JOURNAL OF ECONOMIC SURVEYS

doi: 10.1111/joes.12027

EMBODIED CARBON IN TRADE: A SURVEY OF THE EMPIRICAL LITERATURE

Misato Sato

Grantham Research Institute on Climate Change and the Environment, London School of Economics

Abstract. This paper critically reviews the literature on embodied carbon in trade and evaluates our present empirical understanding of these flows. A careful comparison of quantitative results from this literature exposes significant inconsistencies. For instance, estimates for emission embodied in world trade in 2004 range between 4.4 Gt and 6.2 Gt CO2, the difference corresponding to around half of Europe's annual emissions. A few consistent themes do nevertheless emerge from the literature. Most importantly, emissions in trade constitute a large and growing share of global emissions. Uncertainty about country-level embodied emissions remains large, however, which presents severe limitations for the practical application of embodied carbon principles in climate policy.

Keywords. Climate policy; Embodied carbon; Embodied emissions; International trade

1. Introduction

To what extent do trade and consumption contribute to rising global greenhouse gas (GHG) emissions? Will strengthening domestic climate policy measures lead to real reductions in GHG emissions or relocation of industry and emissions to countries with lax regulation? Who is responsible for the emissions from China's export sectors – the Chinese producers, or the consumers abroad?

In an effort to provide empirical support to such policy debates around the design of GHG mitigation policies for industry emissions and the wider environmental impacts of consumption, there has been a recent boom in the literature which quantitatively examines the embodied carbon content of trade. Typically, these studies measure and contrast the volumes of embodied emissions in a country's imports versus their exports, thereby estimating a country's balance of embodied emissions in trade.

These studies form an extension to the discourse that began in the 1970s, around the geographical displacement of pollution and resource use as a consequence of trade. Previous to carbon, quantitative assessments of embodied pollution and resources have been carried out for water (Wichelns, 2001; Oki and Kanae, 2004; Hoekstra and Hung, 2005), methane (Subak, 1995), energy (Proops, 1977; Herendeen, 1978) and land use (Lenzen and Murray, 2001).

Studies on embodied carbon have thus far found large and growing volumes of embodied emissions in trade (EET; Figure 1), in line with the growth in global trade volumes[1] and international integration of supply chains over the past decade. Studies have found some 4 Gt–6 Gt of CO_2 embodied in global trade in 2004 (equivalent to 15–25% of annual GHG emissions) and 7.8 Gt for 2006 (equivalent to around 30% of global emissions).

The problem is not in the volumes of embodied emissions in trade *per se*, but in the lack of mechanisms to account for the emissions that are produced in one country and consumed in another. The lack of policy measures that regulate the carbon emissions embodied in trade is, in turn, a natural consequence of the

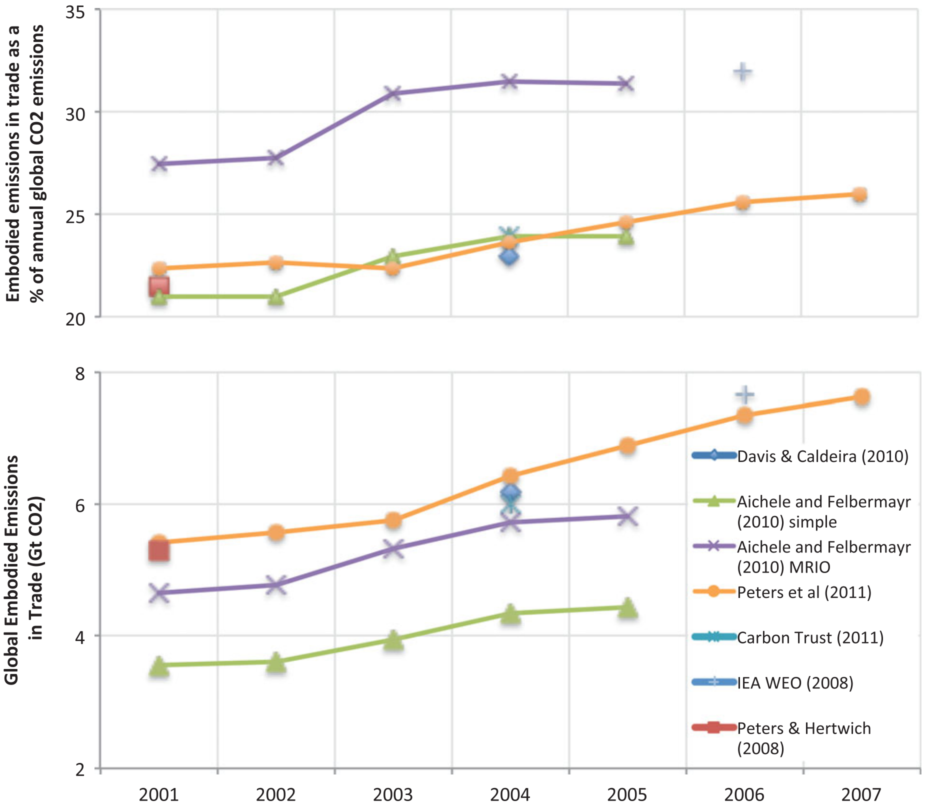

Figure 1. Embodied Emissions in Global Trade: Estimates from the Literature.

Source: Author.

Notes: The bottom graph plots the estimated global EET volumes by study, expressed in absolute volume. The top graph plots the corresponding share relative to global annual CO_2 emissions (right, secondary axis). Studies included: Aichele and Felbermayr (2010) (The emissions embodied in trade between 1995 and 2005 were reported in a previous version dated 2009, and have since been removed in updated versions), Carbon Trust (2011d), Davis and Caldeira (2010), IEA (2008), Peters and Hertwich (2008) and Peters *et al.* (2011b).

convention of conducting GHG accounting and inventory based on the *production based* approach which measures emissions using the territorial system boundary.[2]

Indeed the body of literature quantifying embodied carbon in trade has provided important evidence, highlighting that Annex I countries tend to be net importers of EET, and thus exposing the limitations of the conventional production based perspective. This literature has also prompted debates around an alternative, consumption based approach to carbon accounting (e.g. Munksgaard and Pedersen, 2001; Bastianoni *et al.*, 2004; Rodrigues *et al.*, 2006), questioning what is a fair allocation of mitigation responsibility in the presence of trade, as well as the validity, efficacy and fairness of climate change policies founded on the conventional production based emissions accounting and inventory.

As more quantitative analyses emerge, however, issues around definitions, robustness and uncertainty of EET measurement are gradually coming to light. A large variance across the estimated volumes of EET is problematic because they can be used to support different interpretations with potentially profound implications for environmental and trade policy making. For example, Yan and Yang (2010) find relatively small volumes of embodied emissions in China's imports (0.45 Gt CO_2 relative to 1.18 Gt in exports in 2005) and advocates the consumption based CO_2 accounting system on the basis of fairness. Weber (2008) on the other hand finds substantial volumes embodied in China's imports and concludes that 'if China does not want to take responsibility for its exported emissions, it must at least be held responsible for what it imports' (p. 3576).

Previous reviews of this literature have focused on methodology (e.g. Lutter *et al.*, 2008; Liu and Wang, 2009; Wiedmann *et al.*, 2009, 2011; Hertwich and Peters, 2010; Peters and Solli, 2010). Yet, syntheses of the quantitative results have been relatively few. The contradicting pictures emerging from the growing body of research suggests that it is timely for results to be subject to careful comparative evaluation. The central purpose of this paper is to compare the quantitative results reported across studies and to discuss methodological and data issues that contribute to the variability of results. In doing so, it assesses the extent to which this literature provides a consistent empirical understanding of trade embodied carbon flows. Based on these assessments, it evaluates the strengths of the conclusions and policy implications drawn in this literature.

The paper is structured as follows. Section 2 provides a typology of papers that quantify EET, including scale of analysis and estimation methodology. Section 3 then collates reported results across studies for select countries, in terms of reported volumes of embodied emissions in exports, imports, and the balance. To better understand what drives the differences in estimations across studies, Section 4 examines the various sources of uncertainty involved in EET estimation. In light of these, Section 5 examines the literature in terms of the strength of the conclusions and interpretations of the results. Section 6 offers conclusions.

2. Typologies of Quantitative Embodied Carbon Research

This review covers over 50 papers quantifying embodied carbon in trade, from both the grey and academic literature. This section provides some key typologies.

2.1 *Scales*

Quantification of embodied carbon at the *macro-scale* involves estimating the embodied emissions in imports and exports at the level of a country or a region. A key enquiry pursued at the *macro-scale* is whether a particular country is a net importer or exporter of embodied carbon emissions, and how the consumption based emissions change over time, with respect to production based emissions.

Analysis at the *meso-scale* on the other hand, entails quantifying sector level embodied carbon in trade. Analyses at this scale are often motivated by questions around mitigation in industry sectors exposed to international trade. *Micro-scale* quantification considers the embodied carbon of a product, household or a firm. Carbon footprinting of products are in this vein, typically using methods that apply life cycle assessment (LCA) procedures in relation to carbon. These include the World Resource Institute (WRI)/World Business Council on Sustainable Development (WBCSD)'s GHG Protocol, the ISO 14064 and the British Standard Institution (BSI)'s Publicly Available Specifications-2050 (PAS 2050).[3]

Tukker *et al.* (2009) notes that action at one level can have important ripple effects at another (e.g. EU climate policy applied to specific sectors may impact China's emissions as a country). Indeed, the continuum of methods that allows a broad assessment and ripple effects between the different scales, has received some attention in recent literature (e.g. Wiedmann *et al.*, 2009; Peters and Solli, 2010). Section 5 will discuss the importance of the policy context and the type of analysis conducted. This review focuses primarily on *macro-scale* analysis.

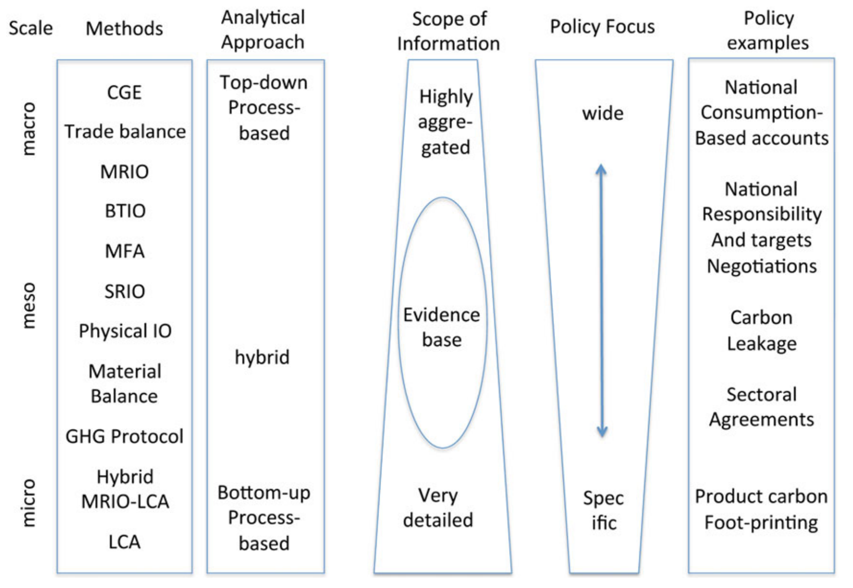

Figure 2. Methods for Calculating Embodied Emissions. *Notes:* Adapted from Wiedmann *et al.* (2009).

2.2 *Methods*

Figure 2 relates methods to scales of analysis (vertical axis), as well as policy relevance and information needs. At the *meso-* and *macro-scale*, three approaches based on environmentally extended input–output (IO) analysis[4] are widely used to calculate embodied carbon in trade: the Single Region Input–Output (SRIO); Bilateral Trade Input–Output (BTIO) which is also known as Embodied Emissions in Bilateral Trade (EEBT); and Multi-Regional Input–Output (MRIO) models. Critical distinctions between the three models can be made with regards to the system boundary used (the way the imported intermediate goods are treated), assumption about technology and model complexity.

The SRIO model takes a single country and examines the emissions associated with its *total consumption* (also termed *total demand*, including household, government and capital investment), taking account of the embodied carbon in trade with the rest of the world (ROW). By aggregating the ROW as one region, it is generally assumed under this model that the same technology is applied to production both home and abroad (the import substitution assumption). Embodied CO_2 for over 20 countries have been examined using SRIO models so far (as reviewed by Wiedmann (2009)).

The BTIO model also considers emissions associated with the *total consumption* of one country, but decomposes trade by trading partner and applies differentiated emission factors, hence relaxing the import substitution assumption. Separately representing a handful of key trading partner countries using a BTIO model has been a popular quantification strategy. The MRIO model extends the IO analysis to a multi-regional level.

A key point to note is that in both SRIO and BTIO models, all imports are allocated to *total consumption*. In contrast, the MRIO model distinguishes between imports which are directed towards *final consumption* versus those directed towards *intermediate consumption*. The latter can be directed to the production of goods for both domestic consumption and exports. Under the MRIO approach, the allocation of intermediate goods is endogenously determined to meet the final demand in each region. Thus in theory at least, this model is capable of fully capturing the re-export of goods (also termed through-trade or feedback effects).

Several method reviews have concluded that the MRIO model is the most appropriate approach for EET quantification at the country level (Liu and Wang, 2009; Peters and Solli, 2010; Rodrigues *et al.*, 2011). Indeed the MRIO model is theoretically sound and now widely used, with dedicated research groups and projects pioneering methodological developments and building databases (see Section 2.4). Its practical application is far from simple, however, and MRIO modelling has been described as a 'minefield for practitioners desiring fairly accurate numbers' (Weber, 2008, p. 22). Discussions around the multiple sources of uncertainty inherent in MRIO models are beginning to gain pace. These include data and computational requirements and the lack of methodological transparency, and will be discussed in greater detail in Section 4.

