THE LONDON SCHOOL OF ECONOMICS AND POLITICAL SCIENCE

# Carbon Emissions and Bilateral Trade

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## Declaration

I certify that the thesis I have presented for examination for the PhD degree of the London School of Economics and Political Science is solely my own work other than where I have clearly indicated that it is the work of others (in which case the extent of any work carried out jointly by me and any other person is clearly identified in it).



I warrant that this authorisation does not, to the best of my belief, infringe the rights of any third party.

## Abstract

International trade adds a thick layer of complexity to climate change mitigation efforts. Questions such as "Who is responsible for the emissions from China's export sectors?" and "Will strengthening domestic climate policy measures lead to relocation of industry and emissions to countries with lax regulation?" are intensely discussed, both in policy and academic circles.

Robust evidence on these issues remains limited, however. Many studies have quantified the volumes of embodied carbon in international trade using complex models, but the results appear very sensitive to the model specification, and conflicting results are reported across different studies. Similarly, the evidence on trade impacts from emissions reduction policies has so far relied largely on model simulations.

This thesis combines two strands of work. The first part focuses on embodied carbon quantification. It critically reviews and compares the results and methods of existing work then goes on to conduct a first quantification exercise of global embodied carbon in bilateral trade at the product level.

The second part measures the response of bilateral trade to industrial energy prices. It estimates the effect of energy price differences on bilateral trade flows, using a panel dataset covering over 80% of global merchandise trade over 16 years. These estimations are used to infer the effect of carbon price differences on trade.

The first part reveals a complex mapping of global embodied carbon flows, contrary to the simplified picture portrayed by previous studies using aggregated models. Embodied carbon is found to be particularly concentrated in certain products and in regional trade. It suggests that rather viewing it as an Annex I vs non Annex I issue, grouping countries according to patterns of production and consumption may be more relevant in discussions surrounding climate policy and trade.

The second part of the thesis finds evidence that trade tends to develop more between countries with different energy prices. However, this effect is small in magnitude and focused on a few sectors. The findings suggest that measures to 'prevent' carbon leakage may have limited impact on most sectors, and should be targeted to those most likely to face adverse trade impacts.

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All errors are obviously my own.

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# List of Abbreviations

| | |
|---|---|
| A$ | Australian Dollar |
| BEC | Broad Economic Categories |
| BEET | Balance of Embodied Emissions in Trade |
| BTIO | Bilateral Trade Input-Output |
| CEPII | Centre d'Etudes Prospectives et d'Informations Internationales |
| CES | Constant Elasticity of Substitution |
| CGE | Computable General Equilibrium |
| CIF | Cost Insurance and Freight |
| $CO_2$ | Carbon dioxide |
| $CO_2eq$ | Carbon dioxide equivalent |
| Comtrade | Commodity Trade Statistics |
| CPI | Consumer Price Index |
| CPM | Carbon Pricing Mechanism |
| CSEF | Country Specific Emission Factor |
| DTA | Domestic Technology Assumption |
| EEBT | Embodied Emissions in Bilateral Trade |
| EEE | Embodied Emissions in Exports |
| EEI | Embodied Emissions in Imports |
| EE-IOA | Environmentally Extended Input Output Analysis |
| EET | Embodied emissions in Trade |

| | |
|---|---|
| EITE | Energy Intensive Trade Exposed |
| ESDS | Economic and Social Data Services |
| ETS | Emissions Trading Scheme |
| EU | European Union |
| FOB | Free On Board |
| GDP | Gross Domestic Product |
| GFN | Global Footprint Network |
| GHG | Green House Gas |
| GMM | Generalised Method of Moments |
| Gt | Giga tonne |
| GTAP | Global Trade Analysis Project |
| HFC | Hydro Fluoro Carbons |
| HOS | Heckscher-Ohlin-Samuelson |
| HS | Harmonised System |
| IDE-JETRO | Institute of Developing Economies - Japan External Trade Organization |
| IEA | International Energy Agency |
| IOA | Input Output Analysis |
| IPCC | Intergovernmental Panel on Climate Change |
| ITT | Intra Industry Trade |
| IV | Instrumental Variable |
| LCA | Life Cycle Assessment |
| LNG | Liquefied Natural Gas |
| MER | Market Exchange Rate |
| MFA | Material Flow Accounting |
| MWh | Mega Watt hour |
| MRIO | Multi Regional Input Output |

| | |
|---|---|
| Mt | Mega tonne |
| MUV | Manufactures Unit Value |
| NTT | New Trade Theory |
| OECD | Organisation for Economic Co-operation and Development |
| OLS | Ordinary Least Squares |
| PPML | Pseudo Poisson Maximum Likelihood |
| PPP | Purchasing Power Parity |
| SDA | Structural Decomposition Analysis |
| SITC | Standard International Trade Classification |
| SIOT | Symmetric Input Output Table |
| SRIO | Single Region Input Output |
| UN | United Nations |
| UNFCCC | United Nations Framework Convention on Climate Change |
| UNIDO | United Nations Industrial Development Organisation |
| US$ | United States Dollars |
| WAEF | World Average Emission Factors |
| WITS | World Integrated Trade Solutions |
| WTO | World Trade Organisation |
| WBCSD | World Business Council on Sustainable Development |

# Chapter 1

# Introduction

## 1.1 Climate and trade - background

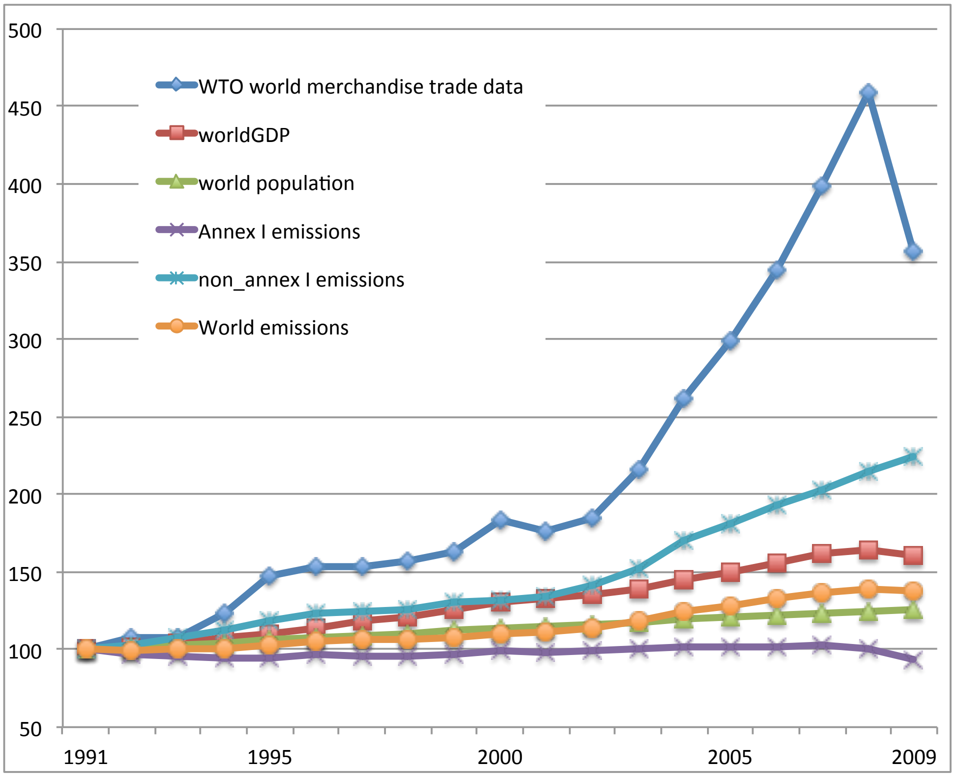
Substantial reductions of $CO_2$ emissions are necessary, in order to meet the objectives of the United Nations Framework Convention on Climate Change (UNFCCC) treaty - to stabilise greenhouse gas concentrations in the atmosphere at a level that reduces the risk of dangerous anthropogenic interference with the climate system. In dealing with the problem of climate change – a global public good problem for which responsibility over mitigation is defined nationally – nations have agreed that these reductions should be achieved in ways that recognise "common but differentiated responsibilities and respective capabilities", with greater responsibility for reducing greenhouse gas (GHG) emissions in the near term resting with industrialised economies (Annex I countries)[1] of the UNFCCC. In accordance with this principle, the Kyoto Protocol adopted a two-tiered mitigation strategy, by establishing legally binding mitigation targets for Annex I countries, whilst no such obligations are required from non-Annex I countries in the interest of their economic development.

Mitigation targets for the Annex I countries are set in terms of a percentage reduction in their GHG emissions by 2012, relative to the 1990 level.[2] Crucially, the emissions considered are the "greenhouse gas emissions and removals taking place within national territory and offshore areas over which the country has jurisdiction" as defined by the Intergovernmental Panel on Climate Change (IPCC, 1996), or those measured using the *production-based* or *territorial based* emissions accounting. Put another way, the Protocol caps the Annex I region's *direct* emissions but so far does not account for *indirect* emissions embodied in trade with non-Annex I countries.

[1] The Annex I Parties to the 1992 UNFCCC are: Australia, Austria, Belarus, Belgium, Bulgaria, Canada, Croatia, the Czech Republic, Denmark, Estonia, European Economic Community, Finland, France, Germany, Greece, Hungary, Iceland, Ireland, Italy, Japan, Latvia, Lichtenstein, Lithuania, Luxembourg, Malta, Monaco, the Netherlands, New Zealand, Norway, Poland, Portugal, Romania, Russian Federation, the Slovak Republic, Slovenia, Spain, Sweden, Switzerland, Turkey, Ukraine, United Kingdom and United States.

[2] The Annex I countries are collectively required to reduce the combined emissions of six GHGs by 5.2% below the 1990 level during the years 2008-2012 under the Kyoto Protocol.

*Figure 1.1:* World merchandise trade, $CO_2$ emissions and macro-variables from 1991 to 2009



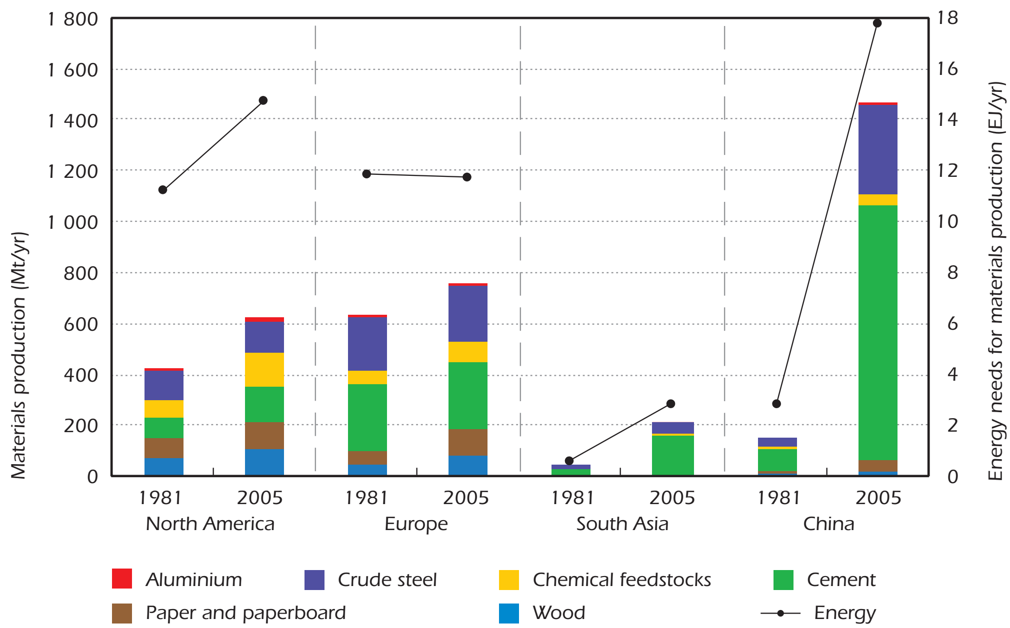

Source: Merchandise trade data from World Trade Organisation (2012), Emissions and macro-economic variables data from IEA (2011a). Notes: Indices 1991 = 100.

The decade following the adoption of the Kyoto Protocol in 1997, however, was characterised by a surge in global trade and rapid economic development in some of the non-Annex I countries. Despite high freight prices, annual growth rates of international merchandise trade far outstripped GDP, population and emissions (Figure 1.1). Industrial output grew rapidly to supply markets home and abroad, notably in China and South Asia (Figure 1.2). The growth of emerging economies fuelled structural changes in commodity markets at the international level (Reinaud, 2008) and increased integration of supply chains internationally (Figure 1.3). While Annex I output and emissions stabilised, non-Annex I emissions increased two fold between 1991 and 2009 (Figure 1.1). These developments present new challenges for the implementation of global environmental policies. This thesis will focus on two particular dimensions: embodied carbon and carbon leakage.

### 1.1.1 Embodied carbon: issues of policy efficacy and responsibility

*Production-based* accounting has obvious advantages, being straightforward to compute and transparent to interpret. However, its adequacy has been questioned as a metric upon which to build and implement fair and effective policy to reduce global emissions. Among the criticisms

*Figure 1.2:* Industrial output growth in world regions.

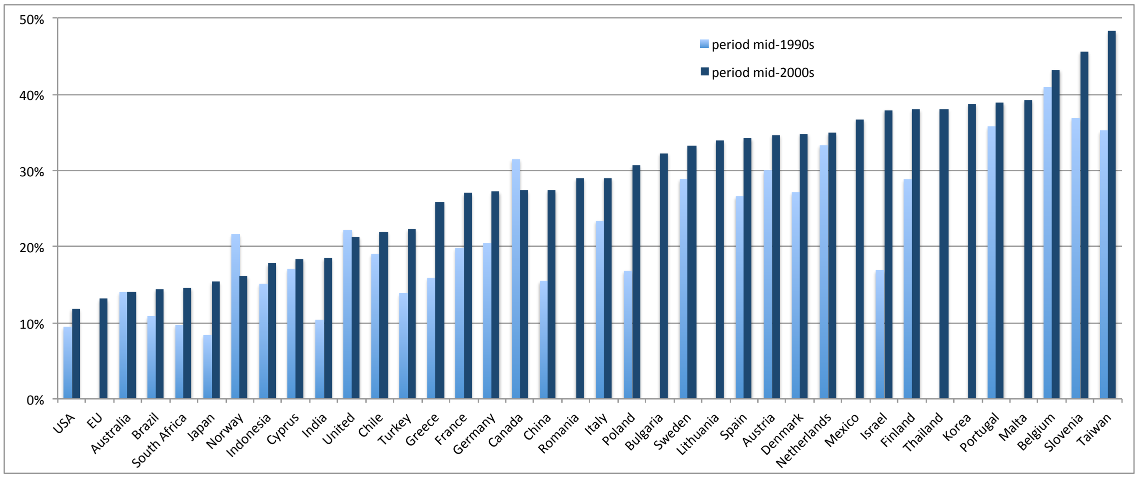



Source: Baron et al. (2007, 7). Note: China accounts for about 80% of the growth in industrial production and of the growth in industrial energy demand between 1981 and 2005.

of the Kyoto Protocol is that it did not lead to serious abatement of global emissions, by ignoring the *indirect* emissions or embodied carbon. As will be demonstrated in this thesis, the scale of embodied emissions imported by Annex I countries is sufficient to dwarf the emissions reductions under the Kyoto Protocol. The possibility of extending the scope of emissions to cover *indirect* emissions (for example by using *consumption-based* carbon accounting) has been discussed as a way to improve the environmental integrity of unilateral climate policy in a globalised world where economic activity and international trade are strongly interrelated.

Combining *production-based* with *consumption-based* metrics on one hand represents an approach to improving policy efficacy. On the other hand, it raises quite separate issues of fairness and responsibility. What fairness principles can be applied to determine the allocation of 'carbon responsibility' across agents or countries along global supply chains? Principles so far put forward range from the view that responsibility lies solely with the consumer, hence countries should be held responsible for the emissions attributable to their consumption (e.g. Kondo et al. (1998) and Munksgaard & Pedersen (2001)). More common, however, are principles of 'shared responsibility' between producers (who benefit in the form of revenues) and consumers (who derive utility). Mathematically, shared responsibility has been defined as a function of the share of value-added in the supply chain (Lenzen et al., 2007), as an even spread across all agents along the chain (Rodrigues et al., 2011) or measured against a country's 'ecological deficit' (the balance

*Figure 1.3:* Import contents of exports of goods and services by country




Source: Input-output based data obtained from OECD (2012). Notes: The import content of export grew over the period 1995 and 2005 for most countries. Data for the EU and Romania in the early period were not available. This indicator captures the degree of transnational fragmentation and vertical specialisation.

between its national emissions and sequestration via sinks) (Ferng, 2003). [3] However, many important aspects including the technical and legal feasibility of *consumption-based* metrics is yet unknown.[4]

These issues of fairness between the producer and consumer form part of a much larger ethical dilemma surrounding the climate change externality. Economic analysis suggests that reconciling ethical arguments is key to forming stable multilateral regimes – they are more likely to succeed when agents involved are able to define the gain to co-operation and share it equitably (e.g. Lange et al. (2010); Tavoni et al. (2011)). However, the distinct features of climate change – the global, intertemporal and highly inequitable nature of its causes and benefits, as well as the extensive reach of impacts on many dimensions of human well-being – necessitates looking at a broader range of ethical arguments and frameworks than in standard welfare economics (Stern, 2007). Whereas the objective of policy under welfare economics is simply to maximise the sum of social utility across individuals within one jurisdiction by a single decision maker, in contrast, climate change necessitate collective action and modelling how people in one country reacts to the impacts of their actions on those in another part of the world. This not only requires agreement on how the welfare of populations with very different standards of living should be assessed and combined in forming judgements on policy; it also requires agreement on principles of distributive fairness, upon which burden sharing agreements can be reached.[5] Fairness principles for burden sharing currently put forward include those based on rights (e.g.

[3]See Section 2.5.1 for further detail.

[4]For example, issues of sovereignty in environmental regulation remain unsolved – what can the EU do about emissions in China, however much they import from China?

[5]Moreover, welfare economics is limited to assessing only the consequences of actions, but not the process by which the outcomes are reached. Frameworks based on other notions of ethics allow for considering process.

egalitarian), freedom (e.g. sovereignty) and justice (e.g. Polluter pays) (Sen, 1999; Ringius et al., 2002). Thus, whether obligations are defined in terms of production or consumption-based emissions adds a further dimension to these debates. As trade volumes grow therefore, so does the need to carefully examine the underlying ethical judgements underpinning policies that control emissions or countries.

### 1.1.2 Carbon leakage

Increased global trade activity also raises concerns about the risk of carbon leakage – whether or not trade acts as a channel by which emissions ’escape’ from regulated to non-regulated entities. This risk exists when a global first-best solution is infeasible and unilateral actions are undertaken in an effort to protect a global public good.

There are a number of definitions of the term ’carbon leakage’, but it is most commonly perceived as the marginal emission changes in country B that is induced by climate policy in country A. The key feature of this definition is the focus on leakage that is ’policy-induced’. This definition is used in this thesis, unless stated otherwise. Applying this definition to the Kyoto Protocol context, the IPCC considers carbon leakage as “the part of emission reductions in Annex I countries that may be offset by an increase of the emissions in the non-constrained countries above their baseline levels” (IPCC, 2007), the “baseline level” being a counterfactual world without Protocol commitments. Peters (2008a) proposes a distinction between this conventional type of carbon leakage which is purely policy-induced (which he terms “strong carbon leakage”), and a broader definition of the term – “soft carbon leakage” – which includes all embodied emissions in trade (EET) regardless of whether they are induced by climate policy or other underlying economic factors that influence trade patterns. He argues that by incorporating a broader scope of emissions, the latter definition better facilitates policy discussions on the growing volumes of embodied carbon in trade. Relatedly, Jacob & Marschinski (2012) discusses the interpretation of “soft carbon leakage”and applies the Laspeyres index to decompose it into four factors: trade balance; economy-wide energy intensity; economy-wide carbon intensity of energy; and trade specialisation.

From a public economics perspective, carbon leakage relates to the effectiveness of unilateral policy. The IPCC lists two policy evaluation criteria that refer to policy effectiveness - environmental and cost effectiveness (IPCC, 2007). The latter criterion is relatively difficult to apply in practice because there is no requirement to collect information on the costs of policy. In terms of the former criterion, because climate change is caused by the global concentration of greenhouse gases, the effectiveness of a mitigation policy depends on its impact on the sum of emissions reductions in all countries world-wide, and not just on the abatement in the country implementing the policy. The net impact on global emissions can be positive or negative, due to international heterogeneity in carbon intensities of production sectors. If climate policy shifts production to a relatively less energy efficient plant in a non-regulated country, this constitutes a 'negative climate spillover' effect as global emissions will increase. Rather confusingly, this represents a 'positive carbon leakage' effect. On the other hand a 'positive climate spillover' results if production is shifted to more efficient plants (e.g. to hydro powered installations), which then constitutes 'negative carbon leakage' (see Branger & Quirion (2013)). The net effect on global emissions is zero if climate policy causes trade to shift between production plants with identical emission intensity.

The potential efficiency problem of carbon leakage makes it a key consideration when evaluating variant unilateral climate policy options. Yet providing empirical support to leakage discussions is a non-trivial task. There are many factors that influence carbon leakage from which the impact of climate policy has to be disentangled and isolated. This is characteristic of 'spillover', 'second-order' or 'side' effects of climate policy, other examples of which include impacts on energy efficiency, product prices and international spillovers such as technology diffusion. These result from the first-order impact of the $CO_2$ control policy which is typically to increase energy costs because most $CO_2$ emissions arise from the burning of fossil fuels.

