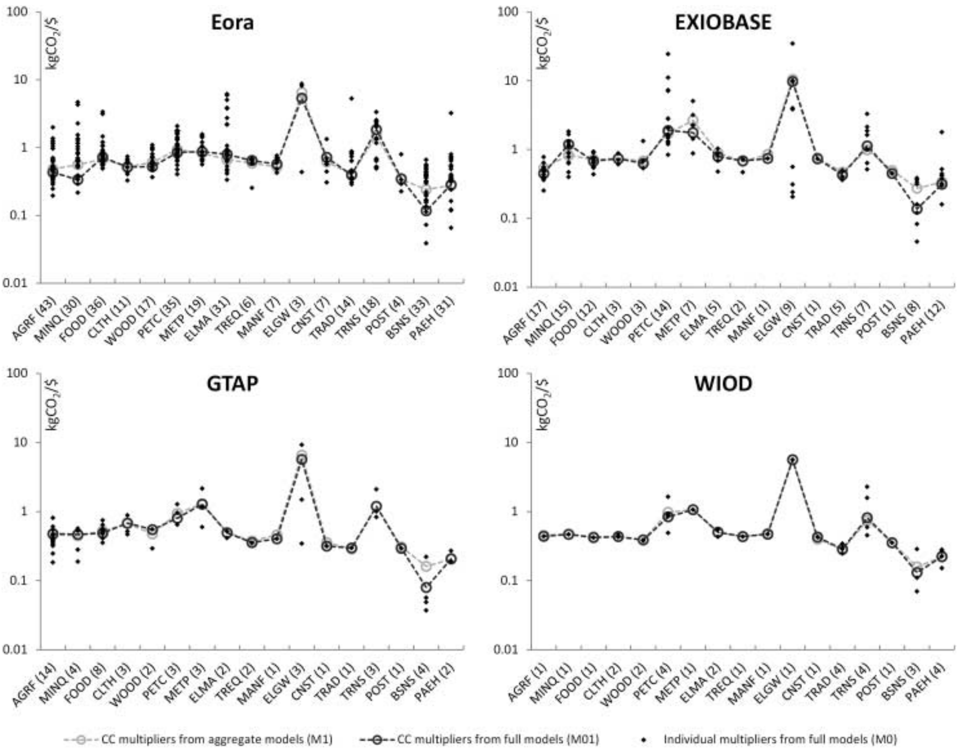
The heterogeneous nature of sectors when it comes to CO_2 intensities is further illustrated in Figure 2, which shows all the multipliers calculated for the example of Australia. As in Figure 1, we compare the CC sector multipliers as calculated using the full ($\mathbf{M}_{01}$) and aggregate ($\mathbf{M}_1$) tables; however, Figure 2 also displays the multipliers of all the Australian sectors that are detailed in the full tables ($\mathbf{M}_0$). In other words, for each CC sector, there is one marker for each sector in the original database that was aggregated into it,

TABLE 3. CO_2 multipliers in full MRIO tables and the effect of aggregation.

	Eora			EXIOBASE			GTAP			WIOD		
s	$\tilde{M}_{01}^{:s}$	$\widetilde{\lvert\delta^{:s}\rvert}$ (%)	+/−	$\tilde{M}_{01}^{:s}$	$\widetilde{\lvert\delta^{:s}\rvert}$ (%)	+/−	$\tilde{M}_{01}^{:s}$	$\widetilde{\lvert\delta^{:s}\rvert}$ (%)	+/−	$\tilde{M}_{01}^{:s}$	$\widetilde{\lvert\delta^{:s}\rvert}$ (%)	+/−
AGRF	0.35	6	12/28	0.52	18	31/9	0.39	13	18/22	0.43	3	36/4
MINQ	0.44	19	8/26	1.14	18	11/28	0.52	11	13/27	0.61	1	24/16
FOOD	0.44	7	3/37	0.61	5	28/12	0.29	3	33/7	0.37	3	38/2
CLTH	0.53	6	6/34	0.58	5	33/7	0.28	3	30/10	0.39	6	39/1
WOOD	0.44	6	15/24	0.73	7	29/11	0.38	6	17/23	0.42	4	33/7
PETC	0.89	10	15/25	1.19	22	36/4	0.52	10	37/3	0.84	12	32/8
METP	0.66	9	16/22	1.26	72	37/2	0.53	23	35/5	0.84	1	28/12
ELMA	0.54	5	5/35	0.73	13	18/22	0.34	5	11/29	0.40	3	37/3
TREQ	0.62	7	5/34	0.78	22	16/24	0.31	5	7/33	0.39	4	37/3
MANF	0.55	9	4/36	0.96	10	33/7	0.33	3	24/16	0.42	5	37/3
ELGW	3.11	5	16/24	5.76	14	29/11	2.96	5	29/11	2.82	0	20/20
CNST	0.51	11	3/37	0.85	10	12/28	0.30	11	8/32	0.43	11	3/37
TRAD	0.31	7	10/29	0.38	7	26/14	0.30	2	21/19	0.24	3	21/19
TRNS	1.42	8	13/27	1.20	13	11/29	1.21	2	22/18	0.68	7	22/18
POST	0.22	9	5/34	0.38	7	28/12	0.16	4	11/29	0.19	3	18/22
BSNS	0.17	11	28/12	0.16	20	34/6	0.11	6	35/5	0.12	7	27/13
PAEH	0.23	9	28/12	0.31	9	35/5	0.18	3	27/13	0.22	3	36/4
Sum +/−	192 (29%)/476 (71%)			447 (66%)/231 (34%)			378 (56%)/302 (44%)			488 (72%)/192 (28%)		

Notes: For each database, the post-aggregated multipliers are shown in absolute values ($kgCO_2$/$) in the first column (median across countries, indicated by a tilde). The remaining two columns show how the multipliers change with aggregation, first as the median relative change, and second as the number of countries for which the multiplier increased/decreased following database aggregation.

FIGURE 2. Aggregate versus original CO_2 multipliers (case of Australia).



Notes: Values in parentheses indicate the number of sectors aggregated from the full to the aggregated database versions.

so that the 'EXIOBASE' panel will have a total of 129 such markers, one for each of the EXIOBASE sectors.

The aggregated multiplier values exhibit largely the same pattern whether the table was constructed from the Eora, EXIOBASE, GTAP, or WIOD database. The 'Electricity, gas, and water' sector has by far the highest CO_2 multiplier across all four CC versions; the multiplier of the 'Transport' sector is also relatively high; and those of the 'Metal products' and the 'Petroleum and chemicals' sectors stand out as notable spikes except in the Eora-based CC table, although they are the third and fourth highest multipliers also in this version.

By comparing the circular markers, we see how each aggregate sector's multiplier is affected by the overall level of table detail. Since the darker circles were calculated from the full versions, they may be considered 'true' multipliers, whereas the light circles representing multipliers of the aggregated tables deviate because of information loss. The more the light circles deviate from the dark ones, the more the additional information available in the full table matters for this sector. Again, in general, the more detail available in the full table, the larger the observed deviation. The CO_2 multipliers of the Australian 'Financial intermediation, business activities' (BSNS) sector suffer particularly from the loss of detail, and the aggregation effect is consistently an overestimation. In the same manner as for the consistent underestimation of the Construction multiplier investigated earlier, a closer inspection of the tables can shed light on the cause of this aggregation error. Whereas the error in the Construction sector's multiplier was caused solely by aggregation of its supplying sectors, the aggregation error for the BSNS sector also comes about from aggregation of this sector itself. Generally, this type of aggregation error arises because the 'true' aggregated multiplier $\mathbf{M}_{01}$ is ultimately a final-demand-weighted average of the multipliers of its full-table subsectors, whereas the aggregation of the full IO table into the CC version implicitly entails a weighting based on gross outputs, because the size of the individual sectors will determine the direct input requirements structure of the aggregated sector. An investigation of the Australian BSNS sector as modeled in WIOD serves to illustrate: The BSNS sector is represented as three different sectors in the full WIOD database: 'Financial Intermediation', 'Real Estate Activities', and 'Renting of Machinery and Equipment (RME) and Other Business Activities'. While the three are of comparable size in terms of gross sales, in terms of sales to final demand, the RES sector is the largest by far. At the same time, the RME sector's CO_2 multiplier is significantly higher than those of the other two.

The multipliers of the 'Mining and quarrying' sector for Eora and EXIOBASE also change significantly when the full systems are used for the calculation rather than the aggregate IO systems. In this case, however, the direction of the error is not the same, the aggregation leading to an overestimation in the Eora case and an underestimation in the EXIOBASE case.

The individual dot markers, showing the original multipliers in the full MRIOs, illustrate the true heterogeneity of the individual sectors (in terms of carbon footprint intensities) that form part of the aggregates. Across all databases and sectors, the internal variability in these sets of aggregated sectors is significant. For the highly detailed Eora and EXIOBASE databases, multipliers falling under the same aggregate sector routinely span an order of magnitude. This means that for a number of environmentally important sectors, using the aggregate table results in a substantial loss of information. For a tabular overview of the spread of original multipliers allocated to the same CC sector, please refer to Table A1 of the appendix.

4. DISCUSSION

The aggregation experiments performed in this study resulted in rather large effects in terms of CO_2 multipliers. In general, the more detailed the original database, the larger the multiplier error when tables were aggregated to the CC system. No particular pattern of how these errors manifested themselves could be identified, although the multipliers of some sectors for some databases were quite consistently (i.e. across regions) too high or too low when the table was aggregated prior to the multiplier calculation. Otherwise, the effect of aggregation on a particular sector's multiplier generally varied significantly across countries, as well as from one table to the next. Only for the Construction sector, which interestingly was not aggregated at all except in some of the more detailed Eora countries, did all the tables show a coherent and significant trend, roughly a 10% underestimation when using the aggregated database versions. Another consistent (but weaker) trend was that the multipliers for the financial and administrative sectors tended to be too high when databases were aggregated. A somewhat surprising finding was a clear tendency for Eora CC multipliers to be reduced when the aggregated version was used, whereas WIOD-derived multipliers showed an equally clear tendency in the opposite direction. No explicit explanation for this was found; however, it may be assumed that one reason lies within the two databases' highly different representation of the RoW region defined in our CC system, which is modeled as only one region in WIOD but as almost 150 individual countries in Eora.

The results of our assessment show that for product carbon footprint accounting, the additional sector detail found in the Eora and EXIOBASE databases adds information that may warrant the additional compilation efforts required, to avoid critical aggregation of sectors with highly different emission structures. Although our comparison has not directly been one of database against database, our results do provide some suggestions in this respect. The findings displayed in Figures 1 and 2 show that the CO_2 multipliers as calculated using the less detailed WIOD or GTAP databases could be significantly different if they had been compiled with more sector detail. If we assume that all four databases are true representations of the global economy, only with different sector and region classifications, the CC multipliers calculated using the full version of the most detailed database would be the most correct representation, and the difference between these and the same multipliers calculated from one of the less detailed databases would represent the gain in multiplier accuracy from compiling more detailed tables.[4]

Figure 2 showcases the true heterogeneity among the individual subsectors contained within each CC sector. The multipliers of the subsectors included within the same CC sector in many cases spanned an order of magnitude; this was found to be true not only in the carbon-intensive ELGW sector, but across the economic spectrum from primary and extractive sectors to administrative and service sectors. This suggests that if a less detailed table is to be used for carbon footprint accounting, the sector classification defined for this study is not ideal, because similarity in terms of economic input structures does not imply similarity in terms of emissions profiles.

We limited our study to CO_2 and found considerable aggregation effects on product multipliers. There is reason to believe that such effects would be even more pronounced for many other environmental extensions. While the CO_2 emissions profile of several sectors is linked at least to some degree to their economic structure through energy use, other environmental interventions such as Pb emissions or water use can be completely unrelated to other common measures of sector similarity, and the intensities may vary even more between sectors than CO_2 emissions intensities do. These factors also speak in favor of higher sector detail in MRIO tables used for environmental assessments.

The implications of our findings are different for different applications of MRIO analysis, and it should here be stressed again that the present analysis has been of multipliers' sensitivity to aggregation, and the results cannot be directly transferred to the case of carbon footprints because these are a product of demand levels as well as multipliers. Nevertheless, for carbon footprinting of individual sectors or products, the results of our analysis can be taken to say that the level of detail could influence results significantly, because of the large differences among the multipliers of specific commodities that are frequently aggregated. However, the adverse effects of this high variability will be dampened in analyses of the total carbon footprint of households or nations, which is determined not from multipliers alone,

[4] Note that this would also require that the environmental extensions are true and equal across all the MRIO models, an assumption that is perhaps more dubious (Peters et al., 2012). For recent work on this, see Owen et al. (2014).

but as the product of multipliers and consumption volumes. For these kinds of analyses, a higher level of detail will have some, but probably more limited, influence.

5. CONCLUSION

For compilers of input–output databases as well as those who want to use them for assessments of various factors embodied in consumption and trade, a recurring question is what level of detail is required for a sufficient degree of accuracy. In this paper, we have approached this question by assessing the sensitivity of CO_2 multipliers to sector detail. Though one of the main challenges for database compilers is now largely overcome, thanks to astounding advances in computational power, the compilation of highly detailed IO tables is still a costly and time-consuming process, and a trade-off will always have to be made, since there is virtually no limit as to the number of firms that could theoretically be included. Although several studies in the past found that a rather limited level of detail was sufficient to achieve acceptable economic output multipliers, the more recent trend of appending various social and environmental extensions to IO tables has led to renewed scrutiny of the effects of aggregation.

Our findings suggest that when conducting carbon footprint assessments using MRIO analysis, a high level of detail can significantly improve the accuracy of the results, because carbon multipliers, one of their determining factors, are sensitive to table detail. In terms of environmental intensities, such as CO_2 emissions or water use, it is clear that industrial processes differ tremendously, whereas in terms of pure economic structure, the variability is generally less. Although a lower level of detail may give acceptable results if firms were grouped into sectors according to their similarity in terms of the environmental stressor under study, the limiting factor these days is not so much computational power for the analysis itself, but rather the time and money spent on compiling large MRIO databases. Compiling databases specifically with, e.g. carbon or water footprinting in mind would be a possible venue; however combined efforts to build versatile and highly detailed databases appear to the authors as a more fruitful way forward.

A note should be made here that the net benefits in accuracy from increased detail suggested above will depend on the detail and accuracy with which additional sectors are added; this has not been quantitatively analyzed here. MRIO data have traditionally been published as face value numbers; uncertainties, though significant, have usually not been well understood. To improve on this situation, the researchers behind Eora have attempted to estimate uncertainties to accompany all values in their database. Such information might improve the credibility of MRIO-based analyses and promote their further use.

The aim of this analysis has been to capitalize on the current situation—historically quite unique—where a suite of MRIO databases with global coverage is available to researchers, to conduct a realistic real-data study on the effects of aggregation in input–output systems. Although we have not aimed to explicitly quantify the aggregation effects attributable to aggregations of sectors versus regions, the focus of the study has been on the overall sensitivity of sector (or commodity) carbon multipliers. Though we conclude that CO_2 multipliers are generally sensitive to table aggregation, we suggest future work should attempt to describe more specifically the relationship between table detail and multiplier accuracy to answer the question of what level of detail is needed to give an acceptable degree of accuracy for various types of analyses.