In light of the differences in system boundaries, scope and level of transparency between the methods, some authors point out that in fact BTIO and MRIO serve different purposes (e.g. Peters, 2008b). While MRIO has the potential to detail consumption-based accounts of the products consumed by a country, the more simple and transparent BTIO model is useful for trade adjusted emission inventories as the *total demand* system boundary it uses is directly comparable to the original statistical source.

Other approaches for quantifying embodied emissions shown in Figure 2 range from complex Computable General Equilibrium (CGE) models to very simple back-of-the-envelope calculations, as well as those using data expressed in physical quantities. On the complex end of the spectrum, Kainuma *et al.* (2000), using a CGE model and accounting for indirect effects such as those induced by changes in socioeconomic structures and production efficiencies, finds significantly lower EET volumes than found under MRIO analyses. On the other extreme, Wang and Watson (2008) uses a crude approach which involves multiplying China's balance of trade by the average CO_2 intensity GDP to estimate China's embodied emissions in exports (trade balance approach, or TBA).

The material balance approach improves upon the latter, by introducing sector disaggregation, drawing sector level intensity factor estimates from bottom-up or LCA studies.[5] For example, Shui and Harriss (2006) examine the carbon content of trade between United States and China from 1997 to 2003 by multiplying the value of trade by sector, with sector carbon intensities derived from the hybrid IO-LCA model (Green Design Institute, 2009).[6] The physical IO and the material flow accounting (MFA) methods use physical quantity data. The latter maps the physical flows of materials, taking account of stock and hence has a dynamic element. The key distinguishing characteristics of the different models are further discussed in Section 4 and summarized in Table 4.

2.3 *Policy vs Methodological Focus*

A distinction can be drawn between studies with an emphasis on drawing policy implications from EET quantification, and those with a stronger emphasis on pursuing methodological contributions to the literature. A stark contrast is apparent, for example, comparing Helm *et al.* (2007) and Wiedmann *et al.* (2008), both of which estimate the U.K.'s consumption based emissions for similar time periods. The former paper uses the simple trade balance approach (multiplying the U.K.'s trade balance and average CO_2 intensity of GDP), whereas the latter uses a much more detailed BTIO model with three key trading regions and 30 economic sectors. Both studies find significant growth in the U.K.'s consumption based emissions and a widening gap between production and consumption based emissions between the early 1990s and 2004.

The two studies compliment one another well: the former uses a simple method to highlight the phenomena, draw policy implications and generate debate; the latter can provide a form of verification by virtue of the fact that they use more sophisticated methods and explore sensitivity of results. The literature as a whole has a heavier emphasis on methodological discussions. Yet the above example begs the questions: to what end are embodied carbon flows quantified? And what are the requirements from decision making in the climate-trade issues? Section 5 will discuss in further detail, the various policy

Table 1. Key Research Groups in the Field of Quantifying Embodied Carbon.

Institution/Projects	Model Focus/Characteristics	Recent Outputs
CICIERO and IndEcol@NTNU and GTAP	GTAP-based MRIO, Strong policy focus	Peters and Solli (2010), Peters *et al*. (2011b), Hertwich and Peters (2009), Peters and Hertwich (2008), Peters *et al*. (2011a)
ISA, Sydney and SEI, York	Detailed SUT-based MRIO, REAP/EORA	Lenzen *et al*. (2010b), Lenzen (2011), Kanemoto *et al*. (2012), Wood and Lenzen (2009), Wiedmann *et al*. (2008, 2010), Dawkins *et al*. (2010), Lenzen *et al*. (2010a)
GDI @CarnegieMellon	US focus. Detailed MRIO using LCA data	Weber and Matthews (2007), Weber and Peters (2009), Weber and Matthews (2008)
SERI	Material extraction, EU focus. GRAM model	Giljum *et al*. (2010, 2008), Bruckner *et al*. (2010)
EXIOPOL	EU focus. Public' disaggregated global SUTs database	Tukker *et al*. (2009), Wiedmann *et al*. (2009), Moll *et al*. (2008), Lutter *et al*. (2008)
OPEN EU Project	GTAP-based water, carbon and ecological footprinting	(Hertwich and Peters, 2010)

Source: Author.

issues surrounding embodied carbon in trade. It will make a distinction between the policy questions where simple calculations suffice, and those where resolution in the embodied carbon estimates matter.

2.4 *Research Groups and Projects Pioneering MRIO Modelling*

Table 1 lists some of the key centres of research and key projects,[7] their models and their focus, along with some recent research outputs.

The symmetric IO tables and the extensions provided by the Global Trade Analysis Project (GTAP) database are widely used as a data source for multi-regional modelling for EET quantification. Researchers at the Norwegian University of Science and Technology (NTNU) played a central role in developing methods to convert the original database into full trade matrices necessary for MRIO modelling.[8] Importantly, empirical analyses using MRIO and other techniques from the NTNU constituency are often framed to address specific policy questions (e.g. Peters *et al.*, 2007; Peters, 2008a) and have made significant contributions to raise the profile of embodied carbon research in the climate debate.

The Stockholm Environment Institute (SEI) at the University of York and the Integrated Sustainability Analysis (ISA) group at the University of Sydney have also pioneered MRIO modelling in the context of environmental pressures. They have produced several analysis tools including the four region U.K.-MRIO model and the Resource and Energy Analysis Programme (REAP) to conduct scenario modelling of the emissions attributable to the U.K.'s consumption, and more recently the global EORA database (see Lenzen *et al.*, 2010a; Kanemoto *et al.*, 2012). The latter aims to achieve the maximum possible disaggregation of MRIO modelling, in terms of country, sectors, valuation margins and the number of years. They simultaneously aim to have a high level of transparency, by using a system of data standardization and automation (Wiedmann *et al.*, 2011).[9]

The research based at the Carnegie Mellon University's Green Design Institute has examined embodied emissions in U.S. trade, using a MRIO model of the United States and seven key trading partners and a time dimension. This model has a detailed breakdown of consumption groups and allows *micro-scale* analysis such as the impact of individual households' consumption on international trade and the role of different socio-economic variables.

The Sustainable Europe Research Institute (SERI) group have an emphasis on the development of indicators on material extraction versus consumption of countries and economic sectors therein, using the Global Resource Accounting Model (GRAM). This model was originally developed as part of the three year European project *petrE*.[10]

The One Planet Economy Network (OPEN) EU research project has multiple partners (including the groups mentioned here) and aims to produce academically robust national carbon, ecological and water footprint indicators, covering 113 countries using GTAP data and an integrated MRIO-footprint model. The IO data from Asian International Input–Output Table by IDE/JETRO and the World Input–Output database by University of Groningen are important resource in this literature.

EXIOPOL – a project under the EU Framework 7 programme – aims to fill gaps in the data availability for analysis on embodied carbon in trade and created SUTs with high-level geographical and sector disaggregation (130 sectors and 43 countries) and many environmental extensions (material flows, land-use, water, energy and externalities are considered, in addition to emissions), using process and LCA data to disaggregate environmentally relevant sectors.

3. Empirical Findings in the Literature

3.1 *EET Estimates at the Global Level*

Figure 1 graphs the estimated volumes of embodied carbon in annual global trade between 2001 and 2006. Most of these estimates are generated from MRIO modelling exercises, with the exception of IEA (2008) which uses the share of exports in GDP to approximate the share of carbon emissions embodied in exports.

Collectively, these estimates show that volumes of embodied carbon in global trade are significant and on a growing trend. Estimates from 2004 range between 4 Gt and 6 Gt CO_2 (roughly 20–30% of global emissions) whereas those for 2006 lie between 7 Gt and 8 Gt CO_2 (around 25–35%). Aichele and Felbermayr (2010) reports a growth rate of EET of around 50% in one decade (1995–2005). Reported estimates for more recent base years confirm this trend – Peters *et al.* (2011b) estimate 7.8 Gt in 2008.

The chart begins to illustrate the non-trivial variation in reported results. In 2004, the lower bound is set at 4.4 Gt CO_2 by Aichele and Felbermayr (2010)'s 'simple' model, and the upper bound by Davis and Caldeira (2010) at 6.2 Gt. The gap of 2.2 Gt between the upper and lower bounds is substantial – equivalent to the EU ETS's annual cap, or around 40% of Europe's CO_2 emissions in 2005.

3.2 *EET Estimates at Country Level*

Tables 2–4 compare the reported levels of emissions for China, the United States, and Japan, respectively, by year and model type, in terms of: production-based emissions; consumption-based emissions; embodied emissions in exports (EEE); the share of EEE relative to production-based emissions; embodied emissions in imports (EEI); the share of EEI relative to production-based emissions; and finally the country's balance

Table 2. EET Estimates from the Literature for China.

Author/Year	Data year	Model	CO_2 Production (Mt CO_2)	CO_2 Consumption (Mt CO_2)	EEE (Mt CO_2)	EEE (%)	EEI (Mt CO_2)	EEI (%)	BEET (%)
Weber *et al*. (2008)	1995	SRIO	3010	3150	570	19	710	24	– 5
Nakano *et al*. (2009)*”	1995	MRIO	2869	2615	318	11	64	2	9
Brukner *et al*. (2010)	1995	MRIO	2759	2152	727	26	120	4	22
Weber *et al*. (2008)	1997	SRIO	3210	3330	580	18	700	22	– 4
Yan and Yang (2010)^	1997	SRIO	3133	2957	314	10	138	4	6
Huimin and Qi (2010)^	1997	BTIO	3219	2871	513	16	165	5	11
Ahmad and Wyckoff (2003)	1997	MRIO	3068	2708	463	15	102	3	12
Huimin and Qi (2010)^	2000	SRIO	2974	2717	623	21	367	12	9
Yan and Yang (2010)^	2000	SRIO	2967	2767	350	12	150	5	7
Nakano *et al*. (2009)*”	2000	MRIO	2904	2645	387	13	128	4	9
Shimoda *et al*. (2008)	2000	MRIO	3221	2537	754	23	71	2	21
Yan and Yang (2010)^	2001	SRIO	3108	2908	380	12	180	6	6
Huimin and Qi (2010)^	2001	BTIO	2454	2271	623	25	440	18	7
Peters and Hertwich (2008)	2001	MRIO	3289	2704	803	24	217	7	18
Weber *et al*. (2008)	2002	SRIO	3620	4030	760	21	1170	32	– 11
Yan and Yang (2010)^	2002	SRIO	3441	3241	400	12	200	6	6
Pan *et al*. (2008)	2002	SRIO	3279	2656	880	27	257	29	19
Huimin and Qi (2010)^	2002	BTIO	2564	2381	733	29	550	21	7
Qi (2008) Upper*	2003	SRIO			800				
Qi (2008) Lower*	2003	SRIO			700				
Yan (2010)	2003	SRIO	4062	3662	700	17	300	7	10
Huimin and Qi (2010)^	2003	BTIO	3667	3373	1027	28	733	20	8
Wang and Watson (2007)	2004	TBA	4732	3623	1490	31	381	8	23
Qi (2008) Upper*	2004	SRIO			1200				
Qi (2008) Lower*	2004	SRIO			900				
Yan and Yang (2010)^	2004	SRIO	4847	4297	950	20	400	8	11
Huimin and Qi (2010)^	2004	BTIO	5044	4567	1393	28	917	18	9
Carbon Trust (2011)	2004	MRIO	4834	3740	1374	28	280	20	23
Davis and Caldiera (2010)	2004	MRIO	5100	3950	1430	28	279	5	23
Atkinson *et al*. (2011)	2004	MRIO	4226	3122	1393	33	290	7	26
Weber *et al*. (2008)	2005	SRIO	5030	5560	1670	33	2200	44	– 11
Yan and Yang (2010)^	2005	SRIO	5429	4699	1180	22	450	8	13
Lin and Sun (2010)	2005	SRIO	5458	4434	2441	45	2333	43	19
Lin and Sun (2010)	2005	BTIO	5458	3370	2441	45	583	11	38
Huimin and Qi (2010)^	2005	BTIO	5699	5039	1760	31	1100	19	12
Nakano *et al*. (2009)*”	2005	MRIO	4508	3921	794	18	207	5	13
Brukner *et al*. (2010)	2005	MRIO	4449	3459	1357	31	366	8	22
IEA WEO 2007	2006	%export**			1600				
Qi (2008) Upper*	2006	SRIO			1650				
Qi (2008) Lower*	2006	SRIO			1250				
Pan *et al*. (2008)	2006	SRIO	5500	3840					31
Yan and Yang (2010)^	2006	SRIO	6018	5018	1500	25	500	8	17
Huimin and Qi (2010)^	2006	BTIO	6423	5580	2163	34	1320	21	13
Yan and Yang (2010)^	2007	SRIO	6499	5362	1725	27	588	9	17
Huimin and Qi (2010)^	2007	BTIO	6672	5829	2493	37	1650	25	13

Notes: EEE% and EEI% refer to the volume of embodied emissions in exports and imports respectively, as a share of total domestic emissions. BEET% is equal to net export (EEE-EEI) relative to domestic production based annual emissions. *Reported in Ellermann *et al*. (2009). **This method uses the share of ex ports in GDP to approximate a share of emissions that are attributable to the production of export goods and services. *”Updated results obtained from authors. ^Results have been extracted from graphs presented in the papers, hence are approximate. In Huimin and Qi (2010), values have been converted from carbon to carbon dioxide.