Carbon leakage can also occur through channels other than trade, for example, via the energy channel. This occurs if domestic emission reductions from climate policy is offset through impacts on international energy markets – induced energy efficiency improvements decreases fossil fuel demand, lowers global fuel prices, increases fuel consumption abroad and global emissions rise. Leakage may also occur via channels of technological change and diffusion, or policy diffusion (Dröge, 2009), or the abatement resource effect (Fullerton et al., 2011). Leakage to unregulated entities is not strictly an extraterritorial phenomenon. If a policy covers only some sectors of the economy, it may shift economic activity to non-regulated sectors of varying carbon intensity. Carbon leakage channels can also be interrelated, for example trade in low-carbon technology products such as wind turbines may impact technology diffusion.

### 1.1.3 Policy relevance

These issues at the nexus of climate and trade are not only interesting for the academic researcher, but are also becoming prominent in political rhetoric and the media.

The Chinese delegation has been particularly vocal at UNFCCC negotiations, that the inclusion of a consumption-based perspective is a 'very important item to make a fair agreement' (Isenhour, 2012). It is also often reported in the Western popular media, that consumers in the rich world should take responsibility for the emissions they are 'outsourcing' to the developing countries (e.g. Watts (2009); The Economist (2011)). As the importance of embodied emissions has become increasingly clear, support for consumption-based measures from the policy sphere has surged. For example, the UK's Department for Environment, Food and Rural Affairs (Defra) produces national consumption-based accounts every year which are reported along side annual territorial emissions. The governments of Switzerland, Sweden and the Netherlands have embarked on similar endeavours. A House of Commons committee recently produced a report (Energy and Climate Change Select Committee, 2012), recommending the government should increase efforts to acknowledge the rise in emissions abroad, driven by the UK consumers' purchasing preferences, in order to achieve greater leverage over those emissions. The consultation period for the recently adopted carbon tax in Australia also explored the possibilities of a carbon-consumption tax (Access Economics, 2009), although in the end, a carbon-production tax was the favoured option.

One of the main debates surrounding climate policy to date has been related to carbon leakage risk, although discussions have centred around what leakage implies for the domestic economy, rather than for global emissions. If positive leakage occurs, not only does it undermine the policy's contribution to global carbon reduction, but it also presents a cost to the national economy if jobs, revenue or investment is shifted abroad (loss of competitiveness). With the implementation of the Emissions Trading Scheme in Europe (EU ETS), one of the key political debates concerned the impacts on European firms that compete internationally with companies located in regions with relatively lax regulations. The Directive (2009/29/EC) on the revision of the EU ETS in Phase III[6] responds to these concerns, by identifying energy intensive and trade exposed (EITE) sectors and exempting them from buying allowances through auctioning with free allocation provisions (European Commission, 2010). Similarly in the California cap-and-trade system that launched its first trading phase on 1 January 2012, potential leakage is addressed by issuing free allowances for the first compliance period (2012 to 2015). Australia's Carbon Pricing Mechanism (CPM) includes an industry assistance package [7], under which the EITE firms will receive the majority of emission allowances for free.

However, the experience with the EU ETS showed that allocating allowances for free has several economic and political drawbacks (Grubb & Neuhoff, 2006). While justified on the grounds of competitiveness and carbon leakage risks, the granting of free allowances to trade-exposed sectors constitutes an implicit subsidy. This policy not only raised issues of State Aid (Johnston, 2006), but also opened the way for large-scale lobbying, generated major inefficiencies, and resulted paradoxically in a source of windfall (on the scale of billions of Euros) for major emitters (Sandbag, 2011) undermining the credibility of the scheme. Governments are seeking better ways to address potential carbon leakage that avoids over-compensation and inefficiencies in the system. There has been active discussions about combining trade-based anti-leakage measures with reduced free allocation (greater share of allowances allocated through auctioning), particularly in Europe.[8] In the US, the requirement for importers to purchase emission allowances has been discussed as a key element of prospective climate change legislation. It was included in the now defunct "American Clean Energy and Security Act of 2009" (Waxman-Markey-Bill Section 766). The Bingaman/Specter bill (S. 1766 "Low Carbon Economy Act") included a weak form

[6] 2013 to 2020.

[7] This has been rebranded as the "Jobs and Competitiveness Program"

[8] The academic literature has examined legal aspects (e.g. Ismer & Neuhoff, 2007; Lockwood & Whalley, 2010; Holzer, 2010; Tamiotti, 2011; Zhang, 2010; Pauwelyn, 2007) and trade impacts (e.g. Monjon & Quirion, 2011; Dong & Whalley, 2010; Gros & Egenhofer, 2011; Fischer & Fox, 2009).

of border adjustment by requiring importers to have emissions permits when the emissions in the unregulated (or underregulated) producing country sector increase above a baseline level (Fischer & Fox, 2009). However, such anti-leakage trade measures are highly sensitive politically. In a globalising world where there is competition between countries to gain export shares, attract foreign direct investment and retain manufacturing sector jobs, any trade provisions can be seen to have a strategic role.

### 1.1.4 Existing research

There is strong agreement in the literature that carbon dioxide emissions from the production of traded goods and services are non-trivial (around 30% of global carbon emissions in 2006) and that they are growing in importance over time.[9] There is less agreement in the empirical literature, about the relationship between climate change policy and trade patterns.

Embodied carbon in trade has been quantified on several scales thus far. Many studies focus on an individual country (e.g. Druckman et al., 2008; Ferng, 2003; Bicknell et al., 1998; Cruz, 2002; Lenzen & Murray, 2001; Machado et al., 2001; Sánchez-Chóliz & Duarte, 2004; Nijdam et al., 2005; Westin & Wadeskog, 2002),[10] and quantify embodied emissions in imports, exports, and trade balance. Several quantifications in time-series have also appeared, for example for China (Peters et al., 2007; Yan & Yang, 2009; Huimin & Ye, 2010) as a net exporter of embodied carbon and the UK (e.g. Wiedmann et al., 2010; Baiocchi & Minx, 2010; Helm et al., 2007) as a net importer. Several global analyses using multi-regional input-output (MRIO) modelling have quantified macro-level flows of embodied carbon (e.g. Bruckner et al., 2010; Nakano et al., 2009; Davis & Caldeira, 2010; Davis et al., 2011; Peters et al., 2011b; Wilting & Vringer, 2009). On the other extreme, embodied carbon has also been assessed at the level of a firm or product (e.g. Carbon Trust, 2011a,c; Hayami & Nakamura, 2007; Steinberger et al., 2009). Whilst some papers test the sensitivity of results to the underlying assumptions, it can be said that the literature as a whole has so far paid limited attention to the robustness of the measurement of embodied carbon. Discussions around the multiple sources of uncertainty inherent in existing methodologies, and the inconsistencies in the reported estimates across studies are only recently beginning to gain pace (e.g. Wiedmann et al., 2011; Ellermann et al., 2009; Kenny & Gray, 2009).

Analyses of carbon leakage and industrial competitiveness impacts of climate policy, have thus far largely relied on *ex-ante* simulation approaches using general or partial equilibrium modelling (e.g. Babiker, 2005; Gerlagh & Kuik, 2007; Burniaux & Martins, 2000). An extremely wide range of leakage rates have been reported from these studies (from -14% to 130%), although the central estimates fall into smaller ranges (5% to 30%) (Dröge, 2011; Lanz et al., 2011). Recently, some empirical analyses examining the historical relationship between energy price and trade, has

[9] Figure 2.1 on page 29 collates estimates across studies.

[10] Over 40 of such studies have been found covering the USA, China, India, Japan, UK, Brazil, Italy, Taiwan, Portugal, Denmark, Norway, Sweden, Australia, New Zealand, the group of Eastern European countries and the Netherlands to name but a few.

helped to address the profound empirical shortcoming in this area (Aldy & Pizer, 2011, 2012; Mulatu et al., 2010; Gerlagh & Mathys, 2011). Exploiting the positive relationship between energy and carbon costs for industry provides a way to conduct *ex-post* analysis on the impact carbon policy on trade. More studies in this new line of analysis, as well as methodological and data improvements, will help strengthen the empirical basis to these issues.

## 1.2 Aims and methods

This research aims to make contributions to the ongoing policy debates described above. Two separate but related strands of research are conducted: the first on the quantification of embodied carbon and the second strand on the trade impacts of asymmetric climate policy. Both strands aim to offer new insights from detailed empirical analyses.

### 1.2.1 Embodied carbon in trade: A survey of the empirical literature

Measuring embodied emissions in trade has seen a resurgence in recent years. Chapter 2 provides a critical and comparative review of this literature and evaluates the existing level of empirical understanding of embodied carbon flows in trade. It subjects the quantitative results reported in this literature to careful comparative evaluation, and discusses methodological and data issues that contribute to the variability of results. In doing so, it assesses the extent to which this literature overall provides a consistent empirical understanding of embodied carbon flows.

Several methodological reviews of the literature have already been conducted (e.g. Lutter et al., 2008; Liu & Wang, 2009; Turner et al., 2007; Wiedmann et al., 2007), but this review focuses on the quantitative results reported by the studies. In doing so, it exposes for the first time, the magnitude of the discrepancies in the estimates of embodied carbon across the numerous quantifications conducted to date. Based on the assessment of the range of available EET estimates and the sources of uncertainty in estimation, it evaluates the strength of the conclusions drawn for policy discussions surrounding the climate and trade nexus.

### 1.2.2 Product-level embodied carbon flows in bilateral trade

As increasingly complex modelling approaches are used to quantifying embodied carbon in trade, the lack of transparency and disaggregation have been identified as some of the key weaknesses. The research presented in Chapter 3 represents a first quantification exercise of global embodied carbon in bilateral-trade at the product level. The objective is to gain insights into the nature of the flows that were previously masked when using aggregated models.

The detailed quantification of embodied carbon flows is conducted by using the material balance approach, whereby bilateral trade flows expressed in physical quantities are multiplied by product pollution intensities. This involved an extensive data search, collecting product carbon intensity factors from multiple data sources (Life Cycle Assessment databases and scientific studies). Although it has limitations of its own, this method overcomes a number of key sources of uncertainty in existing studies, identified in Chapter 2. It also represents a transparent approach to obtaining detailed product level embodied carbon flow estimates.

Covering bilateral trade between 195 countries and broken down at the level of 970 products, this detailed mapping of EET flows represents a first of its kind. This database of embodied carbon, as well as the database on product level carbon intensities built during the process will be made public upon completion of this thesis. The data has already been proved useful for research at the London School of Economics, for detailed firm and product level analysis of the impact of climate policy on innovation and trade.

### 1.2.3 Asymmetric industrial energy prices and international trade

As countries implement carbon pricing policies of differing ambition and speed, there is considerable interest around the potential impacts on international trade patterns. Whilst carbon pricing is a nascent phenomenon, Chapter 4 uses an innovative strategy to understand the potential impacts of asymmetric carbon prices. Namely, it statistically tests the hypothesis that heterogeneous industrial energy price impacts trade flows, using observed data and econometric methods. The results can be used to infer the effects of carbon pricing on future trade patterns, owing to the close relationship between energy costs and carbon costs for industrial sectors. As mentioned, few quantitative studies of carbon leakage exist to date outside of those using *ex-ante* simulation approaches. The aim of this chapter is to help fill this empirical gap.

The analysis uses dynamic panel data methods within a gravity framework, and is applied to a rich dataset comprising bilateral trade flows and industrial energy prices for 66 sectors across 51 countries (covers countries with varying levels of economic development), for the years 1996-2011. The model specifications take into account the dynamic processes which are important yet often neglected in trade analysis, as well as the issue of zero values in the dependent variable. Using the gravity model of trade framework allows explicit testing of the effect of asymmetries in energy prices between trading partners on bilateral exports.

Analysis of carbon leakage using econometric methods is important because it is possible to scrutinise the statistical relationship, by subjecting the results to multiple checks. This chapter strives to assesses the robustness of the estimation results rigorously. It asks whether the results are driven by the underlying theory, by comparing the results across different model specifications. It also assesses the sensitivity of results to underlying assumptions and variable definitions, and conducts tests to reduce the possibility that the results are driven by alternative factors. By subjecting estimates to these tests, it aims to understand the degree to which they are statistically robust. Sector variation of the trade impact is also explored, by estimating coefficients separately for different sector groups.

### 1.2.4 Net embodied carbon effects from carbon pricing policies

Any unilateral carbon pricing measure, invariably raises concerns about carbon leakage. Before incorporating provisions to address leakage into the policy design, one must ask whether the unilateral carbon price is likely to result in substantial leakage in the first place. Chapter 5 combines findings from Chapters 3 and 4 and applies these results, in order to assess the magnitude of the potential leakage problem, and contributes to how carbon leakage is treated in the policy discourse.

Three steps are involved: The magnitude of the effect of energy price on trade is assessed, using the examples of UK imports from South Korea, French imports from and Indonesia and finally by generalising these examples. Simulations are conducted to predict the near-term impact of carbon prices on bilateral import and export levels, using the estimation results from Chapter 4, for Australia's CPM and a hypothetical unilateral carbon price in the US. The near-term effects on trade are then converted to embodied carbon terms, for the case of Australia's CPM.

## 1.3 Structure of this thesis

This thesis is presented as four research papers and structured as two parts: (I) Embodied carbon in trade and (II) Carbon leakage impacts. An additional final chapter provides a synthesis of the four papers, highlights the key findings and policy implications, and offers suggestions for future research.

# Part I

# Embodied carbon in trade

# Chapter 2

# Embodied carbon in trade: A survey of the empirical literature

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## 2.1 Introduction

To what extent do trade and consumption contribute to rising global greenhouse gas (GHG) emissions? Will strengthening domestic climate policy lead to real reductions in GHG emissions or to the 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?

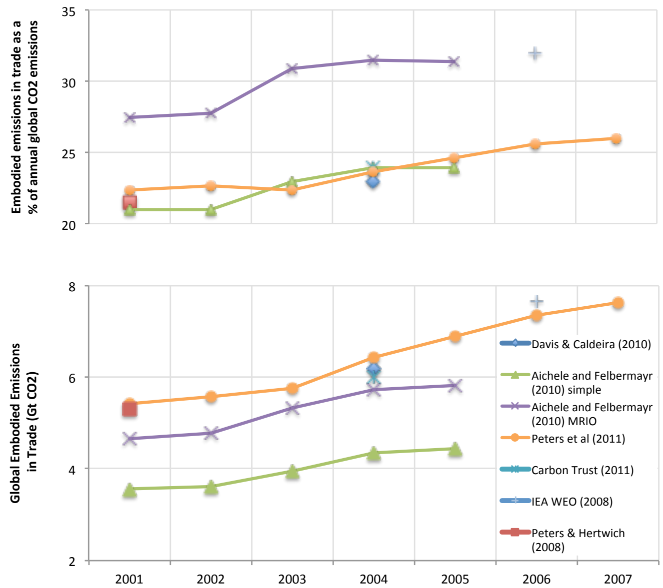
In an effort to provide empirical support to such policy debates around the design of GHG mitigation policies for industry emissions and the wider environmental impacts of consumption, there has been a recent boom in the literature which quantitatively examines the embodied carbon content of trade. Typically, 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; Hoekstra & Hung, 2005; Oki & Kanae, 2004), methane (Subak, 1995), energy (Proops, 1977; Herendeen, 1978) and land use (Lenzen & Murray, 2001).

Studies on embodied carbon have thus far found large and growing volumes of embodied emissions in trade (EET) (Figure 2.1), in line with the growth in global trade volumes[1] and

[1] As shown in Figure 1.1 on page 17, the world has seen a rapid growth in global merchandise trade by 460% in

*Figure 2.1:* 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 annal $CO_2$ emissions. Studies included: Aichele & 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 & Caldeira (2010); IEA (2008); Peters & Hertwich (2008); Peters et al. (2011b)

international integration of supply chains over the past decade. Studies have found some 4-6Gt of $CO_2$ embodied in global trade in 2004 (equivalent to 15-25% of annual GHG emissions) and 7.8Gt 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 convention of conducting GHG accounting and inventory based on the *production based* approach which measures emissions using the territorial system boundary.[2] Whilst this approach has obvious advantages being relatively straightforward to compute and to interpret, the body of literature quantifying embodied carbon in trade has also exposed the limitations of the conventional production based perspective. For example, by showing that Annex I countries tend to be net importers of EET, it highlights the efficacy problem of policies that fail to incorporate indirect emissions as well as the need to resolve issues around responsibility over manufacturing sector emissions in the presence of trade (as discussed

value terms between 1990 and 2008. During the same period, population and global GDP grew by 21% and 64% respectively.

[2]According to Lenzen et al. (2007, pp. 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".

in Section 1.1.1).

This literature has prompted debates around alternative, consumption based approaches to carbon accounting (e.g. Munksgaard & Pedersen, 2001; Bastianoni et al., 2004; Rodrigues et al., 2006). Yet as more quantitative analyses emerge, issues around data quality, 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 & Yang (2009) find relatively small volumes of embodied emissions in China's imports (0.45Gt $CO_2$ relative to 1.18Gt 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; Wiedmann et al., 2009; Hertwich & Peters, 2010; Liu & Wang, 2009; Wiedmann et al., 2011; Peters & 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 Chapter 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 Chapter is structured as follows. Section 2.2 provides a typology of papers that quantify EET, including scale of analysis and estimation methodology. Section 2.3 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 2.4 examines the various sources of uncertainty involved in EET estimation. In light of these, Section 2.5 examines the literature in terms of the strength of the conclusions and interpretations of the results. Section 2.6 offers conclusions.

## 2.2 Typologies of quantitative embodied carbon research

This review covers over 100 papers quantifying embodied carbon in trade, from both the grey and academic literature. This section provides some key typologies including scale of analysis and methodology used. Another key distinguishing feature is between studies with a methodological or a policy focus.

### 2.2.1 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 & Solli, 2010). Section 2.5 will discuss the importance of the policy context and the type of analysis conducted. This review focuses primarily on *macro-scale* analysis.

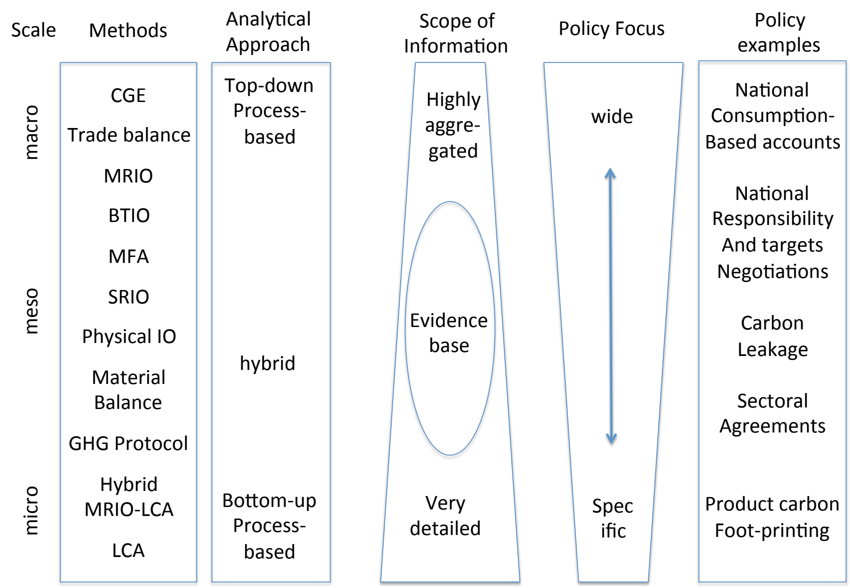
### 2.2.2 Methods

Figure 2.2 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 (EEIO) 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.

[3] Reviewing these methods are beyond the scope of this paper. Matthews et al. (2008) 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 (Miller & Blair, 1985; Leontief, 1970; Ayres & Kneese, 1969). 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.

*Figure 2.2:* Methods for calculating embodied emissions




Notes: Adapted from Wiedmann et al. (2009)

The SRIO model takes a single country and examines the emissions associated with its *total consumption* (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 at 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 input-output 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 feed-back effects).

Several method reviews have concluded that the MRIO model is the most appropriate approach for EET quantification at the country level (Liu & Wang, 2009; Rodrigues et al., 2011; Peters & Solli, 2010). 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 2.2.4). 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 2.4.

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 2.2 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 & 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) which is clearly inadequate.

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 & Harriss (2006) examine the carbon content of trade between US 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 input-output 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 2.4 and summarised in Table 2.5.

[5]Mathematically, the material balance approach is a special case of a generalized physical IO formulation (Wiedmann & 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.

### 2.2.3 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 UK's consumption based emissions for similar time periods. The former paper simply multiplies the UK's trade balance and average $CO_2$ intensity of GDP (i.e. uses the extremely crude TBA approach), 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 UK's consumption based emissions and a widening gap between production and consumption based emissions between the early 1990s and 2004.

The two studies complement one another: the former uses a simple method to highlight the issue of embodied carbon in trade, 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 2.5 will discuss in further detail the various policy 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.

### 2.2.4 Research groups and projects pioneering MRIO modelling

Table 2.1 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 input-output 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, 2008a; Peters et al., 2007) and have made significant contributions to raise the profile of embodied carbon research in the climate debate.

The Stockholm Environment Institute (SEI) formally at the University of York (which has now moved to Leeds University) and the Integrated Sustainability Analysis (ISA) group at the

[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.

*Table 2.1:* Key research groups

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

Source: Author

University of Sydney have also pioneered MRIO modelling in the context of environmental pressures. They have produced several analysis tools including the four region UK-MRIO model and the Resource and Energy Analysis Programme (REAP) to conduct scenario modelling of the emissions attributable to the UK's consumption, and more recently the global EORA database.[9] 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 standardisation and automation (Wiedmann et al., 2011).[10]

The research based at the Carnegie Mellon University's Green Design Institute has examined embodied emissions in US trade, using a MRIO model of the US 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*.[11]

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 input-output 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 supply-use tables (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.