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SUPPLEMENTAL DATA

Supplemental material for this article is available via the supplemental tab on the article's online page at http://dx.doi.org/10.1080/09535314.2014.934325

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Appendix D: Paper IV

Steen-Olsen, K.; Wood, R.; Hertwich, E. G., A carbon footprint time series of Norwegian households. *Submitted for publication in Journal of Industrial Ecology.*

The Carbon Footprint of Norwegian Household Consumption 1999-2012

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Abstract

Environmentally extended input-output analysis is the prevailing method for national environmental footprint accounting; however its practical usefulness for consumers and policymakers suffers from lack of detail. Several extensive global multiregional input-output (MRIO) databases have recently been released. A standard framework for linking such databases with the highly detailed household expenditure surveys that are conducted regularly by national statistics offices has the potential of providing analysts in countries worldwide with a powerful tool for in-depth analyses of their national environmental footprints.

In this article we combine the Norwegian consumer expenditure survey with a global MRIO database to assess the carbon footprint of Norwegian household consumption in 2012, as well as its annual development since 1999. We offer a didactic account of the practical challenges associated with the combination of these types of datasets and the approach taken to address these, and we discuss what barriers still remain before such analyses can be practically conducted and provide reliable results.

We find a carbon footprint of 22.3 tCO_2e/hh in 2012, a 26% increase since 1999. Transport, housing and food were the expenditures contributing the most towards the total footprint. An expenditure elasticity of 1.14 for the Norwegian household carbon footprint was found, particularly due to very low residential direct emissions and high transport expenditures in the higher-income deciles.

Introduction

To achieve large-scale carbon emissions reductions, consumer-side strategies such as demand reduction and lifestyle changes will be required in parallel with industry-side changes (Fischedick et al. 2014). Carbon-emitting industrial processes are ultimately driven by consumer demands for goods and services, most of which comes from private household consumption (Tukker and Jansen 2006; Hertwich and Peters 2009).

Effective consumer-directed mitigation strategies require a reliable analytical framework for analyzing life-cycle carbon emissions embodied in consumption, so-called 'carbon footprints'. Carbon footprints can be calculated using various assessment frameworks that account for indirect emissions. These include process-based approaches such as life cycle assessment (LCA), however assessments of total household environmental impacts have mostly been based on environmentally extended input-output analysis (IOA) (Hertwich 2005; Tukker et al. 2010). Though life-cycle assessments are coveted for their high level of detail, estimating complete household carbon footprints based on LCA is challenging as there is a lack of studies for many household activities and purchases. Analyses based on input-output (IO) tables have an advantage in that they take a top-down approach, thus avoiding the problem of truncation errors (Lenzen and Dey 2000; Majeau-Bettez et al. 2011).

The results of IO-based assessments are quite useful for understanding the overall carbon footprint, its relationship to the consumption pattern, and its development over time. Due to the coarseness of the approach, differences in specific products such as organic versus conventional vegetables or mass-produced versus luxury apparel cannot be resolved. As a result, IO-based assessments are neither able to resolve the effects of some specific lifestyle choices nor to assess the efficacy of some improvement proposals.

Consumer expenditure surveys (CES) are conducted regularly by national statistical offices, providing a wealth of data on household purchases at a detailed product level. Social scientists analyze consumer expenditure surveys to understand household consumption behavior (Fernández-Villaverde and Krueger 2007). CES have also been used to understand how different socioeconomic and demographic factors affect household energy use and carbon footprints (Lenzen et al. 2006; Ornetzeder et al. 2008; Jones and Kammen 2014). By extending input-output analyses with CES data, more detailed analyses of household consumption can be conducted, paving the way towards further environmental analyses of specific consumer patterns or lifestyles to identify strategies for transitioning towards the sustainable society. Furthermore, the highly detailed description of actual household characteristics and their consumption patterns contained within the consumer expenditure surveys facilitate cross-sectional analyses along the same vein of research.

The contribution of this article

The idea of combining CES and IO data to quantify the environmental impacts of household consumption is not new. In a seminal article, Herendeen and Tanaka (1976) utilized the highly detailed 1960-61 US CES together with the US IO table to analyze the direct and indirect energy requirements of various types of households, and found an energy elasticity of income <1, mostly due to relatively stable levels of direct energy use. In a follow-on article, Herendeen (1978) adopted this method for a similar analysis for Norwegian households using the 1973 CES, and found the same tendencies.

Since then, a range of studies has been published that attempt to reap the benefits of detail and accuracy of CES with the complete upstream analysis capabilities of IOA to analyze household environmental impacts from various angles (Sastry et al. 1989; Wier et al. 2001; Lenzen et al. 2006; Roca and Serrano 2007; Weber and Matthews 2008; Wood and Garnett 2009; Grainger and Kolstad 2010; Jones and Kammen 2011), see also the overviews provided by Kok et al. (2006) and Lenzen et al. (2006). These have mostly been cross-sectional analyses, attempting to unearth correlations between environmental pressures embodied in consumption and various explanatory variables available in the CES, such as income, age, household size or level of education. For want of any standard framework, the combination of the datasets has mostly been performed ad hoc, often with limited details provided on the procedure chosen. Given that IO tables are assembled by national statistical offices to represent a single national economy, they do not reflect increasingly globalized patterns of production. Due to significant differences between energy systems and production patterns, national-level assessments can be misleading (Peters and Hertwich 2006).

Recently, several extensive global multiregional input-output (MRIO) databases have emerged, some of them freely available online (Dietzenbacher and Tukker 2013). In light of this, we foresee that future CES-IO studies will increasingly apply systems like these rather than single- or few-region IO systems, as the accuracy gain of a proper and detailed trade representation is potentially significant (Proops et al. 1999; Lenzen et al. 2004; Wiedmann 2009). To encourage a coherent methodological approach by the research community, allowing comparison across studies, we here outline a practical approach for combining a standard CES dataset with one such global MRIO database (see also the related work of Mongelli et al. (2010)). To inspire further discussion of the approach taken, and to facilitate the use and understanding of results of CES-IO analyses also to non-experts of IOA, we offer an exposition of the practical and methodological challenges encountered and a step-by-step procedure to combine the two datasets. We apply our method to construct a time series of carbon footprint accounts for Norwegian households from 1999 to 2012 and discuss how such analyses can serve as a practical tool for policymakers to investigate their national footprint developments. Finally, we identify limitations and weakness of our approach, and outline what major methodological challenges remain to be addressed to ensure acceptable levels of confidence.

The remainder of the article is outlined as follows. In the following section, we provide a brief methodological account of MRIO-based assessments of footprints embodied in consumption, present the IO and CES datasets used in our analysis, and discuss the main practical and methodological challenges involved in the combination of IO and CES data, in general terms as well the specific approach taken here to construct the Norwegian CF accounts. In section 3, we present and discuss the Norwegian household carbon footprint (hhCF) development over the period, including an in-depth investigation of the 2012 hhCF. Section 4 concludes, discussing challenges and opportunities for future research on the environmental impact of household consumption and strategies for its abatement.

Materials & methods

IOA-based footprints of consumption

Environmental pressures caused in the production of goods and services can be allocated to the final demand driving them through life-cycle type models. An input-output table enumerates the total annual sales by all n sectors of an economy to all other sectors as well as to d groups of final consumers in an interindustrial transactions matrix $\mathbf{Z}$ and a final demand matrix $\mathbf{Y}$, respectively. In environmentally extended IO tables, a matrix $\mathbf{F}$ tallying total environmental interventions (e.g. CO_2 emissions) by each sector accompanies the transactions matrix. In multiregional input-output (MRIO) tables, domestic IO tables for m regions are interlinked with bilateral trade data to form a single composite IO table with international trade endogenized.

The central tenet in input-output analysis is that a sector's purchases from other sectors over a year, as well as its total environmental interventions, represent direct requirements to produce what was its gross output that year. Mathematically, this allows the construction of a direct requirements matrix ($\mathbf{A}$) from the transactions matrix and the vector of gross sector outputs ($\mathbf{x}$):

$$\mathbf{A} = \mathbf{Z}\hat{\mathbf{x}}^{-1} \tag{1}$$

Table 1. Matrices, vectors, and indices used

Symbol	Dimension	Explanation	Element (i, j)
	m	Number of regions	
	n	Number of sectors in each region	
	v	Number of CES commodities	
	d	Number of final consumption groups	
	s	Number of environmental interventions	
	p	Number of commodity groups in price	

		index tables	
	t	Number of years of CES data	
$\mathbf{Z}$	$mn \times mn$	Interindustrial transaction matrix	i's total sales to j
$\mathbf{Y}$	$mn \times md$	Final demand matrix	i's total sales to final consumption group j
$\mathbf{y}$	$mn \times 1$	Total final demand vector	Total (global) final demand of product i
$\mathbf{x}$	$mn \times 1$	Gross output vector	i's total sales to industries and final consumers combined
$\mathbf{A}$	$mn \times mn$	Direct requirements matrix	j's required input from i per unit produced
$\mathbf{F}$	$s \times mn$	Total emissions matrix	j's annual emissions of pollutant i
$\mathbf{S}$	$s \times mn$	Emissions coefficient matrix	j's direct emissions of pollutant i per unit produced
$\mathbf{d}$	$s \times 1$		Accumulated environmental interventions associated with the total final demand $\mathbf{Y}$
$\mathbf{I}$	$mn \times mn$	Identity matrix	
$\mathbf{H}$	$v \times t$	Original CES data	Average annual expenditures per household on product i in year j
$\mathbf{P}$	$p \times t$	Price indices	Consumer price index of product i in year j relative to the IOT base year $t = 0$
$\mathbf{G}$	$v \times p$	Price indices – CES concordance matrix	1 if CES product j is allocated to price indices commodity group i, 0 otherwise
r		Exchange rate, base year (EUR/NOK)	
$\mathbf{h}$	$1 \times t$	Number of households by year	
ε		CES deficit to IO household demand	
$\mathbf{C}$	$v \times n$	CES-IO concordance matrix	% of CES commodity i allocated to IO product j
M		Set of IO margins sectors	
N		Set of IO non-margins sectors	
$\boldsymbol{\alpha}$	$n \times 1$	Product taxes paid by product	
$\boldsymbol{\beta}$	$n \times 1$	Margins paid/received by product	
α_{i}		Tax for product i, as share of pp	
β_{i}		For $\mathrm{i} \in \mathrm{N}$: margins on product as share of pp; For $i \in \mathrm{M}$: margins received by sector i, as share of total margins to all sectors	
$\mathbf{y}^{\mathrm{NO}}$	$n \times 1$	Norwegian household final demand from the IOT	Total household demand of product i

$\tilde{\mathbf{Y}}$	$n \times t$	IO household final demand by year as estimated from CES	Total household demand of product i in year j
bp		basic prices	
pp		purchasers' prices	

By inserting this into the IO standard production balance ($\mathbf{Zi} + \mathbf{Yi} = \mathbf{Zi} + \mathbf{y} = \mathbf{x}$) which states that for each sector, total output equals sales to industries plus sales to final consumers, an expression for total output as a function of final demand can be derived:

$$\mathbf{x} = (\mathbf{I} - \mathbf{A})^{-1}\mathbf{y} \tag{2}$$

Assuming that the requirements matrix is independent of the level and composition of the final demand, equation (2) can be used to determine the gross output by sector arising from any final demand imposed on the system.

The total emissions matrix $\mathbf{F}$ can be converted to coefficient form $\mathbf{S}$ analogously as in equation (1). The vector of total environmental impacts associated with a certain final demand, representing the environmental footprint of consumption, is then simply:

$$\mathbf{d} = \mathbf{Sx} = \mathbf{S}(\mathbf{I} - \mathbf{A})^{-1}\mathbf{y} \tag{3}$$

For a further and more detailed mathematical account of IOA in general and of IO-based footprint accounting in particular, the reader is referred to (Miller and Blair 2009) and (Peters and Hertwich 2004), respectively.

In this analysis, we apply the EXIOBASE 2 MRIO database (Wood et al. 2013), which represents the global economy in 2007, distinguishing 43 countries plus an additional 5 aggregate regions constituting the rest of the world[1]. Each region consists of 163 industries and 200 products, yielding a total of 9,600 unique region-products. The very high level of product detail was the rationale behind the choice to use EXIOBASE 2 over other available global MRIO systems (see Dietzenbacher and Tukker (2013) for an overview), none of which model the Norwegian economy with more than around 60 industries/products. We deemed this factor to be more important to the present analysis than the advantage of a time series of MRIO tables offered by some other systems[2]. EXIOBASE 2 also includes an extensive set of environmental extensions.

[1] EXIOBASE 2 is the latest version of the database presented by Tukker et al. (2013) in the special issue cited in Section 1.1 (Dietzenbacher and Tukker 2013).
[2] The next update of the EXIOBASE database, scheduled to be released in 2015, will also feature a time series.

For our case study we focus on greenhouse gas emissions, aggregated to the common unit of CO_2-equivalents (CO_2e).

The Norwegian CES

The Norwegian CES is organized by Statistics Norway (SSB), and is publically available on the SSB website (SSB 2013). The survey was compiled for the first time in 1958, and from 1974 to 2009 it was conducted annually. In each survey, about 2,200 individuals were randomly[3] selected from the Norwegian population, and the households they belonged to made up the survey sample. Since 2009 a new scheme has been adopted with more comprehensive surveys with longer intervals. So far there has been one, conducted in 2012 with an original sample consisting of 7,000 households instead of 2,200. Each household participating in the survey is provided with a diary to record all their purchases over a 14-day period. The households are assigned different 14-day periods over the year in order to even out seasonal variations. Additionally, participating households are invited to an in-depth interview after the reporting period to complement the survey (Holmøy and Lillegård 2014). For surveys up to and including 2009, because of limited sample sizes the survey presented for each year is a three-year average with the previous two years' surveys; for instance the presented survey results for 2009 is in fact composed of data from 2007-2009, converted to 2009-prices.

Since 1999 the expenditures in the Norwegian CES have been classified according to the UN 'Classification of Individual Consumption by Purpose' (COICOP) classification system. Under COICOP, household consumption is classified within 12 divisions, which in turn are subdivided into groups, classes and sub-classes. This hierarchical system allows individual countries to adopt custom COICOP sets with varying levels of detail while still subscribing to a common framework. At its most detailed level the CES dataset published by SSB distinguishes 183 unique COICOP commodities (see the Supporting Information). In addition, the database contains results for households grouped according to various characteristics such as household size and income, though these breakdowns come with a somewhat reduced level of product detail in order to maintain statistical confidence.

Methodological challenges of combining CES and IO data

Combining CES data with IO tables entails some practical and theoretical challenges that must be considered.

I. An immediately apparent challenge for analysts is the use of different commodity **classification** schemes. CES data are usually available in a very detailed format, whereas

[3] From 1974-2009, persons 80 years or older were excluded from the population before the sample was drawn; in 2012 this age limit was raised to 85. Persons living permanently in institutional households (e.g. nursing homes) were also excluded.

economies can be represented in IO tables with anywhere from a couple of dozen to several hundred commodities. Even if the IO table is fairly detailed, constructing the link between CES and IO commodities can be difficult, because IO commodities are not defined with household purchases in mind. Rather, they represent economic sectors, of which potentially only a few deliver goods and services directly for final consumption.

II. Just as CES are typically more detailed than IO tables, they are typically also more up to date and often available on an annual basis. The compilation of IO tables, especially fully trade-linked multiregional tables, is time and labor consuming, which often means they are released with a **time lag** of several years and are not always updated annually. This in turn means that an analysis for a particular year might be forced to use an older IOT together with more recent CES data, which entails additional reconciliation steps and additional uncertainties.