Table 3. EET Estimates from the Literature for the USA.

Author/Year	Data Year	Model	CO_2 Production (Mt CO_2)	CO_2 Consumption (Mt CO_2)	EEE (Mt CO_2)	EEE (%)	EEI (Mt CO_2)	EEI (%)	BEET (%)
Nakano *et al*. (2009)*”	1995	MRIO	4673	4672	283	6	282	6	0
Brukner *et al*. (2010)	1995	MRIO	4170	4510	460	11	801	19	−8
Webber and Matthews (2007) MER^	1997	BTIO			450		600		
Webber and Matthews (2007) PPP^	1997	BTIO					500		
Webber and Matthews (2007) MER^	1997	MRIO			500		850		
Webber and Matthews (2007) PPP^	1997	MRIO					620		
Ahmad and Wyckoff (2003)	1997	MRIO	5421	5684	289	5	552	10	−5
Nakano *et al*. (2009)*”	2000	MRIO	5278	5400	277	5	399	8	−2
Shimoda *et al*. (2008)	2000	MRIO	6058	5797	609	10	349	6	4
Peters and Hertwich (2008)	2001	MRIO	6007	6446	499	8	937	16	−7
Webber and Matthews (2007) MER^	2002	BTIO			450		1100		
Webber and Matthews (2007) PPP^	2002	BTIO					600		
Webber and Matthews (2007) MER^	2002	MRIO			520		1400		
Webber and Matthews (2007) PPP^	2002	MRIO					800		
Weber and Matthews (2008)	2004	CES***		4693					
Webber and Matthews (2007) MER^	2004	BTIO			480		1300		
Webber and Matthews (2007) PPP^	2004	BTIO					750		
Webber and Matthews (2007) MER^	2004	MRIO			550		1800		
Webber and Matthews (2007) PPP^	2004	MRIO					1000		
Weber and Matthews (2008)	2004	MRIO		6694					
Davis and Caldeira (2010)	2004	MRIO	5800	6500	520	9	1220	21	−12
Atkinson *et al*. (2011)	2004	MRIO	4999	5561	627	13	1188	24	−11
Brukner *et al*. (2010)	2005	MRIO	4719	5973	423	9	1678	36	−27
Nakano *et al*. (2009)	2005	MRIO	5418	5762	228	4	571	11	−6

Notes: EEE% and EEI% refer to the volume of embodied emissions in exports and imports respectively, as a share of total domestic emissions. BEET% is equal to net export (EEE-EEI) relative to domestic production based annual emissions. ***An approach based on the data from the U.S. Consumer Expenditure Survey. *”Updated results obtained from authors. ^Results have been extracted from graphs presented in the papers, hence are approximate.

of embodied emissions in trade (BEET). Tables A1–3 compare similarly for the United Kingdom, Denmark and Brazil, and India, respectively.

Comparing the reported results across studies, stark discrepancies are observed, even for the ‘reference’ territorial (production-based) emissions, reflecting the different scope of emissions taken into account in the models as well as different sources of data. As shown in Table 2, for China’s production-based

Table 4. EET Estimates from the Literature for Japan.

Author/Year	Data Year	Model	CO_2 Production (Mt CO_2)	CO_2 Consumption (Mt CO_2)	EEE (Mt CO_2)	EEE (%)	EEI (Mt CO_2)	EEI (%)	BEET (%)
Kanemoto and Tonooka(2009) MER	1995	BTIO	1258	1387	147	12	276	22	−10
Kanemoto and Tonooka(2009) PPP	1995	BTIO	1258	1221	147	12	110	9	3
Ahmad and Wyckoff (2003)	1995	MRIO	1100	1287	102	9	289	26	−17
Nakano *et al*. (2009)*”	1995	MRIO	1051	1220	59	6	229	22	−16
Brukner *et al*. (2010)	1995	MRIO	978	1409	107	11	537	55	−44
Kanemoto and Tonooka(2009) MER	2000	BTIO	1308	1423	188	14	303	23	−9
Kanemoto and Tonooka(2009) PPP	2000	BTIO	1308	1251	188	14	131	10	4
Nakano *et al*. (2009)*”	2000	MRIO	1076	1214	69	6	207	19	−13
Shimoda *et al*. (2008)	2000	MRIO	1051	1134	132	13	214	41	−8
Nansai *et al*. (2008)	2000	MRIO		939			291		
Peters and Hertwich (2008)	2001	MRIO	1291	1489	187	15	385	30	−15
Davis and Caldiera (2010)	2004	MRIO	1310	1600	185	14	420	32	−18
Atkinson *et al*. (2011)	2004	MRIO	940	1200	185	20	468	50	−30
Kanemoto and Tonooka(2009) MER	2005	BTIO	1335	1450	288	22	403	30	−9
Kanemoto and Tonooka(2009) PPP	2005	BTIO	1335	1249	288	22	202	15	6
Nakano *et al*. (2009)*”	2005	MRIO	1114	1232	114	10	232	21	−11
Brukner *et al*. (2010)	2005	MRIO	1070	1450	211	20	592	55	−36

Notes: EEE% and EEI% refer to the volume of embodied emissions in exports and imports respectively, as a share of total domestic emissions. BEET% is equal to net export (EEE-EEI) relative to domestic production based annual emissions. *”Updated results obtained from authors.

emissions in 2005, the difference between the highest and lowest estimates across six studies exceeds 1 Gt (4.4 Gt and 5.7 Gt CO_2). China is no exception, for example, studies on the United Kingdom (Table A1) report varying levels of production-based emissions—in 1995 this ranged from Bruckner *et al*. (2010)’s estimate of 411 Mt CO_2, to Wiedmann *et al*. (2008)’s estimate of 593 Mt CO_2.[11]

Wider variations are found for the estimated volumes of consumption-based emissions, EEE, and EEI. This reflects the more data intensive nature of calculations, which entails more assumptions. China’s consumption-based emissions range between 3.1 and 4.6 Gt CO_2 for 2004, and between 3.4 and 5.6 Gt CO_2 in 2005.

Turning to the volume of embodied emissions in China’s exports, this quantity is of particular interest in the context of calculating national emission targets, as pressure mounts for the world’s largest emitter to undertake legally binding mitigation targets. Contrasting two studies that use MRIO models and data for 2005, Nakano *et al*. (2009) estimates 794 Mt CO_2 embodied in China’s exports (18% of China’s production-based emissions) whereas Bruckner *et al*. (2010) estimates around twice as much at 1.4 Gt (31%). As shown in Table A4, both studies use the same data—OECD input–output tables and IEA energy and emissions data—but the aggregation levels vary.[12] The former has 48 production sectors and 87 regions, whereas the latter has only 17 and 41, respectively.

Studies using SRIO models find higher volumes of embodied emissions in China’s exports. Yan and Yang (2010) report a lower-end estimate at 1.2 Gt (22%) using a SRIO approach assuming U.S. carbon

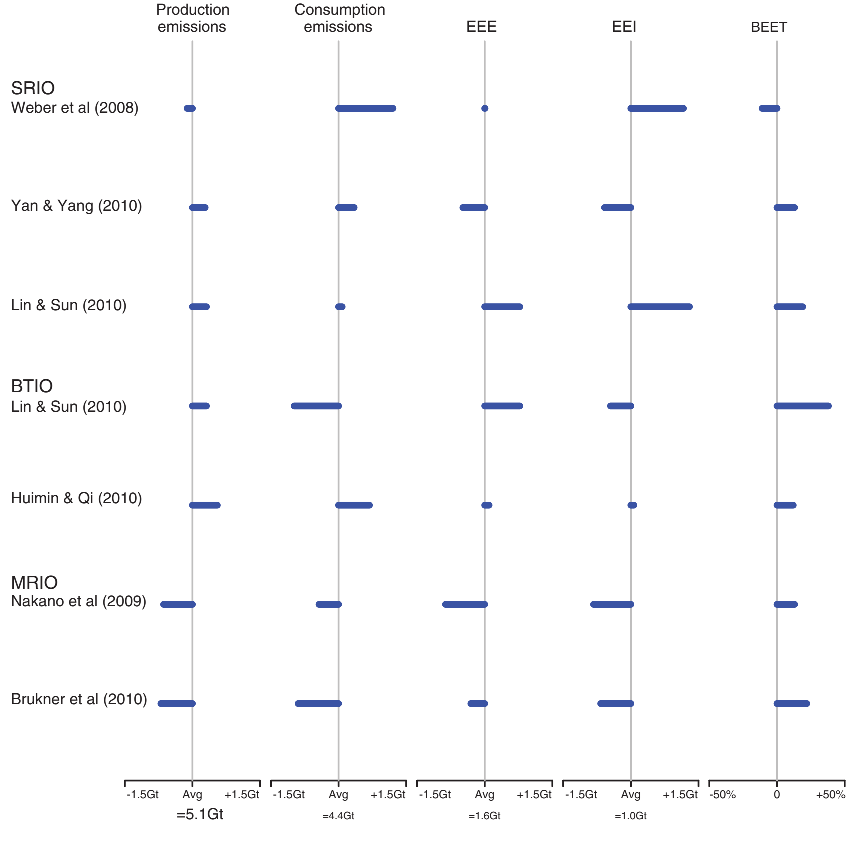

Figure 3. Comparison of EET Estimates from the Literature for China in 2005.

intensity factors for the ROW and using PPP exchange rate adjustments, whereas Lin and Sun (2010) find 2.4 Gt (45%). Such twofold differences in the estimates are not uncommon with these estimations, as the tables show. Recall that in contrast to the system boundary under the MRIO model which distinguishes between imported and domestic input materials, the EEE estimates under the SRIO and BTIO models include the emissions attributable to the production of export goods, whether the input materials are sourced domestically or from abroad.

Attention has also been drawn to the embodied emissions in China's imports, particularly as Chinese demand for intermediate goods and raw materials imports rise with consumption and industrial growth. As shown in Table 2, estimates of EEI vary considerably both within and across different model types. For 2005 in China, two studies by Weber *et al.* (2008) and Lin and Sun (2010) using the SRIO model and assuming import substitution (imports are produced with domestic technology) report significant volumes of EEI, exceeding 2 Gt CO_2 (over 40% of production-based emissions). Huimin and Ye (2010) using a BTIO model with 36 regions and differentiated technology estimates China's EEI at 1.1 Gt CO_2 (equivalent to 19%). Studies using MRIO models (and accounting for through trade) report much less: 0.2–0.4 Gt (5–8%).

To illustrate the variation across studies, Figure 3 graphically compares a set of seven results for China's embodied emissions in 2005. Focusing on the first two columns from the left, they plot for each study and model type, the deviation of the results from the average value of the seven studies, in terms of China's production-based and consumption-based emissions (averaging 5.5 Gt and 4.4 Gt, respectively, as indicated on the *x*-axis). As expected, there is wider variation in the estimates for consumption-based emissions. The next two columns show the deviation from the average for EEE and EEI estimates (while recalling that we are not comparing like for like due to difference in system boundaries). The last column plots not the deviation from the average, but the estimates of the BEET for each study. The first study by Weber *et al*. (2008) finds that China is a net importer of EET, whereas the others find that China is a next exporter (but to varying degrees). This figure highlights the discrepancies across reported results in the literature that are not small in magnitude. In this example there is not one study that stands out as performing close to the average across the five variables.

Perhaps a corollary of China's large embodied emissions in exports is the large volumes of embodied carbon in the U.S.'s imports (Table 3). Weber and Matthews (2007) use an MRIO model with both market exchange rate (MER) and purchasing power parity (PPP) assumptions and find 'bestestimates for CO_2 embodied in U.S.'s imports doubled from 0.6 to 1.3 Gt between 1997 and 2007, which represents 3–5% of world CO_2 emissions in each respective year' (p. 4877). Davis and Caldeira (2010), also using a MRIO model based on GTAP data, find large volumes of EEI in 2004 exceeding 1.2 Gt. They report 'emissions imported to the United States exceeds those of any other country or region, primarily embodied in machinery (91 Mt), electronics (77 Mt), etc.' (p. 5688). Yet again, the table shows that the differences in reported results across studies are nontrivial.

Turning now to Japan, like the United States, it is also found to be a net importer of embodied emissions in general (Table 4 and Figure 4). However, Kanemoto and Tonooka (2009) demonstrate how measuring the embodied carbon content in Japan's imports is extremely sensitive to assumptions about exchange rate. Specifically, when PPP is used to translate countries' input–output tables into Japanese yen, the volume of EEI imported into Japan (particularly the emissions embodied in imports from China which constitutes the largest sources of imports) approximately halves. This shifts the balance of EET such that Japan becomes a net exporter of EET as a result. Figures 3 and 4 collectively show that BTIO tends to overestimate EEE and MRIO underestimates EEE, an expected effect of the system boundary difference. However, in contrast to Figure 3, Figure 4 shows that different studies using MRIO models can report wide ranging results. Atkinson *et al*. (2011) and Davis and Caldeira (2010), for example, use GTAP 7 data but the former study leads to markedly conservative estimates. The authors attribute this divergence to several factors including the omission of government and household demand in their modeling, the lower share of global emissions that their model reattributes as embodied carbon in trade, and the difference in country carbon accounts data used.

Overall, the broad picture emerging from the comparison of the results reported in the set of papers studied show large and growing volumes of embodied carbon emissions in global trade. This picture underlines the deepening of the global economic integration process since the Kyoto Protocol was adopted in the 1990s. In line with the empirical trade literature (e.g. Backer and Yamano, 2008), it portrays a pattern of increasing intermediate goods trade and spatial fragmentation in production and consumption. It shows that notable and growing volumes of embodied carbon traded to and from both new and old centers of production and consumption. As summarized by Hertwich and Peters (2010): 'high density OECD countries had higher emissions embodied in imports than exports, while for materials exporters like Russia, Canada, Australia, Finland, Norway, and South Africa, the situation was the reverse. Emerging economies specializing in manufacturing, like China and India also had higher emissions in embodied exports and in imports.' (p. 16).