[9] See Lenzen et al. (2010a); Kanemoto et al. (2012)

[10] To do this, standardised matrix balancing approaches for the use of supply-use tables (SUT) in a MRIO framework have been explored to avoid the use of aggregated symmetric input-output tables.

[11] The model extends the monetary core model (a global, multi-regional, environmental input-output model based on OECD IO tables) with a global dataset on material inputs in physical units. http://www.petre.org.uk/

## 2.3 Empirical findings in the literature

### 2.3.1 EET estimates at the global level

Figure 2.1 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 very crudely approximate the share of carbon emissions embodied in exports using the share of exports in GDP.

Collectively, these estimates show that volumes of embodied carbon in global trade are significant and on a growing trend. Estimates from 2004 range between 4Gt and 6Gt $CO_2$ (roughly 20-30% of global emissions) whereas those for 2006 lie between 7Gt and 8Gt $CO_2$ (around 25-35%). Aichele & 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 7.8Gt in 2008.

The chart begins to illustrate the non-trivial variation in reported results. In 2004, the lower bound is set at 4.4Gt $CO_2$ by Aichele & Felbermayr (2010)'s 'simple' model, and the upper bound by Davis & Caldeira (2010) at 6.2Gt. The gap of 2.2Gt $CO_2$ 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.

### 2.3.2 EET estimates at country level

Tables 2.2,2.3 and 2.4 compare the reported levels of emissions for China, the USA 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 of embodied emissions in trade (BEET). A sample of 13 studies which quantify China's embodied emissions in trade for the years 2004 and 2005 are summarised in Table 2.9 of the Appendix. It shows that 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. In addition, Tables 2.6 , 2.7 and 2.8 in the Appendix compare similarly for the UK, 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 2.2, for China's production based emissions in 2005, the difference between the highest and lowest estimates across six studies exceeds 1Gt (4.4Gt and 5.7Gt $CO_2$). China is no exception, for

*Table 2.2:* EET estimates from the literature for China

| Author/Year | Data year | model | CO2 production (Mt CO2) | CO2 consumption (Mt CO2) | EEE (Mt CO2) | EEE (%) | EEI (Mt CO2) | 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 & Yang (2010)^ | 1997 | SRIO | 3133 | 2957 | 314 | 10 | 138 | 4 | 6 |
| Huimin & 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 & Qi (2010)^ | 2000 | SRIO | 2974 | 2717 | 623 | 21 | 367 | 12 | 9 |
| Yan & 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 & Yang (2010)^ | 2001 | SRIO | 3108 | 2908 | 380 | 12 | 180 | 6 | 6 |
| Huimin & Qi (2010)^ | 2001 | BTIO | 2454 | 2271 | 623 | 25 | 440 | 18 | 7 |
| Peters & 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 & 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 & 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 & 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 & Yang (2010)^ | 2004 | SRIO | 4847 | 4297 | 950 | 20 | 400 | 8 | 11 |
| Huimin & 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 & Yang (2010)^ | 2005 | SRIO | 5429 | 4699 | 1180 | 22 | 450 | 8 | 13 |
| Lin & Sun (2010) | 2005 | SRIO | 5458 | 4434 | 2441 | 45 | 2333 | 43 | 19 |
| Lin & Sun (2010) | 2005 | BTIO | 5458 | 3370 | 2441 | 45 | 583 | 11 | 38 |
| Huimin & 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 & Yang (2010)^ | 2006 | SRIO | 6018 | 5018 | 1500 | 25 | 500 | 8 | 17 |
| Huimin & Qi (2010)^ | 2006 | BTIO | 6423 | 5580 | 2163 | 34 | 1320 | 21 | 13 |
| Yan & Yang (2010)^ | 2007 | SRIO | 6499 | 5362 | 1725 | 27 | 588 | 9 | 17 |
| Huimin & 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 exports 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 papers, hence are approximate. In Huimin & Ye (2010), values have been converted from carbon to carbon dioxide.

example, studies on the UK (Table 2.6) report varying levels of production-based emissions - in 1995 this ranged from Bruckner et al. (2010)'s estimate of 411Mt $CO_2$, to Wiedmann et al. (2008)'s estimate of 593Mt $CO_2$.[12]

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 3.1Gt and 4.6Gt $CO_2$ for 2004, and between 3.4Gt and 5.6Gt $CO_2$ 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 794Mt CO2 embodied in China's exports (18% of China's production-based emissions) whereas Bruckner et al. (2010) estimates around twice as much at 1.4Gt (31%). As shown in Table 2.9, both studies use the same data - OECD input-output tables and IEA energy and emissions data - but the aggregation levels vary. 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 & Yang (2009) report a lower-end estimate at 1.2Gt (22%) using a SRIO approach assuming US carbon intensity factors for the ROW and using PPP exchange rate adjustments, whereas Lin & Sun (2010) find 2.4Gt (45%). Such two fold 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 2.2, 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 & Sun (2010) using the SRIO model and assuming import substitution (imports are produced with domestic technology) report significant volumes of EEI, exceeding 2Gt $CO_2$ (over 40% of production based emissions). Huimin & Ye (2010) using a BTIO model with 36 regions and differentiated technology estimates China's EEI at 1.1Gt $CO_2$ (equivalent to 19%). Studies using MRIO models (and accounting for through trade) report much less: 0.2 to 0.4Gt (5-8%).

To illustrate the variation across studies, Figure 2.3 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

[12] The emissions level given by World Resource Institute's CAIT is 529Mt $CO_2$.

*Figure 2.3:* Comparison of EET estimates from the literature for China in 2005

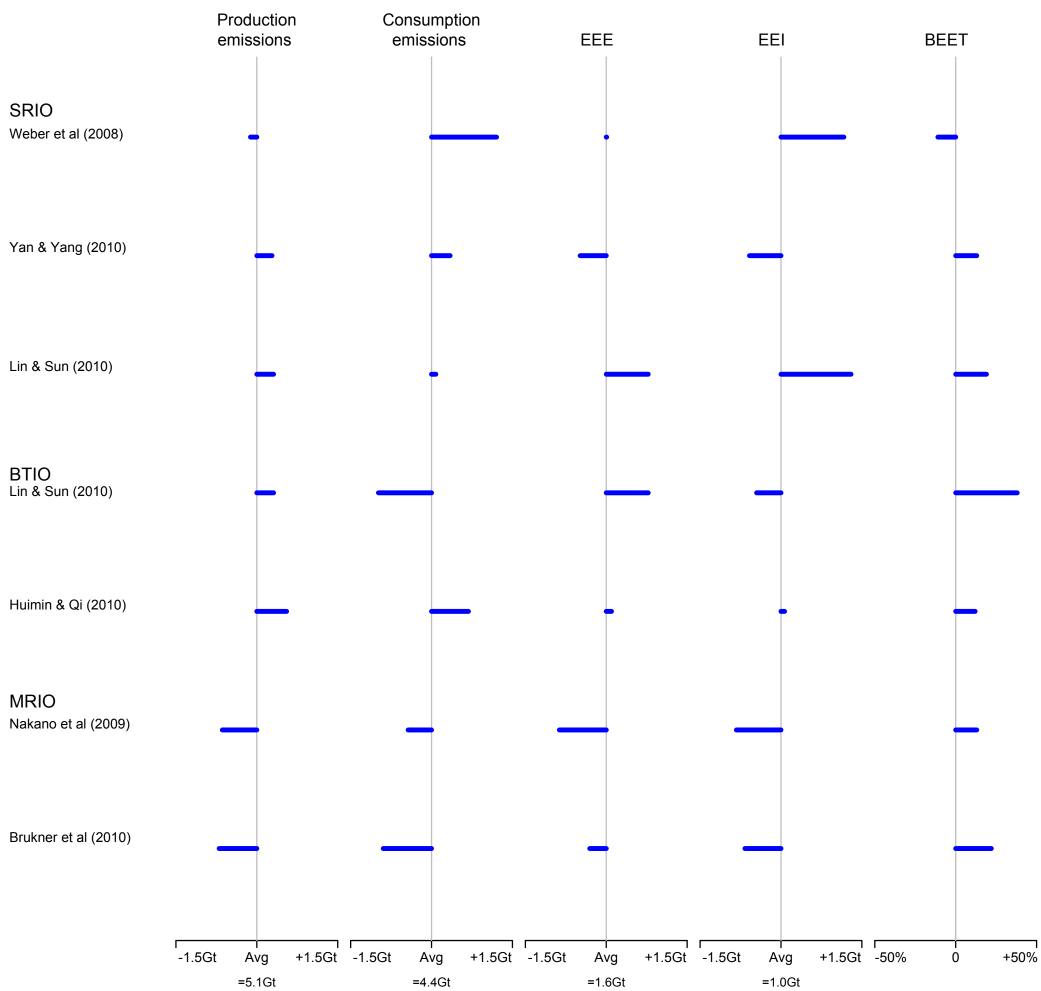


*Table 2.3:* EET estimates from the literature for the USA

| Author/Year | Data year | model | CO2 production (Mt CO2) | CO2 consumption (Mt CO2) | EEE (Mt CO2) | EEE (%) | EEI (Mt CO2) | 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 & Matthews (2007) MER^ | 1997 | BTIO | | | 450 | | 600 | | |
| Webber & Matthews (2007) PPP^ | 1997 | BTIO | | | | | 500 | | |
| Webber & Matthews (2007) MER^ | 1997 | MRIO | | | 500 | | 850 | | |
| Webber & 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 & Hertwich (2008) | 2001 | MRIO | 6007 | 6446 | 499 | 8 | 937 | 16 | -7 |
| Webber & Matthews (2007) MER^ | 2002 | BTIO | | | 450 | | 1100 | | |
| Webber & Matthews (2007) PPP^ | 2002 | BTIO | | | | | 600 | | |
| Webber & Matthews (2007) MER^ | 2002 | MRIO | | | 520 | | 1400 | | |
| Webber & Matthews (2007) PPP^ | 2002 | MRIO | | | | | 800 | | |
| Weber and Matthews (2008) | 2004 | CES*** | | 4693 | | | | | |
| Webber & Matthews (2007) MER^ | 2004 | BTIO | | | 480 | | 1300 | | |
| Webber & Matthews (2007) PPP^ | 2004 | BTIO | | | | | 750 | | |
| Webber & Matthews (2007) MER^ | 2004 | MRIO | | | 550 | | 1800 | | |
| Webber & 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 US Consumer Expenditure Survey. *" Updated results obtained from authors. ^Results have been extracted from graphs presented in papers, hence are approximate.

5.1Gt and 4.4Gt 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 (whilst 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 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 USA's imports (Table 2.4). Weber & Matthews (2007) use an MRIO model with both market exchange rate (MER) and purchasing power parity (PPP) assumptions and find "best estimates for $CO_2$ embodied in U.S. imports doubled from 0.6 to 1.3Gt between 1997 and 2007, which represents 3% to 5% of world $CO_2$ emissions in each respective year" (p. 4877). Davis & Caldeira (2010), also using a MRIO model based on GTAP data, find large volumes of EEI in 2004 exceeding 1.2Gt. They report "emissions imported to the U.S. exceeds those of any other country or region, primarily embodied in machinery (91Mt), electronics

*Table 2.4:* EET estimates from the literature for Japan

| Author/Year | Data year | model | CO2 production (Mt CO2) | CO2 consumption (Mt CO2) | EEE (Mt CO2) | EEE (%) | EEI (Mt CO2) | EEI (%) | BEET (%) |
|---|---|---|---|---|---|---|---|---|---|
| Kanemoto&Tonooka(2009)MER | 1995 | BTIO | 1258 | 1387 | 147 | 12 | 276 | 22 | -10 |
| Kanemoto&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&Tonooka(2009)MER | 2000 | BTIO | 1308 | 1423 | 188 | 14 | 303 | 23 | -9 |
| Kanemoto&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 & 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&Tonooka(2009)MER | 2005 | BTIO | 1335 | 1450 | 288 | 22 | 403 | 30 | -9 |
| Kanemoto&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 |

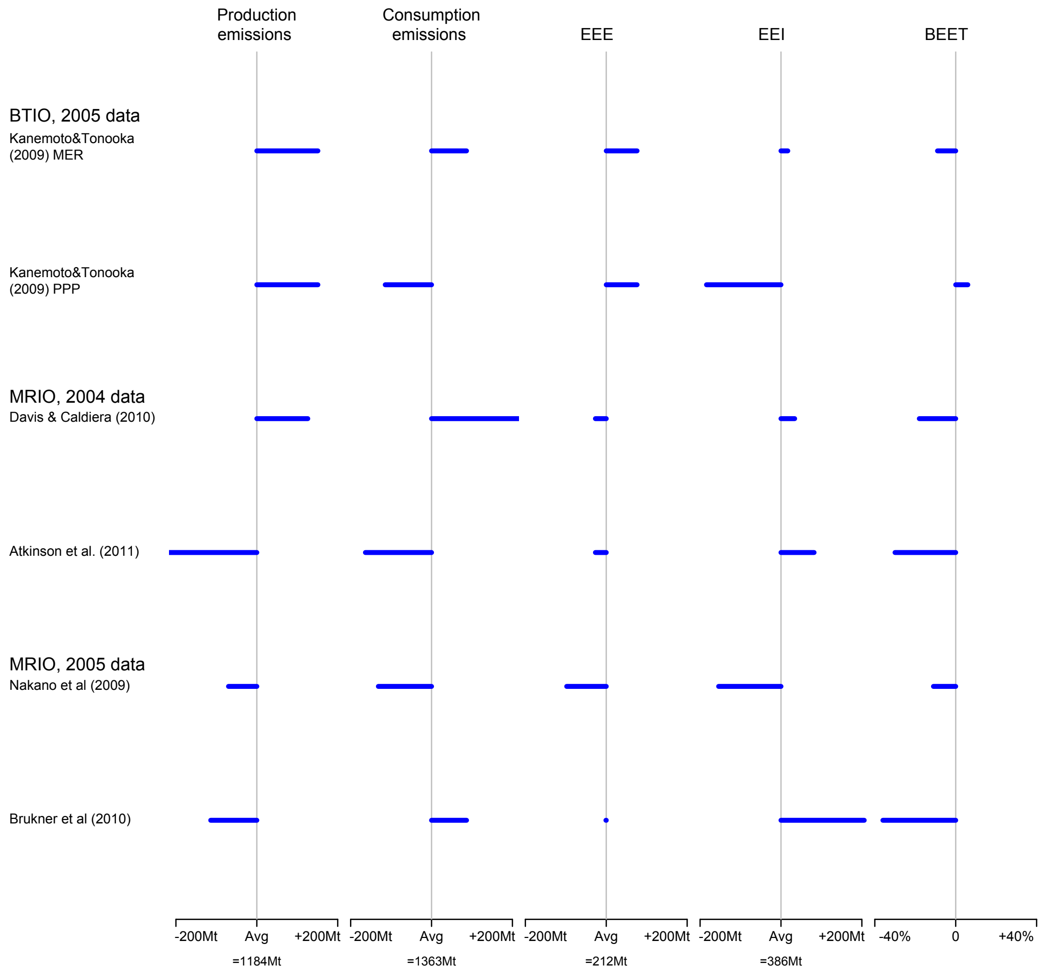
Notes: EEE% and EEI% refer to the volume of embodied emissions in exports and imports respectively, as a share of total domestic emissions. BEET% is equal to net export (EEE-EEI) relative to domestic production based annual emissions. *" Updated results obtained from authors.

(77Mt)..." (p.5688). Yet again, the table shows that differences in reported results across studies are non-trivial.

Turning now to Japan, like the US, it is also found to be a net importer of embodied emissions in general (Table 2.4 and Figure 2.4). However, Kanemoto & 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 2.3 and 2.4 collectively show that BTIO tends to over-estimate EEE and MRIO underestimates EEE, an expected effect of the system boundary difference. However, in contrast to Figure 2.3, Figure 2.4 shows that different studies using MRIO models can report wide ranging results. Atkinson et al. (2011) and Davis & Caldeira (2010), for example, both 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 modelling, 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 shows 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 & Yamano, 2008), it portrays a pattern of increasing intermediate goods trade and spatial

*Figure 2.4:* Comparison of EET estimates from the literature for Japan in 2004-2005




fragmentation in production and consumption. It shows notable and growing volumes of embodied carbon traded to and from both new and old centres of production and consumption. As summarised by Hertwich & 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 specialising 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 a description of the quantification approaches used; in practice many simplifications are necessary to overcome data, methodological and computational constraints in estimating embodied carbon flows. The next section describes these issues that undermine the robustness of existing quantification of embodied emissions.

## 2.4 Issues contributing to uncertainty in EET estimation

Sources of uncertainty in EET estimation derive from data, method or model structure. This section discusses sources of uncertainty in terms of those that are applicable across model structures, and those that are specific to certain models.

### 2.4.1 Generic sources of uncertainty

#### 2.4.1.1 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 input-output data**: The quality of the input-output data depends on both the underlying supply-use tables (SUT),[13] and the procedure used for compiling the symmetric input–output table. Despite large potential uncertainties, there is not a strong tradition of performing uncertainty analysis in IOA due to the relative lack of information on uncertainty distributions (Lenzen & Murray, 2001), particularly comparisons between different studies. Druckman et al. (2008) conducts a simple test on the impact of the IO table compilation procedure on the UK 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 harmonised IO tables used for environmental MRIO modelling are OECD[15] and the Global Trade Analysis Project.[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 & 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 2.2.4), but this involves additional assumptions and uncertainty.

[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 (symmetric input output tables)".

[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 byAhmad & Wyckoff (2003), Nakano et al. (2009),Bruckner et al. (2010), Aichele & Felbermayr (2010) and Giljum et al. (2008).

[16]Used by Kainuma et al. (2000), Rodrigues et al. (2011), Atkinson et al. (2011), Peters & Hertwich (2008) and Wilting & Vringer (2009)

**Trade data:** International trade statistics suffer 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 $cif$ (cost insurance and freight) valuation typically used to record imports, and $fob$ (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 & Walmsley, 2008).[18] The degree of uncertainty associated with such methods are unknown and unverified. Moreover, additional uncertainty is introduced when allocating bilateral trade into importing/ exporting sectors under the MRIO, as will be discussed in Section 2.4.2.3.

**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 & 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% to 20% from UNFCCC data at the national level and may be greater at the sector level (Reinvang & 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 & Wang, 2009).

#### 2.4.1.2 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 & 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 whilst 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 exports as the number of regions increase.

Whilst a multi-regional model may serve better from the perspective of representing technology differences, there are trade-offs to be made with other sources of uncertainty. Andrew et al. (2009) examines the trade-off between complexity and accuracy in MRIO and finds that including only the most important trade partners in terms of emissions embodied in imports and aggregating the

[17] Other differences in reporting practises such as 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 complemented with Global Trade Information Services (GTIS) data.

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 & Murray, 2001). The majority of studies considers 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 are the main explanatory factor for the difference between their own conclusion that Australia is a net exporter of embodied emissions, and that of Common & Salma (1992)'s which find Australia to have a balanced trade.

**Sector Aggregation**: Whilst MRIO models overcome issues with geographical aggregation, there is a trade-off with sector aggregation. The sector resolution of the model tends to become more coarse under MRIO models because of the process of matching datasets. This usually requires taking a lower common denominator of the various levels of disaggregation available – USA and Japan produce tables of about 500 sectors, but Brazil has only 19. Harmonised 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 input-output literature) because input-output tables implicitly assume one industry technology and homogeneity of firms producing for the domestic and export markets (Weisz & Duchin, 2006; Liu & 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 specialised 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, whilst 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,

[19] GTAP has 57 sectors, OECD harmonised tables have 48 sectors, and the Asian database from IDE-JETRO has 76 sectors (maximum). The EU mandates submission every five years, of harmonised 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.

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 & 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).

#### 2.4.1.3 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 (Maurer & Degain, 2012; Reinvang & Peters, 2008).[21] Using basic prices avoids some of the issues, but only to a limited extent (Muradian et al., 2002; Ahmad & Wyckoff, 2003; Weber & 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 market exchange rate (MER) or purchasing power parity (PPP). Studies have repeatedly shown that the results of EET estimation are very sensitive to this assumption. As shown in Table 2.4, Kanemoto & 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 & 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]

[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. Trade data is recorded in either f.o.b. (free-on-board) or c.i.f. (cost-insurance-freight), the latter of which includes tax. In Lenzen et al. (2004), economy-wide basic price-/f.o.b./c.i.f. ratios are used to convert imports into basic prices. Using physical quantities would avoid uncertainties induced by this conversion.

[23] Additionally, Hayami & 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

To overcome problems with monetary data, several studies integrate physical units into the monetary core model (e.g. Machado et al., 2001; De Haan, 2001; Giljum, 2005; Weisz & 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.

### 2.4.2 Model structure specific sources of uncertainty

#### 2.4.2.1 Import substitution assumption

Quantification of EET using MRIO models has shown the importance of accounting for international differences in carbon emission factors (e.g. Peters & Hertwich, 2006; Gaston & Dong, 2008; Nakano et al., 2009; Westin & Wadeskog, 2002; Ahmad & Wyckoff, 2003; Wilting & 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 & 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.

#### 2.4.2.2 Multidirectional feed-back in trade

The growing evidence that cross-border supply chains have become more prevalent in the global economy (Backer & Yamano, 2008) highlights the importance of taking account of feedback effects for estimating embodied carbon flows, particularly for countries like China 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.

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 enterprise 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 & Sun, 2010).