III. IO and CES datasets typically apply different **valuation** schemes. In CES tables, purchases are reported as perceived by the consumer, for instance a purchase of a 1000 NOK[4] pair of shoes is recorded as a 1000 NOK payment in the “Footwear” commodity group. The standard in IO tables is to record the (trade and transport) margins component of a purchase separately as payments to the margins sectors. Furthermore, direct taxes on products are deducted from the purchase sum. With a tax rate of 25%, the purchase in this example would be recorded as a final demand of 800 NOK, distributed as (for example) payments of 500 NOK to the “Clothing and footwear” sector and 300 NOK to the “Trade” sector. These two valuation schemes are referred to as *purchasers’ prices* (pp) and *basic prices* (bp), respectively. The practical implication is that CES consumption data must be converted from pp to bp before the input-output analysis can be conducted. Such a conversion requires detailed information on tax and margin rates, both of which are often available in IO statistics, but also are only available at the aggregated product group level.

IV. Several factors can lead to mismatches in the data sources’ report of overall household consumption levels. Among these are different methods to estimate national totals from survey samples, as well as mismatched definitions of households and their consumption. A third and more important factor however, is a well-known problem of a significant and non-even degree of **underreporting** in CES. In other words, the sum of all expenditures according to the CES will usually be significantly less than the total household consumption as given in macroeconomic statistics. This underreporting is typically biased towards certain product categories, where the products are of such a nature as to make the respondents less likely to correctly report that particular purchase, or less likely to complete the survey at all (Mørk and Willand-Evensen 2004; Heinonen et al.

[4] Norwegian crowns

2013; SSB 2013). Examples include purchases of sweets, of alcohol and drugs, and expenses related to medical emergencies, funeral services and various infrequent purchases.

V. Though the lion's share of a household's carbon footprint is embodied in the products it consumes, there is also a significant portion which consists of **direct emissions** by the household, notably residential fuel use and tailpipe emissions from private cars. In IO systems, the households sector is usually modeled as exogenous to the industrial-economic system. In practice, this means that direct emissions are not calculated in the model, they are simply given as a static quantity. Typically, the direct emissions accounts accompanying an IO table are only provided as economy-wide totals. Thus any calculations of the direct emissions component of hhCF with any detail beyond the national household average must be added separately by the analyst.

VI. In consumer expenditure surveys, only amounts of each product consumed are recorded; no distinction is made of the share of household purchases that are **direct imports**. Though not essential, it is well known from previous MRIO analyses that emissions embodied in the same products manufactured in different countries can be widely different[5]. For this reason, an estimate of direct household imports of certain products can give non-negligible effects on results.

Reconciliation

The following is a sequential account of the practical approach taken here to reconcile the Norwegian CES with the EXIOBASE 2 database, to allow for the calculation of a carbon footprint time series for Norwegian households.

In the first step, the original CES matrix $\mathbf{H}$ showing household expenditures on v COICOP products[6] for t years is converted to constant prices of the IO base year by applying a matrix $\mathbf{P}$ of price indices by product and year. A concordance matrix $\mathbf{G}$ is required to bridge the price indices $\mathbf{P}$ from their own product classification to that of the CES. At the same time, we convert the data to the currency of the IO table using average exchange rate r for the base year, and scale it up to represent the national total by multiplying by the total number of households for each year $\mathbf{h}$ (SSB 2011). The resulting matrix $\mathbf{H}^{\mathrm{pp.0}}$ represents the CES time series expressed in constant prices (indicated by the superscript 0), and scaled up from average expenditures per household to total national household expenditures:

[5] The same of course holds true for many domestic products which fall into the same sector in an IOT but are in reality very different.

[6] Note that the number of products in the CES is in fact $\nu - 1$, the νth element is a dummy product used here to account for underreported consumption.

$$\mathbf{H}^{\mathrm{pp.0}} = (\mathbf{G} \times \mathbf{P}) \circ \mathbf{H} \times \hat{\mathbf{h}} \times r \tag{4}$$

Next, we adjust the CES data for underreporting by comparing the total expenditures in the base year, $\mathbf{H}_{t=0}^{\mathrm{pp,0}}$, to the Norwegian household final demand $\mathbf{y}^{\mathrm{NO,pp}}$ of n IO products according to the IOT:

$$\varepsilon = \sum_{v} \mathbf{H}_{t=0}^{\mathrm{pp,0}} - \sum_{n} \mathbf{y}^{\mathrm{NO,pp}} \tag{5}$$

The (assumed) CES deficit ε is attributed to underreporting. We further assume this underreporting is stable over time in relative terms, and add the corresponding amount for all years as consumption in the νth commodity group:

$$\mathbf{H}_{(\nu,\mathrm{t})}^{\mathrm{pp,0}} = \frac{\varepsilon}{\sum_{\boldsymbol{v}} \mathbf{H}_{\mathrm{t=0}}^{\mathrm{pp,0}}} \times \sum_{\boldsymbol{v}} \mathbf{H}_{\mathrm{t}}^{\mathrm{pp,0}} \tag{6}$$

The next step is to convert the data from the CES classification to the IOT classification. This requires a concordance matrix **C**, in which each CES product is mapped to one or more IO products. The tilde represents IO final demand estimated from CES data:

$$\widetilde{\mathbf{Y}}^{\mathrm{pp}} = \mathbf{C}' \times \mathbf{H}^{\mathrm{pp,0}} \tag{7}$$

where **C** is determined through some optimization process “D” seeking to minimize the difference (e.g. absolute differences or relative differences, not specifically defined here) between the estimated final demand and the true final demand for the base year 0 according to the IO tables:

$$\begin{gathered} \min D\left(\mathbf{y}^{\mathrm{NO,pp}}, \widetilde{\mathbf{Y}}_{\mathrm{t=0}}^{\mathrm{pp}}\right) \\ \text{s.t. } \mathbf{C} \times \mathbf{j}_{\mathrm{n}} = \mathbf{j}_{\nu} \end{gathered} \tag{8}$$

where **j** is a vector of ones of the indicated dimension.

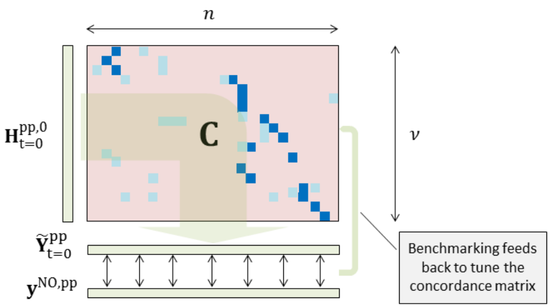

Figure 1. Schematic illustration of the optimization approach D taken in our case study to tune the CES-IO concordance matrix by benchmarking it against IO final demand. In the matrix, the blue elements represent non-zero entries; the dark blue elements represent the assumed closest fit (one per CES product). Each row sums to exactly 1. Note that some columns (IO products) are all-zero, meaning they are not directly purchased by households.

Following this, the final demand estimates must be converted from purchasers' to basic prices using vectors of margins and taxes by IO product. First, we look at the IO final demand vector in purchasers' prices ($\mathbf{y}^{\mathrm{NO,pp}}$) and the accompanying vectors of taxes ($\mathbf{a}$) and margins ($\mathbf{b}$)[7] by product. Furthermore, we assign each IO product i to exactly one of the following two sets: M for margin commodities, or N for non-margin commodities. Note that in the margins vector, we have:

$$\sum_{\mathrm{i}\in\mathrm{M}} \mathbf{b}_{\mathrm{i}} = -\sum_{\mathrm{i}\in\mathrm{N}} \mathbf{b}_{\mathrm{i}} \tag{9}$$

From these tax and margins vectors, the following set of taxes and margins shares (assumed to be constant over time) are defined for each IO product i:

$$\alpha_{\mathrm{i}} = \frac{\mathbf{a}_{\mathrm{i}}}{\mathbf{y}_{\mathrm{i}}^{\mathrm{NO,pp}}} \tag{10}$$

$$\beta_{\mathrm{i}} = \frac{\mathbf{b}_{\mathrm{i}}}{\mathbf{y}_{\mathrm{i}}^{\mathrm{NO,pp}}} \text{ , i} \in \mathrm{N} \tag{11}$$

$$\beta_{\mathrm{i}} = \frac{\mathbf{b}_{\mathrm{i}}}{\sum_{\mathrm{i}\in\mathrm{M}} \boldsymbol{\beta}_{\mathrm{i}}} \text{ , i} \in \mathrm{M} \tag{12}$$

Now, for each product i, we first remove taxes from the total purchases:

[7] To simplify the mathematical notation in this section we assume only one type of margin

$$\widetilde{\mathbf{Y}}_{\mathrm{i}}^{\mathrm{pp}^*} = (1-\alpha_{\mathrm{i}})\widetilde{\mathbf{Y}}_{\mathrm{i}}^{\mathrm{pp}} \tag{13}$$

The asterisk superscript here indicates the intermittent stage where taxes have been removed, but margins still remain to be adjusted. To arrive at basic prices then, we redistribute margins by first subtracting margin payments from the non-margin products:

$$\widetilde{\mathbf{Y}}_{\mathrm{i}}^{\mathrm{bp}} = (1-\beta_{\mathrm{i}})\widetilde{\mathbf{Y}}_{\mathrm{i}}^{\mathrm{pp}}\ , \mathrm{i} \in \mathrm{N} \tag{14}$$

Finally, the matrix $\widetilde{\mathbf{Y}}^{\mathrm{bp}}$ of total household purchases for each year in constant basic prices is obtained by redistributing the subtracted margins across the margins sectors:

$$\widetilde{\mathbf{Y}}_{\mathrm{i}}^{\mathrm{bp}} = \widetilde{\mathbf{Y}}_{\mathrm{i}}^{\mathrm{pp}^*} + \beta_{\mathrm{i}} \times \sum_{i \in \mathrm{N}} \left(\widetilde{\mathbf{Y}}_{\mathrm{i}}^{\mathrm{pp}^*} - \widetilde{\mathbf{Y}}_{\mathrm{i}}^{\mathrm{bp}} \right)\ , \mathrm{i} \in \mathrm{M} \tag{15}$$

Now, to use this in the MRIO framework, import shares and distributions must be estimated for the final demand of each product, to obtain final demand of each product from each supplying region (i.e. with mn rows rather than just n). To obtain this distribution, we use ratios from comparison of the Norwegian household column of the MRIO final demand matrix ($\mathbf{Y}_{\mathrm{NOhh}}$) against its own aggregation over supplying regions ($\mathbf{y}_{\mathrm{NO}}$), again assuming a constant relationship over time:

$$\widetilde{\mathbf{Y}}_{\mathrm{m \cdot n,t}}^{\mathrm{MRIO,bp}} = \widetilde{\mathbf{Y}}_{\mathrm{n,t}}^{\mathrm{bp}} \times \frac{\mathbf{Y}_{\mathrm{m \cdot n,NOhh}}}{\mathbf{y}_{\mathrm{n}}^{\mathrm{NO}}} \tag{16}$$

The subscripts in equation (16) above indicate final demand on product n in region m in year t.

Results

Norwegian household CF, 2012

After adjustments for underreporting, the average Norwegian household spent 511 thousand NOK on consumption of goods and services in 2012, carrying a total carbon footprint of 22.3 tCO_2e/hh. The average CF *multiplier*, that is, the carbon footprint embodied in each NOK of final consumption, was 44 gCO_2e/NOK. The differences among CF multipliers of individual COICOP commodities are large, however. In Figure 2, the Norwegian hhCF is broken down by the twelve COICOP divisions, with the footprint of each visualized as a product of annual expenditures per household and the average CF multiplier of the division. The overall hhCF is dominated by transport, housing and food, but through different mechanisms: While housing contributes significantly mainly from its large share of the overall household budget, the carbon footprint related to transport is almost double, due to the fact that every NOK spent on transport led to emissions of 95 gCO_2e, compared to only 29 gCO_2e/NOK for housing.

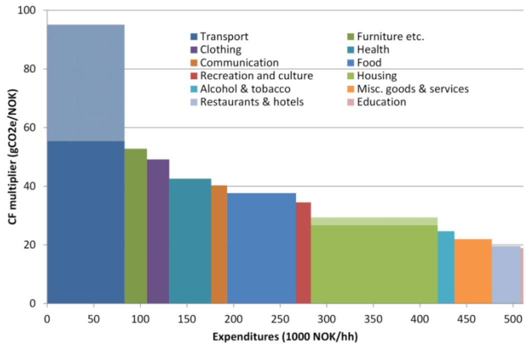

Figure 2. Norwegian household expenditures and the average carbon footprint intensities of each COICOP division, 2012. The lighter shaded parts of the 'Transport' and 'Housing' columns constitute direct emissions by households.

Of the total household CF, direct emissions by households constituted 16.5% or 3.7 tCO_2e/hh. Direct emissions in the housing category are very low in Norway compared to most other industrialized countries as Norwegian households predominantly use electricity for cooking and space heating. The fact that the Norwegian electricity mix is largely based on hydropower serves to further lower the overall CF intensity of housing.

The result that food, transport and housing are the consumption groups contributing the most towards the total hhCF is in agreement with the findings of several previous studies (Tukker and Jansen 2006; Hertwich and Peters 2009; Tukker et al. 2011). Transport is relatively more important in Norway than in other countries for the reasons mentioned above, combined with several factors that serve to increase the travel distances of Norwegians, including low population density, limited rail network, and high affluence. In recent years in particular, there has been a tremendous increase in air travel by Norwegians (Denstadli and Rideng 2012).

A key benefit of MRIO analysis is the ability to quantify how consumption in a certain region leads to environmental effects elsewhere. The trend of increased globalization of supply chains has caused consumers in developed countries, although sustaining high environmental footprints, to be geographically separated from the effects of many of the environmental burdens of consumption, which may occur far upstream. A substantial share of the processing and manufacture of final and intermediate products ultimately delivered for consumption by Norwegian households has been shifted to developing countries, notably China: In 2012, 11% of the emissions contributing to the Norwegian hhCF took place in China (Table 2). In fact, while the majority of the value added embodied in Norwegian household consumption was generated domestically, the share was merely 37% for embodied carbon emissions[8].

Table 2. Consumption-based account of Norwegian household purchases: regional distribution of effects.

	Value added generation	Greenhouse gas emissions
Norway	70%	37%
EU27	15%	20%
USA	5%	3%
China	1%	11%
Other	9%	28%

The big geographical difference observed here between value added and greenhouse gases embodied in final consumption is due to the fact that they largely accumulate at different stages of the supply chains. Whereas embodied CF is typically associated with emissions in secondary (manufacturing) sectors, the majority of value added is generated closer to the end user. Grouping the MRIO sectors in clusters representing primary, secondary, and tertiary sectors showed emissions distributed 10%–49%–22%, respectively, while value added was distributed 5%–24%– 71%.

Across the population, total CF per household increased rapidly with income (Table 3). In the most affluent decile, the average expenditures per household were 4.1 times those in the poorest decile, while their average carbon footprints were 5.1 times higher. Overall, we estimate an expenditure elasticity of carbon footprints of 1.14 (R^2 = 0.999). This was due to the fact that affluent households spent relatively more on the carbon-intensive commodities; in fact, the top three elasticities in Table 3 coincide with the three most CF intensive commodities in Figure 2 (transport, furniture, and clothing).

Table 3. Total expenditures and carbon footprint by COICOP division, 2012. Results for all households, as well as by expenditure levels. The two rightmost columns show expenditure elasticity of CF and associated R^2 values. The CF of each COICOP division i (as well as the total) is regressed to $CF_i = ax^{\epsilon_{d,i}}$, where x represents total expenditures per household.

	All hh	Decile 1	Deciles 2+3	Deciles 4+5	Deciles 6+7	Deciles 8+9	Decile 10	ϵ_d	R^2
Exp. per hh (10^3 NOK)	*511*	*229*	*342*	*410*	*535*	*678*	*949*		
CF per hh	**22,170**	**8,557**	**14,081**	**16,964**	**23,448**	**30,207**	**43,524**	**1.14**	**0.999**

[8] Prell et al. (2014) compared value added and SO_2 embodied in US consumption and found similar results.