Yet the quantities of the embodied carbon flows at country level remain highly uncertain for most countries and years. Significant inconsistencies are found when comparing reported results across the studies surveyed as shown in this section. Why such a large range of estimates are being produced is evident from the description of the quantification approaches used; in practice many simplifications are necessary to overcome data, methodological and computational constraints in estimating embodied carbon

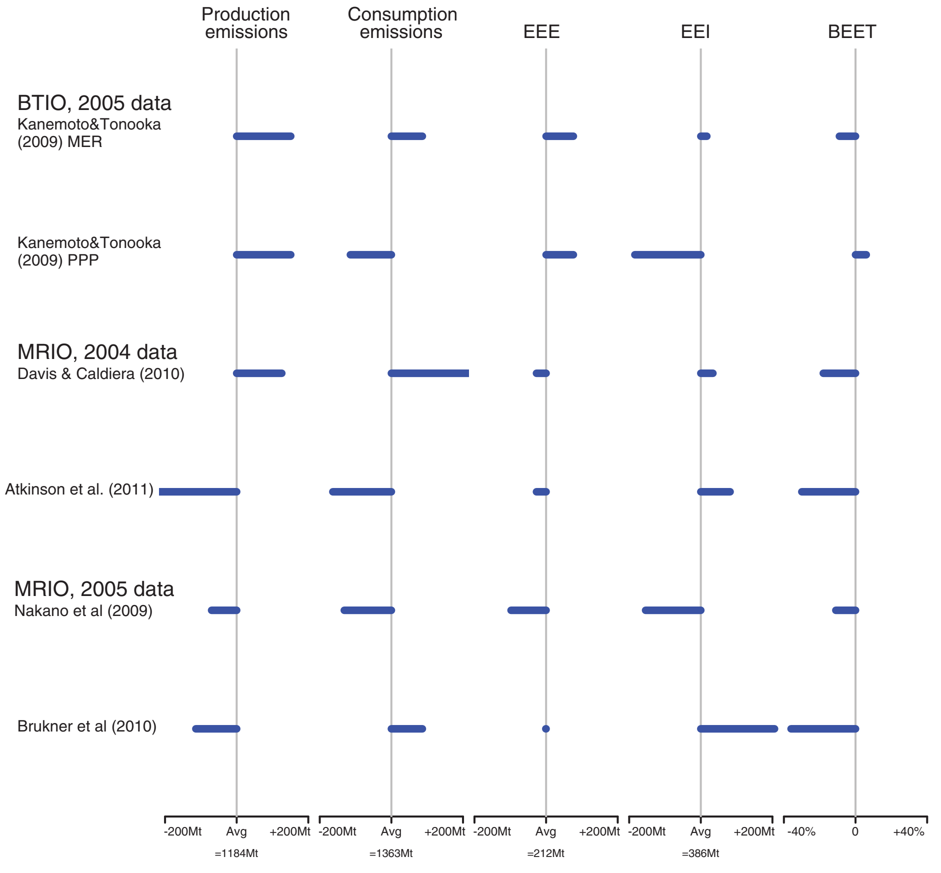

Figure 4. Comparison of EET Estimates from the Literature for Japan in 2004–2005.

flows. The next section describes these issues that undermine the robustness of existing quantification of embodied emissions.

4. Issues Contributing to Uncertainty in EET Estimation

4.1 *Generic Sources of Uncertainty*

4.1.1 *Reliability of Primary Data*

Although the data intensive nature of EET quantification is frequently noted, the reliability of the underlying statistics is often overlooked.

Economic IO data: The quality of the IO data depends on both the underlying supply-use tables (SUT)[13] and the procedure used for complying the symmetric IO table. Druckman *et al.* (2008) conducts a simple test on the impact of the IO table compilation procedure on the U.K. embodied carbon results for 1995, and finds that there is a 'carbon inconsistency' of around 13% between the two methods.[14]

The two main sources of harmonized IO tables used for environmental MRIO modelling are OECD[15] and the GTAP.[16] Additional uncertainties are introduced during the process of interlinking and harmonizing IO tables for MRIO modelling, which requires multiple assumptions and aggregation of sectors (Weber *et al.*, 2008). One paper cautions: '. . . the GTAP database has considerable uncertainty, but it is unknown how big this uncertainty is'. (Reinvang and Peters, 2008, p. 31). Directly using SUTs for MRIO modelling has been the favoured approach by some researchers to increase transparency and disaggregation (e.g. Tukker *et al.*, 2009, see Section 2.4), but this involves additional assumptions and uncertainty.

Trade data: International trade statistics are sufffer from quality issues, in part due to the voluntary nature of reporting trade data. Mirror statistics between two countries often do not match in bilateral trade data, due in part to differences between *c.i.f.* (cost insurance and freight) valuation typically used to record imports, and *f.o.b.* (free on board) valuation for exports (Lenzen *et al.*, 2004).[17] Several procedures have been developed to reconcile non-matching mirror statistics, such as GTAP's reliability index approach (Narayanan and Walmsley, 2008).[18] The degree of uncertainty associated with such methods are unknown and unverified. Moreover, additional uncertainty is induced when allocating bilateral trade into importing/exporting sectors under the MRIO, as will be discussed in Section 4.2.3.

Environmental and emissions data: For the estimation of embodied emissions, reliable emission intensity coefficients are difficult to obtain particularly at a detailed sector level and for developing countries (Liu and Wang, 2009). Peters *et al.* (2007) questions the accuracy of Chinese emission intensity data, in particular highlighting the uncertainty around the decline in energy intensity between 1996 and 2000 and whether this was real or due to under-reporting of coal consumption (see Akimoto *et al.* (2006)). Problems with the GTAP CO_2 emissions data have also been noted – the quality is poor and may vary 10–20% from UNFCCC data at the national level and may be greater at the sector level (Reinvang and Peters, 2008). Moving towards EIO-LCA hybrid models, in theory, allows for more disaggregation of sectors and the capturing of international technology differences. However in practice, the availability of LCA-based carbon intensity data poses serious restrictions (Liu and Wang, 2009).

4.1.2 *Data Coverage and Aggregation*

Geographical coverage and aggregation: Spatial disaggregation has several advantages, including improved representation of trade patterns and technology differences between countries and regions. For example, Su and Ang (2010) estimate China's embodied carbon in exports using three levels of spatial aggregation. The authors find that when aggregated at the country level using national average carbon intensities, emissions from the central coast and east coast regions (with lower carbon intensity) are overestimated whereas those from the northeast and northwest (with higher carbon intensity) are underestimated. The net effect is a drop in total CO_2 embodied in China's export as the number of regions increase.

Although a multi-regional model may serve better from the perspective of representing technology differences, there are tradeoffs to be made with other sources of uncertainty. Andrew *et al.* (2009) examines the tradeoff between complexity and accuracy in MRIO and finds that including only the most important trade partner in terms of emissions embodied in imports and aggregating the rest of the world can substantially reduce the data requirement and achieve a good approximation to more complex models.

Greenhouse gas and *sector coverage* leads to systematic differences in EET estimates, hence studies should make these explicit to aid the interpretation of the results (Lenzen and Murray, 2001). The majority of studies consider only CO_2 emissions from fossil fuel combustion and the most important differences are due to the inclusion/exclusion of process emissions (e.g. from the cement and chemicals sectors) and the service sectors. Some studies consider a much wider scope of emissions – Lenzen (1998) includes CH_4 and N_2O due to fossil fuel consumption in addition to CO_2, as well as CH_4 and C_2F_6 due to industrial processes, solvent use, agriculture, land use change, forestry and waste and fugitive emissions from fossil fuel extraction. The latter study finds that differences in GHG coverage bounds is the main explanatory factor for the difference between their own conclusion that Australia is a net exporter of embodied emissions, and that of Common and Salma (1992)'s which find Australia to have a balanced trade.

Sector aggregation: Although MRIO models overcome issues with geographical aggregation, there is a tradeoff with sector aggregation. The sector resolution of the model tends to become more coarse under MRIO models because of the process of matching data sets. This usually requires taking a lower common denominator, of the various levels of disaggregation available – United States and Japan produce tables of about 500 sectors, but Brazil has only 19. Harmonized tables tend to have around 50 sectors.[19]

Aggregation is also carried out to make the running of models computationally more manageable but can lead to errors in estimates (this is referred to as *aggregation bias* in the IO literature) because IO tables implicitly assumes one industry technology and homogeneity of firms producing for the domestic and export markets (Weisz and Duchin, 2006; Liu and Wang, 2009). This issue is particularly important for sectors with differentiated products such as the 'non-metallic minerals sector' which includes clinker, cement, as well as basic and specialized glass products. Aggregation error is also important where the sector's trade composition does not reflect the production composition, or where technology is differentiated between export-demand and domestic-demand oriented production.[20]

For macro or country level analysis, Tukker *et al.* (2009) argue that at least 100–150 sectors are necessary in order to avoid lumping together important sectors with different emission intensities, whereas Su *et al.* (2010) find that around 40 sectors are sufficient to capture the overall share of embodied emissions in a country's total exports. The extent of disaggregation necessary, is in fact contingent on the policy question at hand. For sector level analysis, the policy question at hand should also guide the level of disaggregation necessary, as the problem of heterogeneity can continue down to the product level – Maurer and Degain (2012) notes that 'even in the most finely disaggregated import and export data, there are large differences in unit values of exports and imports across countries reflecting quality differences that cannot be eliminated by disaggregation' (p. 17).

4.1.3 *Using Monetary Data*

The majority of top-down EET quantification rely on monetary data, to approximate physical flows of goods. This assumes proportionality between monetary and physical flows. This necessitates multiple assumptions which induce additional layers of uncertainty in estimating EET, particularly in sectors where product heterogeneity is important (Reinvang and Peters, 2008; Maurer and Degain, 2012).[21] Using basic prices avoids some of the issues, but only to a limited extent (Muradian *et al.*, 2002; Ahmad and Wyckoff, 2003; Weber and Matthews, 2007).[22] Quantitatively, the error associated with assuming proportionality between monetary and physical trade flows has been found to be significant – up to 40% for Australian energy and greenhouse gas multipliers (Lenzen, 1998).

In addition, the use of monetary data requires assumptions about exchange rates – using MER or PPP. Studies have repeatedly shown that the results of EET estimation are very sensitive to this assumption. As shown in Table 4, Kanemoto and Tonooka (2009) report that using PPP reduces the estimate of Japan's EEI by a third, compared with the same scenario using MER, largely due to the impact of the assumption on EEI from China. Weber and Matthews (2007) finds that 'For most developed countries, the difference between MER and PPP is relatively small, reflecting similar price levels. However, the difference between MER and PPP can be much higher for developing countries – a factor of about 2 for Mexico and 4 for China in 1997 . . . [it is] likely that the true value of EEI falls somewhere between the values calculated using MER and PPP and that the mix varies by commodity, as each commodity's output in each country includes a mix of exports and domestically consumed goods, and the exports are usually valued higher per unit than domestically consumed goods. However, in the absence of physical unit data for thousands of commodities, this uncertainty is difficult to reduce.' (p. 4879).[23]

To overcome problems with monetary data, several studies integrate physical units into the monetary core model (e.g. De Haan, 2001; Machado *et al.*, 2001; Giljum, 2005; Weisz and Duchin, 2006; Giljum *et al.*, 2010). Overall, the large sensitivity of EET estimates to assumptions used on price data suggests that studies that rely on monetary data should at minimum, test the sensitivity of results to the exchange rate assumption made.

4.2 *Methodology Specific Sources of Uncertainty*

4.2.1 *Import Substitution Assumption*

Quantification of EET using MRIO models have shown the importance of accounting for international differences in carbon emission factors (e.g. Westin and Wadeskog, 2002; Ahmad and Wyckoff, 2003; Peters and Hertwich, 2006; Gaston and Dong, 2008; Nakano *et al.*, 2009; Wilting and Vringer, 2009). Applying domestic emission intensity factors (known as the import substitution assumption or domestic technology assumption) can produce outliers. This puts forward a case for using a BTIO framework rather than SRIO, with key trade partners represented within the model.

Recent analysis has shown, however, that technology can vary significantly within countries, as well as across. This is particularly true for large countries like China (Su and Ang, 2010). Others have shown that for the estimation of EET for many countries, the use of world average emission intensities can perform well and reduce data requirements (Andrew *et al.*, 2009). This suggests that explicitly representing differentiated technology is important not for *all,* but for *key* trade partners and trade sectors.

4.2.2 *Multi-directional Feedback in Trade*

The growing evidence that cross-border supply chains have become more prevalent in the global economy (Backer and Yamano, 2008) highlight the importance of taking account of feedback effects for estimating embodied carbon flows, particularly for countries with significant processing trade activity.[24] The MRIO framework addresses this issue to some extent by separating imports into final and intermediate demand. However, this process also introduces new sources of uncertainty, such as the allocation of intermediate demand based on non-survey data, discussed next.

Quantitatively, Peters and Hertwich (2006) and Weber and Matthews (2007) both find that models with and without multi-directional feedback can lead to a difference in excess of 20% in terms countries' net embodied carbon in trade.