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

#### 2.4.2.3 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 & 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 this 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 & Hertwich, 2006; Rodrigues et al., 2011). Using the notation from Rodrigues et al. (2011), this is specified as $t_{ij}^{ab} = imp_{ij}^{*b}\frac{exp_{**}^{a*}}{exp_{**}^{**}}$ 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. Several additional estimation methods are proposed by Rodrigues et al. (2011, p.52), for example using bilateral data to disaggregate imports; using import data to disaggregate bilateral trade, or using aggregated trade shares. Using the same notation, these are mathematically and respectively $t_{ij}^{ab} = imp_{ij}^{*b}\frac{exp_{i*}^{ab}}{exp_{i*}^{*b}}$ and $t_{ij}^{ab} = exp_{i*}^{ab}\frac{imp_{ij}^{ab}}{imp_{i*}^{*b}}$. 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.

### 2.4.3 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 trade-offs, 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 2.5 summarises these trade-offs). Whilst 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.

[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 $r$.

*Table 2.5:* 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 |
| | | Transparency | Medium | Medium | High | High | Low | Low |
| Model construction | | Ability to capture time dimension | High | High | Medium | Medium | Low | Low |
| Model construction | | Level of sector disaggregation | High | Medium | Medium | Medium | Low | High |
| Model construction | regional breakdown | Capturesbilateral trade-partner info. | n | n | n | y | y (non-survey data) | y (non-survey data) |
| Model construction | regional breakdown | Captures differences in carbon intensities by country | n | n | n | y | y | y |
| Model construction | Inter-sectoral trade | domestic | n | n | y | y | y | y |
| Model construction | Inter-sectoral trade | 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 |
| Vulnerability to source of uncertainty | Data issues | IO Harmonisation (e.g. different yearbase) | n/a | n/a | n | n | High | High |
| Vulnerability to source of uncertainty | Data issues | generic trade data issues | Medium | Medium | Medium | Medium | High | High |
| Vulnerability to source of uncertainty | structural | Non-survey estimation of origin of sector's imports | n/a | n/a | n/a | n/a | y | y |
| Vulnerability to source of uncertainty | structural | aggregation error (sectors) | n/a | n/a | Low | Low | High | Medium |
| Vulnerability to source of uncertainty | structural | error due to lack of representation of technology differences | High | High | High | Low | Low | Medium |
| Vulnerability to source of uncertainty | structural | error due to lack of feed-back loops | High | High | High | High | Low | Low |

Source: Author

Moreover, clear statements of system boundaries, underlying assumptions and methodology are noticeably absent in the literature (Wiedmann & Minx, 2008; Peters, 2010). Large variations in the estimates of country level embodied carbon in trade remain 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 trade-offs, 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 that effort is made to address the existence of high levels of re-exports, even in those cases where the model structure does not allow the explicit representation of multi-directional feed-backs in trade (i.e. the MRIO framework is not used). Huimin & Ye (2010), for example, employ 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.

## 2.5 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 of 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 carbon leakage concerns as well as measures to address such concerns. This section summarises 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 Chapter thus far.

### 2.5.1 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, recognising 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; Bastianoni et al., 2004; Ferng, 2003; Huimin & Ye, 2010). Lenzen et al. (2007) for example proposes 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. The authors 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 $\alpha$, which they argue represents an even distribution of responsibility between the up and down streams. However, this is arbitrary and the political and practical feasibility of these allocation methods are yet to be examined, including their legality and data issues.

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 UK's imports and exports and concludes "any progress towards the UK's carbon reduction targets (visible under a production perspective) disappears completely when viewed from a consumption perspective" (p. 594). Peters & Hertwich (2008) highlights the importance of non-Annex I's domestic emissions and export embodied emissions by 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 & 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).[26] Studies quantifying EET have 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 HFC gases – Montreal Protocol – explicitly incorporates a consumption based perspective in the allocation of mitigation responsibility (Ahmad & Wyckoff, 2003). In the case of carbon, however, the methodological and data considerations discussed in Section 2.4 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 have 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 suggest that particularly for some countries with large net manufacturing imports, using consumption based principles as a shadow indicator may be insightful for evaluating the drivers of global emissions or assess the environmental impacts connected to national consumption (e.g. Peters & Solli, 2010).

[26] 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 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.

### 2.5.2 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, whilst '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 e.g. 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 & Hertwich, 2008; Peters, 2008a).

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.[27] 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 (as shown in Section 2.3), the literature (e.g. Wiedmann et al., 2011) also suggests that border measures may in practice have to 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 & 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. Carbon Trust, 2011a,b,c,e; Steinberger et al., 2009). Hayami & 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.

[27] Some of the recent debates can be found in Lockwood & Whalley (2008, 2010)

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 US and UK find considerable diversity in consumption habits particularly at high income levels, hence suggest large potentials for mitigation (e.g. Weber & Matthews, 2008; Druckman & 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.

## 2.6 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 input-output analysis as well as LCA literatures), studies have measured the embodied carbon at the level of the country, sector and 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 chapter 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. This thesis suggests that sensitivity to exchange rate assumptions should be tested at the very minimum.

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[28] as well as ongoing efforts to fill the data gaps reflect a significant

[28] e.g. the OPEN EU project (http://www.oneplaneteconomynetwork.org) and the Reunion Project (Wiedmann et al., 2011).

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 been previously applied in the context of global environmental agreements on HFC gases. Yet, this chapter 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 in 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 chapter, 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 decarbonising these global supply chains, and the design of climate-trade integrated policies to support this. EET quantification at the product level suggests 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 seems especially timely for future investigation.

## 2.7 Appendix

*Table 2.6:* EET estimates from the literature for the UK

| Author/Year | Data year | model | CO2 production (Mt CO2) | CO2 consumption (Mt CO2) | EEE (Mt CO2) | EEE (%) | EEI (Mt CO2) | EEI (%) | BEET (%) |
|---|---|---|---|---|---|---|---|---|---|
| Druckman et al. (2008) | 1990 | SRIO | 643 | 650 | | | | | 1 |
| Druckman & 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 & 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 & 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 |

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.

*Table 2.7:* EET estimates from the literature for Denmark

| Author/Year | Data year | model | CO2 production (Mt CO2) | CO2 consumption (Mt CO2) | EEE (Mt CO2) | EEE (%) | EEI (Mt CO2) | EEI (%) | BEET (%) |
|---|---|---|---|---|---|---|---|---|---|
| Munksgaard & 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 etal (2004) 1 | 1997 | SRIO | 58 | 47 | 30 | 52 | 19 | 32 | 19 |
| Lenzen etal (2004) 2 | 1997 | BTIO | 58 | 58 | 38 | 64 | 37 | 63 | 1 |
| Lenzen etal (2004) 3 | 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 & 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 |

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.

*Table 2.8:* EET estimates from the literature for Brazil and India

| Author/Year | Data year | model | CO2 production (Mt CO2) | CO2 consumption (Mt CO2) | EEE (Mt CO2) | EEE (%) | EEI (Mt CO2) | EEI (%) | BEET (%) |
|---|---|---|---|---|---|---|---|---|---|
| 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 & 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 |
| Mukhopadhyay (2004) | 1993/1994 | SRIO | | | 37 | | 49 | | negative |
| Ahmad and Wyckoff (2003) | 1993 | MRIO | 672 | 623 | 74 | 11 | 24 | 4 | 7 |
| Dietzendbacher et al (2007) | 1996/1997 | SRIO | 920 | 1047 | 93 | 10 | 221 | 24 | -14 |
| 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 |
| Nakano et al (2009)*" | 2000 | MRIO | 907 | 877 | 58 | 6 | 28 | 3 | 3 |
| Peters & 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 |

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.

*Table 2.9:* Summary of methods, data and results from 13 studies of embodied emissions in China's trade for the years 2004 or 2005

| | | | Model | Data | | | Assumptions | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| reference | Country | Year | structure | Economic data | Trade Data | Emisions Data | Aggregation | | | Coverage | | | | | |
| | | | | source | source | source | #production sectors | #regions | #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 Stats & Balances | | 8 | | | | yes | n/a | | |
| Wang & Watson (2008) | China vs ROW | 2004 | trade balance | | NBS | IEA CO2 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, Weber, Liu (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 Year Book | 37 | 2 | | | | | | import sub. adjusted by national avg intensity scalar | |
| Huimin & Ye (2010) | China vs ROW | 1997-2007 | SRIO | NBSInput-output tabels of China, 2002 | NBS Foreign Econ Stat Yrbook 1998-2005, GTAP (imports) | IEA CO2 emis. from fuel combust. | | | | | | | | | |
| Yan & Yang (2010) | China vs ROW | 1997-2007 | SRIO | NBS China input-output table, 1997 | Chinese Statistical Yearbook | from Peters, Weber, Liu (2006) | | | 20 | yes | no | yes | China's exports adjusted by PPP | ROW= USA (data from Carnegie Mellon Uni.GDI) | no |
| Lin & 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 Stats & Balances | 13 | 10 | 3 | no | yes | yes | | | yes |
| Nakano et al (2009) | multiple | 1995, 2000, 2005 | MRIO | OECD harmonised IO tables | OECD Bilateral Trade database | IEA CO2 emis. from fuel combust. | 17 | 41 | | no | yes | yes | MER | using like countries | yes |
| Bruckner et al (2010) | multiple | 1995-2005 | MRIO | OECD harmonised IO tables | OECD Bilateral Trade database | IEA Energy balances | 48 | 55 | 4 | no | yes | yes | | using like countries | yes |
| Peters & Hertwich (2008) | multiple | 2001 | MRIO | GTAP 6 | GTAP 6 | GTAP | 57 | 87 | | | | | | | yes |
| Davis & 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 |

Note: The results column show the values of embodied emissions estimate by the studies, in terms of the total emissions attributable to China's production, consumption, exports and imports. Embodied emissions in exports (EEE) and imports (EEI) are presented both in terms of the absolute volume (in Mt $CO_2$) and in terms of a share of total emissions from production in China.

# Chapter 3

# Product level embodied carbon flows in bilateral trade

## 3.1 Introduction

The industry sectors currently account for around a third of global energy demand and $CO_2$ emissions (IEA, 2007a). Decarbonising industrial production and consumption is therefore critical in achieving long term GHG stabilisation goals. However, in contrast to sectors such as transport, power generation and buildings, the geographic mobility of production facilities adds an additional layer of complexity to the issue of controlling industry sector emissions.

On one hand, the possibility to decouple production and consumption via international trade can facilitate carbon mitigation within production chains. The global aluminium sector, for example, could benefit from concentrating the electricity intensive primary aluminium smelting segment of the production chain in those locations with ample zero-carbon power generation capacity from hydro. On the other hand, trade also provides industries the opportunity to strategically choose production locations to avoid stringent environmental regulations. As countries introduce climate policy measures of varying stringency and global merchandise trade continues to grow[1], there are increasing concerns about the impact on production, investment and carbon leakage.

A large number of studies have quantified embodied emissions in trade (EET), using several different methodologies, as reviewed by a number of papers (e.g. Kitzes et al., 2009; Liu & Wang, 2009; Peters, 2008b; Wiedmann, 2009) and Chapter 2 of this thesis. Most studies use an input-output framework to capture indirect effects, either within a single region context (e.g. Druckman et al., 2008; Ferng, 2003), or within a multi-regional setting (e.g. Peters & Hertwich, 2008; Davis & Caldeira, 2010; Atkinson et al., 2011). Alternative approaches include simplified

[1] Merchandise trade grew 460% in value between 1991 and 2008, outstripping population and global GDP growth of 21% and 64% respectively (see Figure 1.1 on page 17).

methods using average carbon intensity of GDP multiplied by trade balance (e.g. Helm et al., 2007; Wang & Watson, 2008), material balance methods using physical rather than monetary data (e.g. Muradian et al., 2002) as well as computable general equilibrium models (e.g. Kainuma et al., 2000). These approaches are grouped into the category of top-down methods, in contrast to the bottom-up methods used for the calculation of embodied emissions in products (e.g. Life cycle analysis (LCA)).

The literature overall has provided some broad conclusions. In general, large volumes of embodied emissions are found in global trade, with around 4-6Gt $CO_2$ for years 2004-2006 (see Figure 2.1 on page 29), equivalent to around a third of global annual $CO_2$ emissions. Global trade embodied emissions are also growing rapidly over time (Peters et al., 2011b; Wiedmann et al., 2010), and most industrialised countries[2] are net importers of EET, while the trend is reversed for many of the emerging economies and resource rich countries like China, Russia and South Africa.

However, thus far studies quantifying embodied carbon in trade have had limited impact on policy making. Reasons for this include the high uncertainty surrounding measurement – as recent reviews highlight, all underlying data, methodology and choice of methods suffer issues with accuracy but to different degrees such that comparing across studies reveals a large variation in the estimations of EET (e.g. Wiedmann (2009); Lutter et al. (2008); Liu & Wang (2009) and in Section 2.4 of this thesis). The issue of uncertainty has been compounded by the lack of methodological transparency, reliability and reproducibility of the information, particularly from studies using multi-regional input-output (MRIO) analysis (Wiedmann et al., 2011). Moreover, while carbon leakage is widely understood as a sectoral issue, relatively few global studies have examined embodied carbon at the sector level. Weber & Matthews (2007) examines sectoral EET but only for the US, as does Weber et al. (2008) for China. Peters et al. (2011b) provides a detailed analysis using a disaggregated model with 113 regions and 57 countries, but his sectoral results are aggregated for global trade, or the trade between Annex I and non-Annex I, whereby bilateral trade by country information is lost.

This chapter quantifies global embodied carbon in bilateral trade between 195 countries, disaggregated at the level of 970 products for the year 2006, and describes the results. To the author's knowledge, this detailed mapping of EET flows represents a first of its kind. It does so by constructing and combining two large data sets: product level global bilateral trade in physical quantities and carbon intensities of products. The methodological principal of the material balance approach is then applied to this data to estimate EET. This method was previously applied in analyses on a smaller geographical scale in ecological foot-printing research. It has the advantage of offering a transparent way of quantifying EET. It also overcomes a number of key sources of uncertainty implicit in the more commonly applied input output methods discussed in Section 2.4.

[2]Industrialised countries are defined here as the countries included in Annex I of the Kyoto Protocol.

However, for data reasons, this analysis relies on the use of world average emission factors (WAEF), defined in physical terms (kg $CO_2$/kg product). At this disaggregated level of product definition, availability of carbon intensity is limited to few countries where LCA is more commonly conducted, and to few time periods hence few country specific emission factors (CSEF) are available. Currently, the extent to which using WAEF affects the accuracy of results is poorly understood. This chapter will explore the sensitivity of the results to the WAEF assumption using a case study of cement in Section 3.5. It is also important to note the limitations of the interpretation of the EET quantification in this analysis, due to the WAEF assumption. In particular, the results are not able to say whether or not trade has increased overall global emissions. Additional analysis such as case studies for sectors where country-specific carbon emission factors can be obtained, will be necessary to assess, for example, the impact of policies such as a border tax on carbon leakage.

The objective of the study is to provide insights into the nature of carbon flows that were previously masked under quantification exercises conducted using aggregated models. It builds on recent studies, by further disaggregating estimations using high resolution bilateral trade information at the product level (Weber & Matthews, 2007; Peters et al., 2011b). This more detailed quantification enables the identification of sectors, and products within sectors, where global EET flows are concentrated. It also highlights the bilateral trade routes where significant levels of embodied carbon are exchanged. The complex picture emerging from the detailed analysis challenges the existing literature, which provides a more simplistic perspective which focuses on the exchange of embodied carbon between two large groups – Annex I vs non-Annex I.

This paper is structured as follows. Section 3.2 describes the methodology and the key assumptions. Section 3.3 gives details of the data collected and used to develop worldwide product level estimates of embodied emissions in trade. Section 3.4 presents results in terms of four key findings, with regards to the geographical and sectoral distribution of EET, the heterogeneity across countries (China, EU and US), as well as how countries can be characterised, in terms of their trade embodied carbon from a global supply chain perspective. Section 3.5 asks to what extent the results are sensitive to key assumptions, and illustrates using the examples of aluminium and steel sectors. Section 3.6 summaries the insights from the detailed quantification exercise.

## 3.2 Quantification strategy

### 3.2.1 Material balance methodology

The material balance methodology was developed within the ecological footprinting literature as an alternative input-output framework for calculating footprint trade (Kitzes et al., 2009).

'Footprint' or 'intensity' multipliers usually derived from life cycle analysis (LCA)[3] are combined with isolated values of imports and exports by sectors (weight or value), in order to estimate ecological footprints embodied in traded goods (e.g. Bicknell et al., 1998; Muradian et al., 2002; Bagliani et al., 2005; Turner et al., 2007). Bilateral trade flows expressed in total weight terms are combined with carbon intensity multipliers:

$$EEE_j^{r,s} = \sum_{r \neq s} X_j^{r,s} * EF_j^w \tag{3.1}$$

Here, $CO_2$ embodied in exports from country $r$ to country $s$ $(s = 1, 2, 3, ....., S)$ via product $j$ is calculated by multiplying country $r$'s export matrix $X$ of good $j$ (where goods $j$ = 1,2,3,..., $J$) expressed in physical quantities, by a vector of world average emission factors $EF_j^w$ expressed also in physical terms (kg $CO_2$/kg product). The $CO_2$ intensity factors are derived from engineering based techniques using large amounts of primary data. Specifically, intensity factors calculated using the *cradle-to-gate* system boundary are used, thus covering emissions from a partial product life cycle, from manufacture (cradle) to the factory gate i.e., before it is transported to the consumer. $EEE_j^{r,s}$ thus reflects the *embodied* carbon emissions attributable to the production of the good throughout the production chain including the production of inputs. This is in contrast to carbon emissions factors using alternative system boundaries such as *gate-to-gate*, *cradle-to-grave* (including the use phase and disposal phase of the product) and *cradle-to-cradle* (including recycling).

Mathematically, the material balance method represents a special case of a generalised physical input–output formulation. Yet in practice, data availability and necessary simplifying assumptions under both methods restricts their equivalence (Wiedmann & Lenzen, 2007). Importantly the *cradle-to-gate* carbon intensity coefficients under the material balance approach, assume that all production inputs are sourced domestically.[4] This issue is problematic from a general equilibrium point of view, as it gives rise to the double-counting of emissions. It is less important from the country-level EET quantification perspective, in the case of large economies such as the US, the EU, Australia, Brazil and Japan. For these countries, the import content of exports in the period mid-2000 was relatively low at around 10% to 15%. Hence double-counting is contained to small levels (see Figure 1.3 in Appendix 3.7). Yet careful interpretation of the results is necessary for countries with skewed trade structures such as Taiwan, Korea and Portugal.

In addition, even when restricted to the *cradle-to-gate* system boundary, the material balance approach can still suffer from truncation errors, or a lack of full coverage of indirect upstream

[3]LCA is designed to evaluate the environmental impacts of a given product or service and is similar in philosophy to input-output analysis as a method to calculate embodied emissions in products, but differs in several important respects. It is a process-based bottom-up technique used to examine the production process of a specific product in detail, unlike the top-down input-output approach which obstructs from analysis of specific materials or products. The latter captures all indirect effects (e.g. within the economy) whereas LCA imposes boundaries. LCA guidelines are given by the ISO standards.

[4]This assumption is also made in input-output analysis using the BTIO or EEBT framework, but not under MRIO.

flows (Lenzen, 2001). Over and under counting is possible, due to a lack of standardised boundary setting principles among process-flow LCA studies. Methods to address outlier observations are therefore deployed, as is described in Section 3.3.2.

Key advantages over methods using the input–output framework are as follows. First, it enables a more detailed examination of sectors, hence avoiding issues with coarse sector aggregation discussed in the literature (e.g. Lenzen et al., 2004; Tukker et al., 2009). Second, by using physical trade data, it avoids inherent problems with using monetary data to approximate physical flows of goods. These problems are related to assumptions about valuation, prices and exchange rates, for example (see for example Maurer & Degain, 2012; Reinvang & Peters, 2008). Third, the method is more transparent and closer to source data in contrast to the popular MRIO modelling analysis involves considerable adjustments to the source data, aggregation of sectors and regions, and inherently suffers from the lack of methodological transparency (Weber et al., 2008).

## 3.3 Data

### 3.3.1 Bilateral trade

The level of sectoral and geographical disaggregation used in this investigation go beyond that of previous work. Trade data is taken from UN Commodity Trade (COMTRADE) statistics which contains detailed bilateral import and export statistics. Despite all of the criticism it attracts, COMTRADE remains one of the most comprehensive trade databases available, recording trade of commodities classified under several systems.[5] It has been used for a variety of research themes and is the data underlying most trade models including the Global Trade Analysis Project,[6] as well as many studies that quantify EET (e.g. Peters & Hertwich, 2008; Weber & Matthews, 2007; Atkinson et al., 2011).

The sample data covers 970 sectors (SITC revision 3 classification, 4 digit resolution) and 195 countries for the year 2006. This includes all traded commodities, including food and fuel. It excludes electricity, passenger transport equipment (cars) and live animals.[7]

In two cases, 4-digit sectors were further disaggregated at 5-digit level – the 4-digit sector 8841 that combines contact lenses, optical glasses, sunglasses and optical fibre was disaggregated to the 5-digit level, as well as sector 6610, to distinguish between Portland cement, lime and

[5] SITC (Rev.1 from 1962, Rev.2 from 1976 and Rev.3 from 1988), the Harmonized System (HS) (from 1988 with revisions in 1996 and 2002) and Broad Economic Categories (BEC).

[6] However, GTAP undertakes significant aggregations and modifications to the COMTRADE data and the degree of these changes are not always explained (Tukker et al., 2009).

[7] Electricity is excluded because there is missing data for the majority of countries. Animals are excluded also because of missing data, and there are also limited estimates of their carbon intensity. The issue of car trade data is discussed below.

cement clinker. This was done to address the variation in the carbon intensity data for these products.