01 Food	3,018	1,390	1,862	2,386	3,376	4,145	5,209	0.98	0.986
02 Alc. & tobacco	333	198	257	265	356	412	551	0.72	0.983
03 Clothing	1,162	529	536	771	1,152	1,730	2,717	1.26	0.932
04 Housing	4,088	1,744	2,713	3,720	4,215	4,879	8,041	1.02	0.976
05 Furniture etc.	1,280	408	788	983	1,325	1,763	2,666	1.29	0.994
06 Health	632	421	470	581	679	758	915	0.57	0.978
07 Transport	7,864	1,776	5,083	5,569	8,421	11,335	15,923	1.48	0.955
08 Communication	589	457	383	434	640	762	995	0.65	0.791
09 Recreation	1,883	1,091	1,139	1,242	1,957	2,596	3,884	0.97	0.906
10 Education	26	26	13	17	24	37	51	0.70	0.475
11 Restaurants	484	212	316	383	471	676	937	1.05	0.995
12 Misc.	811	305	523	614	832	1,116	1,635	1.17	0.998

The two lowest-income groups in Table 3 exhibit some interesting differences. The households in the poorest decile of Norwegian households typically do not own a car, nor do they own their house, both in contrast to the large majority of Norwegian households overall (see tables 10444 and 10448 in (SSB 2012)). For this reason, the hhCF in the lowest-income decile is disproportionally small for these consumption categories. The reduced transport CF in particular contributes to a low CF per NOK spent overall for the lowest-income cohort. Lower car and house ownership rates leave more income disposable for other consumption, which reduced this effect to some degree: For the COICOP divisions Clothing (03), Communication (08), and Recreation (09), the CF of decile 1 is similar or even higher than those of deciles 2-3.

The observed CF elasticity of expenditures of 1.14 is an unexpected result, in contrast with the findings of Herendeen (1978) and most studies since, which have generally found $\epsilon < 1$ (Lenzen et al. 2006). Two limitations of the present cross-sectional analysis could potentially affect our result: First, the CES broken down by income deciles as published by SSB features less product detail than the full survey due to reduced sample sizes, and the deciles are aggregated so our elasticity calculation is performed over only six income groups. Furthermore, the emissions model applied here does not distinguish luxury products, a luxury car at twice the price of an average car is for example assumed to carry twice the carbon footprint (Hertwich 2005)[9]. The same limitations, however, also affect most other studies reviewed by Lenzen et al. (2006). On the other hand, some important underlying factors support the result of a high elasticity. First, in stark contrast to most countries, emissions from direct household energy use are almost

[9] Girod and de Haan (2010) take a different assumption, converting monetary expenditures to consumption of functional units to account for quality changes, which is also not without problem. They find that a significant portion of the expenditure increase with household income is attributable to increased price per functional unit consumed; however it is not clear how the environmental impact scales with the price of the product.

negligible in Norway, where energy for cooking and space heating is overwhelmingly based on electricity from hydropower. Direct energy use in households is generally inelastic; this has been an important reducing factor for the overall CF elasticity of income in previous studies (Vringer and Blok 1995; Lenzen et al. 2006; Jones and Kammen 2011). Second, consumption in the travel and transport category is associated with high carbon intensities and high elasticities; in a recent case study on German consumers, Aamaas et al. (2013) found much higher climate impact from travel for the higher income cohorts, with an elasticity of 1.17 for air travel.

Footprint development 1999-2012

The time series analysis showed an increase in the carbon footprint of Norwegian household consumption from 1999 to 2012 by 25%, corresponding to an average of 340 kgCO_2e per year. Over the same period consumption volumes rose by 26%. Since our analysis is based on a detailed consumption time series coupled with a static technology-emissions model, we expect the overall carbon footprint development to match that of the real expenditures fairly well; however, the detailed results show the increase was neither linear nor monotone (Figure 3). Much of the growth occurred over three years, from 2004 to 2007, while two years (2003 and 2008) had slightly reduced CF compared to the previous year.

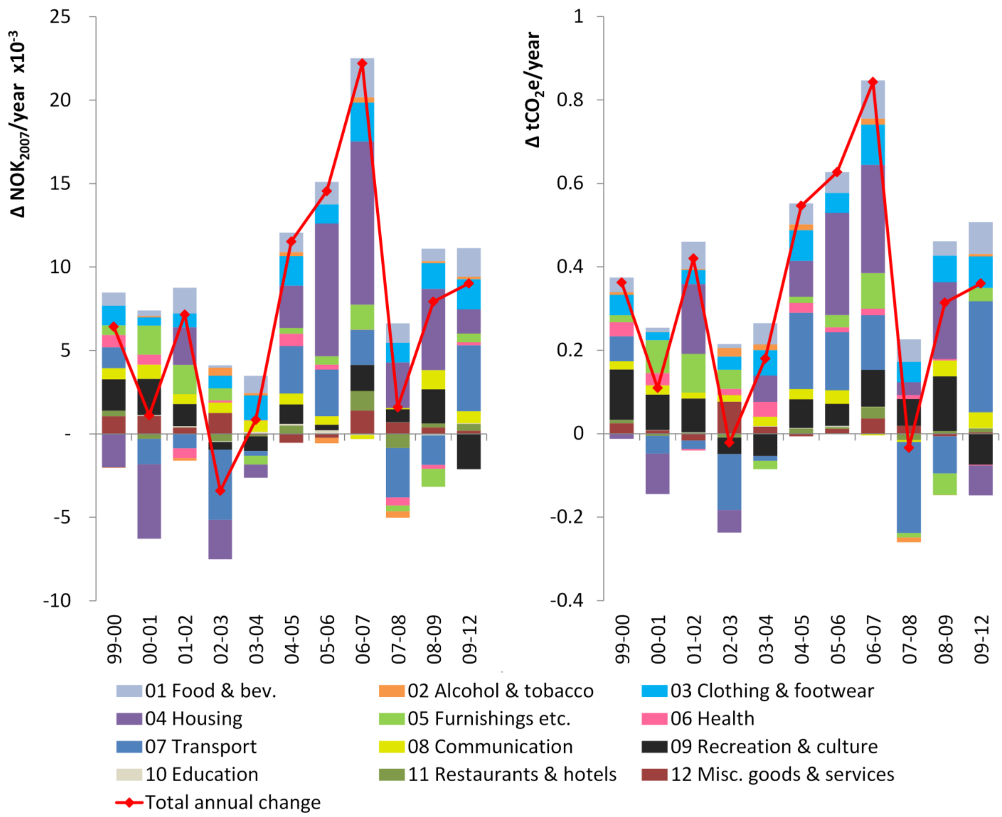

Figure 3. Annual change in expenditures and their associated carbon footprints by COICOP division, 1999-2012. Changes in the final three-year period were divided by three to get average annual change. Average household size fluctuated from 2.19 p/hh in 1999 through a maximum of 2.24 p/hh in 2002 to a minimum of 2.12 p/hh in 2012.

The CF of the three consumption categories highlighted here and in previous studies as the main contributors towards the total hhCF (food, shelter, mobility) all grew significantly over the period. Despite an overall standstill in consumption levels over the first five years, the CF of **housing** expenditures still increased by a total of 8%. This was the result of a shift in real housing expenditures from rent towards material goods such as furniture as well as tools and materials for renovation. In the remainder of the period, particularly from 2009 to 2012, this effect was reversed, leading to growths in both expenditures and carbon footprints.

A considerable part of the overall increase in the CF of housing may have come from holiday homes. Since the end of World War II the trend has been a steady increase in living area per person; however this has been halted over the last decade or so due to increased urbanization, combined with rapidly increasing house prices (Vestlandsforsking 2012). The share of Norwegian households who also own a vacation home or cottage has remained stable at 21-24% over the last three decades (SSB 2013). At the same time however, the average vacation home has grown bigger, and the traditional primitive cottage is increasingly being replaced by holiday homes of similar or even higher standards than regular homes. As of 2007, a newly constructed holiday home was on average as large as a newly constructed regular residence; by comparison, as late as the mid-eighties the difference was almost a factor of three (SSB 2009).

Due to its overall high CF intensities (Figure 2), expenditure shifts within the **transport** category explain much of the differences observed between the developments of hhCF versus real consumption over the period. The changes in consumption over the years have for a large part consisted of car sales, which showed significant fluctuation over the period (SSB 2014a). As could be expected, total fuel consumption, which contributes significantly more to the overall emissions, remained stable by comparison. Aviation contributed increasingly to the overall transport CF of Norwegian households over the period. The number of flights per person grew rapidly; in particular, the number of flights to/from foreign destinations were almost doubled in the decade from 2001 to 2011 (Vågane 2012). Still, the CF of air travel of 623 kgCO2e/hh is likely too low, due to the difficulty of accounting of emissions from international transport in national emissions accounts (Arvesen and Hertwich 2007).

The hhCF of **food** and non-alcoholic beverages grew steadily over the years, with no major fluctuations. Overall, the CF of food and beverages grew by 33%. The CES dataset showed a significant increase in meat consumption per capita over the period: in physical terms this was increased by 13%. At the same time, consumption of staple foods such as potatoes and bread were reduced by 39% and 8%, respectively (SSB 2013). Recent statistics on the development of Norwegian household diets estimates that the gross (wholesale) annual meat consumption was increased by 23 kg per person between 1989 and 2008 (Johansson 2011).

The results suggest that the increase in CF from food consumption was not purely an effect of shifts in dietary preferences, however. Overall, the data show a tendency of increase in volumes purchased: Real expenditure levels increased in 54 out of the 62 product classes included within the Food and beverages division, and reductions in the remaining eight had little effect on the total. As the overall food consumption levels per person are fairly constrained, this suggests an increase in food waste by households[10].

The dramatic increase observed in the CF of **clothing** and footwear warrants further comments. Though a general shift towards the high-end segment has quite possibly led to an

[10] Household sizes must also be taken into consideration, however this in fact showed a slight decrease, from 2.19 p/hh in 1999 to 2.12 p/hh in 2012.

overestimation of this footprint, it is clear that there has been a large growth in the consumption of clothing also in volume terms. According to the CES time series, the share of the Norwegian household budget spent on clothing has remained more or less constant. In current prices, household expenditures on clothing increased by 57% while the price of clothing was decreased by 48%, suggesting a rough tripling of consumption levels in volume terms. SSB import statistics, however, show a more modest growth of around 40% in physical terms (SSB 2014b). One reason for this discrepancy could be a shift towards increased purchases of luxury brands, not fully captured by the consumer price index.

The ceteris paribus assumption implied in the present analysis might entail substantial errors for certain product types, as changes in technology or international trade and consumption patterns can vary significantly over a period of 13 years. Hence it must be stressed again that the temporal development observed are from changes in consumption volumes and patterns alone. Previous structural decomposition analyses have found that technological improvements have generally led to reduced carbon emissions intensities, but that these have not been sufficient to offset the emission growth caused by increased affluence (Peters et al. 2007; Guan et al. 2008).

Discussion and Conclusions

Despite widespread media attention and public concern about climate change prospects, the carbon footprint of Norwegian households increased across all COICOP divisions from 1999 to 2012. Fundamentally, this was related to the sustained growth in the real income of Norwegian households seen over the past decades, particularly since the turn of the millennium, which for the most part was realized as increased consumption in general (Vrålstad and Melby 2009).

In order to lessen the environmental burdens of private consumption, a solid and detailed understanding of the underlying links between consumption and overall impacts is required. With the recent efforts to construct databases that are economically, environmentally and geographically detailed, up-to-date and reliable, MRIO analysis remains the best suited tool for consumption-based assessments of household environmental impacts including supply-chain effects. Still, pure IO-based assessments have thus far been of limited practical value for specific policy design due to a lack of detail. The approach taken here to combine consumer expenditure surveys with input-output models provides a straightforward solution–albeit partial–to the traditional IO challenges of product detail and timeliness.

Detailed temporal analyses of household carbon footprints have a significant potential for informing the public debate and policy on climate change mitigation. The present analysis highlighted the diversity in the set of household activities that constitute the overall Norwegian household carbon footprint, with many of the COICOP divisions contributing significantly towards the total. This reflects the pervasiveness of carbon emitting processes in society, suggesting the need for large-scale emission abatement strategies to be economy-wide and comprehensive, aiming for overall footprint reduction through a combination of a series of contributing “wedges” across economic sectors rather than “silver bullet” strategies.

In the interest of reducing the carbon footprint of household consumption, two key challenges should in the authors’ view be the focus of future studies in this vein. First, further efforts should be made to establish a commonly accepted standard framework for environmental footprint analyses using detailed local data combined with comprehensive global trade models. Such a framework should be straightforward enough to allow and encourage analyses also by non-experts of IOA, in the interest of promoting consumption-based accounting of impacts to complement the traditional approach.

Second, a significant limitation to the analysis undertaken here remains. In order to perform the environmental assessment of the household expenditures as described in the CES, the data must be converted to fit the input-output system. This aggregation of detailed CES purchases into a usually limited number of relevant IO products entails a considerable loss of detail. While the method used here allows for comparison over time including structural shifts across IO products, there is no way of distinguishing environmental effects of various CES products allocated to the same IO sector. This can be addressed by moving towards hybrid models that capitalize on product-specific results from life cycle databases. This has the potential to greatly improve the ability to distinguish between functionally similar but environmentally different commodities. Still, the lack of an overarching standard for such hybrid models, potentially leading to a range of models using different data and assumptions which may produce results that are more or less in conflict, may hamper the acceptance of such analyses as a basis for policy.

The work presented here is intended to contribute to the expansion of the knowledge base required for analyses of life-cycle environmental impacts of specific purchases and activities at the level of individual households. The transition to a sustainable society will require a multifaceted strategy, with regulations and incentives directed at individual consumers as well as at producers and emitters. With further improvements in the representation of household purchases, multiregional input-output analysis offers significant potential as a tool for analyzing environmental impacts of consumption. Given the existence of up-to-date and detailed consumer expenditure surveys in most countries, the potential gains of combining CES and IOA for informing national environmental policies are large.

Supporting Information

Detailed CES data; CES and MRIO product classifications.

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Appendix E: Supporting Paper AI

Ewing, B.; Hawkins, T.; Wiedmann, T.; Galli, A.; Ercin, A. E.; Weinzettel, J.; Steen-Olsen, K., Integrating Ecological and Water Footprint Accounting in a Multi-Regional Input-Output Framework. *Ecological Indicators* **2012,** *23*, 1-8.

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Integrating ecological and water footprint accounting in a multi-regional input–output framework

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A B S T R A C T

Carbon, ecological, and water footprints (CF, EF, and WF) are accounting tools that can be used to understand the connection between consumption activities and environmental pressures on the Earth's atmosphere, bioproductive areas, and freshwater resources. These indicators have been gaining acceptance from researchers and policymakers but are not harmonized with one another, and ecological and water footprints are lacking in their representation of product supply chains. In this paper we integrate existing methods for calculating EF and WF within a multi-regional input–output (MRIO) modelling framework that has already been successfully applied for CF estimation. We introduce a new MRIO method for conserving the high degree of product detail found in existing physical EF and WF accounts. Calculating EF and WF in this way is consistent with the current best practice for CF accounting, making results more reliable and easier to compare across the three indicators. We discuss alternatives for linking the MRIO model and the footprint datasets and the implications for results. The model presented here is novel and offers significant improvements in EF and WF accounting through harmonization of methods with CF accounting, preservation of product-level detail, comprehensive inclusion of sectors of the global economy, and clear representation of flows along supply chains and international trade linkages. The matrix organization of the model improves transparency and provides a structure upon which further improvements in footprint calculation can be built. The model described here is the first environmentally extended MRIO model that harmonizes EF and WF accounts and aligns physical unit data of product use with standard economic and environmental accounting.