4.2.3 *Allocation of Imports to Intermediate and Final Demand*

To trace embodied carbon flows in trade, information is required about the spatial origin of intermediate and final imports. Further, this information must be disaggregated by consuming sector (e.g. government, investment or industry sector). Survey data for this level of information is often not available, however. This is due to the considerable cost, time and resources that are associated with conducting international industry surveys (Lenzen and Murray, 2001). To construct multi-regional models, therefore, the inter-regional intermediate trade component[25] must be estimated, based on known variables and analytical assumptions.

The standard non-survey approach used to estimate this is the *trade share method*, which uses a region's share in total global exports, and applies to all entries along the row of the imports matrix, for all using domestic industries and imported final demand vectors (Lenzen *et al.*, 2004; Peters and Hertwich, 2006; Rodrigues *et al.*, 2011).[26] Other methods are used by Rodrigues *et al.* (2011, p. 52) which uses three additional estimation approaches[27] and the project EXIOPOL which uses an alternative non-survey approach which is based on Oosterhaven *et al.* (2008), as described in Tukker *et al.* (2009). The extent of adjustment in the bilateral trade data to match the estimated intermediate trade component is unknown, however.

4.3 *Summary*

The data intensive exercise of estimating embodied carbon in trade involves multiple methodological and data issues. Researchers in this field are faced with many tradeoffs, for example between regional and sectoral detail, or between policy relevance, cost, complexity and ease of estimation as well as robustness of the results (Table 5 summarizes these tradeoffs). Although some papers test the sensitivity of EET estimates to assumptions made in their analysis, it can be said that the literature as a whole has so far paid little attention to ensuring the measurement is sufficiently robust.

Moreover, clear statements of system boundaries, underlying assumptions and methodology are noticeably absent in the literature (Wiedmann and Minx, 2008). The large variations in the estimates of country level embodied carbon in trade remains prevalent. As an increasing number of governments endorse the potential role of flow based indicators for environmental policy evaluation and decision making, it is hoped that more structured analysis of the tradeoffs, as well as the suitability of different methods and system boundaries for the evaluation of different policy issues will emerge.

Assessing the accuracy of the reported volumes of EET is difficult because the results are not always directly comparable to available survey data (the BTIO model is more comparable to national trade balances whereas MRIO models are not (Peters, 2008b)). Nonetheless, the evaluation of the different sources of uncertainty in this section suggest some minimum requirements for EET quantification analysis. For example, to address the fact that EET estimations are very sensitive to the assumption about technology, at minimum, the key trading partners' technologies should be accounted for. The import substitution assumption can lead to extreme results, hence there is a strong case for using BTIO over SRIO. Similarly, for country level estimations, it appears important to capture an appropriate amount of sector detail, such that the important trading sectors are represented. It is not clear what the optimum aggregation level is, but the literature suggests that good representation of the *key* trading partners and sectors is more important than disaggregation and detail *per se*. The appropriate level of sector disaggregation will also depend on the motivating policy question.

In terms of system boundary, for countries with a high share of processing trade, the distinction between using total and final demand is important. For such countries, it is important even in those cases where the model structure does not allow the explicit representation of the multi-directional feedback in trade (i.e. the MRIO framework is not used), that efforts is made to address the existence of high levels of re-exports. Huimin and Ye (2010), for example, apply a simple method in their study of China's embodied carbon, using the share of processing trade and applying this to embodied emissions.

Although some of the issues associated with using monetary data are difficult to overcome, one that can and should be addressed is the assumption made when applying currency exchange rates – using MER or PPP. This assumption in particular has been proven repeatedly to strongly affect EET estimation levels. Sensitivity analysis should be conducted at minimum, to make a case for robustness of the results.

5. What Does this Mean for Policy?

Embodied carbon in trade has been a subject of substantial interest in the academic and political spheres. Estimates of EET flows can inform many policy questions, which can be grouped into two broad levels. At a *higher level*, empirical understanding of embodied carbon in trade can help shape thinking around issues such as fairness in the allocation of responsibility between producers and consumers. At a *lower level*, more specific policy elements can be evaluated using EET estimates, for example, discussions around the

Table 5. The Characteristics of Existing EET Quantification Approaches.

System Boundary	Total Demand				Final Demand	
Model Type	Trade * Intensity (Physical)	Trade * Intensity (Monetary)	SRIO	BTIO/EEBT	MRIO	Hybrid MRIO-LCA
Model construction						
Transparency	Medium	Medium	High	High	Low	Low
Ability to capture time dimension	High	High	Medium	Medium	Low	Low
Level of sector disaggregation regional breakdown	High	Medium	Medium	Medium	Low	High
Capturesbilateral trade- partner info.	n	n	n	y	y (non-survey data)	y (non-survey data)
Captures differences in carbon intensities by country	n	n	n	y	y	y
Inter-sectoral trade						
domestic	n	n	y	y	y	y
international	n	n	n	n	y	y
Vulnerability to source of uncertainty						
Data issues						
error due to SUT conversion to IO	n/a	n/a	Medium	Medium	High	High
IO Harmonization (e.g. different yearbase)	n/a	n/a	n	n	High	High
generic trade data issues	Medium	Medium	Medium	Medium	High	High
structural						
Non-survey estimation of origin of sector's imports	n/a	n/a	n/a	n/a	y	y
aggregation error (sectors)	n/a	n/a	Low	Low	High	Medium
error due to lack of representation of technology differences	High	High	High	Low	Low	Medium
error due to lack of feed-back loops	High	High	High	High	Low	Low

Source: Author.

carbon leakage concerns as well a measures to address such concerns. This section summarizes the policy contexts in which embodied carbon have been measured, focusing on the *higher level*. It also evaluates the extent to which the existing literature can assist these debates, in light of the degree of uncertainty involved in the quantification as highlighted in this paper thus far.

5.1 *Insights for Higher Level Policy Elements*

Embodied carbon in trade has informed discussions around the *fair allocation of responsibility between the producers and the consumers* of emissions that are emitted throughout the multi-country processes linked by trade. There are a variety of views about the notion of fairness from a theoretical perspective. On the one extreme, some authors advocate the full attribution of responsibility to the consumer. Other authors are in favour of shared responsibility principles, recognizing that there are benefits accrued to both producers (e.g. value-added, jobs) and consumers (e.g. utility) along the chain (e.g. Kondo *et al.*, 1998; Ferng, 2003; Bastianoni *et al.*, 2004; Huimin and Ye, 2010). Lenzen *et al.* (2007) for example propose an allocation to each segment of the supply chain, depending on the share of value-added. Rodrigues *et al.* (2011) also proposes a method to distribute responsibility along the chain, suggesting an even spread.[28]

Relatedly, the empirical literature on EET has evaluated the *validity, efficacy and fairness of using the production based approach to emissions accounting* particularly *as a basis for international burden sharing agreements* such as those under the Kyoto Protocol. For example, Druckman *et al.* (2008) quantify the volume of embodied emissions in U.K.'s imports and exports and concludes 'any progress towards the U.K.'s carbon reduction targets (visible under a production perspective) disappears completely when viewed from a consumption perspective' (p. 594). Peters and Hertwich (2008) using a global MRIO model find that 'from 1996 to 2006 global CO_2 emissions have increased by 35% even though Annex I countries are still on target for a 5% reduction in 1990 GHG emissions by 2008–2012.' (p. 1406). The latter paper also evaluates how the embodied carbon balances of countries may affect their incentives to participate in international agreements on climate change. They argue that barriers to participation (as well as problems of carbon leakage) may be overcome by encouraging international coalition formation in defining emissions mitigation objectives. However, it is unclear what incentives are necessary to induce countries into such coalition building.

The *assessment of sustainable development* is another central motivation behind quantifying embodied emissions in trade at a *higher level* (e.g. Lenzen and Murray, 2001; Hong *et al.*, 2007). Resource flow based indicators for the global impacts of production and consumption activities are officially endorsed by the European Union and OECD to support environmental-economic decision making and to improve material flow and resource productivity, for example under EU's Sustainable Development Strategy (European Commission, 2004) and the EU Action Plan on Sustainable Consumption and Production (European Commission, 2008).[29] Studies quantifying EET has also helped shape thinking around the impact of trade on *natural resource dependency* and *supply chain security*. For example, Giljum *et al.* (2008) quantifies the embodied resource content of trade from a North–South perspective and finds 'trade pattern of net imports to the North is particularly visible for the EU25, which faces the strongest dependence on resource imports of all investigated world regions, in particular regarding fossil fuels and metal ores.' (p. 18). Machado *et al.* (2001) use estimates of Brazil's embodied carbon and energy to highlight the adverse impact of trade promotion policies on export dependency and energy security.

To help address these *higher level* issues, suggestions have been made for presenting the consumption based indicator alongside the usual territorial accounts to the UNFCCC (e.g. Wiedmann *et al.*, 2011). Interestingly, the international agreement on hydrofluorocarbons (HFCs) gases – Montreal Protocol – explicitly incorporates a consumption based perspective in the allocation of mitigation responsibility (Ahmad and Wyckoff, 2003). In the case of carbon, however, the methodological and data considerations discussed in Section 4 limit the practical application of consumption based accounting in climate policy in a serious way. Indeed attempts in public policy to deviate away from the conventional production based carbon accounting approach to account for EET has been met with hard opposition. For example, the Canadian 'clean energy exports credit' proposal to the Kyoto Protocol was rejected (Zhang, 2004), as was Denmark's plea to the European Union to deduct from their national accounts, the emissions for electricity which was consumed by Norwegian consumers (Lenzen *et al.*, 2004). Nonetheless, these studies put forward a strong case for incorporating consumption based principles (for example as a shadow indicator) into strategies for CO_2 mitigation, for example to evaluate the drivers of global emissions or assess the environmental impacts connected to national consumption (e.g. Peters and Solli, 2010).

5.2 *Insights for Lower Level, Detailed Policy Elements*

At a *lower level*, the literature quantifying EET makes contributions towards more specific policy issues, in particular, the discourse on carbon leakage. Peters (2008a) suggests the *distinction between 'strong' and 'weak' carbon leakage.* The former, narrower definition considers only the geographical shift in production (and its associated emissions) in direct response to climate policy, whereas 'weak' carbon leakage extends the term to cover all trade embodied emissions, whether the changes in trade level are driven by policy or by underlying economic factors, for example international differences in labour price, industrial capacity, technology, environmental standards and demand. It is argued the latter definition is more conducive to discussing possible fruitful synergies between climate change and trade policies (Peters, 2008a; Peters and Hertwich, 2008).

As an extension to the carbon leakage debate, quantifying EET has also enabled *the evaluation of policies to regulate cross-border embodied emissions*, such as border carbon measures.[30] For example, by quantifying existing EET volumes and modelling different mitigation and carbon price scenarios, Mattoo *et al.* (2009) assess the carbon leakage and welfare effect of a border tax adjustment and find potential for large international transfers due to such trade measures – in the direction from exporting to consuming countries. This suggests that countries with export industries may benefit from collecting a carbon tax domestically and redistributing the revenue internally. By highlighting the difficulty of measuring embodied carbon (e.g. Wiedmann *et al.*, 2011), the literature also suggests that border measures may in practice be based on averaged, rather than the actual carbon content of traded goods, which in turn is likely to impact incentives for importers and exporters (Monjon and Quirion, 2011).

EET quantification has also led authors to *advocate a sectoral perspective to approaching emissions mitigation*. Weber *et al.* (2008, p. 3577) and Carbon Trust (2011b) identify the inefficient and coal dominated electricity production in China as the main source of embodied carbon in consumption around the world. These authors suggest that policies promoting technology transfer in these carbon intensive industries may be more direct and effective than efforts to reduce trade (e.g. with a border carbon tax), partly because of the large indirect role of the same industries in supplying each other, and also because of the potential magnitude of problems involved in agreeing a trade treaty.

Embodied carbon quantification has been shown to be a useful tool from the perspective of identifying carbon hotspots in a global supply chain (e.g. Steinberger *et al.*, 2009; Carbon Trust, 2011a, b, c, e). Hayami and Nakamura (2007) using a case study on PV cell production in Japan and Canada finds that while it is desirable for countries to clean up production, it may be more desirable for them to ensure that the intermediate input goods they import from abroad are made with clean technology, in order to reduce the total carbon footprint of consumption.

Several studies examine the *role of the consumer in GHG mitigation and potential role for policy to promote more sustainable consumption* as an approach for countries to reduce their carbon footprints and support wider global emissions reductions. Studies on the carbon footprints of households in the United States and United Kingdom find considerable diversity in consumption habits particularly at high income levels, hence suggest large potentials for mitigation (e.g. Weber and Matthews, 2008; Druckman and Jackson, 2010). They put forward a case for incorporating consumption based perspectives for emissions mitigation policies, particularly for countries with high level of net imports of embodied carbon.

6. Conclusions

As the saying goes, 'That which can be measured can be improved'. Quantification of embodied emissions in trade has seen a resurgence in recent years, and has provided insights into a variety of policy issues surrounding the climate and trade nexus. Using several distinct approaches (notably those arising from the IO analysis as well as LCA literatures) studies have measured the embodied carbon at the level of the country, sector or city as well as firm and products.

Thanks to the increasing number of databases and studies that report EET at country level, the estimates can be compared against the methodologies and data sources used. This paper sought to provide a critical and comparative review of this literature focusing on the quantitative reported results, in order to evaluate the existing level of empirical understanding of embodied carbon flows in trade. Overall, the literature finds large and growing volumes of carbon dioxide emissions embodied in global trade. However, quantities of EET at the country level remain highly uncertain for most countries and years. Significant inconsistencies are apparent when comparing reported results across the studies surveyed. For example, estimates for emissions embodied in China's exports in 2005 range between 18% to 45% of their production emissions, whereas that embodied in China's imports in the same year range between 5% to 44%.