The raw trade data from COMTRADE, requires considerable cleaning before use in analysis as explained in detail by Moran et al. (2009). This is largely due to the reliance on self-reporting and the lack of standardised procedures in trade data collection (Narayanan & Walmsley, 2008). These issues can be observed in discrepancies between mirror statistics – that is to say the two records for the same trade transaction recorded by the importing country and the exporting country do not match (in theory $Trade_j^{r,s} = Trade_j^{s,r}$). A number of underlying issues have been used to explain these discrepancies – misreporting, difference in practices with respect to the inclusion/exclusion of transport costs,[8] as well as variations in reporting dates. However, few systematic biases have thus far been found. Accuracy of trade data can vary by product, for the same reporting country. In aggregate the asymmetry is 4% (for cases where both are available i.e. no missing data) but in individual case, it can exceed 30%.[9]

The issue of non-matching mirror statistics with COMTRADE data is dealt with in this study using a standard rule. It prioritises observations with the larger value i.e. by taking the maximum. For the sample year 2006, COMTRADE reports quantity values for over 92% of records. If neither observation in a pair records quantity information, the observation with the maximum monetary value is taken. Then the corresponding quantity is estimated for that observation, based on an average price per unit of quantity, in that product category over all countries.

As a robustness check, the results using this method are compared with two alternative methods: taking only exports and only imports or taking an average. Using only the exports or imports lead to a substantial increase in the number of missing trade for the observations (around 37% and 29% more respectively) making it necessary to estimate traded volumes using monetary value data. Taking the maximum is favourable to taking the average, also because of the presence of missing values in the data set, and the inability to distinguish between missing data and no-trade which are both represented by a number zero.

The notable gap in the trade data in physical quantities is that of passenger transport equipment. Where the data on automobile trade in value terms exists, it is problematic to convert them into quantities of cars because of the wide variance in car prices. Significant embodied emissions have been found, particularly due to car exports from Japan, China and Europe to the US in a study by the Carbon Trust (2011b).[10] Additionally, the data has been examined carefully to detect notable

[8]There are known differences in *cif* (cost insurance and freight) valuation typically used to record imports and *fob* (free on board) valuation used.

[9]In the GTAP, a rating system is developed whereby for a certain product, a country is rated highly, if their import and export records match well with their counter part country. Data reported by that country is prioritised for that product, whether they are reporting imports or exports. This method is based on the observation that countries tend to report trade accurately for certain products which are a high priority, for example, records of textile exports are recorded with high accuracy in China, reflecting its importance relative to other export goods (Narayanan & Walmsley, 2008).

[10]The study found that in 2004, automobile production (700 million vehicles) accounted for 5% to 6% of global GHG emissions and around 40% of these emissions moved across international borders.

misreporting. The Solomon Islands, for instance despite having no steel industry appears in the data as a large exporter of steel, and this misreporting is confirmed by contrasting export with production data. For significant products and sectors in the reported results (e.g. Australia's ore exports), the magnitude of embodied emissions in exports (EEE) have been compared with domestic production levels.

### 3.3.2 Carbon intensity factors

The most important priority in using the material balance approach is to locate robust product carbon intensity information ($EF_j^w$), ideally country-specific (Kitzes et al., 2009). Carbon intensities of products have been estimated for industrial and manufactured goods using bottom-up approaches such as LCA, which have been applied more than a million times to estimate life-cycle environmental impacts (Matthews et al., 2008). An extensive data search was conducted and product carbon intensity factors were collected from multiple data sources (see Table 3.1). The database on product level carbon intensities built during the process will be made public upon completion of this thesis. These include the Global Footprint Network (GFN) which provides a comprehensive set of estimates of carbon intensity factors by 4-digit trade category (under SITC Revision 1).[11] The European Union's ELCD is a core database comprising of Life Cycle Inventory (LCI) data from various EU business associations and other sources, mainly for key materials and energy carriers. Similarly, the Carbon Footprint of Products database is an initiative by the Japanese Ministry of Economy, Trade and Industry to improve data availability and transparency for LCA, and covering a range of heavy industrial sectors. Altogether, some 700 carbon intensities were found for around 400 products.

However, carbon intensity estimates are available only for select years, countries and products, due to the costly nature of bottom-up analysis. Moreover, differences in system boundaries remain a main source of variation in the measurement of carbon intensities in bottom-up methods, despite the many efforts to harmonise methods, for example by the International Organization of Standardization (ISO), the World Resource Institute (WRI) and the World Business Council for Sustainable Development (WBCSD). Studies combining LCA with top-down input-output models have shown how results from LCA product analysis are sensitive to the inclusion or exclusion of certain flows (e.g. lack of upstream representation, transport and use phase emissions) (Suh et al., 2004; Lenzen, 2001; Kitzes et al., 2009).

In order to determine a best-available estimate of world average emission factors $EF_j^w$ in light of these issues, the strategy adopted here is to collect as many available product level carbon

[11] Correspondence tables from COMTRADE were used to match carbon intensity estimates for SITC Revision 1 to Revision 3. They are global average figures, based on embodied energy estimates (from GFN internal data) and multiplied by "World Electricity and Heat Carbon Intensity" from International Energy Agency's $CO_2$ Emissions from Fuel Combustion Database 2007 (Global Footprint Network, 2006, p.69). The GFN data has been used for analyses on embodied emissions and ecological footprint in trade (e.g. Moran et al., 2009) and discussed in detail in (Kitzes et al., 2009).

intensities as possible strictly restricting to those using the *cradle-to-gate* system boundary, then taking an average excluding outliers.[12] Section 3.5 will put to test the extent to which this assumption drives the results.

As verification in cases where an estimate is available from only one source (GFN), a test is conducted at the broader 3-digit resolution of SITC classification to determine its reliability.[13] If for another 4-digit product in the same 3-digit category the GFN estimates falls within ±25% of the available range, then the GFN estimate is deemed reliable for all 4-digit products in that category. Otherwise, the same test is conducted at 2-digit level. If the test is rejected at 2-digit level, or if no other estimates are available at 2-digit level product classification, then an average carbon intensity factor for all categories is used (2.58 $CO_2$ per kg product) as the best-guess estimate (the GFN estimates were found to lie at the upper-end of estimates). The latter average factor was applied to the majority of down-stream products such as electrical equipment and machinery, due to the lack of LCA estimates for these products. Summary statistics of the resulting vector of carbon intensities are provided in Table 3.2 in Appendix 3.7.

## 3.4 Quantification results

On a global level, this study explains 7.3Gt of $CO_2$ embodied in trade (including fuel and food), representing roughly a quarter of annual global $CO_2$ emissions in 2006. This is in line with the estimates of EET found in the literature: Davis & Caldeira (2010) finds approximately 6.2Gt of $CO_2$ (23%) for the year 2004, and Peters et al. (2011b) finds around 7.8Gt $CO_2$ (26%) in 2008. This section presents four key findings (marked by each sub-heading) that emerge from the quantification of product level embodied carbon in bilateral trade.

A large share of papers quantifying carbon embodied in trade describe the state of the world in 2004, due to the lack of availability of multi-regional input output database beyond that time. This study describes results for the year 2006, in an attempt to both update existing findings and to avoid the shock in trade data, around the period 2007 to 2009 in which the world saw a sharp decline in gross domestic product (GDP) brought about by the global financial crisis.

### 3.4.1 The geographical distribution of embodied emissions in trade reflects regional dependencies.

So far, China's large net exports of embodied emissions to the US has received considerable attention. However, this study's estimates show the South to North flow of EET is only part of the story. In fact trade within non-Annex I regions (South to South) accounts for 32% of total

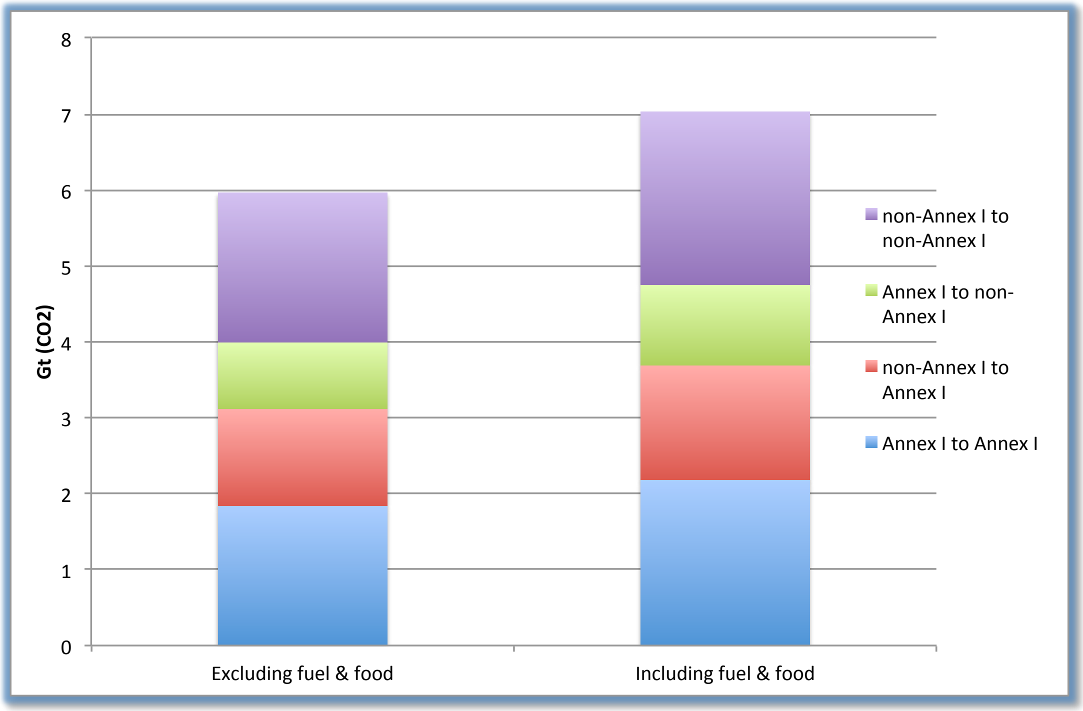
[12] Where several estimates were available for one 4-digit product category, outliers are defined statistically using inter-quartile range.

[13] This approach is inspired by methods used in advanced computational aspects of LCA calculations, to validate estimates of missing flows that are truncated (Suh, 2001).

*Table 3.1:* Carbon Intensity Databases

| | Authors | Database | Sector coverage |
|---|---|---|---|
| 1. | Global Carbon Foot Print Network | Carbon Footprint database | All SITC sectors at 4-digit level |
| 2. | EU Commission, Joint Research Centre | European Life Cycle Database | Comprehensive |
| 3. | CPM Chalmers | CPM LCA database | Comprehensive |
| 4. | Aarhus University, Faculty of Agricultural Science | carbon footprint database | Food |
| 5. | Hammond & Jones (2008) | Inventory of Carbon & Energy | Building materials |
| 6 | Bergmann et al. (2007) | Imposing a unilateral carbon constraint on European energy intensive industries and its impact on their international competitiveness - data & analysis | |
| 7. | Moll et al. (2005) | Iron and steel - a materials system analysis | iron & steel |
| 8. | GEMIS | Global Emission Model for Integrated Systems Version 4.6 | comprehensive |
| 9. | British Geological Survey | World Minerals Statistics | Industrial minerals, mine products |
| 10. | U.S. Life Cycle Inventory Database | National Renewable Energy Laboratory | comprehensive |

*Figure 3.1:* Global embodied carbon in trade by Annex-I and non-Annex I, 2006




Source: Author's own calculations. These estimates include food and agricultural products.

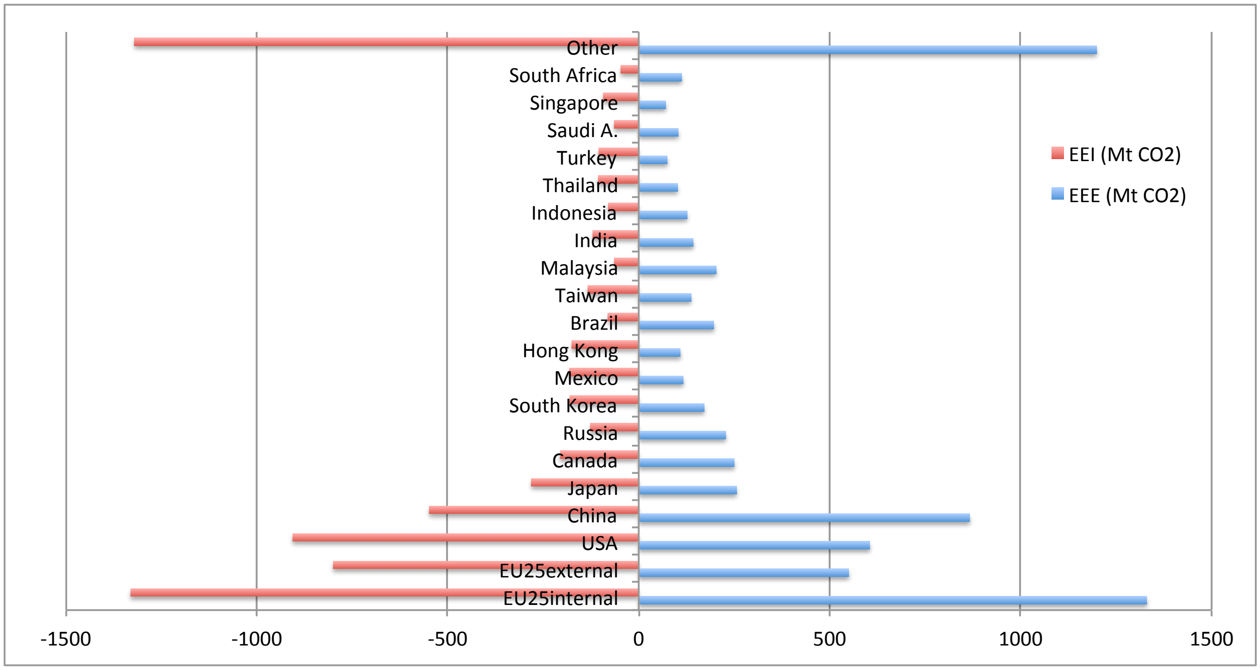
EET flows, exceeding by a small margin the trade within the Annex I region (North to North) at 31%, as is shown in Figure 3.1. The other third is split between the flow of EET from non-Annex I to Annex I (22% in red) and vice versa (15% in green).

Breaking down the flows by country, Figure 3.2 shows the level of embodied emissions in imports and exports for 20 countries with the largest total volumes in 2006. Notable is the substantial volume of EET attributable to the EU internal trade (around 1.3Gt $CO_2$), although this has previously received limited attention in the literature. This evidence supports the general case for having a uniform carbon price in large free-trade blocks. Emissions embodied in the EU25's external trade are large in volume also, with import volumes exceeding exports (800Mt and 550Mt $CO_2$ respectively). Volumes for the US trade are larger but comparable to EU 25 external trade, also with imports exceeding exports, respectively at around 900Mt and 600Mt $CO_2$. These estimates are comparable to others in the literature - between 571Mt and 1800Mt $CO_2$ for embodied emissions in imports (EEI) (Bruckner et al., 2010; Nakano et al., 2009; Weber & Matthews, 2007; Atkinson et al., 2011) and between 227Mt and 630Mt $CO_2$ for EEE.[14]

China stands out as the largest single exporter of embodied carbon with around 870Mt $CO_2$. On the other hand, China also imports large volumes of embodied emissions, which is estimated

[14]As described in Section 2.4.1.3, large volumes of EEE estimated for China in the literature (but not this study) can be due to the use of monetary data and more specifically, the use of Market Exchange Rate, without adjusting for PPP. By using data in physical quantities, this study avoids this issue.

*Figure 3.2:* Embodied Emissions in Imports (EEI) and Exports (EEE) by country in 2006




Source: Author's own calculations. These estimates include food and agricultural products.

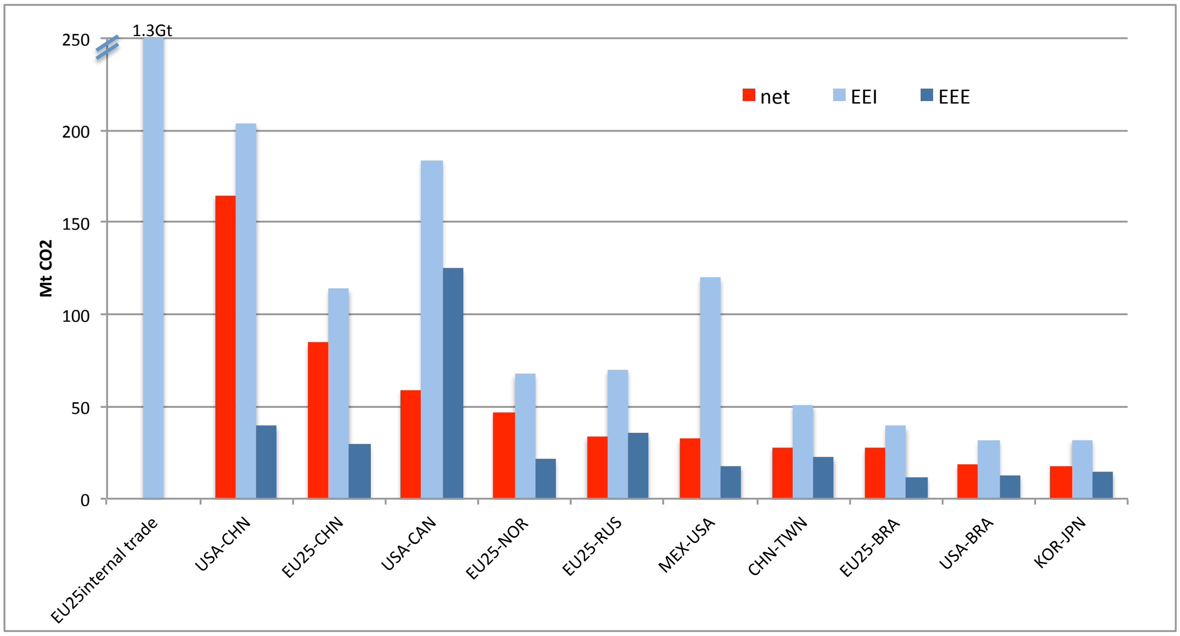
to be around 550Mt in this study.[15] Other countries with relatively large EET volumes include other industrialised export oriented economies and emerging economies in East Asia, such as Japan, Taiwan, Korea and Malaysia, as well as natural resource rich countries such as Canada, Russia, Australia, Indonesia and Brazil.

EET flows can be further broken down by bilateral trade routes. Figure 3.3 shows 12 bilateral trade routes with the largest volumes of net EET flow (red bar), and the corresponding absolute volumes behind the net figures. For example, the US imports around 160Mt of embodied $CO_2$ in trade from China, and in return exports around 40Mt resulting in a net flow of 120Mt $CO_2$. Malaysian imports from Japan are notable - a large share of this is in the form of iron & steel and steel products for Japanese car manufacturing in Malaysia (Asuka et al., 2010). The extent to which embodied emissions flow from Malaysia to other countries via cars cannot be observed in this analysis due to the lack of trade data for passenger vehicles.

Focusing on the red bars (net trade) in Figure 3.3, the highest red bar indicates large net EET imports by US from China. The EU25 also has a large net import from China. Yet looking at the absolute volumes of embodied carbon in trade instead (the blue bars) reveals that US's trade with its neighbouring countries like Canada and Mexico is just as important. This result that trade in embodied emissions tends to be greater between neighbouring regions was confirmed when examining bilateral EET flows between 11 regions.[16] For example, Australia-Asian countries

[15]This is on the lower end of the estimates found in the literature as discussed below in Section 3.4.3.1.
[16]Results at the 11 region level are not presented in this thesis in the interest of space.

*Figure 3.3:* Net vs aggregate embodied carbon in bilateral trade – some key country pairs. 2006




Source: Author's own calculations. Notes: For each pair, the first indicates importer and the latter, the exporter. For the pair USA-CHN, EEI is the US imports from China, and EEE is US exports to China. Similarly EU internal trade is 1.3Gt $CO_2$.

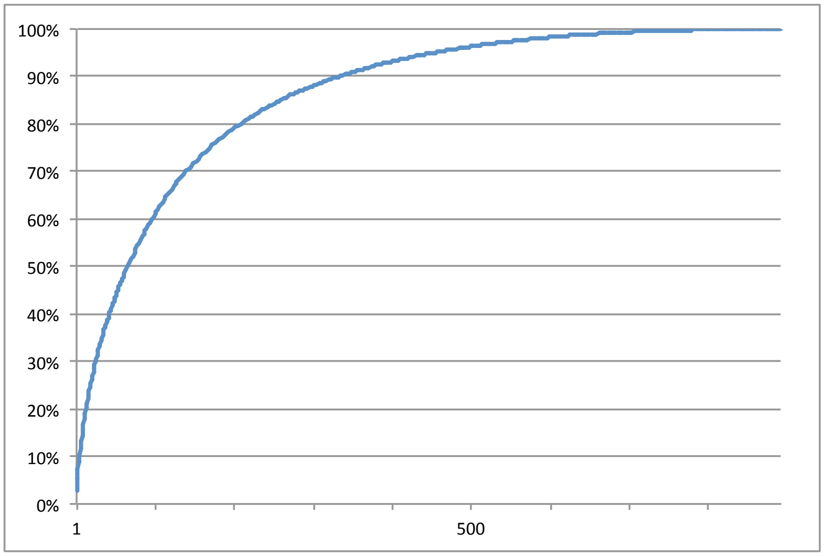
import and export more embodied carbon with Pacific regions such as South-East Asia and China, compared to places farther away. Again, these results suggest that harmonising $CO_2$ mitigation policy across neighbouring countries with strong trade links will go a long way to address potential adverse impacts on trade. Indeed, in Europe, differences in the rules of emissions allowance allocation during the first two phases of the EU ETS attracted strong criticism from industry, and the European Commission has sought to increase the degree of harmonisation through guidance notes (del Río González, 2006).