1. Introduction

Leaders and decision-makers face the challenge of interpreting a wide variety of information from a broad range of sources to inform policy choices and investment decisions. They rely on selected indicators that are easy to understand and communicate but which tend to over-simplify or omit some factors involved in complex systems. This can skew decision-making toward greater consideration of certain factors at the expense of others. For example, issues such as societal well-being or environmental integrity are often under-emphasized compared to the measurement of gross domestic product (GDP) as a widely adopted indicator of the performance of an economy.

In the attempt to move its policy process "Beyond GDP" and couple economic indicators with indicators of social and environmental issues, the European Union (EU) and other regions are seeking indicators more closely related to societal goals such as improving quality of life and well-being and minimizing environmental impacts (European Commission, 2009). For environmental issues, three important indicators are the carbon, ecological, and water footprints (CF, EF, and WF), which have been recently grouped into a "footprint family" suite of indicators (Galli et al., 2012). Each footprint indicates a particular class of impacts associated with the activities of an individual or group. Potential

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for global warming is indicated by the CF (Wiedmann and Minx, 2008; Wright et al., 2011), effects on water availability and quality in terms of total volume of freshwater consumed or polluted are indicated by the WF (Hoekstra et al., 2011), and appropriation of the regenerative capacity of the biosphere expressed in global average bioproductive hectares are indicated by the EF (Rees, 1992; Wackernagel and Rees, 1996). The measurement of regenerative capacity by the EF is unique among many indicators, including the CF and WF, since human demand for biological resources is directly compared with the biosphere's capacity.

Decision makers have become increasingly interested in understanding the impact of current life-style and consumption patterns upon global ecosystems (see e.g. Kastner et al., 2011; Kissinger and Rees, 2010). This change in policy focus has brought about an increased interest in the use of footprint indicators for policy assessment. In contrast to the territorial, production-based approach used for international agreements such as the Kyoto Protocol and adopted by most environmental statistics, footprint indicators account for impacts induced by consumption, including production as well as trade flows (Galli et al., 2012).

Historically, national EF and WF accounts have used an approach based on physical flows, which allow for the incorporation of a large number of commodities tracked in UN production and trade statistics. However, a number of assumptions had to be made with respect to national and international production and supply chains that diminished the value of using highly disaggregated physical datasets. For example, the omission of trade in services (especially transport services) and upstream impacts of energy goods (fossil fuels) as well as the use of generic embodied energy factors were previously identified as limitations in the EF approach (Wiedmann, 2009a).

In recent years multi-regional economic input–output (MRIO) models have improved to the point they can more easily be applied to environmental accounting frameworks and address limitations associated with purely physical datasets (Wiedmann et al., 2011a). Environmental MRIO modelling has already been used to inform discussions in global climate and resource use policies about allocation of responsibility (Hertwich and Peters, 2009; Peters and Hertwich, 2008; Wiedmann, 2009b). EF and WF practitioners have been slower in adopting a national IO or global MRIO approach primarily due to concerns about the loss of detail in the underlying accounts (Wiedmann et al., 2007). The coarse grouping of economic sectors introduces the potential for misrepresentation of EF and WF of individual products (Lenzen, 2001). These problems are primarily caused by the grouping of unlike production processes or inadequate geographic distinction (Andrew et al., 2009; Lenzen, 2011; Su and Ang, 2010; Su et al., 2010; Williams et al., 2009; Feng et al., 2011).

In this paper we demonstrate how, for the first time, a high level of detail in commodity classification can be maintained when integrating existing physical unit production and trade datasets which underlie EF and WF accounts with the more complete, but less specific, MRIO framework. By bringing together the detailed EF and WF accounts with a MRIO model, our methodology allows for the calculation of direct demand for land and water at the level of specific products as well as calculation of indirect impacts throughout the supply chains of product sectors. Benefits of this approach include the harmonization of the CF, EF, and WF accounts, alignment of physical unit data of product use with standard economic and environmental accounting, allocation of intermediate product trade to final consumption activities, and calculation of service sector footprints that were previously excluded from EF and WF accounts. These improvements of detail, consistency between accounting methods, alignment of these accounts with international standards, and allocation final demand clearly increase the usefulness of the approach adopted in this work.

The paper proceeds with Section 2, which sets out the theoretical foundations of an integrated MRIO-Footprint (MRIO-F) model. This is followed by a Section 3 where we elaborate on the strengths and weaknesses of the approach, consider uncertainty, and discuss policy applications. Section 4 concludes. Supporting Information includes a detailed explanation of data and calculations as well as a numerical example of the suggested MRIO-F model.

2. Method: an integrated MRIO-Footprint model

The incentive for the integrated MRIO-F model presented in this paper is to maintain the product detail found in traditional EF and WF accounting while avoiding the large data burden that would be required for full disaggregation of the MRIO sectors. This is possible through the addition of a satellite account and complementary matrix-based physical model to the monetary MRIO model. By incorporating a large amount of additional detail related to primary products such as crops, forestry products, livestock, and fish, and by tracking these products in physical units, the MRIO-F model enables calculation of direct footprints at the individual product level.

We begin by presenting a basic method which allows the assignment of footprints to detailed physical demand, but which does not yet allow the user to distinguish the indirect contributions of specific physical products within MRIO sectors. This is followed by a modification where more detailed satellite accounts are incorporated to allow the contributions of more specific physical products to be distinguished.

For definitions of ecological and water footprint accounting and related background information the reader is referred to Supporting Information of this article.

2.1. *Accounting conventions and method for calculating ecological and water footprint*

For transparency it is best to maintain a harmonized accounting of the EF and WF associated with production processes in each region. To do this we begin by arranging the available production data in physical units into a matrix (P) with primary products in rows and producing countries in columns. Henceforth we refer to *primary biological products* simply as *products*. Next, we define two 3-dimensional data structures to store the data required to estimate the EF (L) and WF (W) associated with the production processes in each country. These have dimensions of products by producing country by bioproductive area or water type. Each value represents a portion of the direct EF or WF. The bioproductive areas associated with EF are cropland, pasture/grazing land, forest, infrastructure, fishing grounds, and carbon uptake land (Ewing et al., 2010). The types of water appropriation are green, blue, and grey. Blue refers to surface and ground water, green refers to rainwater stored in the soil as soil moisture, and grey relates to polluted water (Hoekstra et al., 2009, 2011).

We define a matrix describing an arbitrary demand for a given product produced in a given country (D^*), which is provided as an input to the model. Using these matrices we can calculate direct bioproductive area appropriation (L^*) in hectares (ha) by land type and WF (W^*), in m^3, associated with the arbitrary *physical demand* (D^*), in tonnes.

$$l^*_{i,j,k} = l_{i,j,k} p^{-1}_{i,j} d^*_{i,j} \quad \text{and} \quad w^*_{i,j,k} = w_{i,j,k} p^{-1}_{i,j} d^*_{i,j} \tag{1}$$

In these equations, $l_{i,j,k}$, $w_{i,j,k}$, $p_{i,j}$, $d^*_{i,j}$, $l^*_{i,j,k}$, and $w^*_{i,j}$ are elements of L, W, P, D^*, L^*, and W^* respectively and *i*, *j*, and *k* are indices describing product, country, and type. Note that we use asterisks to distinguish model inputs and outputs from the underlying dataset. The actual direct EF in global hectares (gha) can be derived from L^*

by summing the components of each land type. Likewise, the direct WF can be obtained by summing the types of water appropriation in W^*.

2.2. Calculation of the total ecological footprint and water footprint per broad sector in the MRIO-F model

The simplest approach to integrate footprints into a satellite account is to allocate all footprints of primary products to their producing MRIO sectors. There are three increasingly complex options: (1) vectors of total EF and WF with dimensions 1 by monetary sector, (2) matrices such that rows describe the various types of EF and WF, or (3) 3-dimensional matrices including the contribution of each physical product. The model structure for options 2 and 3 are depicted in Fig. 1a and b respectively. In practice, the first two options can be created by aggregating the matrices of option 3.

We proceed with option 3 by using the EF (L) and WF (W) 3-dimensional data structures previously described. First the number of columns in L and W must be increased to include not only producing countries but also the producing industry/process by assigning detailed products to their producing MRIO sector. These data structures are placed below the intermediate transactions matrix of the IO model. The result is the option-3-matrix where rows represent physical products by country, columns represent sectors of the IO model and the country where they are located, and layers represent EF or WF type.

Option 2 is achieved by summing across products, leaving matrices of EF and WF by MRIO sector in each country and footprint type. These option-2-matrices, referred to as L^M and W^M, are part of the calculation of indirect EF and WF. The superscript M is used to distinguish these matrices from those with columns representing only total production by country.

2.3. Calculation of indirect ecological and water footprints

To calculate the indirect effects associated with the demand D^* specified in terms of physical production we must translate D^* into a final demand column vector consisting of monetary values corresponding to consumption within each product sector in each country of the MRIO model. This can subsequently be used to 'drive' the MRIO model with an equivalent monetary demand ($y^{*,D}$). Thus D^* is transformed via a three-step process by which

(1) D^* values are converted from physical production values to monetary values using a matrix of prices (Π),
(2) D^* rows are transformed from physical commodities to the products of the monetary IO model using a correspondence matrix ($C_{M,P}$), and
(3) the matrix is converted into a column vector via a matrix conversion process (Ψ) which essentially places each column below the previous one.

This transformation is calculated as

$$y^{*,D} = \Psi(C_{M,P}(\Pi \cdot D^*)) \tag{2}$$

A more complete discussion of the construction of the price matrix, Π, and a discussion of how demand specified in terms of products and their locations of consumption can be transformed to an equivalent D^* is provided in Supporting Information.

Indirect land use is then calculated as follows

$$L^{*,M} = (L^M\hat{x}^{-1})(I - A)^{-1}Ay^{*,D} \tag{3}$$

where $L^{*,M}$ is the resultant indirect EF, L^M is the bioproductive hectares per monetary sector, x is the total output per monetary sector, I is the identity matrix, and A is the direct requirements or the technical or IO coefficients matrix – derived by normalizing the transaction matrix by total sector output (Miller and Blair, 2009). $L^{*,M}$ and L^M both have dimensions of type of bioproductive area appropriation (i.e. cropland, pasture, forest, built up land, etc.) by sector and country of the MRIO model. These matrices differ from those described in the previous section because they are related to the sectors and monetary units used in the MRIO model and are thus distinguished with the superscript M.

On the right side of Eq. (3), the first term represents the bioproductive area and water appropriation of individual economic sectors per monetary output of these sectors, the second term is the Leontief Inverse specifying the total requirements per unit output, and the third term provides the direct requirements needed to meet the originally specified final demand. Recall that $y^{*,D}$ represents demand for the same products as D^* but now in monetary terms and aggregated to MRIO commodities. Thus the direct requirements associated with $y^{*,D}$ – represented by the term $Ay^{*,D}$ – are used as the final demand in the model to prevent double counting of direct EF, which has already been accounted for using the physical model described in Sections 2.1 and 2.2.

Similarly, indirect WF can be calculated as

$$W^{*,M} = (W^M\hat{x}^{-1})(I - A)^{-1}Ay^{*,D} \tag{4}$$

where $W^{*,M}$ is the resultant indirect WF, W^M is the total WF per monetary sector, and the rest of the variables follow the same interpretation used for the indirect land use calculation above.

2.4. Hybrid method for calculating indirect ecological and water footprint

Up until this point, we have described a basic method for calculating indirect EF and WF. However, the method described in the previous sections does not take full advantage of the detailed EF and WF accounts which relate directly to the products measured in production mass rather than the broader monetary product sectors. The method described below represents a new approach to maintain the product detail using a hybrid method for calculating indirect EF and WF.

The simplest potential hybrid option is to perform calculations using satellite accounts disaggregated by physical products (as described in option 3 in Section 2.2). The benefit of this approach is that product detail can be added with minimal additional effort. The drawback is that individual products are only associated with their producing MRIO sector. The use of individual products is still determined by the associated MRIO sector. Thus, for example, within the *cereal* grains sector, the footprints of different grains for animal consumption, human consumption, and inputs to other processes would all be averaged according to monetary flows of the corresponding sector.

The most challenging potential hybrid option is to use the physical data to add detail to the IO use matrix so the detailed EF and WF accounts can be related directly to the more detailed breakdown. The advantage of this option is that it results in a two-layer dataset for sectors with available mass unit data, where one layer contains monetary transactions and the second contains the associated physical flows. The disadvantages of this option relate to current data limitations required to fully disaggregate a sector in the IO table. Disaggregating an IO table requires additional detail both on the row, which describes the specific use of a product, and on the column, which describes the inputs required for production of a product. Information for disaggregating rows can often be found in product-flow accounts. Information for disaggregating columns (including the environmental extensions data) is more analogous to that found in life cycle assessment (LCA) studies (see e.g. Wiedmann et al., 2011b). However, neither product flow nor LCA data are available for all EF or WF product categories and across all countries.

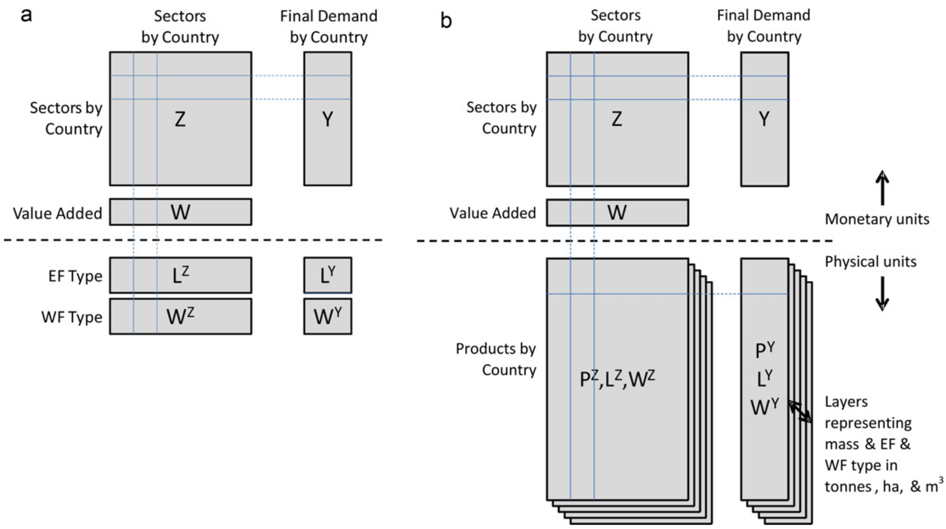

Fig. 1. Simplified (a) and hybrid (b) accounting frameworks for integrating physical and monetary data for calculating EF and WF in a MRIO-F model.

Even if they were, disaggregating IO sectors is a resource-intensive and time-consuming task; for instance, if the sectors of all countries were disaggregated into the required level of detail, the matrix would exceed the possibility to be inverted by normal computers due to the immense size of the dimensions.

We have chosen a compromise. When additional information is available along certain product rows, but little or no additional information is available to inform new sub-sector columns, it can be simpler to create a more detailed use account in mass flows alongside the monetary use account but without altering the original structure of the monetary account. The benefit of such an account is that one can take advantage of the additional detail contained in the mass-unit product account while maintaining transparency and integrity in the less detailed monetary dataset.

Adopting this approach, we create an account of the use of physical products (P_{use}), which is written below the monetary use table where rows are the physical products and producing country and columns are the sectors and consuming countries as defined in the monetary use table. Fig. 1b depicts the structure of the combined system of monetary and physical use tables. Entries in this table represent the use of a physical product by the industry, service sector, or final demand category.