Sources of uncertainty in EET estimations include both data limitations and some methodological issues. The assumptions involved when using international trade in monetary terms, as well as the attribution of intermediate trade to intermediate and final consumption, are among the key problems. Although some of the issues associated with using monetary data are difficult to overcome, one that can and should be addressed is the assumption made when applying currency exchange rates – using MER or PPP. This assumption in particular has been proven repeatedly to affect EET estimation levels. Sensitivity analysis should be conducted at minimum, to make a case for robustness of the results.

Although the level of uncertainty around quantitative results from any one study remains large, collectively, they appear reasonable and useful. The application of increasingly sophisticated modelling techniques (particularly in MRIO modelling), discussions around the creation of a meta-database for MRIO data[31] as well as ongoing efforts to fill the data gaps reflect a significant level of interest invested in the potential for embodied carbon measurement for political and corporate decision making.

In fact, embodied carbon in trade arises in a variety of policy discourse surrounding climate and trade, which can be grouped broadly into two levels. At a *higher-level* of policy discussions, EET quantified at the country level has been used as a tool to deliberate issues around the fair allocation of mitigation responsibility in the presence of trade, as well as the validity, efficacy and fairness of climate change policies founded on the conventional production based emissions accounting and inventory. Explicitly incorporating consumption based principles can, in theory, improve fairness of outcomes in terms of the distribution of responsibility across producers and consumers. These principles have ben previously applied in the context of global environmental agreements on HFC gases. Yet, this paper argued that in the case of carbon,the methodological and data considerations limit the practical application of consumption based accounting in climate policy in a serious way. However, there may be a case for incorporating consumption based principles, for example as a shadow indicator, into strategies for CO_2 mitigation for certain countries with large net imports of embodied carbon.

At a *lower-level*, EET flows quantified at the sector level have facilitated discussions around the carbon leakage concerns that surrounds the implementation of unilateral climate change policies. Although a review of the sector, firm or product level quantification of EET was beyond the scope of this paper, their potential policy implications were discussed. It was found that the empirical understanding of embodied carbon at the sector or supply chain level can provide useful insights for the potential design, functioning and distributional consequences of measures to address these concerns. It also opens new questions with regards to the role of trade in decarbonizing these global supply chains, and the design of climate-trade integrated policies to support this. EET quantification at the product level suggest that policies promoting sustainable consumption can complement existing approaches to drive down emissions in a production (through to consumption) chain.

Scope remains for further research at many levels – methodological, and empirical – in the quantification of embodied carbon. Sector level analysis seem especially timely for future investigation.

Acknowledgements

I would like to thank Richard Perkins, Giles Atkinson, Philippe Quirion, Raphael Calel and two anonymous referees for many helpful comments. Financial support has come from the Grantham Foundation and the ESRC through the Centre for Climate Change Economics and Policy.

Notes

1. The world has seen a rapid growth in global merchandize trade by 460% in value terms between 1990 and 2008 (World Trade Organisation, 2012). During the same period, population and global gross domestic product (GDP) grew by 21% and 64%, respectively.
2. Production based emissions are relatively straightforward to compute and to interpret. Under the United Nations Framework Convention on Climate Change (UNFCCC), for example, countries are currently required to measure their annual emission levels 'including all green house gas emissions and removals taking place within national (including administered) territories and offshore areas over which the country has jurisdiction' (IPCC, 1996, p. 5). According to Lenzen *et al.* (2007, p. 27), this accounting norm is in line with the 'tendency of economic policy in market driven economies not to interfere with consumer's preferences that the producer centric representation is the dominant form of viewing the environmental impacts of industrial production'.
3. Reviewing these methods are beyond the scope of this paper. Pandey *et al.* (2010) discusses some of the differences across carbon footprinting methodologies.
4. The IO analysis is a top-down technique to attribute pollution or resource use to a final demand in a consistent framework (Ayres and Kneese, 1969; Leontief, 1970; Miller and Blair, 1985). Symmetric EEIO tables can be derived from national supply-use tables (SUTs) extended with environmental data. It describes the annual transaction between different sectors within an economy (the output of one sector is an input of another) and also how the sectors trade externally. IO tables are compiled by national statistics offices to map the circular flows of money, labour, goods, services, payments, wages, rents from households, firms, sectors, import, export, government and investment.
5. Mathematically, the material balance approach is a special case of a generalized physical IO formulation (Wiedmann and Lenzen, 2007) although in practice, imperfect data availability and the resulting simplifications leads to inconsistent results from the two methods. Additionally, the implication of using carbon intensity factors determined exogenously is that the results are vulnerable to LCA issues such as lack of full coverage of indirect upstream flows (system boundary issues), over and under counting and truncation errors (Lenzen, 2001).
6. This model expands the technical coefficient matrix by selectively disaggregating industry sectors in the IO table using information from process-based accounts.
7. As the table shows, some research centres and projects overlap in terms of researchers and models used.
8. This involves developing methods to approximate the off-diagonal blocks (intermediate trade flow matrix) which is necessary because the original data does not include the full trade matrices between all countries. Correction of inconsistencies in the original database is also necessary to enable MRIO modelling with GTAP data.
9. To do this, standardized matrix balancing approaches for the use of SUT in a MRIO framework have been explored to avoid the use of aggregated symmetric IO tables.

10. The model extends the monetary core model (a global, multi-regional, environmental IO model based on OECD IO tables) with a global data set on material inputs in physical units. http://www.petre.org.uk/.
11. The emissions level given by World Resource Institute's CAIT is 529 Mt CO_2.
12. A sample of 13 studies which quantify China's embodied emissions in trade for the years 2004 and 2005 are summarized in Table A4 of the Appendix. It shows several methodologies have been applied using different assumptions, with data drawn from varying sources: Chinese National Bureau of Statistics (NBS), OECD, GTAP, IEA and UN sources. Sector aggregation ranges from zero to 57, and regional aggregation from two (China vs. ROW, or rest of the world) to 113.
13. On the quality of SUTs, Thage (2005, p. 14) notes 'the size of sampling and non-sampling errors associated with the primary data on which the SUT is based, and the fact that a considerable part of the data contents of the SUT is usually obtained by grossing-up methods, extrapolations, estimates of a more or less subjective nature and even model calculations, should be taken into account when choosing the compilation method for the SIOT'.
14. The consistency check here for the estimated IO table from SUT gives the percentage difference between the left and right-hand sides of the relationship $x = (I - A)^{-1}y$ where x is output and y is final demand.
15. Used by Ahmad and Wyckoff (2003), Nakano *et al.* (2009), Bruckner *et al.* (2010), Aichele and Felbermayr (2010), Giljum *et al.* (2008).
16. GTAP is used by Kainuma *et al.* (2000), Rodrigues *et al.* (2011), Atkinson *et al.* (2011), Peters and Hertwich (2008), Wilting and Vringer (2009).
17. Other differences in reporting practises such as definition of sectors and products, minimum levels and time periods, as well as the treatment of unallocated or confidential trade also lead to discrepancies (Guo *et al.*, 2009).
18. GTAP trade data is based on UN COMTRADE and complimented with Global Trade Information Services (GTIS).
19. GTAP has 57 sectors, OECD harmonized tables have 48 sectors, and the Asian database from IDE-JETRO has 76 sectors (maximum). The EU mandates submission every five years, of harmonized tables (60 products and 60 industries), however, there are some key gaps in the data availability.
20. Lenzen *et al.* (2004) examines Denmark's EET using a 128 sector model or an aggregated 10 sector model. For the uni-directional trade scenario, the authors find that total emissions produced remains the same in the closed framework but aggregation results in a different distribution of EET across sectors. For the multi-regional trade scenario, the CO_2 embodied in domestic final demand increases, mainly because the CO_2 intensity of the aggregated 'electricity, gas and water' sector increases. This is, however, offset by the decreases of the CO_2 intensity of manufactured goods.
21. Even in the case where products are identical in a physical sense, they are often different in an economic sense in that they may be sold at different prices to different purchasers due to the existence of market power or long term price contracts, as well as differences in the way transportation costs are invoiced, or in the way taxes or subsidies on production are accounted for.
22. Basic prices tend to be more stable over time. The difference between basic prices and trade data in *f.o.b.* and *c.i.f.* is that includes tax. In Lenzen *et al.* (2004), economy-wide basic price-/*f.o.b.*/*c.i.f.* ratios in order to convert imports into basic prices. Using physical quantities would avoid uncertainties induced by this conversion.
23. Additionally, Hayami and Nakamura (2007) note that using monetary units and the industry-technology assumption means that the aggregation error is never really eliminated, even if you have a high-resolution disaggregation of sectors. This is because almost always, firms produce multiple products, but the common overhead costs get spread across the different output products.
24. This is officially defined as 'business activities in which the operating enterprize imports all or part of the raw or ancillary materials, spare parts, components, and packaging materials, and re-exports

finished products after processing or assembling these materials/parts'. In 2007, processing trade accounted for 45% of China's total international trade (Lin and Sun, 2010).

25. This is usually represented by A^{rs}, or the inverse of matrix A of intermediate consumption of imported products from region s to region r to s.
26. Using the notation from the latter, this is specified as $t_{ij}^{ab} = im_{ij}^{*b} \frac{ex_{**}^{a*}}{ex_{**}^{**}}$, where t_{ij}^{ab} describes the flow from sector i in region a to sector j in region b, $*$ denotes the sum of all values and imp and exp denote imports and exports, respectively.
27. These are: $t_{ij}^{ab} = imp_{ij}^{*b} \frac{ex_{i*}^{ab}}{ex_{i*}^{*b}}$; $t_{ij}^{ab} = ex_{i*}^{ab} \frac{im_{ij}^{ab}}{im_{i*}^{*b}}$ and $t_{ij}^{ab} = ex_{i*}^{ab} \frac{im_{*j}^{**}}{im_{**}^{**}}$.
28. They define for each country or stage k, the total downstream embodied emissions E_k^D and a symmetrical E_k^U which is the total upstream embodied emissions. They define total carbon responsibility of a country k as $E_k = \alpha E_k^U + (1 - \alpha) E_k^D$, suggesting a value of a half for α, hence an even distribution of responsibility between the up and down streams.
29. Carbon footprint indicators extend from previous literature on ecological footprinting including *carrying capacity, bioproductivity* and *land disturbance*. The ecological footprint was developed as an intuitively simple and elegant method for comparing the amount of productive land required to support the consumption of a given population indefinitely (Wackernagel *et al.*, 1993). To measure the sustainability of a given population, this land area is compared with the actual available land area.
30. Some of the recent debates can be found in Lockwood and Whalley (2008, 2010).
31. For example, the OPEN EU project (http://www.oneplaneteconomynetwork.org) and the Reunion Project (Wiedmann *et al.*, 2011).

References

Ahmad, N. and Wyckoff, A. (2003) Carbon dioxide emissions embodied in international trade of goods. OECD Science, Technology and Industry Working Papers 2003/15, OECD Publishing.

Aichele, R. and Felbermayr, G. (2010) *Kyoto and the carbon content of trade*. FZID Discussion Papers 10-2010, University of Hohenheim, Center for Research on Innovation and Services (FZID).

Akimoto, H., Ohara, T., Kurokawa, J. and Horii, N. (2006) Verification of energy consumption in China during 1996-2003 by using satellite observational data. *Atmospheric Environment* 40: 7663–7667.

Andrew, R., Peters, G.P. and Lennox, J. (2009) Approximation and regional aggregation in multi-regional input-output analysis for national carbon footprint accounting. *Economic Systems Research* 21(3): 311–335.

Atkinson, G., Hamilton, K., Ruta, G. and Van Der Mensbrugghe, D. (2011) Trade in 'virtual carbon': empirical results and implications for policy. *Global Environmental Change* 21(2): 563–574.

Ayres, R.U. and Kneese, A.V. (1969) Production, consumption, and externalities. *The American Economic Review* 59(3): 282–297.

Backer, K.D. and Yamano, N. (2008) The measurement of globalisation using international input-output tables. In OECD (ed.), *Staying Competitive in the Global Economy: Compendium of Studies on Global Value Chains*, Chapter 2 (pp. 37–64). Paris: OECD.

Bastianoni, S., Pulselli, F.M. and Tiezzi, E. (2004) The problem of assigning responsibility for greenhouse gas emissions. *Ecological Economics* 49(3): 253–257.

Bruckner, M., Giljum, S., Lutz, C. and Wiebe, K. (2010) Consumption-based CO_2 emissions and carbon leakage: results from the Global Resource Accounting Model GRAM. In *Proceedings of the International Conference on Economic Modeling (Ecomod 2010)*, Istanbul.

Carbon Trust (2011a) *International Carbon Flows: Aluminium*. CTC 790, London: Carbon Trust.

Carbon Trust (2011b) *International Carbon Flows: Automotive*. CTC 792, London: Carbon Trust.

Carbon Trust (2011c) *International Carbon Flows: Cotton*. CTC 794, London: Carbon Trust.

Carbon Trust (2011d) *International Carbon fzlows: Global Flows*. CTC 795, London: Carbon Trust.

Carbon Trust (2011e) *International Carbon Flows: Steel*. CTC 791, London: Carbon Trust.

Common, M.S. and Salma, U. (1992) Accounting for changes in Australian carbon dioxide emissions. *Energy Economics* 14(3): 217–225.

Davis, S.J. and Caldeira, K. (2010) Consumption-based accounting of CO_2 emissions. *Proceedings of the National Academy of Sciences* 107(12): 5687–5692.