### 3.4.2 Around 15% of products account for 70% of embodied carbon in trade.

Sector heterogeneity was highlighted in the debates around the trade impacts of the EU ETS. Partial equilibrium modelling studies showed how trade impacts are likely to be sector-specific, depending on their varying levels of carbon intensity of production, ability to pass through abatement costs to consumers, as well as different levels of sensitivity to multiple barriers of trade e.g. product differentiation, service differentiation, transport costs, capacity constraints and import restrictions (Hourcade et al., 2007; Demailly & Quirion, 2008). Arguments have been made in favour of policy measures tailored specifically to each sector, rather than generalised solutions (Dröge & Cooper, 2010).

The product level evaluation of embodied carbon in this analysis finds that, of the 970 products examined, around 15% of the products account for around 70% of global EET (Figure 3.4). This

*Figure 3.4:* Distribution of EET by product category



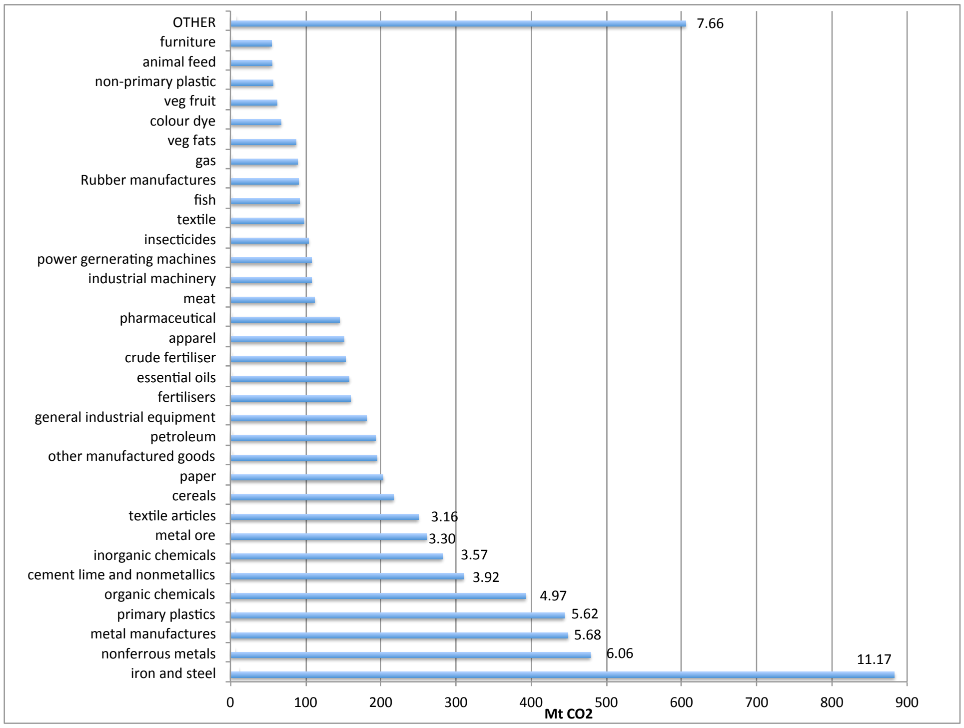

Source: Author's own calculations. These estimates include food and agricultural products.

suggests that focusing mitigation efforts and trade-measures on the products in this group of 15% would be an effective approach to address potential carbon leakage.

Of the 970 products studied, among the highest ranking in terms of their contribution to global EET were upstream or basic products from a focused group of industrial production sectors such as non-ferrous metal, fuels, cement, iron & steel, organic chemicals, primary plastics, inorganic chemicals, and fertiliser sectors. Aggregating the 970 product categories (SITC Rev 3, 4-digit level) into 60 sectors (3-digit level), Figure 3.5 shows the level of embodied carbon by sector. The highest shares are attributable to sectors known as the 'basic' or 'heavy' industry sectors. The iron & steel sector accounted for around 11% of all EET in 2006. This is followed by the non-ferrous metal sector, the metal manufactures sector and the primary plastic sector all at around 6%, then the organic chemicals sector at 5%. Trade intensities tend to be higher for these heavy industry sectors relative to other sectors in this category such as cement, lime and non-metallics (which has a 4% share), which may partly explain the difference in their contributions to global EET. In addition to the heavy industry sectors, there are some light manufacturing sectors present in the higher ranks such as textile articles and machinery and industrial equipment. This is due to the trade-intensive nature of these goods.[17]

[17] Hourcade et al. (2007) finds for the UK's trade intensity from Non-EU countries, that textiles and electrical equipment sectors have the highest trade intensity (over 45%) followed by aluminium. In contrast, trade is less intensive for iron & steel and chemicals (between 20-30%), and even less so for cement and paper (5-10%).

*Figure 3.5:* Embodied emissions in trade by sector in 2006




Source: Author's own calculations. These estimates include food and agricultural products. The numbers on the end of the bars indicate the percentage share of total EET attributable to the sector.

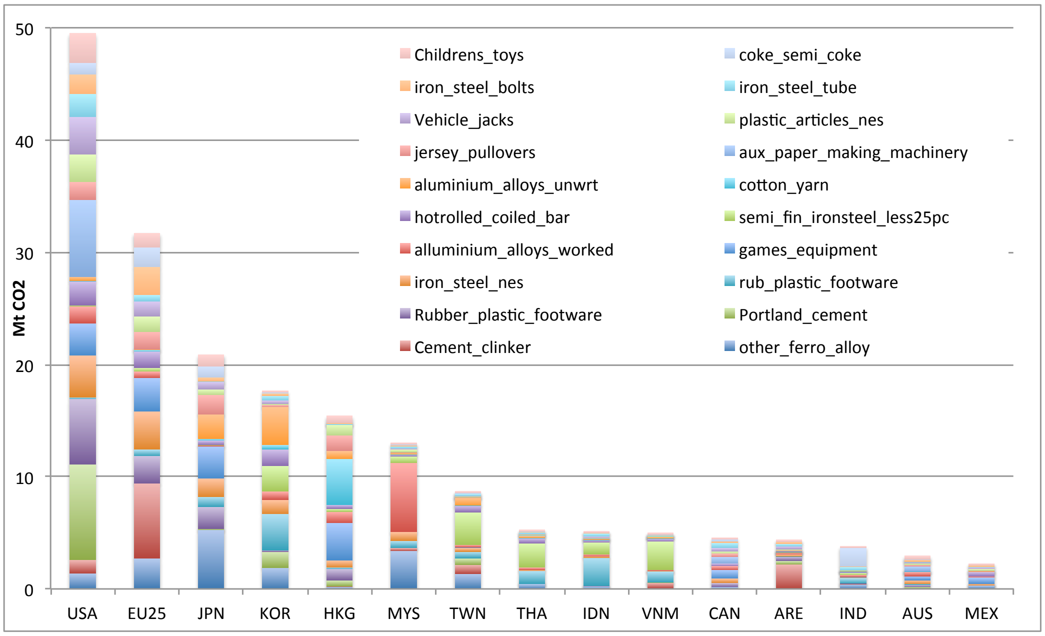
### 3.4.3 There are striking differences in the origin and destination of countries' embodied emissions in imports and exports, as well as the product compositions.

Using EET estimates in China, the US and the EU as examples, this section illustrates how this data reveals striking differences between the EEI and EEE in terms of both the product composition and trade partners.

#### 3.4.3.1 China

China is the world's largest producer of commodities such as ammonia, cement and iron and steel (IEA, 2008). These sectors contribute substantially to both global trade and annual emissions, and are at the centre of the debate on carbon leakage and embodied emissions in trade (e.g. Pan et al., 2008; Liu & Wang, 2009; Peters & Hertwich, 2008; Qi et al., 2008). A number of examinations of China's embodied carbon in trade point out the differences between the sector composition that make up their EEE and EEI (Wang & Watson, 2008; Weber et al., 2008). Here we take a close examination, focusing on the 20 products that account for the largest shares of China's EEE and EEI. The multi-coloured bars in Figures 3.6 and 3.7 plot the volume of embodied emissions attributable to these 20 products, by destination country (the horizontal

*Figure 3.6:* China's product level EEE for key products and trading partners, 2006




Source: Author's own calculations. From the axis origin moving right-ward, the countries are: United States; European Union, Japan, South Korea, Hong Kong, Malaysia, Thailand, Indonesia, Vietnam, Canada, UAE, India and Australia.

axis indicates the trading partners). The small layers that compose the bar illustrate the complex product composition of the embodied carbon trade.

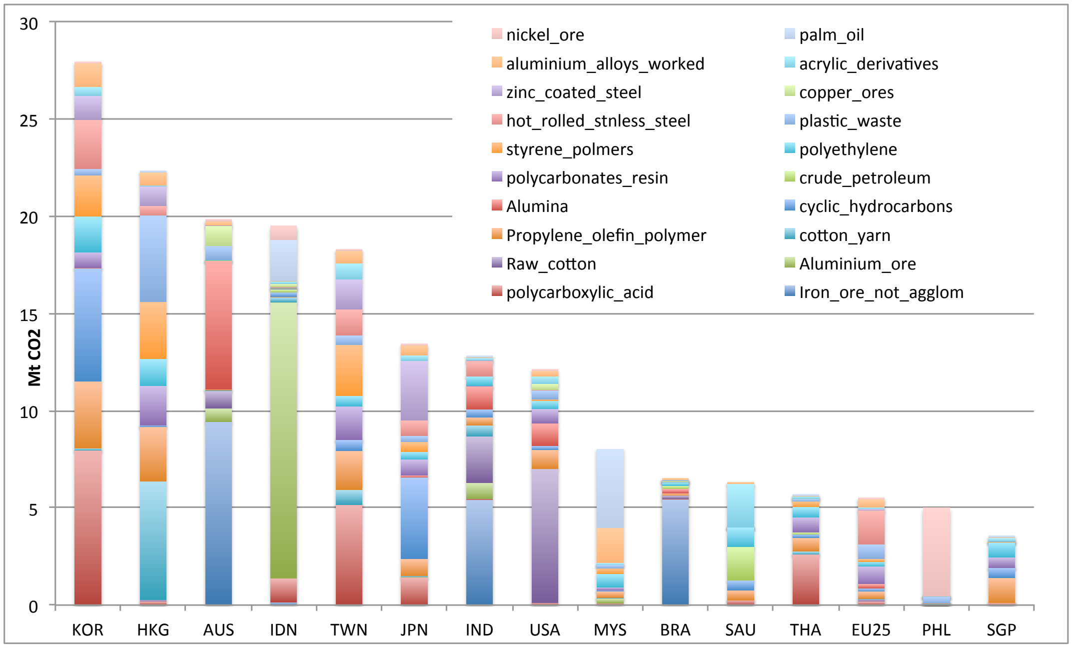
Drawing attention to the legend and the horizontal axis, the products via which China imports and exports embodied carbon, and the trading partners are strikingly different. Carbon imports are embodied in primary products such as iron and aluminium ores (from Australia, Indonesia, Brazil), raw cotton (primarily from US) and basic chemicals and plastics (from Korea and Taiwan). Other key sources of EEI include Hong Kong (largely as re-exports), India, Thailand, Saudi Arabia and Malaysia (Figure 3.7). In contrast, China's carbon exports are embodied primarily in down-stream consumer goods such as games and toys, furniture and apparel products to countries such as the US, EU, Japan and Korea, Hong Kong and Thailand. Also important are upstream industrial products such as basic steel products, chemicals, cement and cement clinker.

Aggregated at the 60 sector level, Figure 3.8 shows the sector composition of China's EET, this time in terms of volumes of net trade. Large EET surpluses are found for metal manufacture, other manufactured foods, apparel, iron & steel, cement & lime, industrial equipment and textile articles sectors. On the other hand, China has a negative net balance of trade in sectors such as metal ore, primary plastic, organic chemicals and textile.

The key sectors identified in China's embodied carbon in exports (e.g. electronics, metal products, apparel) are coherent with previous studies (Weber et al., 2008; Wang & Watson, 2008).[18]

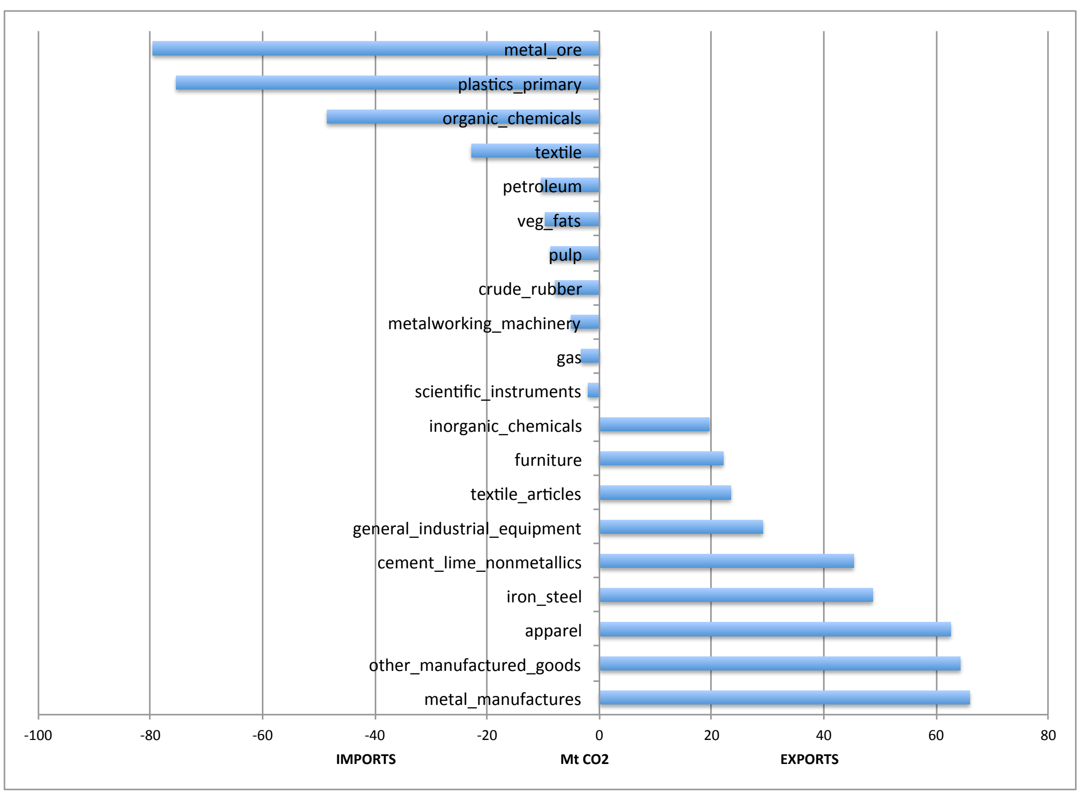
[18] Weber et al. (2008, p. 3574) analyses the change in China's sectoral composition of EEE over time, and reports:

*Figure 3.7:* China's product level EEI for key products and trading partners, 2006




Source: Author's own calculations. From the axis origin moving right-ward, the countries are: South Korea, Hong Kong, Australia, Indonesia, Taiwan, Japan, India, USA, Malaysia, Brazil, Saudi Arabia, Thailand, EU25; Philippines and Singapore.

*Figure 3.8:* China's imbalanced sectors in terms of embodied carbon trade, 2006




Source: Author's own calculations.

However, this analysis offers new insights into the sectoral composition of China's EEI, as well as the geographical distributions of their EEE and EEI. While the destination of Chinese export embodied emissions are focused in the industrialised rich countries (US, Europe, Japan), the origins of China's EEI are more varied. They include mainly neighbouring Pacific countries both rich and poor, but mostly resource rich.

For purpose of verification, on the country level, this study finds China's EEE in 2006 to total around 0.83Gt $CO_2$, around 15% of China's total emissions and EEI to be around 0.6Gt, or 10% of China's domestic emissions. Compared with other studies (see Figure 3.20 in the Appendix), the EEE lies in the lower end of the range from the literature, which is attributed to the use of world average carbon intensity factors. Nonetheless, the insights into the relative importance of products for China's EEE and EEI and the key trade routes still hold.

#### 3.4.3.2 EU

The key products in Europe's export of embodied carbon (external EU trade) include a range of upstream and semi-finished industrial products such as basic iron and steel products (hot rolled steel, hot rolled alloy steel and angled iron and steel), semi-finished steel products (e.g. seamless tubes, seamed pipes), aluminium products, upstream chemical products (carbonates and peroxide carbonates, nitrogen fertilisers), primary plastic products (propylene polymer) glass products, paper products, as well as some downstream manufacturing products such as general machinery and motor engines (Figure 3.9). The key recipients of the EEE from Europe for the select products are the US, Turkey, Russia, Switzerland, China, Brazil and Norway.

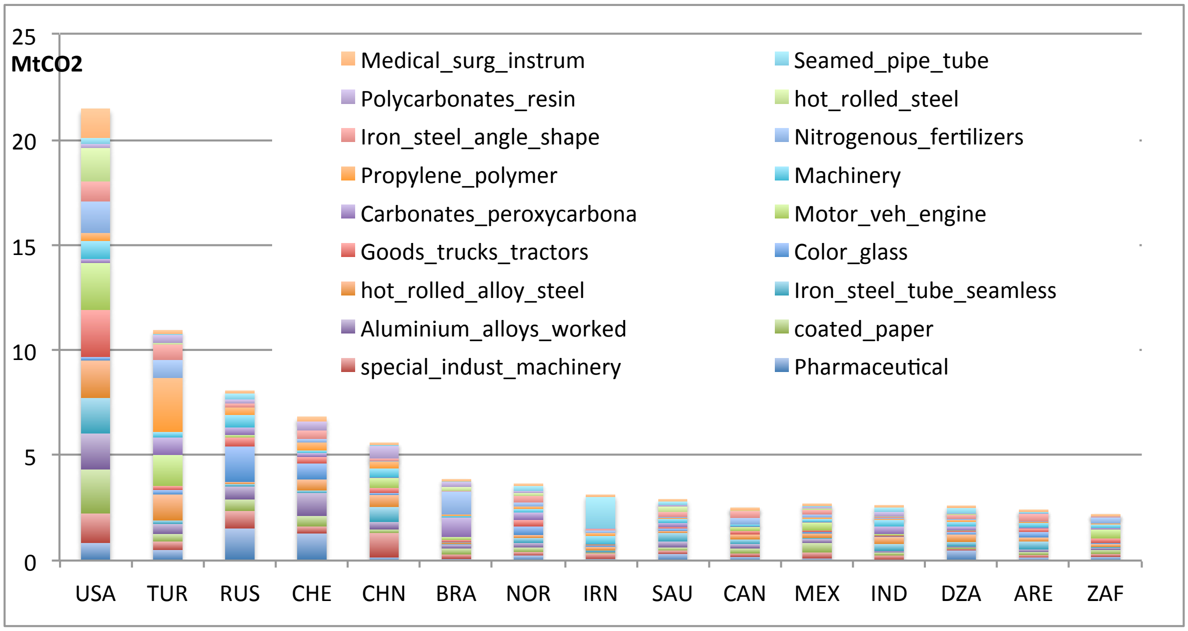
The key products for Europe's imports of embodied carbon, on the other hand, include mined industrial inputs such as iron ore, copper ore, coke, aluminium ore, petroleum oil and other coal products (Figure 3.10). It highlights that Europe's dependence on imports for resource inputs not only has implications for supply chain security but also for the embodied carbon trade balance.[19]

These findings are in line with Giljum et al. (2008) which quantifies the embodied resource content of trade from a North-South perspective: "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). Other

"Emissions embodied in primary product exports (including here: all mining, raw timber, raw chemicals, and basic metals) have decreased from between 20% to 24% in the early years of the analysis (1987–1992) to only 13% in 2002–2005 as the Chinese economy has developed into producing higher value-added items."

[19] Supply chain security has gained significant importance in European Union policies in the past decade. This is emphasised, for example, in the revised EU Sustainable Development Strategy, the Thematic Strategy on the Sustainable Use of Natural Resources and in the upcoming EU Action Plan on Sustainable Consumption and Production: "More than ever, Europe needs to import to export. Tackling restrictions on access to resources such as energy, metals and scrap, primary raw materials including certain agricultural materials, hides and skins must be a high priority. Measures taken by some of our biggest trading partners to restrict access to their supplies of these inputs are causing some EU industries major problems" (European Commission, 2006).

*Figure 3.9:* EU25's product level EEE for key products and trading partners




Source: Author's own calculations. From the axis origin moving right-ward, the countries are: USA, Turkey, Russia, Switzerland, China, Brazil, Norway, Iran, Saudi Arabia, Canada, Mexico, India, Algeria, United Arab Emirates and South Africa.

key products for Europe's EEI include upstream industrial products such as chemical wood pulp, metals and non-ferrous metals (ferro manganese, iron and steel articles and aluminium alloys) and chemical products (nitrogen fertiliser, ether peroxides).