The physical use table can be used to calculate indirect EF and WF in a stepwise approach, by first calculating the use of physical products associated with a given demand.

$$p^{use,*} = (p^{use}\hat{x}^{-1})[(I - A)^{-1}y^{*,D}] \quad (5)$$

and then calculating the EF and WF associated with each physical product (i) and producing country (j).

$$l^*_{i,j,k} = l_{i,j,k}p^{-1}_{i,j}P^{use,*}_{i,j} \quad \text{and} \quad w^*_{i,j,k} = w_{i,j,k}p^{-1}_{i,j}P^{use,*}_{i,j} \quad (6)$$

Here l is an element of the 3-dimensional data structure containing EF, L, and w is an element of the WF, W, data structure. The index k describes the components of EF and WF such as grazing land and blue water, respectively.

Generally, not all of the data required to fully specify the use of physical products along rows are available. Nonetheless, any additional detail improves on the implicit assumption that all detailed products follow the same use pattern as the general MRIO sector.

The physical use table (P_{use}) can be estimated from the adjusted exports matrix ($M^{adj}_{tot\,exp}$) defined in Section 2.5. The quality or level of detail of the estimated physical product use table can be adjusted depending on the amount of additional information collected to inform the distribution of products across columns. At the simplest level, values in a column of the exports matrix are allocated across users, tracked along columns, by allocating each entry according to shares of the monetary transactions of the associated product involving the industry, service, and final demand columns of that country in the monetary use table. Additional data can be added in a step-wise manner. For example, when available, FAO statistics can be used to adjust consumption of physical products by households or specific sectors such as *livestock cattle*. Instances where it is known that no flow takes place can be specified uniquely for each physical product. Trade data can be used to separate domestic consumption from exports. Physical product flows are then allocated across the appropriate monetary transactions. In a more sophisticated implementation, additional constraints such as prices or certain known product flows could be imposed and the system resolved by optimization with respect to a target such as least deviation between the monetary rows and the associated physical products combined with prices.

Note the relationship between footprints and sectors is altered slightly in this method of calculation as the EF and WF are associated with product flows and product flows are pushed forward to the sector by which they are consumed. This is in contrast to the method described in Section 2.3 where EF and WF are tied directly to the producing MRIO sector.

A numerical example is available in Supplementary Information to illustrate how additional information describing interactions between physical sectors can be incorporated into the MRIO model.

2.5. *Potential for utilizing additional input structure data associated with producing physical products*

In Section 2.4 we described how physical accounts could be used to add detail to an MRIO-F model without going as far as to fully disaggregate the monetary MRIO sectors. That approach relies on adding product-level detail in a satellite account. However, existing national EF and WF accounts include limited information regarding the input structures associated with producing certain physical products. Ideally, this information could be incorporated so the integrated MRIO-F model takes advantage of all detailed data available.

For example, national EF and WF accounts include the use of crop products used as feed for specific livestock types. The methods previously described do not allow for exchanges between the detailed physical sectors. Thus, inputs of crops to livestock would be aggregated in MRIO sectors such as *cattle*, *pigs*, or *milk* rather than more detailed categories provided by the FAO or other sources. Similarly, some products of interest such as farmed fish are difficult to place within multiple MRIO sectors without using the additional physical structure data.

In these cases a matrix describing the use of *primary* products (such as sorghum) to produce *secondary* products (such as beef cattle) can be created. This matrix has dimensions of primary products and country by secondary products and country. Trade data would be used to resolve *primary* product imports used to produce *secondary* products. This matrix would be used to convert final demand for physical *secondary* products into complimentary final demand vectors for MRIO sector and physical *primary* product output.

This approach is complicated and care must be taken to avoid double counting by additional adjustments. If using the most challenging potential hybrid MRIO option described in Section 2.4 where the satellite account is defined in terms of physical product use by MRIO sector, the direct footprint can be calculated based on the physical product inputs and the indirect footprint calculated by driving the MRIO model with a demand equal to the direct requirements associated with the secondary product(s) demanded. If using the simplest potential hybrid MRIO option described in Section 2.4 where the satellite account is defined in terms of physical product production or footprint driven directly by MRIO sector it is necessary to make an additional adjustment. This adjustment is trivial if: (1) the secondary physical products together represent the entire production of the corresponding MRIO sector and (2) the more detailed inputs are fully represented in the MRIO sector direct inputs. When these conditions hold, the direct footprint can be calculated based on the physical product inputs and the indirect footprint calculated by driving the MRIO model with a demand equal to the direct requirements associated with the secondary product(s) demanded, but then the direct footprint associated with the demand driven by the physical *primary* products must be subtracted from the indirect footprint.

If performed correctly this approach offers the possibility of adding valuable detail to analyses involving consumption of secondary agricultural products such as livestock.

3. Discussion

3.1. Calculation benefits of MRIO and the integrated MRIO-F model

An environmentally extended MRIO framework allows for an efficient measurement of environmental flows along the complete supply chains of products and services, thus highlighting the direct link between economic activities and their environmental consequences (Wiedmann, 2009b; Wiedmann et al., 2007). Utilizing the MRIO-F framework is a crucial step in making EF and WF accounting consistent with standard Integrated Environmental and Economic Accounting (SEEA) (United Nations, 2003) and is in line with good practice in national environmental statistics (de Haan and Keuning, 1996). Furthermore, consolidating data in matrix form and performing calculations using matrix algebra reduces the likelihood of error (see Supporting Information for details).

The proposed MRIO-F model harmonizes CF, EF, and WF calculations, maintains product-level detail, and provides new insight into the complex web of interactions between international economies and the accompanying resource and waste flows that comprise the EF and WF indicators. The MRIO-F model allows for a more refined calculation of direct EF and WF than existing input–output models by incorporating the detailed physical accounts explicitly as a satellite account to the MRIO model. As such, the MRIO-F model constitutes a hybrid method between bottom-up physical resource accounting and top-down economic modelling. Various research groups worldwide are progressing MRIO development at a fast pace (Wiedmann et al., 2011a) and we developed the model and documentation to seamlessly allow for the utilization of these progressions in future iterations of the MRIO-F model.

3.2. Benefits of MRIO-F for policy analysis

The MRIO-F model opens the way for a new set of analyses and comparisons among the three footprint indicators. At the same time, using a common calculation framework significantly reduces the burden on a decision-maker to understand three independent models. The consumption-based footprint accounting complements traditional accounting of resource, land, or water use, which is based on a production perspective. Using the MRIO-F framework for footprint accounting offers a clear mechanism for storing direct footprints and allowing the calculation of either producer- or consumer-based aggregates. Both understanding local impacts of production of individual products (e.g. water use of specific crops) as well as tracing (international) supply chains of manufactured goods and services, is essential for the successful formulation of sustainable consumption and production (SCP) policies.

Integration of production *and* consumption as well as environmental *and* economic interactions makes the MRIO-F framework the best choice for footprint scenarios describing future changes brought about by policy interventions. Simulation of the combined effects of economic, social, and environmental policies aids in identifying strategies which best reconcile competing goals. Changes in production efficiencies, technologies, infrastructure, natural resource use, sector transformation, trade flows, and patterns of consumption are examples of variables that can be explored in MRIO-based scenario simulations (Wilting et al., 2008). Dynamic modelling offers opportunities for exploring the effects of fiscal policies such as taxation, trade tariffs, carbon trading, or economic stimulus.

Including multilateral trade flows provides insight into the environmental trade-offs driven by differences in inter-industry interdependencies and trade (Wilting and Vringer, 2009). The use of MRIO provides opportunities for measurement of feedback effects whereby production changes in one region are caused by intermediate demand changes in another region. In addition, MRIO offers the possibility of delineating supply chain footprints into mutually exclusive and collectively exhaustive portions of responsibility which can be shared by all actors in each economy (Lenzen et al., 2007) which is particularly relevant if a price is associated with an environmental impact such as through carbon taxes or trading schemes. In such a way this cost could be allocated according to agreed-on rules to the appropriate actors in an economy.

Finally, sophisticated analytical tools are available for the MRIO-F modeller such as structural decomposition analysis, contribution analysis, and structural path analysis. Structural decomposition analysis identifies drivers responsible for changes in the environmental performance of the economy and characterization of ex-post or ex-ante effects such as eco-efficiency, demand composition shifting, and output growth (see e.g. Baiocchi and Minx, 2010; Weinzettel and Kovanda, 2011; Wood, 2009).

Contribution analysis identifies the specific processes or regions contributing most significantly to the overall impacts (Hertwich, 2011; Peters et al., 2011) and structural path analysis identifies specific hot-spot nodes within supply networks by sector, product, and region. The latter technique allows for the investigation of specific international supply chains and is ideally suited to prioritise hot spots of environmental impacts of consumption by linking to specific production locations (Peters and Hertwich, 2006; Wood and Lenzen, 2009).

3.3. Limitations and uncertainty

A key concern about the integration of footprint accounts with the relatively less detailed MRIO model in monetary units is the introduction of uncertainty. The integration of the footprint accounts with the MRIO model proposed here involves a tradeoff between the benefits noted in Sections 3.1 and 3.2 and the drawbacks associated with product aggregation, regional aggregation, and price inhomogeneity.

3.3.1. Product aggregation

The level of MRIO product detail in primary sectors such as agriculture, fishing and forestry is less than that available in the detailed physical data. For food products, often more detailed physical data are available. In the case of other products and service sectors, often the level of detail available in MRIO models is comparable or greater than that available in internationally harmonized physical accounts.

By keeping as much product and sector detail as possible and by separating direct and indirect footprint calculations, the MRIO-F model minimizes potential errors associated with product aggregation. Previous studies have shown that utilizing specific product detail can help to achieve the minimal level of uncertainty (Hawkins et al., 2007; Lenzen, 2011; Su et al., 2010). Williams et al. (2009) demonstrated the range of supply-chain energy use in iron/steel-related sectors increases by almost a factor of five if the number of sectors in the model is increased from 32 to 399. This expanded range is a result of the improved reflection of differences in the life cycle inventories of individual products. A similar effect can be expected for the MRIO-F model and quantitative investigations will be the subject of future research.

3.3.2. National and regional aggregation

Similar to product aggregation, aggregation on the national level and across countries masks variation within a category. In linking physical tables to the MRIO, it is sometimes necessary to estimate indirect footprints for countries for which physical unit data are available but specific monetary MRIO data are not by using a common "rest of" region. As a result, direct footprints can be calculated from the country-specific FAO data while indirect supply chain effects can only be calculated from the generalized regions, thus introducing the potential for inaccuracy within the indirect portion of the overall result.

Even though most global MRIO datasets aggregate minor economies into world regions, recent developments have resulted in the creation of a MRIO model that distinguishes approximately than 200 countries (Wiedmann et al., 2011a). The effect of regional aggregation in MRIO modelling on national CF results has recently been demonstrated by Andrew et al. (2009) and Su and Ang (2010) (for regions of China).

There are differences in the variation inherent in aggregation of countries such as Switzerland or Estonia versus Russia, China, Canada, or the United States. The aggregation of the farming practices of southern California, Florida, and Iowa results in variation within a calculated result. The loss of resolution associated with WF aggregation has been discussed for non-IO-based approaches by Pfister et al. (2009) who suggests that aggregation by watershed is more appropriate for characterization of WF. Unfortunately, institutional considerations often result in provision of data at the national level.

As is the case with product aggregation, where the indirect contribution to the footprint is small, generalization should not significantly affect results. However, if the indirect contribution is large and if the supply chain footprints vary significantly across products and countries, the uncertainty should be addressed in future work through the collection of additional data. Integrating footprint accounts based on physical datasets with MRIO has drawbacks associated with data vintage. While many types of physical and trade data are available on an annual basis, MRIO models have so far been only produced for specific years. Recent developments in MRIO compilation, however, have brought up-to-date time series data within reach (Wiedmann et al., 2011a).

3.3.3. Price inhomogeneity

Representing flows in monetary units introduces uncertainties associated with inhomogeneity in prices over time, between transactions, and between different physical products (Lenzen and Murray, 2001; Weisz and Duchin, 2006). Aggregation across products, geographic regions, time, and individual transactions results in variation in the prices associated with the underlying transactions that make up a value in the MRIO. Price inhomogeneity also comes into play when physical demand is converted to monetary demand to be used as an input to the MRIO model. For example, a small amount of a high value product such as *cashew nuts* would induce the same demand in the MRIO model as a large amount of a low value product such as *dry beans* since both are associated with the same MRIO sector *vegetables, fruit, and nuts*. In this way, indirect footprint results involve a monetary allocation of impacts and will scale linearly with price. An alleviating factor is that prices used in the MRIO-F model are averaged over at least one year which smoothes short-term price fluctuations.

3.3.4. Trade-off between loss of product detail in trade and improved accounting for intermediate products

Understanding the trade-offs between loss of product detail in trade and improved accounting for intermediate products is a crucial issue when considering the overall value of integrating the footprint accounts with MRIO. Without the MRIO model the indirect contribution of service sectors is altogether ignored, the footprints of intermediate products are allocated to the country into which they are directly imported, and the contributions to overall footprints are not related to the products actually consumed by households (compare to (Wiedmann, 2009a)). To include services and properly allocate impacts to final consumption, for now we must accept the inherent inhomogeneity associated with aggregation across products, within countries, and across countries and accept a monetary allocation of indirect impacts. Moving forward, we hope more detailed datasets will be available to reduce this inhomogeneity.

4. Conclusions

In this paper, we have described a new method to calculate national and regional EF and WF values by utilizing a MRIO framework. The general advantages of utilizing MRIO to measure EF and WF include aligning physical unit data of product use with standard environmental-economic accounting, systematic calculation of service sector footprints, complete system boundary coverage for entire economies, enumeration of the indirect impacts associated with international supply chains, and utilization of sophisticated environmentally extended MRIO modelling techniques to identify key nodes within supply networks that are responsible for changes in the environmental performance of the economy.

The MRIO-F method described in this paper provides a new approach for researchers and practitioners that preserves the product-level detail of the physical EF and WF accounts and harmonizes the CF, EF, and WF methodologies. Product-level detail for direct footprints is accomplished through the addition of a satellite account and complementary matrix-based physical model to the monetary MRIO model. By including additional detail related to primary products and by tracking these products in physical units, the MRIO-F model enables a more accurate calculation of direct footprints at the individual product level.

The MRIO-F model opens the way for a new set of sustainable consumption and production analyses simultaneously among the three footprint indicators. Most notably, the MRIO-F model facilitates the development of scenario analyses that will analyze the changes in production efficiencies, technologies, infrastructure, natural resource use, sector transformation, trade flows, and patterns of consumption.

Based on the methodological framework described in this article, the footprint model EUREAPA has been created through the EU-FP7 project OPEN:EU, enabling the evaluation of policy scenarios on a national and international level (http://www.oneplaneteconomynetwork.org).

Supporting information

Additional information on background, method, data as well as a numerical example and an expanded discussion are available in Supporting Information. This document also includes a list of variables, acronyms, and special terms used.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.ecolind.2012.02.025.

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Appendix F: Supporting Paper AII

Steen-Olsen, K.; Hertwich, E. G., Life cycle assessment as a means to identify the most effective action for sustainable consumption. In *Handbook of Research on Sustainable Consumption*, Reisch, L.; Thøgersen, J., Eds. Edward Elgar Publishing. In Press.

Life cycle assessment as a means to identify the most effective action for sustainable consumption

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Introduction

What kind of responses would you get if you took a poll to the streets asking people what actions they should take to reduce the environmental burden of their lifestyle? It is likely that common replies would involve installing energy efficient light bulbs and water efficient shower heads, switching to reusable shopping nets, driving electric cars, and so on. But how much, if at all, do these individual actions actually contribute? The sustainable society, though hardly controversial as a vision for the global community, is as challenging as it is ambitious, and it does not help that sustainability is intrinsically hard to measure.