Dawkins, E., Roelich, K. and Owen, A. (2010) *A consumption approach for emissions accounting – the REAP tool and REAP data for 2006*. SEI Project Report, Stockholm Environment Institute, Stockholm.

De Haan, M. (2001) A structural decomposition analysis of pollution in the Netherlands. *Economic Systems Research* 13(2): 181–196.

Druckman, A., Bradley, P., Papathanasopoulou, E. and Jackson, T. (2008) Measuring progress towards carbon reduction in the UK. *Ecological Economics* 66(4): 594–604.

Druckman, A. and Jackson, T. (2010) The bare necessities: how much household carbon do we really need? *Ecological Economics* 69(9): 1794–1804.

Ellermann, C., Oliver, P. and Xingyu, L. (2009) Competitiveness and carbon leakage: implications for China and the world. In *Proceedings of the Conference on the International Dimensions of Climate Policies*, Bern.

European Commission (2004) Communication from the Commission to the Council and the European Parliament – stimulating technologies for sustainable development: an environmental technologies action plan for the European Union. Communication COM/2004/0038 final, European Commission, Brussels.

European Commission (2008) *Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions on the sustainable consumption and production and sustainable industrial policy action plan {SEC(2008) 2110}{SEC(2008) 2111}*. Communication COM/2008/0397 final, European Commission, Brussels.

Ferng, J.-J. (2003) Allocating the responsibility of CO_2 over-emissions from the perspectives of benefit principle and ecological deficit. *Ecological Economics* 46(1): 121–141.

Gaston, C. and Dong, L. (2008) CO_2 emissions from Chinese imports. In *Proceedings of the International Input-Output Meeting on Managing the Environment*, Seville.

Giljum, S. (2005) Quantifying indirect natural resource requirements with a global multicountry input-output model. In *Proceedings of the Conference on Environmental Accounting and Sustainable Development Indicators*, The Charles University Environmental Center, Prague, pp. 183–189.

Giljum, S., Lutz, C., Jungnitz, A., Bruckner, M. and Hinterberger, F. (2008) Global dimensions of European natural resource use: first results from the Global Resource Accounting Model (GRAM). SERI Working Paper 7, Sustainable Europe Research Institute, Vienna.

Giljum, S., Polzin, C. and Lutz, C. (2010) Global implications of a european environmental tax reform. SERI Working Paper 10, Sustainable Europe Research Institute, Vienna.

Green Design Institute. Economic Input-Output Life Cycle Assessment (EIO-LCA) [Online]. (2009) Carnegie Mellon University. Available from: http://www.eiolca.net/.

Guo, D., Webb, C. and Yamano, N. (2009) Towards harmonised bilateral trade data for inter-country input-output analysis: statistical issues. OECD Science, Technology and Industry Working Papers 2009/4, OECD Publishing.

Hayami, H. and Nakamura, M. (2007) Greenhouse gas emissions in Canada and Japan: sector-specific estimates and managerial and economic implications. *Journal of Environmental Management* 85(2): 371–392.

Helm, D., Smale, R. and Phillips, J. (2007) Too good to be true? The UK's climate change record. [Online], Dieter Helm CBE. Available from: http://www.dieterhelm.co.uk/sites/default/files/Carbon_record_2007.pdf.

Herendeen, R.A. (1978) Input-output techniques and energy cost of commodities. *Energy Policy* 6(2): 162–165.

Hertwich, E.G. and Peters, G.P. (2009) Carbon footprint of nations: a global, trade-linked analysis. *Environmental Science and Technology* 43(16): 6414–6420.

Hertwich, E.G. and Peters, G.P. (2010) *Multiregional input-output database*. OPEN: EU Technical Document, One Planet Economy Network.

Hoekstra, A.Y. and Hung, P.Q. (2005) Globalisation of water resources: International virtual water flows in relation to crop trade. *Global Environmental Change* 15(1): 45–56.

Hong, L., Dong, Z.P., Chunyu, H. and Gang, W. (2007) Evaluating the effects of embodied energy in international trade on ecological footprint in China. *Ecological Economics* 62(1): 136–148.

Huimin, L. and Ye, Q. (2010) Carbon embodied in international trade of China and its emission responsibility. *Chinese Journal of Population, Resources and Environmen* 8(2): 24–31.

IEA (2008) *World Energy Outlook 2008*. Paris: International Energy Agency (OECD/IEA).

IPCC (1996) *Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories*. Workbook (Volume 2), Intergovermental Panel on Climate Change, Geneva.

Kainuma, M., Matsuoka, Y. and Morita, T. (2000). Estimation of embodied CO_2 emissions by general equilibrium model. *European Journal of Operational Research* 122(2): 392–404.

Kanemoto, K., Lenzen, M., Geschke, A. and Moran, D. (2012) Building Eora: a global multi-region input-output model at high country and sector resolution. In *Proceedings of the 20th International Input-Output Conference*, Bratislava.

Kanemoto, K. and Tonooka, Y. (2009) Embodied CO_2 emissions in Japan's international trade. *Journal of Japan Society of Energy and Resources* 30(2): 15–23.

Kondo, Y., Moriguchi, Y. and Shimizu, H. (1998) CO_2 emissions in Japan: influences of imports and exports. *Applied Energy* 59(2–3): 163–174.

Lenzen, M. (1998) Primary energy and greenhouse gases embodied in Australian final consumption: an input-output analysis. *Energy Policy* 26(6): 495–506.

Lenzen, M. (2001) Errors in conventional and input-output-based life-cycle inventories. *Journal of Industrial Ecology* 4(4): 127–148.

Lenzen, M. (2011) Aggregation versus disaggregation in input–output analysis of the environment. *Economic Systems Research* 23(1): 73–89.

Lenzen, M., Kanemoto, K., Geschke, A., Moran, D., Muñoz, P., Ugon, J., Wood, R. and Ting, Y. (2010a) A global multi-region input-output time series at high country and sector detail. In *Proceedings for the 18th International Input Output Conference*, Sydney.

Lenzen, M., Murray, J., Sack, F. and Wiedmann, T. (2007) Shared producer and consumer responsibility – theory and practice. *Ecological Economics* 61(1): 27–42.

Lenzen, M. and Murray, S.A. (2001) A modified ecological footprint method and its application to Australia. *Ecological Economics* 37(2): 229–255.

Lenzen, M., Pade, L.-L. and Munksgaard, J. (2004) CO_2 multipliers in multi-region input-output models. *Economic Systems Research* 16(4): 391–412.

Lenzen, M., Wood, R. and Wiedmann, T. (2010b) Uncertainty analysis for multi-region input-output models – a case study of the UK's carbon footprint. *Economic Systems Research* 22(1): 43–63.

Leontief, W. (1970) Environmental repercussions and the economic structure: an input-output approach. *Review of Economics and Statistics* 52(3): 262–271.

Lin, B. and Sun, C. (2010) Evaluating carbon dioxide emissions in international trade of China. *Energy Policy* 38(1): 613–621.

Liu, X. and Wang, C. (2009) Quantitative analysis of CO_2 embodiment in international trade: an overview of emerging literatures. *Frontiers of Environmental Science and Engineering in China* 3(1): 12–19.

Lockwood, B. and Whalley, J. (2008) Climate change-related border tax adjustments. CIGI Policy Brief 4, The Centre for International Governance Innovation, Ontario.

Lockwood, B. and Whalley, J. (2010) Carbon-motivated border tax adjustments: old wine in green bottles? *World Economy* 33(6): 810–819.

Lutter, S., Giljum, S., Wilting, H., Widemann, T. and Palm, V. (2008) Interim-report on the results of the evaluation of methodologies assessed with the RACER framework. EIPOT Work Package 3, Scientific Knowledge for Environmental Protection (SKEP) Environmental Impact of Trade (EIPOT) Project.

Machado, G., Schaeffer, R. and Worrell, E. (2001) Energy and carbon embodied in the international trade of Brazil: an input-output approach. *Ecological Economics* 39(3): 409–424.

Mattoo, A., Subramanian, A., van der Mensbrugghe, D. and He, J. (2009) *Reconciling climate change and trade policy*. Working Paper WP 09-15, The Peterson Institute for International Economics, Washington, DC.

Maurer, A. and Degain, C. (2012) Globalization and trade flows: what you see is not what you get! *Journal of International Commerce, Economics and Policy* 03(03), 1250019, doi:10.1142/S1793993312500196

Miller, R.E. and Blair, P.D. (1985) *Input-Output Analysis: Foundations and Extensions*. Englewood Cliffths, NJ: Prentice-Hall.

Moll, S., Acosta, J., Giljum, S., Lutter, S., Pulles, M.T. and Kuenen, J. (2008) Creating a global environmental database for input-output applications environmental extensions in the EXIOPOL project. In *Proceedings of the Intermediate Input-Output Meeting on Managing the Environment*, Seville.

Monjon, S. and Quirion, P. (2011) A border adjustment for the EU ETS: reconciling WTO rules and capacity to tackle carbon leakage. *Climate Policy* 11(5): 1212–1225.

Munksgaard, J. and Pedersen, K.A. (2001) CO_2 accounts for open economies: producer or consumer responsibility? *Energy Policy* 29(4): 327–334.

Muradian, R., O'Connor, M. and Martinez-Alier, J. (2002) Embodied pollution in trade: estimating the 'environmental load displacement' of industrialised countries. *Ecological Economics* 41(1): 51–67.

Nakano, S., Okamura, A., Sakurai, N., Suzuki, M., Tojo, Y. and Yamano, N. (2009) *The measurement of CO_2 embodiments in international trade: evidence from the harmonised input-output and bilateral trade database*. OECD Science, Technology and Industry Working Papers 2009/03, OECD Publishing.

Narayanan, G.B. and Walmsley, T.L. (2008) Global trade, assistance, and production: the GTAP 7 data base. Center for Global Trade Analysis, Purdue University.

Oki, T. and Kanae, S. (2004) Virtual water trade and world water resources. *Water Science and Technology* 49(7): 203–209.

Oosterhaven, J., Stelder, D. and Inomata, S. (2008) Estimating international interindustry linkages: non-survey simulations of the Asian-Pacific economy. *Economic Systems Research* 20(4): 395–414.

Pandey, D., Agrawal, M. and Pandey, J.S. (2010) Carbon footprint: current methods of estimation. *Environmental Monitoring and Assessment* 1(78): 135–160.

Peters, G., Webber, C. L. and Liu, J. (2006) Construction of chinese energy and emissions inventory. NTNU Program for industriell Økologi Working paper, (Rapport nr: 4/2006).

Peters, G. (2008a) Reassessing carbon leakage. In *Proceedings of the 11th Annual Conference on Global Economic Analysis "Future of Global Economy"*, Helsinki.

Peters, G.P. (2008b) From production-based to consumption-based national emission inventories. *Ecological Economics* 65(1): 13–23.

Peters, G. and Hertwich, E. (2008) CO_2 embodied in international trade with implications for global climate policy. *Environmental Science and Technology* 42(5): 1401–1407.

Peters, G. and Solli, C. (2010) *Global Carbon Footprints: Methods and Import and Export Corrected Results from the Nordic Countries in Global Carbon Footprint Studies*. Copenhagen, Norden: The Nordic Council and Nordic Council of Ministers.

Peters, G.P. and Hertwich, E.G. (2006) Pollution embodied in trade: the Norwegian case. *Global Environmental Change* 16(4): 379–387.

Peters, G.P., Weber, C.L., Guan, D. and Hubacek, K. (2007) China's growing CO_2 emissions—a race between increasing consumption and efficiency gains. *Environmental Science and Technology* 41(17): 5939–5944.

Peters, G.P., Andrew, R. and Lennox, J. (2011a) Constructing an environmentally-extended multi-regional input-output table using the GTAP data. *Economic Systems Research* 23(2): 131–152.

Peters, G.P., Minx, J.C., Weber, C.L. and Edenhofer, O. (2011b) Growth in emission transfers via international trade from 1990 to 2008. *Proceedings of the National Academy of Science* 108(21): 8533–8534.

Proops, J.L.R. (1977) Input-output analysis and energy intensities: a comparison of some methodologies. *Applied Mathematical Modelling* 1(4): 181–186.

Reinvang, R. and Peters, G. (2008) Norwegian consumption, Chinese pollution. Technical report, WWF Norway, WWF China Programme Office, Norwegian University of Science and Technology.

Rodrigues, J.A., Domingos, T., Giljum, S. and Schneider, F.A. (2006) Designing an indicator of environmental responsibility. *Ecological Economics* 59(3): 256–266.

Rodrigues, J.F.D., Domingos, T.M.D. and Marques, A.P. (2011) *Carbon Responsibility and Embodied Emissions: Theory and Measurement*. Routledge: Taylor and Francis Group.

Shimoda, M., Watanabe, T., Ye, Z. and Fujikawa, K. (2008) *An empirical study on in- terdependency of environmental load and international IO structure in the Asia-Pacific region*. Technical report, IIOA.

Shui, B. and Harriss, R.C. (2006) The role of CO_2 embodiment in US-China trade. *Energy Policy* 34(18): 4063–4068.

Steinberger, J.K., Friot, D., Jolliet, O. and Erkman, S. (2009) A spatially explicit life cycle inventory of the global textile chain. *International Journal of Life Cycle Assessmen* 14: 443–455.

Su, B. and Ang, B.W. (2010) Input-output analysis of CO_2 emissions embodied in trade: the effects of spatial aggregation. *Ecological Economics* 70(1): 10–18.

Su, B., Huang, H.C., Ang, B.W. and Zhou, P. (2010) Input-output analysis of CO_2 emissions embodied in trade: the effects of sector aggregation. *Energy Economics* 32(1): 166–175.