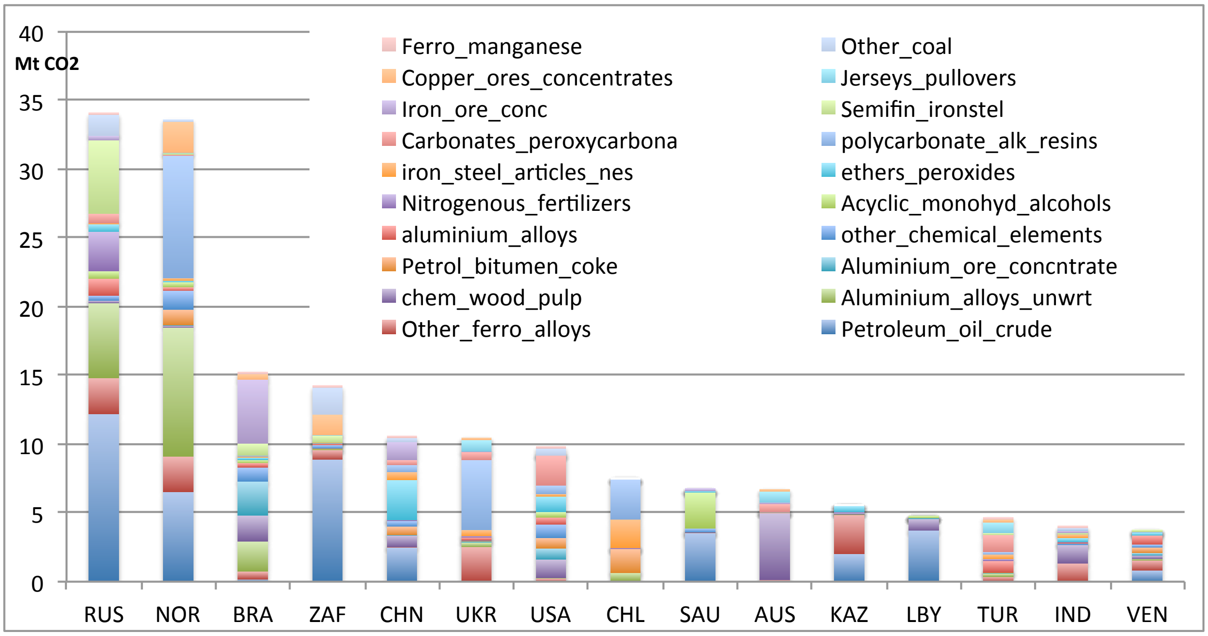
From the horizontal axis of Figure 3.10, it is shown that at least for the selected 20 products, embodied emissions into Europe originate from many resource-rich countries such as Russia, Norway, Brazil, South Africa, China, Ukraine and the US. Comparing the bars in Figures 3.9 and 3.10, Europe's EEE tends to be fairly evenly distributed (perhaps indicating the diversity of Europe's industrial production), whereas imports can be concentrated from particular countries.

In terms of net EET by sector (when aggregated into 60 sectors) Europe's EET imbalances are smaller relative to that of China (comparing Figure 3.11 and 3.8), nonetheless, large surpluses are observed for sectors including metal manufacturing, primary plastic, industrial machinery, general industrial equipments and pharmaceuticals. Large net deficits are found for non-ferrous metals, metal ores, petroleum and apparel.

#### 3.4.3.3 US

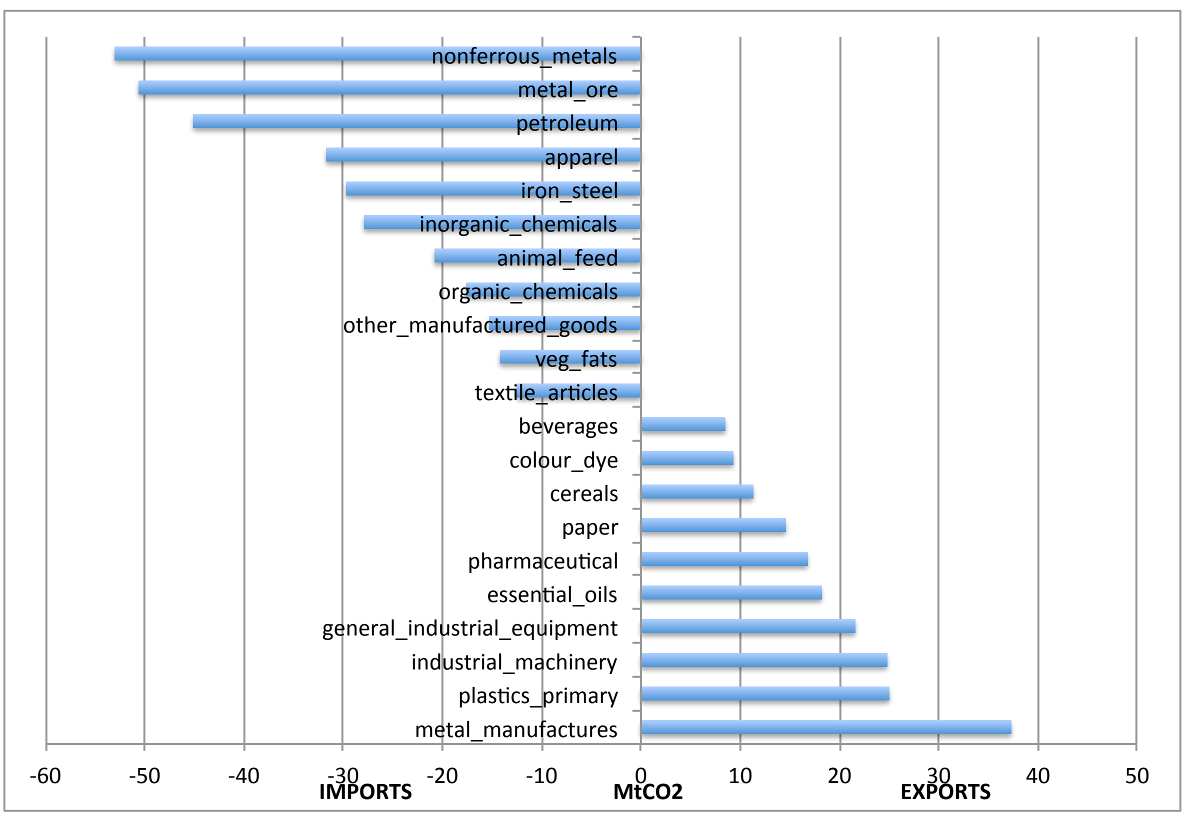
Figure 3.12 shows the key products for the embodied carbon in US export include cotton (particularly to China), other chemical elements (particularly to Japan) carbonates (particularly to Mexico and Brazil), motor vehicle engines (particularly to Canada) aluminium alloys (particularly

*Figure 3.10:* The EU 25's product level EEI for key products and trading partners



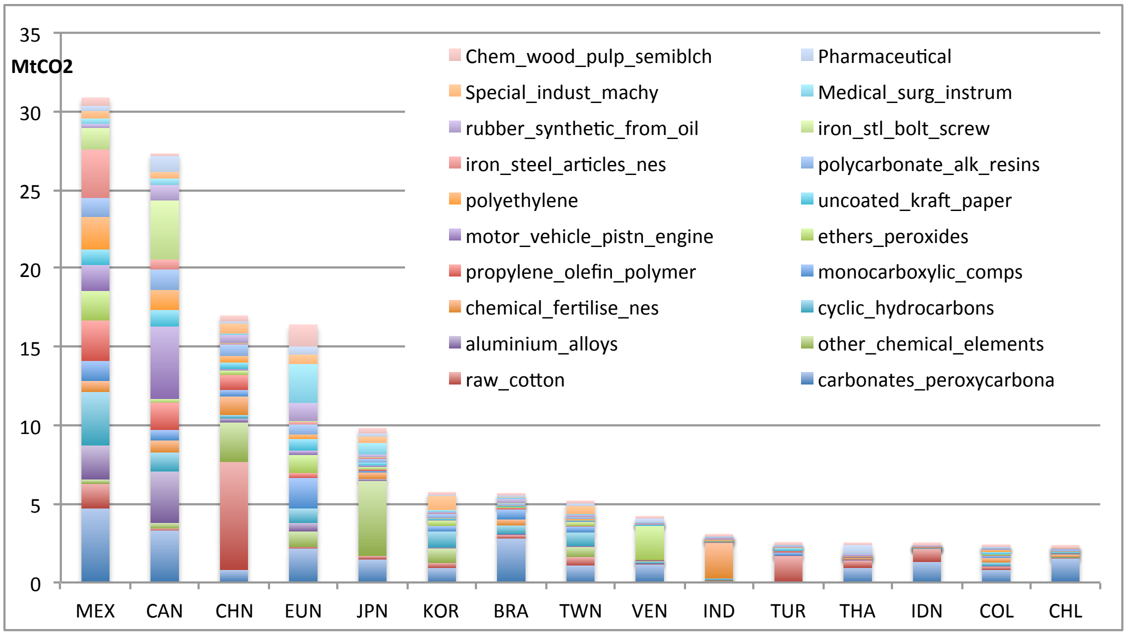

Source: Author's own calculations. From the axis origin moving right-ward, the countries are: Russia, Norway, Brazil, South Africa, China, Ukraine, USA, Chile, Saudi Arabia, Australia, Kazakhstan, Libya, Turkey, India and Venezuela.

*Figure 3.11:* EU25's imbalanced sectors in terms of embodied carbon trade, 2006




Source: Author's own calculations.

*Figure 3.12:* US product level EEE for key products and trading partners




Source: Author's own calculations. From the axis origin moving right-ward, the countries are: Mexico, Canada, China, European Union, Japan, South Korea, Brazil, Taiwan, Venezuela, India Turkey, Thailand, Indonesia, Columbia and Chile.

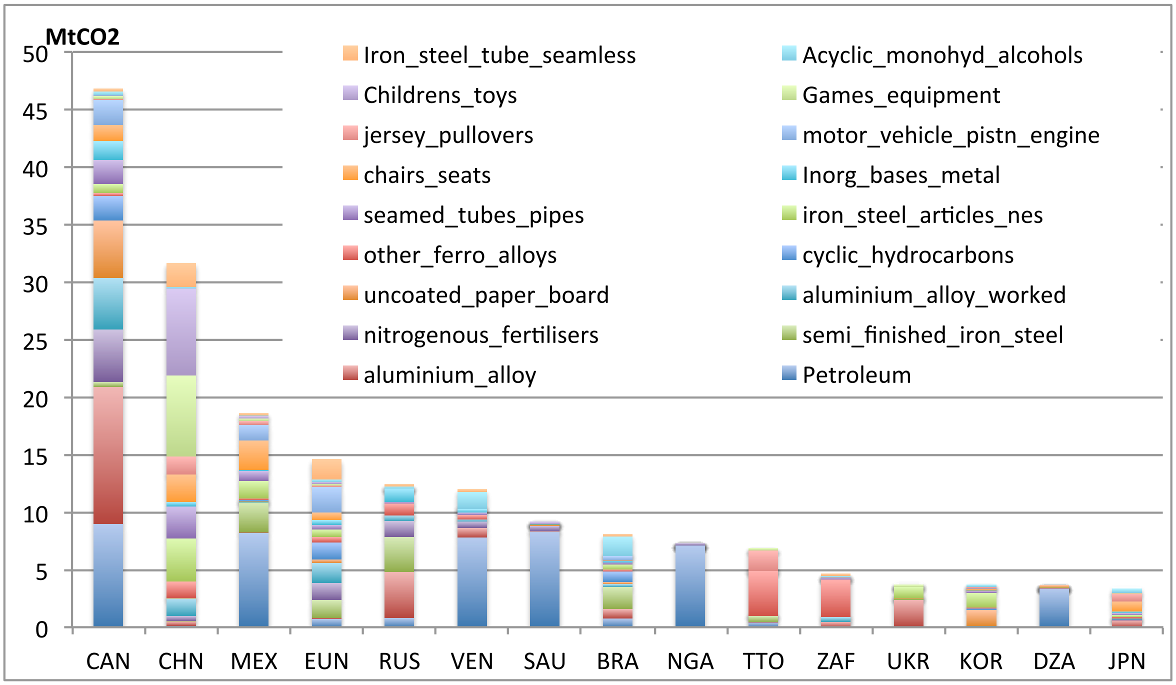
to Canada) and ether peroxides (particularly to Venezuela). This list is in sharp contrast to the key products via which China exports embodied carbon, which tend to be down-stream consumer products.

The products where the EEI are focused for the US include several steel and non-ferrous metal products (aluminium alloys, semi-finished steel, seamed tube pipes, other ferro alloys, iron and steel articles) as well as chemical products (cyclic hydrocarbons, acyclic mono-hydro alcohols and nitrogen fertilisers) as well as petroleum and paper products (uncoated paper). Relative to the top 20 products for China and the EU's EEI, the presence of down-stream consumer goods is notable, including games equipment, children's toys and apparel products (jerseys and pull-overs) (Figure 3.13). The US imports EET via these products from countries such as Canada, China, Mexico, Europe, Russia, Venezuela, Saudi Arabia and Brazil.

In terms of net EET at the 60 sectors level, given the large trade deficit in oil, it is not surprising that the US has a net deficit in embodied carbon trade in petroleum (crude) production (Figure 3.14). However, given the sizeable production output of the US iron and steel industry, it is rather more surprising that net EEI is large in this sector. On the other hand, large net surpluses are found for cereals and inorganic chemicals. The fact that the products in the cereals sector do not appear in Figure 3.12 is likely due to the heavy differentiation of products in this sector at the product level (detailed definition of types of maize, cereal, corn).

In sum, looking at the origins of EEI for China, Europe and US reveals the important role played

*Figure 3.13:* US product level EEI for key products and trading partners




Source: Author's own calculations. From the axis origin moving right-ward, the countries are: Canada, China, Mexico, European Union, Russia, Venezuela, Saudi Arabia, Brazil, Nigeria, Trinidad and Tobago, South Africa, Ukraine, South Korea, Algeria, Japan

by resource rich countries such as Russia, Australia, Brazil, and Canada, in contributions to the carbon flows through global supply chains.

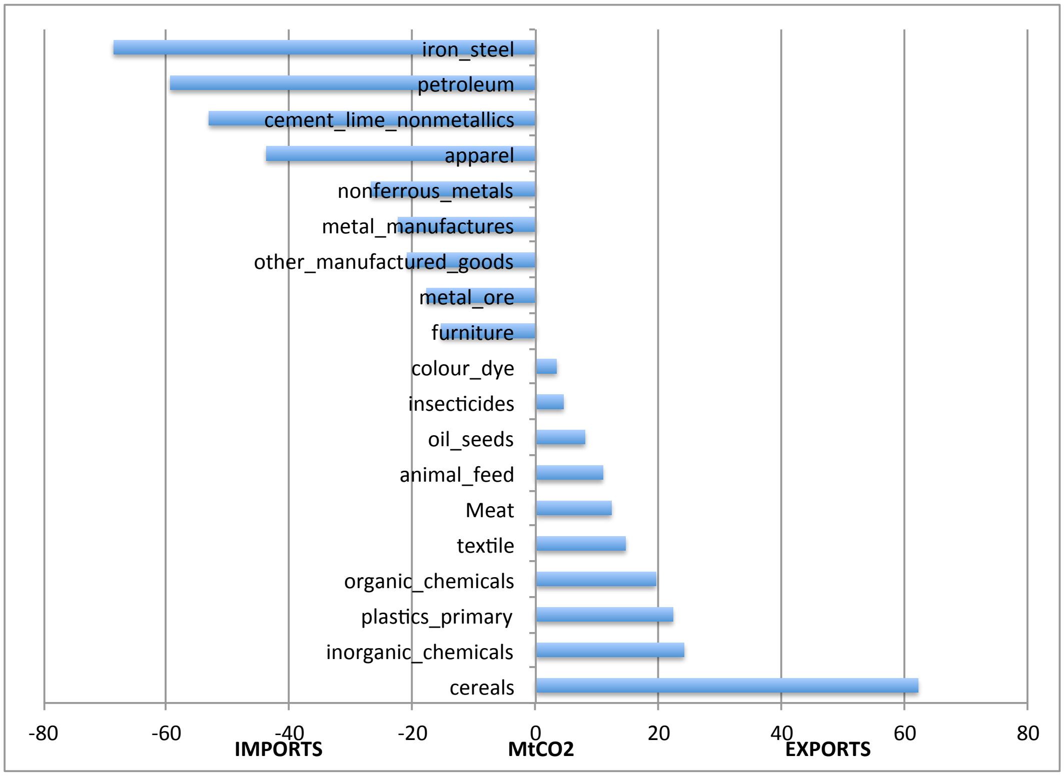
### 3.4.4 Three country types can be identified, in terms of global supply chain positioning.

Further examining cross-country differences in the sector composition of embodied carbon, the EET estimates are evaluated in terms of "supply-chain stages". Here, the product level flows are grouped into seven aggregated stages: food and beverages; fuel and mined products; raw materials; upstream industrial products; industrial processing (mid-stream industrial products); industrial and transport equipment/ machinery; and final consumer goods. [20] Figure 3.15 plots the EEE (blue bars) and EEI (red bars) by supply-chain stage, for four Annex I countries, and Figure 3.16 for four non-Annex I countries to illustrate.

Very different outcomes can be observed. Among the former group, the US stands out because in almost every supply chain stage, their EEE exceeds EEI. This imbalance is particularly true for fuel and mined products, and final consumer goods. Japan is distinct, in that EEE are almost exclusively focused in industrial products from all four stages (upstream, mid stream, machinery

[20] See Table 3.3 in Appendix for sector groupings.

*Figure 3.14:* US imbalanced sectors in terms of embodied carbon trade, 2006



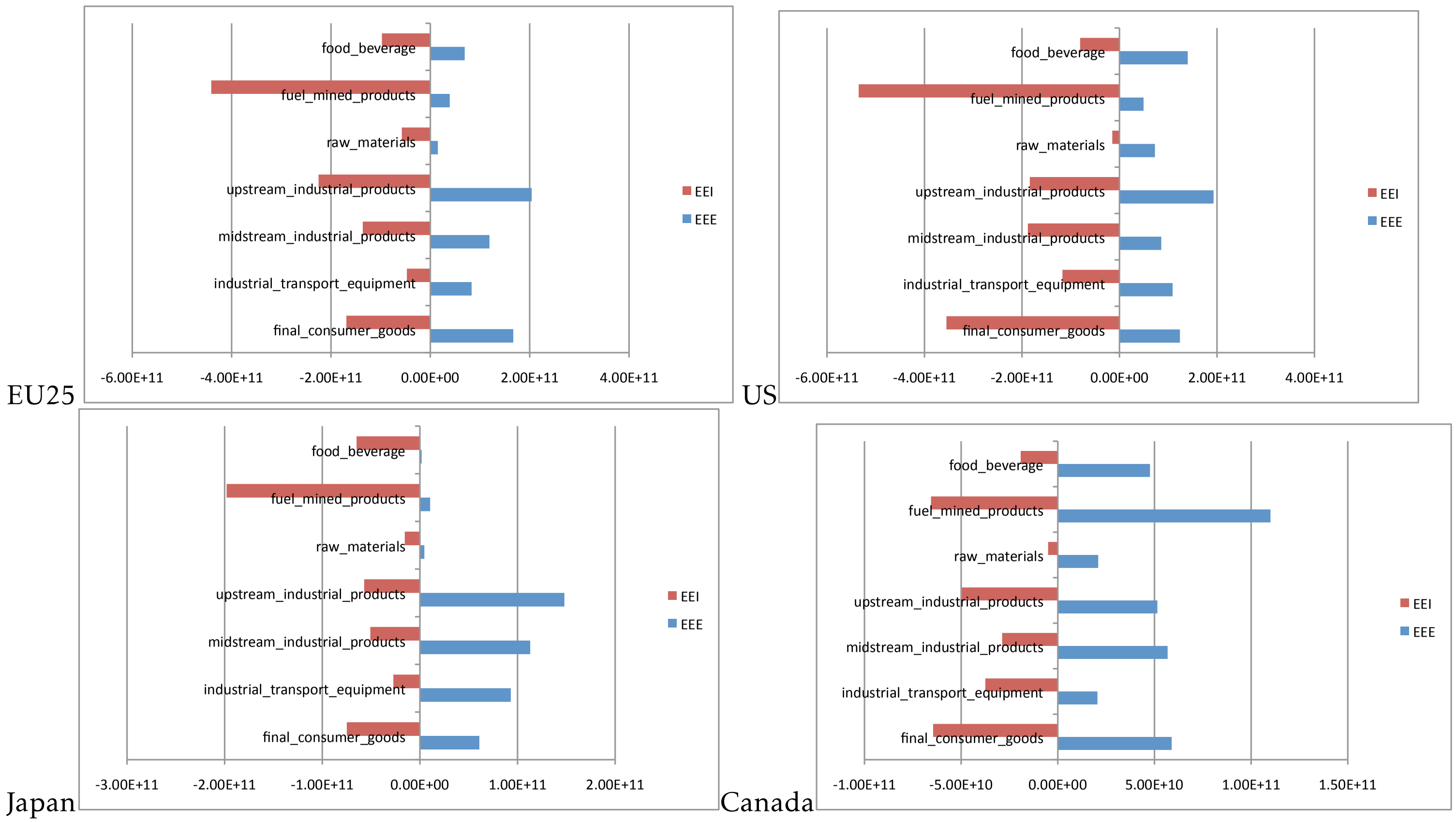

Source: Author's own calculations.

and final consumer goods). Their EEI on the other hand are mostly focused in the primary stages (food and beverages, fuel and mined products and final consumer goods). Canada has a large net surplus in the fuel and mining stage, and the other stages have roughly even amounts of EEE and EEI. The EU appears to be a relatively evenly balanced on the two sides of the Y-axis for almost every stage, except for fuels and mined products. Therefore the net balance of EET is small, although the absolute volumes are large.

Now looking at Figure 3.16. Brazil appears as an archetypal resource-based economy, exporting significant levels of embodied emissions via agricultural products, mined products, raw materials and upstream industrial products. China also exports significant embodied emissions, but in sharp contrast to Brazil, upstream industrial products and final consumer goods account for a large share of EEE. Korea closely resembles Japan and to some extent the EU's position. Russia, rather similar to Canada, has large net surpluses at the fuel and mining stage, as well as the upstream and mid-stream industrial production stages.