One challenge for consumers is the plethora of information they are exposed to from media, official agencies, commercial actors, friends, and family. Even for those who are motivated to do so, changing behavior on environmental grounds takes cognitive effort, which consumers economize (Stern et al. 2010). As such, it is in the interest of all parties that those behavior changes that are successfully introduced to the public carry as much weight for the environment as possible.

In research on environmental psychology and sustainable consumption, the term ‘environmentally significant behavior’ has emerged to highlight the need for studies to take into account both the size of the environmental impact and the possibility of changing that impact through behavior change (behavioral plasticity, see e.g. (Stern 2000)). Not all types of environmental behavior matter equally; for instance, major infrequent decisions such as what kind of car to buy, or whether to buy a car at all, typically have larger environmental impacts than whether or not people reuse their shopping bags (Stern 2000).

Clearly, a consumer-oriented environmental policy measure that is successful in changing consumer behavior will not be effective if the net impact per person that changes their behavior is low. Gatersleben et al. (2002) report highly variable correlations between the perceived and actual environmental impact of different types of household consumption. The overall environmental impact of a certain purchase or activity is usually not intuitively obvious, but will be a combination of several effects that can be different in a number of ways:

- Temporally: In addition to the direct effects incurred during its use, a purchased product might have caused environmental harm in the past, or it is committed to do so in the future. A nuclear power plant may deliver virtually emission free electricity in the present, but will have to deal with the handling of radioactive waste in the future. A fine whisky purchased today contributed to emissions two decades ago.
- Spatially: In contrast to preindustrial times, when trees logged for ship building and firewood would come from (and visibly diminish) local forests, the globalized economy of today has increasingly separated consumers from producers, and retail goods currently available to consumers in the western world can contain parts designed, manufactured and assembled in dozens of countries.
- Chemically: The production, use, and disposal of consumer goods frequently involve emissions of thousands of chemical compounds, each affecting the environment in different, often unknown ways.
- Mechanistically: There might be several causal pathways between emitted substances and ultimate adverse effects. For instance, inhalation of mercury vapor has been known for centuries to be hazardous, but today we know that mercury also accumulates in the food chain and represents an additional health risk to humans and large predators through ingestion.
- Categorically: The release of pollutants and wastes to the environment, and the intake of resources from it, during the lifetime of a product may affect the

environment in many different ways. One such harmful impact category is adverse climate change from anthropogenic greenhouse gas emissions; other examples include increased cancer incidence or species extinction.

Life cycle assessment (LCA) is a well-established method to quantify the overall environmental impacts of consumption activities, put forward by the United Nations as a central tool to achieve sustainable consumption (United Nations 2002). LCA describes the impacts associated with consumption of services provided by a product system, where the product system includes the production, distribution, operation, maintenance and disposal of the product. LCA can hence facilitate a consumption-based approach to understanding environmental problems. The life of a product in this respect starts with the extraction of natural resources such as iron ore from the ground, and ends with their return to nature as landfill or as emissions.

The consideration of indirect (upstream) emissions is highly important when addressing sustainable consumption. Roughly half of the energy use of households is indirect, and the share increases with increased affluence (Moll et al. 2005; Hertwich 2011). For other impact categories, such as terrestrial and marine ecotoxicity, the share will be much higher because emissions occur in the producing and extracting industries, far away from the consumer.

The published assessments of the overall environmental impacts of households generally agree on which household consumption activities contribute more or less towards the total, when consumption is broken down in broad categories. Specifically, researchers single out food, shelter and mobility as being most important, overall (Tukker et al. 2008; Hertwich and Peters 2009; Tukker et al. 2010; Hertwich 2011). In the remainder of this chapter, following a section devoted to an introduction to the general structure of a life cycle assessment, these three household consumption categories will be dealt with in turn, in an attempt to summarize some general insights in each of them from the LCA literature over the last decade or so. The chapter is concluded with a general discussion of the potential contributions of LCA to the international debate on sustainable consumption, along with a word of caution as to its limitations.

The structure of an LCA

Life cycle assessments are standardized under ISO 14040, which outlines four key assessment phases to be conducted in an LCA (International Organization for Standardization 2006): 1) Goal and scope definition, 2) Inventory analysis, 3) Impact assessment, 4) Interpretation. In this section, these four elements will be introduced briefly.

Any LCA should start with defining the **goal and scope** of the analysis. To ensure that LCA studies are not taken as evidence to support practices or products without sufficient grounds, and to avoid comparison of LCA studies that are in fact not comparable, it is of utmost importance that the researchers set out by clearly and unambiguously defining the goal and scope of the study. A central part of this is the definition of a 'functional unit' as the fundamental unit under study. It specifies the service or product to be delivered to the customer, and for which the total embodied environmental impacts is to be quantified. Since LCAs are frequently used to compare the environmental performance of two or more alternative ways of delivering a service, it is important that the functional unit is described precisely, and that the focus is on the service required rather than on a specific type of product. For instance, rather than defining the functional unit simply as a desk lamp, it may be defined as the provision of adequate working light for one office worker during the course of one standard working year; or rather than a can of soda, the functional unit might be specified as a certain volume of packaged carbonated beverage (see, e.g. (Amienyo et al. 2013)).

Also important in the goal and scope phase is specifying the system boundaries, that is, which processes are included in the analysis and which are not. The supply chain of a functional unit can in principle be modeled with extreme complexity, because every industrial process is indirectly dependent on a wide range of inputs in addition to the direct inputs of production. The assembly of a desk lamp directly requires the lamp components, but the manufacturing in practice also requires a factory, labor inputs, energy to heat and illuminate the factory, roads and other infrastructure to service the factory and so on. Since the supply chain of any product hence is in principle infinitely long, any LCA implies a system boundary outside of which potentially relevant processes are excluded, whether it is explicitly defined or not.

Inventory analysis is the compilation of an inventory of primary flows, i.e. material, energy and other exchanges with nature, along the life cycle of the product system. Exchanges with nature include uses of natural resources as well as emission and waste flows back to nature. This stage, which is usually the most time consuming, involves the organization of a system of industrial processes and the quantification of their material and energy exchanges with nature as well as with other industrial processes. In an LCA the industrial system that ultimately delivers the functional unit is modeled as a network of processes that take material and energy inputs from each other and nature in order to deliver their output product. For the analysis, the process diagram is reformulated in matrix form following the Leontief model.

In practice, analysts will typically model a *foreground system* consisting of the most relevant processes for the system under study, and link this to a *background system* in the form of commercial LCI databases of generic products and services. For instance, an analyst assessing a solar cell panel would probably focus on modeling the individual components of the panel, but would rely on a background database to provide life cycle results for the steel in its frame.

Impact assessment is the process of translating the result of the inventory analysis into environmental consequences. The life cycle inventory gives a quantitative account of all the environmental interventions (sometimes enumerating emissions of thousands of chemicals) resulting from the functional unit over its life cycle. This includes total resource use and emissions incurred directly and indirectly in the supply chain, as well as in the production, use and disposal of the product under study. As such, the LCI could be viewed as a complete analysis result; however, for most purposes this is not very useful as an environmental assessment. In the impact assessment phase, knowledge of the various environmental mechanisms is applied to aggregate the contributions of individual primary flows to each one of a set of environmental impact categories, measured in units of some reference compound or damage indicator. For instance, emissions of CO_2 and CH_4 are both counted towards global warming potential (GWP) impact, however since CH_4 is a more potent greenhouse gas (GHG), it is assigned a weight to allow the two emissions to be added and expressed in the single unit kilograms of CO_2-equivalents.

Finally, ISO14040 defines **interpretation** as a separate phase to be conducted concurrently throughout the study, to provide guidance to an iterative process between the first three stages. The interpretation part of an LCA should serve to inform readers about the relevance, validity, and conclusions of the study, and should discuss the sensitivity analysis of the results, if any has been performed (International Organization for Standardization 2006).

LCA insights on some important consumption categories

Food

Studies of the environmental impacts of household consumption consistently highlight food consumption as one of the most important contributing consumption categories towards total impacts of consumption (Tukker et al. 2008). Food is one of the areas where LCA has contributed significantly to the understanding of the environmental impacts, on different scales. A range of assessments have been published, see e.g. Roy et al. (2009) or de Vries and de Boer (2010) for reviews of recent life cycle assessments of food products.

Within the food category, meat and dairy products are singled out as the most significant products in terms of environmental impacts (Tukker and Jansen 2006; Carlsson-Kanyama and González 2009; Hertwich et al. 2010). Meat carries a lot of embodied resources and energy due to the rather inefficient transfer of energy from feed to meat; the differences in said conversion efficiencies also explain why beef generally has higher environmental impacts than pork or chicken (Carlsson-Kanyama et al. 2003; Weber and Matthews 2008; de Vries and de Boer 2010). Adding to the effect of lower conversion efficiency, a lower reproduction rate for cattle compared to chickens or pigs also drive up the life cycle land and energy requirements (de Vries and de Boer 2010). In terms of global warming potential (GWP), meat is different from most other products in that CH_4 and N_2O

emissions contribute more towards the total than does CO_2 (de Vries and de Boer 2010). CH_4 emissions are particularly important for meat from ruminants, due to emissions from enteric fermentation in the rumen, whereas N_2O emissions are primarily due to fertilizer use[1].

In the dairy category, milk and cheese are important products. In an LCA of industrial milk production, Eide (2002) compared dairies of various sizes and found the agricultural stage to be the overall most important in terms of environmental impacts; however, milk produced from the smallest dairies carried very high life cycle energy requirements due to electricity consumption at the dairy itself. This efficiency difference led to total life cycle energy requirements of 6.3 MJ/liter for milk from the smallest dairy versus only 3.6 MJ/liter for the largest. Though milk is consumed in greater quantities than cheese (103 kg/capita and 18 kg/capita, respectively, in Norway, 2010 (Johansson 2011)), LCA results reported by Carlsson-Kanyama and González for Sweden show 11 times higher life cycle GWP per kg cheese compared to milk when both are domestically produced (2009); in fact cheese is more GHG-intensive per kg than pork and chicken meat.

The promotion of organically farmed food as a more sustainable option has spurred some public debate, due to doubts of the net effect of organic products when factors such as production efficiency is taken into account. This is the kind of research question which LCA is ideally suited to answer, and there have been some attempts to do so. Williams et al. (2006) compared organic to conventional meat production, and found

[1] This applies to all agriculture, not only meat production.

that in UK conditions, organic beef had a higher carbon footprint; however, for mutton the result was the opposite. Pork had similar carbon footprint whether organically or conventionally grown. Wood et al. (2006) compared organic to conventional farming in Australia and conclude that though the organic farms had higher on-farm energy use, the conventional farms carried much higher embodied energy use upstream, leading to higher life cycle energy requirements. Overall, LCA research seems to confirm the notion that the benefits of reduced use of pesticides and mineral fertilizer comes at the price of somewhat reduced efficiency, particularly in terms of land area requirements (Cederberg and Mattsson 2000; Haas et al. 2001). Organically produced milk exhibits a similar trade-off pattern as meat (Cederberg and Mattsson 2000; Williams et al. 2006).

Related to the debate of organic versus conventional food products, there have been campaigns promoting locally produced food to reduce environmental impacts of food transport. However, an influential study on the climate impact of food in the United States concluded that in general, the transport stage is of minor importance compared to the production stage, contributing only 11% of the total food-related carbon footprint for the average household (Weber and Matthews 2008). Milà i Canals et al. (2007) studied apples consumed in Europe, and found lower primary energy requirements for domestically grown apples when the apples were in season, but variable results out of season; other studies support this notion of favoring local produce only when it is in season (Carlsson-Kanyama et al. 2003). Among imported products, the distance and mode of transport can significantly influence life cycle results, especially air-transported food carries high embodied impacts (Jungbluth et al. 2000).

In order to limit global warming to 2°C, emissions related to food and diets will have to be addressed, as one of the major sources of GHG emissions. The energy and greenhouse gas intensity of meat and meat products seems to suggest vegetarian diets are preferable from a sustainability perspective. In a comprehensive review of life cycle inventories of 84 food items, González et al. (2011) concluded that plant-based food was overall much more efficient than animal-based foods in terms of protein delivery per unit energy use or GHG emitted. Girod et al. (2013) estimate that life cycle GHG emissions per calorie will have to be reduced from a global average of 1.3 to 0.37 gCO2e/kcal from 2000 to 2050, which will put strict limitations on the share of animal-based products in our diets (see Figure 1).

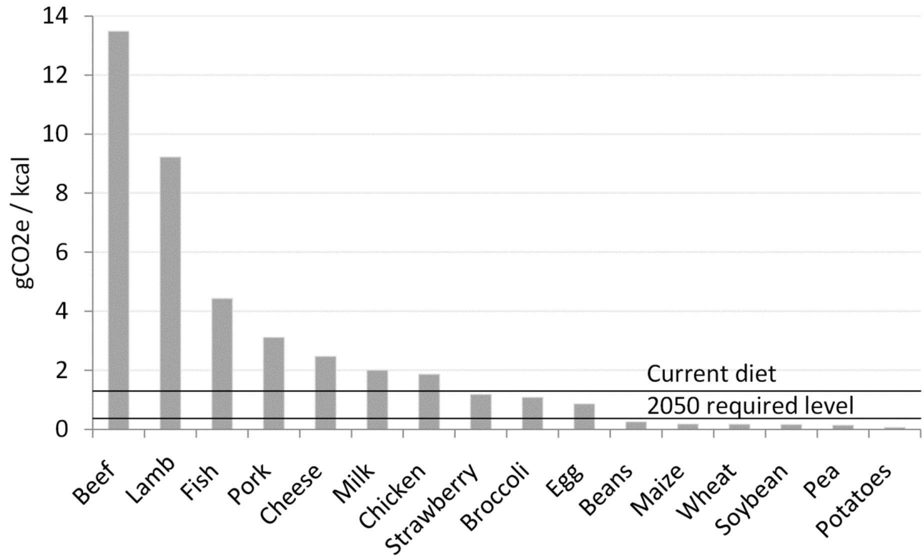

Figure 1. Life cycle GHG emissions per kcal of some food items, based on (González et al. 2011). Current global average and 2050 target levels shown for reference, based on (Girod et al. 2013). gCO2e = grams of CO_2-equivalents

When comparing environmental impacts of different types of food using LCA, the selection of functional unit is not obvious. For instance, on a mass basis, milk has lower land use requirements than meat, because of the high water content of milk (de Vries and de Boer 2010). Since the primary function of food is to provide nutrition for the body, comparisons based on fixed amounts of protein or energy might be better suited than mass-based comparisons; see Schau and Fet (2008) for an overview of some food LCAs and the various functional units chosen in them. Roy et al. (2009) found chicken to be the most environmentally efficient meat if protein was chosen as the functional unit, whereas pork performed better per calorie delivered. Due to this possible ambiguity, Carlsson-Kanyama (1998) recommends comparisons to be made on complete meals or diets with comparable nutritional qualities, rather than on single products.

Shelter

The 'shelter' category in analyses of household environmental impacts concerns dwellings, including the construction, maintenance and ultimate demolition stages; and the energy required for space and water heating (or cooling). When assessing the environmental impacts of household consumption over a year, the shelter category entails some additional considerations in contrast to analyses of food consumption.

What are the environmental impacts of the occupancy of a house over a year? Aside from the energy used for space and water heating, cooking, cleaning, etc.; the lifetime energy requirements of the building itself are significant. However, most of this energy use is generally required in the construction phase with only smaller additional requirements for structural maintenance and upgrades over the years. Obviously, these high initial energy requirements must be distributed over the years the dwelling will be used, and the assumed lifetime of buildings will significantly affect the annualized embodied energy requirements.