Subak, S. (1995) Methane embodied in the international trade of commodities: implications for global emissions. *Global Environmental Change* 5(5): 433–446.

Thage, B. (2005) Symmetric input-output tables: compilation issues. In *Proceedings of the 15th International Input-Output Conference*, Beijin.

Tukker, A., Poliakov, E., Heijungs, R., Hawkins, T., Neuwahl, F., Rueda-Cantuche, J.A.M., Giljum, S., Moll, S., Oosterhaven, J. and Bouwmeester, M. (2009) Towards a global multi-regional environmentally extended input-output database. *Ecological Economics* 68(7): 1928–1937.

Wackernagel, M., McIntosh, J., Rees, W.E. and Woollard, R. (1993) *How Big is our Ecological Footprint? A Handbook for Estimating a Community's Appropriated Carrying Capacity*. Vancouver: University of British Columbia. Discussion draft of the UBC Task Force on Planning Healthy and Sustainable Communities.

Wang, T. and Watson, J. (2008) China's carbon emissions and international trade: implications for post-2012 policy. *Climate Policy* 8: 577–587.

Weber, C.L. (2008) Uncertainties in constructing environmental multiregional input-output models. In *Proceedings of the International Input-Output Meeting on Managing the Environment*, Seville.

Weber, C.L. and Matthews, H.S. (2007) Embodied environmental emissions in U.S. international trade, 1997–2004. *Environmental Science and Technology* 41(14): 4875–4881.

Weber, C.L. and Matthews, H.S. (2008) Quantifying the global and distributional aspects of American household carbon footprint. *Ecological Economics* 66(2-3): 379–391.

Weber, C.L. and Peters, G.P. (2009) Climate change policy and international trade: Policy considerations in the US. *Energy Policy* 37(2): 432–440.

Weber, C.L., Peters, G.P., Guan, D. and Hubacek, K. (2008) The contribution of Chinese exports to climate change. *Energy Policy* 36(9): 3572–3577.

Weisz, H. and Duchin, F. (2006) Physical and monetary input-output analysis: what makes the difference? *Ecological Economics* 57(3): 534–541.

Westin, J. and Wadeskog, A. (2002) The environmental impact of swedish trade. MI71 Environmental accounts MI71ÖP0202, Statistiska centralbyrån (SCB) – Statistics Sweden.

Wichelns, D. (2001) The role of 'virtual water' in efforts to achieve food security and other national goals, with an example from Egypt. *Agricultural Water Management* 49(2): 131–151.

Wiedmann, T. (2009) A review of recent multi-region input-output models used for consumption-based emission and resource accounting. *Ecological Economics* 69(2): 211–222.

Wiedmann, T. and Lenzen, M. (2007) On the conversion between local and global hectares in ecological footprint analysis. *Ecological Economics* 60(4): 673–677.

Wiedmann, T. and Minx, J. (2008) A definition of 'carbon footprint'. In C. Pertsova (ed.), *Ecological Economics Research Trends* (pp. 1–11). Hauppauge, NY: Nova Science Publishers.

Wiedmann, T., Wilting, H., Lutter, S., Palm, V., Giljum, S., Wadeskog, A. and Nijdam, D. (2009) Development of a methodology for the assessment of global environmental impacts of traded goods and services. Final report, Scientific Knowledge for Environmental Protection (SKEP) Environmental Impact of Trade (EIPOT) Project.

Wiedmann, T., Wilting, H.C., Lenzen, M., Lutter, S. and Palm, V. (2011) Quo vadis MRIO? methodological, data and institutional requirements for multi-region input-output analysis. *Ecological Economics* 70(11): 1937–1945.

Wiedmann, T., Wood, R., Lenzen, M., Minx, J., Guan, D. and Barrett, J. (2008) *Development of an embedded carbon emissions indicator – producing a time series of input-output tables and embedded carbon dioxide emissions for the UK by using a MRIO data optimisation system*. Report to the UK Department for Environment, Food and Rural Affairs, by the Stockholm Environment Institute and the University of Sydney, Defra, London.

Wiedmann, T., Wood, R., Minx, J.C., Lenzen, M., Guan, D. and Harris, R. (2010) A carbon footprint time series of the UK – results from a multi-regional input-output model. *Economic Systems Research* 22(1): 19–42.

Wilting, H.C. and Vringer, K. (2009) Carbon and land use accounting from a producer's and a consumer's perspective – an empirical examination covering the world. *Economic Systems Research* 21(3): 291–310.

Wood, R. and Lenzen, M. (2009) Structural path decomposition. *Energy Economics* 31(3): 335–341.

World Trade Organisation (2012) WTO Statistics Database.

Yan, Y. and Yang, L. (2010) China's foreign trade and climate change: a case study of CO_2 emissions. *Energy Policy* 38(1): 350–356.

Zhang, Z.X. (2004) Open trade with the US without compromising Canada. *Journal of World Trade* 38(1): 155–182.

Appendix A

Table A1. EET Estimates from the Literature for the United Kindom.

Author/Year	Data Year	Model	CO_2 Production (Mt CO_2)	CO_2 Consumption (Mt CO_2)	EEE (Mt CO_2)	EEE (%)	EEI (Mt CO_2)	EEI (%)	BEET (%)
Druckman *et al*. (2008)	1990	SRIO	643	650					1
Druckman and Jackson (2009)	1990	SRIO	810	854					−6
Ahmad and Wyckoff (2003)	1995	MRIO	536	549	110	21	123	23	−2
Nakano *et al*. (2009)*”	1995	MRIO	488	516	58	12	86	18	−6
Brukner *et al*. (2010)	1995	MRIO	411	633	102	25	325	79	−54
Wiedmann *et al*. (2008)	1995	MRIO	593	652	222	37	281	47	−10
Nakano *et al*. (2009)*”	2000	MRIO	479	535	62	13	117	25	−12
Wiedmann *et al*. (2008)	2000	MRIO	609	681	218	36	290	48	−12
Peters and Hertwich (2008)	2001	MRIO	619	721	132	21	234	38	−17
Wiedmann *et al*. (2008)	2001	MRIO	625	732	229	37	336	54	−17
UK Carbon Trust (2006)	2002	SRIO	606	647					
Wiedmann *et al*. (2008)	2002	MRIO	610	730	222	36	343	56	−20
Helm (2007)	2003	TBA	720	1060	200	28	540	75	−47
Wiedmann *et al*. (2008)	2003	MRIO	625	764	242	39	380	61	−22
Druckman *et al*. (2008)	2004	SRIO	693	748					−8
Druckman and Jackson (2009)	2004	SRIO	730	914					−24
Davis and Caldiera (2010)	2004	MRIO	555	808	95	17	348	63	−46
Wiedmann *et al*. (2008)	2004	MRIO	631	762	242	38	374	59	−21
Carbon Trust (2011)	2004	MRIO	632	845	125	20	338	53	−34
Minx *et al*. (2009)	2004	MRIO	560	934					−27
Nakano *et al*. (2009)*”	2005	MRIO	488	549	59	12	121	25	−13
Brukner *et al*. (2010)	2005	MRIO	486	718	157	32	389	80	−48

Table A2. EET Estimates from the Literature for Denmark.

Author/Year	Data Year	Model	CO_2 Production (Mt CO_2)	CO_2 Consumption (Mt CO_2)	EEE (Mt CO_2)	EEE (%)	EEI (Mt CO_2)	EEI (%)	BEET (%)
Munksgaard and Pedersen (2001)	1994	SRIO	63	56	12	18	7	11	7
Nakano *et al*. (2009)*”	1995	MRIO	56	65	6	11	16	29	−17
Lenzen *et al*. (2004)	1997	SRIO	58	47	30	52	19	32	19
Lenzen *et al*. (2004)	1997	BTIO	58	58	38	64	37	63	1
Lenzen *et al*. (2004)	1997	MRIO	58	59	38	65	38	66	−1
Ahmad and Wyckoff (2003)	1997	MRIO	58	57	22	38	21	36	2
Peters *et al*. (2010)	1997	MRIO	76	71	37	49	32	42	7
Nakano *et al*. (2009)*”	2000	MRIO	48	60	7	14	20	41	−27
Peters and Hertwich (2008)	2001	MRIO	75	85	26	34	36	48	−14
Peters *et al*. (2010)	2001	MRIO	83	84	47	56	47	56	−1
Peters *et al*. (2010)	2004	MRIO	94	100	49	52	55	58	−6
Nakano *et al*. (2009)*”	2005	MRIO	45	61	7	16	23	51	−35

Table A3. EET Estimates from the Literature for Brazil and India.

	Author/Year	Data Year	Model	CO_2 Production (Mt CO_2)	CO_2 Consumption (Mt CO_2)	EEE (Mt CO_2)	EEE (%)	EEI (Mt CO_2)	EEI (%)	BEET (%)
Brazil	Machado *et al*. (2001)	1995	SRIO	364	351	50	10	36	13	4
	Nakano *et al*. (2009)*”	1995	MRIO	221	228	21	9	28	13	−3
	Ahmad and Wyckoff (2003)	1996	MRIO	258	266	24	9	32	12	−3
	Nakano *et al*. (2009)*”	2000	MRIO	278	283	25	9	31	11	−2
	Peters and Hertwich (2008)	2001	MRIO	321	319	63	20	61	19	1
	Atkinson *et al*. (2011)	2004	MRIO	232	230	73	31	70	30	1
	Davis and Caldiera (2010)	2004	MRIO	341	313	88	26	60	18	8
	Nakano *et al*. (2009)*”	2005	MRIO	300	303	38	13	41	14	−1
India	Mukhopadhyay (2004)	1993/1994	SRIO			37		49		negative
	Ahmad and Wyckoff (2003)	1993	MRIO	672	623	74	11	24	4	7
	Nakano *et al*. (2009)*”	1995	MRIO	723	684	51	7	12	2	5
	Brukner *et al*. (2010)	1995	MRIO	718	630	131	18	42	6	12
	Dietzendbacher *et al*. (2007)	1996/1997	SRIO	920	1047	93	10	221	24	−14
	Nakano *et al*. (2009)*”	2000	MRIO	907	877	58	6	28	3	3
	Peters and Hertwich (2008)	2001	MRIO	1025	954	134	13	64	6	7
	Atkinson *et al*. (2011)	2004	MRIO	918	876	161	18	119	13	5
	Davis and Caldiera (2010)	2004	MRIO	1360	1260	206	15	107	8	7
	Nakano *et al*. (2009)*”	2005	MRIO	1063	965	121	11	23	2	9
	Brukner *et al*. (2010)	2005	MRIO	1163	1028	277	24	142	12	12

Table A4. Summary of Methods and Data from 13 Studies Quantifying China's EET.

			Model	Data			Assumptions								
				Economic Data	Trade Data	Emisions Data	Aggregation			Coverage					
Reference	Country	Year	Structure	Source	Source	Source	No. Production Sectors	No. Regions	No. Energy Types	Process Emissions	International Transport	Service Sectors	Exchange Rates	Foreign Technology	Feed-back Loop
IEA (2008)		2006	% export in GDP		UNCTAD	IEA world energy statistics and balances		8				yes	n/a		
Wang and Watson (2008)	China vs. ROW	2004	trade balance		NBS	IEA CO_2 emissions from fuel combustion							actual terms		no
Weber *et al*. (2008)	China vs. ROW	1987–2005	SRIO	NBS for 1997–2005	GTAP for 1987–2002, China statistical year book for 2005	From Peters *et al*. (2006)	42	2	20	yes	no	yes	n/a	Import substitution assumption	no
Pan *et al*. (2008)	China vs. ROW	2002–2006	SRIO	NBS	UNCOMTRADE	China statistical yearbook	37	2						Import sub. adjusted by national avg intensity scalar	
Huimin and Ye (2010)	China vs. ROW	1997–2007	SRIO	NBS Input–output tabels of China, 2002	NBS Foreign Econ statistical yearbook 1998–2005, GTAP (imports)	IEA CO_2 emissions from fuel combustion									
Yan and Yang (2010)	China vs. ROW	1997–2007	SRIO	NBS China input–output table, 1997	Chinese statistical year book	From Peters *et al*. (2006)			20	yes	no	yes	China's exports adjusted by PPP	ROW = USA (data from Carnegie Mellon Uni.GDI)	no
Lin and Sun (2010)	China vs. ROW	2005	BTIO	NBS		China Statistical year book	15			yes				Adjusted	
Shimoda *et al*. (2008)	multiple	2000	BTIO	IED Asian international IO data	IED Asian international IO data	IEA World Energy statistics and Balances	13	10	3	no	yes	yes			yes
Nakano *et al*. (2009)	multiple	1995, 2000, 2005	MRIO	OECD harmonized IO tables	OECD bilateral trade database	IEA CO_2 emissions from fuel combustion	17	41		no	yes	yes	MER	Using like countries	yes
Bruckner *et al*. (2010)	multiple	1995–2005	MRIO	OECD harmonized IO tables	OECD bilateral trade database	IEA energy balances	48	55	4	no	yes	yes		Using like countries	yes
Peters and Hertwich (2008)	multiple	2001	MRIO	GTAP 6	GTAP 6	GTAP	57	87							yes
Davis and Caldeira (2010)	multiple	2004	MRIO	GTAP 7	GTAP 7	GTAP, matched with CDIAC	57	113		yes	no				
Atkinson *et al*. (2011)	multiple	2004	MRIO	GTAP 7	GTAP	GTAP	19	15		no			MER		yes

Notes: Partial MRIO models use country-specific emission factor and make some adjustment for re-exports.