Increased focus on energy consumption in buildings has led to significant advances in the energy efficiency of buildings in recent years, ranging from small-scale improvements of furnishings and appliances such as energy efficient light bulbs and hot water taps, to larger structural changes such as improved thermal insulation in walls and ceiling, and triple-glazed windows. Buildings with especially low energy requirements are referred to as low-energy buildings or passive houses, while self-sufficient or zero-energy houses are even better insulated and also generate their own electric or thermal energy so that they are not dependent on an external energy supply at all.

Due to the structural complexity of low- and zero-energy buildings compared to conventional houses however, there have been concerns about their overall life cycle impacts. Following this, several LCAs have been published, attempting to quantify life cycle energy requirements and emissions by including also the emissions incurred in the construction, maintenance and demolition stages. Though there are considerable differences in structural characteristics and climatic conditions among them, the numerous case studies published (see (Sartori and Hestnes 2007; Ramesh et al. 2010; Hertwich 2011; Ürge-Vorsatz et al. 2012) for overviews of some relevant studies) seem to allow at least two general conclusions to be drawn. Firstly, even after accounting for embodied energy use, the operation stage dominates the life cycle energy use of residential buildings, and the bulk of this energy use is associated with space conditioning (heating and cooling) and hot water provision. Most of the remainder consists of the embodied (construction-phase) energy, with maintenance and demolition contributing only to a smaller degree to the overall requirements (Fay et al. 2000). As houses become more energy efficient, the contribution of heating and other direct energy use will decrease. The LCA of a low-energy compared to a standard house by Blengini and Di Carlo

(2010) suggests that after the heating requirements have been reduced, there is no longer a clear culprit, which means further improvements will be more challenging and require a more systematic approach.

Secondly, though low-energy houses have indeed been found to carry higher embodied energy, this (slight) energy cost is several times outweighed by the lifetime energy savings gained in the operations stage through lower heating requirements (Thiers and Peuportier 2008; Blengini and Di Carlo 2010; Dodoo et al. 2010; Dahlstrøm et al. 2012). However, the marginal net life cycle savings diminishes as increasing steps are taken to lower the operational energy requirements of the buildings; in fact, the annualized total energy requirements of the most efficient low-energy houses may be lower than those of completely self-sufficient houses (Sartori and Hestnes 2007). Hernandez and Kenny (2010) highlight the same trade-off effect for insulation levels in low-energy buildings, and find that the energy ratio, defined as the ratio of the decrease in annual energy use to the increase in annualized embodied energy, diminishes steadily with increased insulation thickness and after a few increments drops below 1, implying negative lifetime energy savings on the margin.

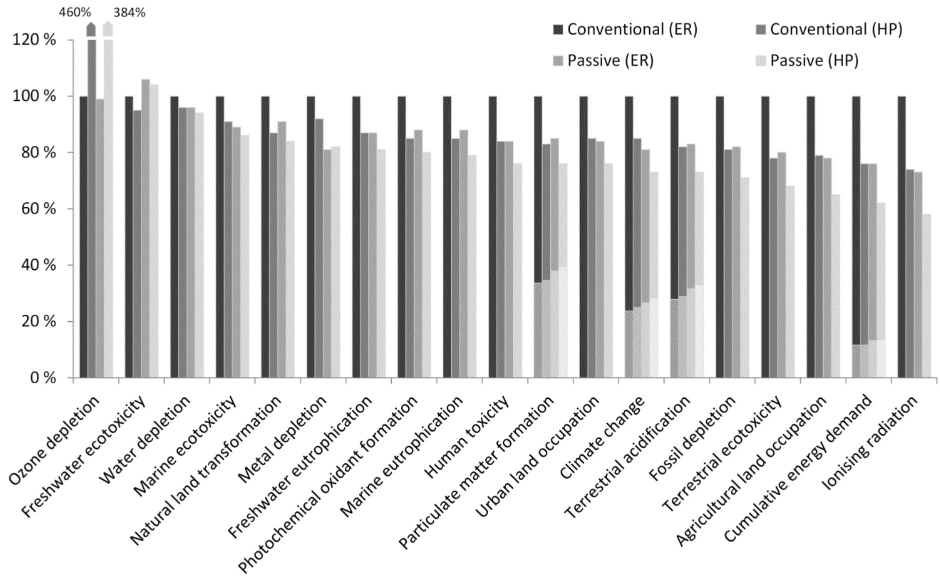

Figure 2. Various life cycle impacts of conventional versus passive house using either electric resistance (ER) or heat pump (HP) for space heating. Impacts are shown relative to conventional (ER). For four of the impact types, the contribution of embodied (construction, materials and demolition) impacts are indicated as shaded areas. The high ozone depletion potential of the HP cases is due to refrigerant leakages from the heat pumps. Based on (Dahlstrøm et al. 2012)

The choice of materials obviously impacts the energy use and environmental impacts embodied in buildings. The production of concrete entails significant energy use and CO_2 emissions; in a comparative LCA of embodied emissions Börjesson and Gustavsson (2000) find concrete buildings to have 60-80 % more embodied primary energy than a wooden equivalent. Wood also offers a potential for in-stock carbon storage, as well as energy recovery after demolition (Ürge-Vorsatz et al. 2012).

In the environmental assessment of most goods and products, a pertinent question as new and more energy and environmentally efficient technologies and standards of some product evolve is whether or when to replace the product in order to minimize the environmental impacts. Kim et al. (2006) apply a life cycle optimization model to determine optimal lifetimes of household refrigerators in the US to minimize energy, GWP and costs, and conclude from model runs from 2004 to 2020 that ‘current owners should replace refrigerators that consume more than 1000 kWh/year of electricity (typical mid-sized 1994 models and older) as an efficient strategy from both cost and energy perspectives’.

Along the same lines, life cycle type assessments have been used to determine whether existing buildings should be retrofitted to comply with current standards or whether it would be preferable to demolish and rebuild entirely. Itard & Klunder (2007) perform a comparative life cycle assessment of four cases of increasing levels of interventions, from maintenance through consolidation and transformation through redevelopment, and find transformation to be environmentally preferable to demolition and rebuilding. Dodoo et al. (2010) assess several retrofitting measures on an existing wooden house, and find significant potential for life cycle energy savings, however the net gains depend largely on the heating technology of the house, with the greatest savings for electrical resistance heating and lower savings for houses using heat pumps or district heating.

Mobility

Household energy requirements in the developed world have increased greatly over the past century, and mobility, particularly private car use, has been one of the major contributors to this development (Fuglestvedt et al. 2008). Across the world, though the onset came at different times throughout the 20th century, as soon as private cars have become publically available and affordable the per capita stocks soon soared; in the United States the ownership rate is approaching 0.8 cars per capita (Pauliuk et al. 2011). To consumers, car use is perhaps the most intuitively obvious private consumption activity that contributes to household environmental impacts. Combustion of gasoline simultaneously entails resource use, energy use, and emissions of CO_2 plus a range of other pollutants. As such, the mobility category offers significant potential for consumption-oriented strategies and policies for sustainability. In this section, we have chosen to focus on private car use, which is the most important of the various transport modes included in the mobility category.

When evaluating strategies to reduce the environmental impacts of private car use, LCA can serve to avoid the pitfall of environmental *problem shifting*. Problem shifting refers to situations where an effort to reduce environmental impacts of one process leads to other adverse impacts instead. Problem shifting can occur as simple relocations of fundamentally equivalent emissions, either geographically (by importing rather than producing emission intensive components) or within the production chain. Electric vehicles are sometimes marketed as emission-free because there are no tailpipe emissions; however the generation of the electricity with which their batteries are charged will require emissions at the power plant instead, thus the vehicle emissions are effectively relocated further up the supply chain.

Other than relocations of a specific type of emission, problem shifts can also manifest themselves in terms of other types of environmental impacts. In the impact assessment stage of LCA, the system's impacts in terms of several different types of environmental impacts are usually assessed, thus allowing a simultaneous assessment of widely different potential impacts. Such a problem-shift has been identified for biofuels, which have been suggested as potential substitutes for fossil based vehicle fuels to mitigate climate change. Using life cycle type assessments, several studies have identified important trade-offs: Anticipated increases in demand for biofuels would imply increased land use requirements, leading to pressures on existing agricultural land (thus leading to increased food prices in some areas) as well as on rainforests and other natural land (Bringezu et al. 2012).

In LCAs of vehicles, there are two main subsystems to consider. The *vehicle cycle* includes the life cycle of the vehicle itself, while the *fuel cycle* (also known as 'Well-to-Wheel') includes the conversion of fuel to vehicle movement ('Tank-to-Wheel') and the complete upstream process of delivering fuel to the vehicle ('Well-to-Tank'). As was the case for housing, the LCA literature unsurprisingly shows that for conventional ICE (internal combustion engine) vehicles fueled by gasoline or diesel, the operating stage is the most important in terms of energy requirements and GWP (MacLean and Lave 2003a).

However, for the various low-carbon alternatives available, indirect impacts become more important in relative, probably also in absolute terms (Schäfer et al. 2006).

Several LCAs evaluate different vehicle fuel options. MacLean and Lave (2003b) reviewed a dozen studies on alternative fuels and propulsion technologies for cars and light trucks, and conclude that no one technology can be singled out as preferable, because they have different advantages and disadvantages. Ultimately, the question is how we should weight the different impact types and environmental problems against each other.

Bioethanol has been used in fuel blends in Brazil for decades already, and has been put forward as a low-hanging fruit for decarbonization of car fuels, because existing conventional cars can run on gasoline blends with up to 10 % ethanol (E10) without any modifications (MacLean and Lave 2003a). Niven (2005) assess the sustainability of ethanol/gasoline blends in Australia based on a review of available literature, and concludes that for such E10 blends, from a life cycle perspective the GHG reductions are small and come at a significant price in terms of other environmental impacts, notably soil and groundwater contamination, air pollution, and land use. In another review article, von Blottnitz and Curran (2007) evaluate 47 published LCAs of bioethanol compared to conventional fuel, and find large variations depending on the Well-to-Thank side of the bioethanol. Using tropical sugar crops as feedstock appears preferable, however the authors point to potential land-use conflicts. In terms of energy use and GWP, bioethanol performs well throughout, while for other impact categories the authors find divergent or unclear results across studies.

The case of electrical versus conventional passenger cars was recently assessed by Hawkins et al. (2013), who performed a comparative LCA of the environmental impacts of 1 km driven with a typical European small car powered with either a conventional internal combustion engine or a battery-electric powertrain (see Figure 3). Assuming a common vehicle glider (i.e. the car except powertrain) and vehicle and battery lifetimes of 150,000 km, the authors quantify lifecycle impacts per km driven within ten different impact categories. They assess different fuels for the conventional vehicles, and different battery types as well as different electricity sources for the electric vehicles, and find overall a 10% to 24% decrease in GWP when assuming the present European electricity mix. However, the electric vehicles perform worse in terms of several other impact categories, such as human toxicity potential, freshwater ecotoxicity and eutrophication potentials, and metal depletion potentials, chiefly originating from battery production. It should be emphasized that vehicle batteries are still an emerging technology and that changes in production methods are likely to affect environmental impacts of future battery production.

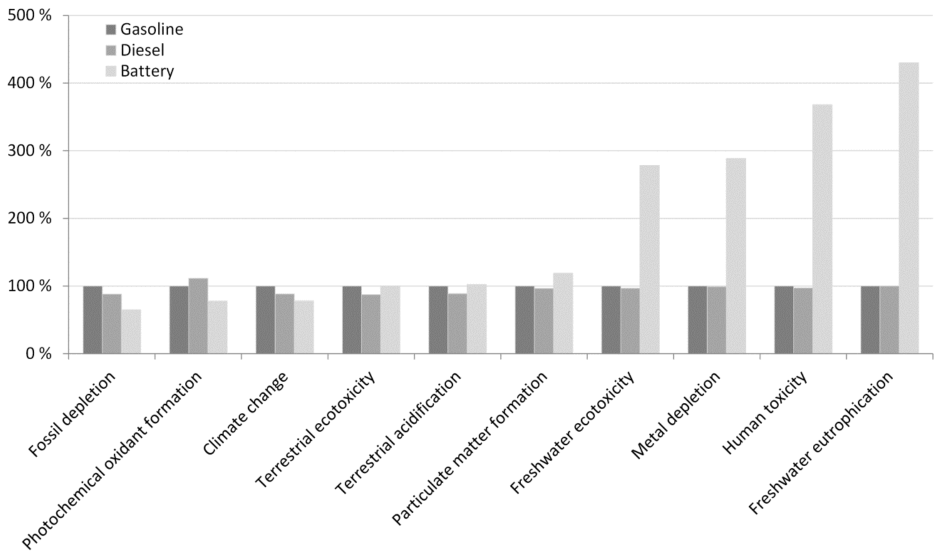

Figure 3. Various life cycle impacts of a battery electric vehicle (BEV) versus a comparable conventional vehicle (gasoline or diesel fueled) shown as relative to gasoline vehicle results; based on (Hawkins et al. 2013). BEV results are averages across two types of Li-ion batteries with similar results, assuming a European average electricity mix

Discussion: The Role of LCA in societal sustainability assessments

The global growth in overall material and energy use per capita as well as in absolute numbers; the increased material and technological complexity of products and services and their international supply chains; the progressing scientific understanding of the interconnectedness of natural systems and the various types of environmental challenges associated with them; all these factors point to the need for a holistic, life cycle type approach to sustainability. However, life cycle assessments alone cannot uncover the path towards a sustainable future. An important limitation of LCAs in this respect lies on the consumer side. For instance, the overall impacts of LCA findings on the sustainability of different types of travel modes, car types and fuel types may be diminished by the fact that consumers may ultimately prefer big, powerful trucks with loud engines (MacLean and Lave 2003a). Furthermore, there is good reason why the ISO standard names interpretation as one of the four main steps of any LCA. Two examples highlight the importance of interpreting LCA results, especially when attempting to draw conclusions on the bigger picture. Firstly, significant advances in technology and management have led life cycle GHG emissions of air travel to be comparable to those of cars per passenger-kilometer. However, in the detail of this functional unit (which is perfectly standard to use) lies a devil: Airplanes travel many times faster than cars, allowing consumers to travel vast distances on a regular basis. Long-weekend intercontinental trips are becoming popular among an increasing group of well-off people, something that would be unthinkable with any other mode of transport. In other words, on a per time basis, air travel is much more emissions intensive than cars, allowing much higher overall per capita carbon footprints. Secondly, in an interesting article Cullen and Allwood (2009) illustrate the “washing machine effect”, whereby a synthesis of individual analyses may lead to a distorted picture of the overall system. The name of the effect was drawn from an example of a washing machine, where individual energy-use LCAs of separate parts including the machine itself, the detergent, the process of washing clothes, all conclude that the operations stage, i.e. the energy used in the washing cycle, is clearly the most important stage, while a combination of the three to one larger system revealed the materials and production stages to be equally important.

As illustrated in this chapter, life cycle assessment is a powerful tool for analyzing complex process chains and the way industrial processes interact with each other and nature. Using LCA, analysts can simultaneously assess a system's impacts across a range of different impact types and identify benefits and trade-offs between similar products. The points made in the previous paragraph however, illustrate that the results of life cycle assessments must be carefully interpreted before they are used to create environmental policies. For design of efficient policies and strategies for sustainability, LCA results should also be evaluated in light of the growing body of literature from the fields of consumer and behavioral research, since, as pointed out by Stern (2011), the overall effect of an attempted shift in consumer patterns to lessen environmental impacts is ultimately the product of the technical potential for reduction and the share of the population able and willing to make the change.

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