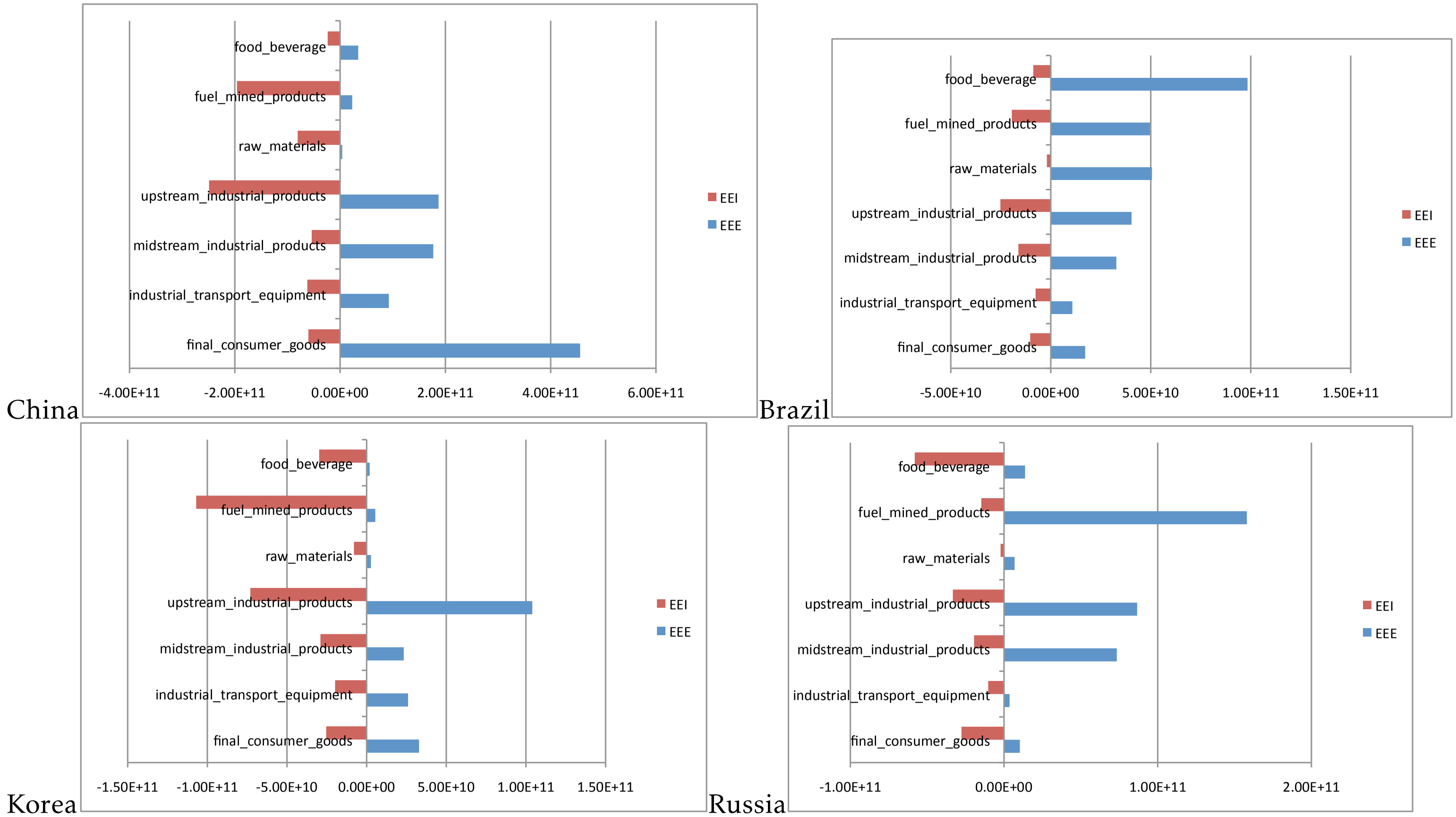
This perspective highlights how the conventional grouping of countries into industrialised countries versus developing countries (Annex I vs non-Annex I) is rather too simplistic of limited relevance now, in the context of climate change policy and trade. This analysis instead suggests that a more relevant grouping of countries may be according to patterns of production and consumption:

*Figure 3.15:* EEE and EEI by supply-chain stage, for EU25(external), US, Japan and Canada




Source: Author's own calculations.

*Figure 3.16:* EEE and EEI by supply-chain stage, for China, Brazil, Korea, Russia




Source: Author's own calculations.

- Production centres – Resource-rich, countries which export large volumes of EEE via 'upstream', industrial feed-stock products, including mined, energy and basic industrial products as inputs to industrial production globally.

- Consumption centres – Heavily service industry oriented countries with significant merchandise or 'downstream' trade imports.

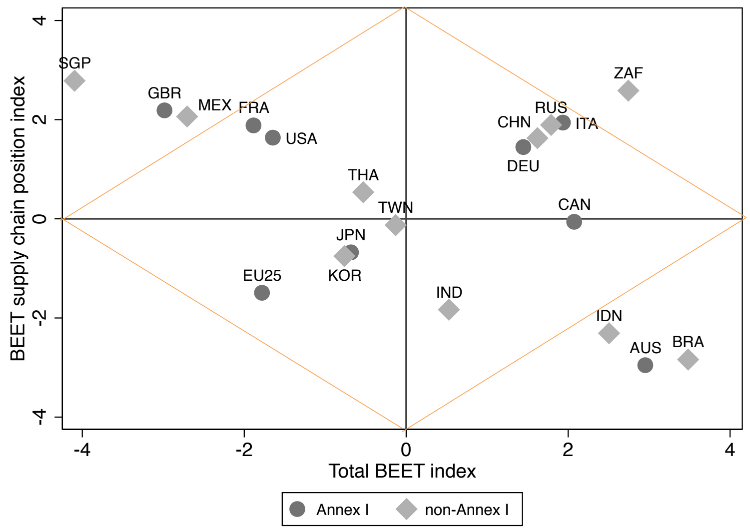
- Production & consumption centres – Importers of upstream or 'mid-stream' industrial products and specialise in assembly or other down-stream industrial processes.

In an attempt to formally characterise the suggested grouping, two simple indicators are developed and applied (Figure 3.17). On the horizontal axis is an index of a country's total BEET for all sectors combined. It is measured by the BEET (total EEE- total EEI), normalised (divided) by the country's production-based emissions to allows for comparison. It is expressed in natural logs, or in the case of net imports (negative) the natural log of the absolute value. On the extreme or 'unbalanced' ends, Singapore and Brazil have an the most negative and positive values of this indicator with the highest relative shares of net imports and net exports of EET respectively.

On the vertical axis is an index of a country's balance of EET in terms of their position on a supply chain (vertical axis). It tries to compare the relative importance of each supply chain stage for any one country, in terms of the stages illustrated in Figures 3.15 and 3.16. It gives indication about whether the imbalance in EET is greater in upstream segments (represented by the bottom of the y-axis) such as fuel and ore production, or slightly along the supply chain in 'mid-stream' products such as basic industrial materials (e.g. cement, pulp, basic metals and basic chemicals), or final consumer goods such as apparel and toys (top of axis). This is measured by a simple summation. I first take the absolute value of the BEET for the 'middle' sectors of the supply chain (such as paper, metal products and plastic manufacturing), subtract that of the 'upstream' sectors (fuel, food and raw materials), then add that of the 'downstream' sectors (consumer foods and transport equipment). Those close to the horizontal line therefore have greater net EET volumes in the 'mid-stream' sectors.

Figure 3.17 combines these two indices. In terms of the three groups named above, the countries closer to the top left corner of the chart can be characterised as "consumption centres" (e.g. UK and Singapore), as net importers of EET with emphasis of EET on the upstream sectors. On the opposite side, countries closer to the bottom right corner represent "production centres" characterised by resource rich economies. Countries that lie closer to the origin can be grouped as "production & consumption centres". This picture suggests that a further distinctions can be made between "production & consumption centres". Very close to the origin are four countries (Thailand, Taiwan, Japan and Korea) which appear to exhibit very similar EET characteristics – small negative balance of overall EET and greatest EET activity in 'mid-stream production stages'. These represent countries with high levels of processing trade (manufacturing of export goods using imported inputs). The USA and France are similar to this group, except that the

*Figure 3.17:* Positioning of countries according to their balance of their trade embodied emissions




Source: Author's own calculations. On the x-axis, the negative values indicate the size of net import and positive values indicate the size of net exports. On the y-axis, negative values indicate 'upstream' and positive values indicate 'downstream'.

negative balance of total EET is likely to be due to the importing of 'down-stream' goods (for the USA, this is consistent with Figure 3.15). Mirroring this, Germany, China, Russia and Italy form a cluster of net exporters of total EET, which is likely due to the export of 'down-stream' goods.

Some observations can be drawn from this perspective from embodied carbon trade balance at the country level. First, it emphasises how the convention of grouping countries into Annex I vs non-Annex I in climate policy debates is unhelpful. Instead, this alternative grouping of countries suggested in this subsection may provide a useful perspective on countries characteristics, and aid discussions around the climate and trade nexus. Second, according to the calculations in this chapter, the majority of large emitters fall into the category of "production & consumption centres". That is to say, on a country level, emission levels are comparable when using the production-based vis-à-vis the consumption-based accounting methods, because they tend to import as much as they export or vice versa. Of course the same cannot be said for the balance of EET at the sector or product level. This suggests that the role of consumption-based accounting methods may be limited at the country level, for example in the context of multilateral burden sharing agreements. Given the large uncertainties surrounding EET measurement as highlighted in Chapter 2, it is likely that the costs of reaching international agreement on a reasonable range of estimates may far outweigh the gains from incorporating consumption-based metrics into such already politically sensitive decisions. On the other hand, the role of consumption-based accounting methods may be important at the sector level, particularly for key energy-intensive and trade-intensive sectors. Efforts to improve the estimations of EET flows for such sectors is likely to add more value than repeating country-level estimations (as has been the trend in the literature to date), more for the discussions about carbon leakage than about fairness and responsibility. For data reasons, this analysis is not able to explicitly quantify 'strong' carbon leakage effects as explained in Section 3.1. However, it is hoped that by identifying the key products and trade routes of EET, this analysis has helped in the process of narrowing down the geographical and sectoral scope, for future work to target.

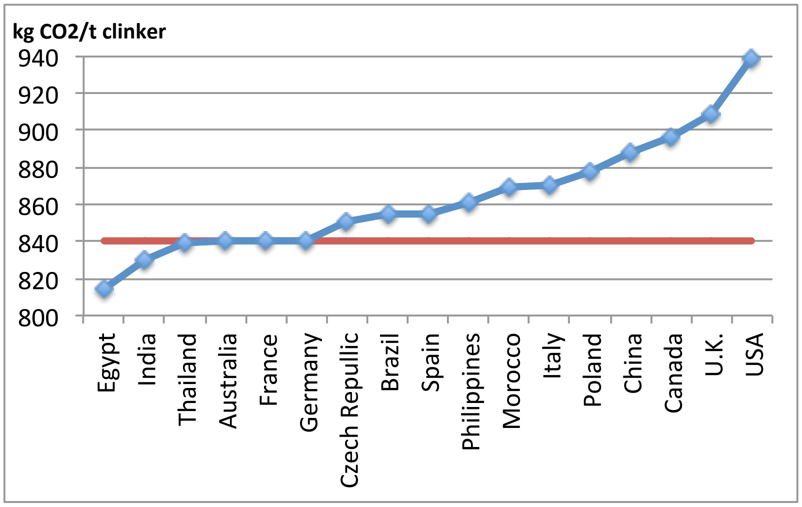
## 3.5 Sensitivity analysis - A comparison of the WAEF and CSEF assumptions in the case of cement clinker trade

In this subsection, sensitivity of the results to the use of world average emission factors is explored using a case study of the cement clinker trade. Currently, the extent to which using WAEF affects the accuracy of results is poorly understood. MRIO analysis has shown that the assumptions about carbon intensity matter, usually by comparing EET estimates when using country-specific emission factors vis-à-vi the domestic technology assumption (DTA) i.e. assuming imports are produced using the same technology as domestic production. In the case of Norway, applying the DTA can underestimate emissions by up to a factor of 2.5 (Peters & Hertwich, 2006). Andrew et al. (2009) compares the WAEF assumption relative to DTA when estimating EET within a SRIO framework and rather unsurprisingly, finds the WAEF perform better particularly for smaller or energy-intensity 'outlier' countries. To the author's knowledge, the relevant comparison between using WAEF and country-specific emission factors has yet to be made. Previous comparisons have also been based on estimation using data expressed in monetary terms (kg $CO_2$/USD) rather than in physical quantity terms.

Cement manufacturing accounts for around 5-7% of global emission (Benhelal et al., 2013), and clinker production is the most energy-intensive step , accounting for around 80% of the energy used. International differences in carbon intensity of clinker production is driven mainly by the thermal efficiency of plants (which strongly relates to kiln technology type and age of installations) and the carbon intensity of the fuel mix (fossil fuels, waste and biomass). Relative to the most efficient plant type (preheater kilns with precalciner or PH-PC), long dry kilns consume around 33% more thermal energy and the old wet kilns consume up to 85% more (Cement Sustainability Initiative, 2009). In addition, capacity utilisation rate and asset rationalisation (turnover and asset renewal times) can strongly influence the regional average thermal consumption. Operating an installation at just a small fraction of its design capacity increases the energy consumption per ton clinker produced.

Using 2006 bilateral trade dat in cement clinker (sector 66121 using SITC Revision 3 classification) between the 17 countries in the sample, I estimate EET volumes for each country pair and both directions of trade. This gives a sample of 176 flows for which the EET estimates can be compared. Using the WAEF, the embodied emissions in bilateral trade between these countries totalled 11.9Mt $CO_2$, whereas using CSEF, it totalled 12.3 Mt $CO_2$. The latter is higher because in this sample there are more countries with CSEF greater than WAEF as shown in Figure 3.18. For

*Figure 3.18:* Weighted average $CO_2$ (excluding $CO_2$ from electric power) emission per tonne clinker by country in 2006

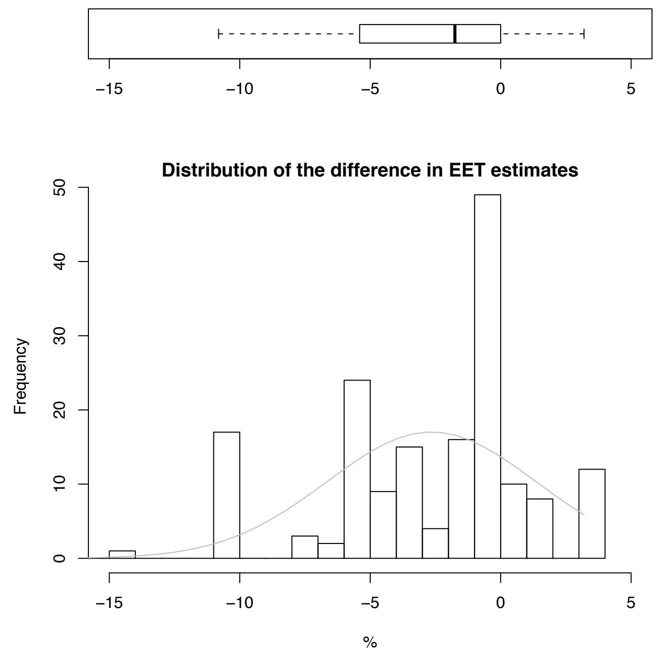



Source and Notes: The red line shows the WAEF used in this analysis and the blue line shows the CSEF obtained from Cement Sustainability Initiative (2013).

each EET flow, I took the difference between the two estimated EET volumes, and divided it by the estimate using CSEF, in order to calculate the impact of the WAEF assumption in percentage terms. The results are described in Figure 3.19 in which the histogram shows the distribution of the inconsistency across the 176 flows, and the box-plot above shows the quartile ranges. I find that the WAEF on average underestimates embodied emissions in clinker by 2%. On the more affected end, EEE from the US, UK and Canada are systematically underestimated by 6-10%. This is a relatively small sensitivity in the context of EET measurement, where assumptions can swing estimate results by orders of magnitude discussed in Section 2.4.

Figure 3.18 shows how the country-averages diverged from the world average emission factor in 2006 – WAEF was 840kg $CO_2$ /tonne of clinker as shown by the red line and the CSEFs ranged between 814-939kg $CO_2$/t clinker across 17 countries. The data is obtained from the "Getting the Numbers Right" (GNR) database, which is high quality environmental and production data collected by the WBCSD's Cement Sustainability Initiative. The coverage of plants in this database is more comprehensive (>70%) for Europe, North America, Central America and Brazil but varies for the rest of the world (Cement Sustainability Initiative, 2013). The high average carbon intensity in the USA shown in the Figure is due to the relatively large number of wet, semi-wet and long dry kilns. This is in turn due to the slow asset renewal driven by low energy prices and lengthy procedures for new kiln permits. Preheater kilns with or without precalciner are more dominant in China, India and rest of Asia and Australia reflecting the growing cement market and relatively young assets. The average thermal efficiency is about 10% better in the non-Annex 1 region than in the Annex 1 region, reflecting the generally newer, more efficient equipment in non-Annex 1 countries.

*Figure 3.19:* Sensitivity analysis - inconsistency in EET estimates using WAEF and CSEF for the case of bilateral trade in clinker, 2006




Source and Notes: The red line shows the WAEF used in this analysis and the blue line shows the CSEF obtained from Cement Sustainability Initiative (2013).

Of course, the sensitivity of the EET estimates to the WAEF assumption varies across products. Greater sensitivity may be found for products such as aluminium and steel which exhibit large heterogeneity in carbon intensities across production plants. In the case of aluminium, this is a function of the source of electricity (from zero carbon hydro or nuclear to high-carbon coal plants), as well as the share of recycled aluminium. For steel, the electric arc furnace (EAF) plants typically use 30-40% of the energy required for the blast oxygen furnace (BOF) plants (Hourcade et al., 2007). The available data was insufficient to conduct sensitivity analysis for these sectors.[21]

This case studies also provides some insights into the use and adjustments of carbon intensities in general. One way to address the lack of country specific carbon intensity data is to systematically adjust world average coefficients, according to weights that reflect an average technology level of a country, typically measured by the average carbon intensity of GDP. This approach has been applied by the GTAP to fill data gaps, but it requires the assumption that the technology level does not vary across sectors within a country. Having 'country specific' carbon intensities has obvious advantages for the analysis of carbon leakage. Yet the cement sector shows that this may be a rather arbitrary way to adjust emission factors. At 1041 $tCO_2$/ Million $GDP,

[21] For the case of aluminium, data on the production share of primary and secondary aluminium was available for many countries, but not the carbon intensities of primary and secondary production by country. For steel, the carbon intensities for BOF and EAF were available at the regional level, but not the share of production.

China has a much higher carbon intensity of GDP relative to others such as Australia (760 $tCO_2$/ Million $GDP), Egypt (504 $tCO_2$/ Million $GDP), USA (451 $tCO_2$/ Million $GDP), UK( 271 $tCO_2$/ Million $GDP) and France (204 $tCO_2$/ Million $GDP). Yet as shown in Figure 3.18, China's carbon intensity in the cement clinker sector is lower than the UK or the US. This analysis shows that such simple adjustment does not leads to improvements in emission factors. Indeed, it has been shown that this assumption contributes to inconsistent country level annual emission volumes when comparing to the UNFCCC and IPCC data (Reinvang & Peters, 2008). The majority of multi-regional analysis of embodied carbon, carbon leakage and related studies on impacts from border adjustments (e.g. Mattoo et al. (2009)) rely to varying degrees on such artificially adjusted emission factors, and this should be an important caveat to their results. This chapter suggests that obtaining reliable country specific emission factors for the key products identified in Section 3.4.2 will go a long way to improve the reliability of such analyses.

## 3.6 Discussion and conclusion

High resolution product level bilateral trade data from the COMTRADE was combined with carbon intensity coefficients, to obtain a detailed mapping of global embodied carbon trade. Like previous studies, this analysis finds that significant volumes of carbon emissions are traded between countries. However, thanks to the level of disaggregation that was not available in previous studies, this paper has revealed new insights into the nature of these flows.

For example, considerable emphasis has been put on China's large surplus and the US's large deficit in the embodied carbon literature. This study shows that for the US, the embodied carbon trade flows with neighbouring countries such as Canada and Mexico are also important. The emissions attributable to EU internal trade is also substantial. It suggests that regional harmonisation of climate mitigation policy should be a priority, perhaps more so than efforts farther afield. Focusing too much on the Annex I and non-Annex I imbalance of embodied carbon in trade invites simplistic and problematic interpretations of EET estimates. It is often combined, for example, with a literal interpretation of classical trade theory based on the notion of comparative advantage, giving rise to interpretations such as "rich countries are outsourcing carbon-dioxide emissions" (The Economist, 2011).

In terms of the distribution of global EET across products, of the 970 products examined, around 15% of the products account for around 70% of global EET. This suggests that focusing mitigation efforts and trade-measures on the products in this group would be an effective approach to address potential trade related distortions, and will also help decarbonising international supply chains. Such product-specific measures could be better justified on environmental grounds, and less vulnerable to criticism of applying trade protectionist measures. As a first step in this direction, it narrows down the products for which rectifying data constraints about their carbon footprints should be a priority.

Examining product level bilateral trade in EET revealed striking differences in terms of the product composition of a country's EEE and EEI. China's carbon imports are typically embodied in primary inputs to industrial production: mined products such as iron and aluminium ores, raw cotton, and basic chemicals and plastics. In contrast, significant volumes of embodied carbon are exported via manufactured products such as games and toys, furniture and apparel products, and also upstream industrial products such as basic steel products, chemicals, cement and cement clinker. The origin and destination of countries' EEI and EEE are also very different. This shows that product and country coverage is therefore key to the impact and effectiveness of measures designed to address carbon leakage.

Looking at the origins of EEI for China, Europe and US in this paper revealed the important role played by resource rich countries such as Russia, Australia, Brazil, and Canada, in contributions to carbon flows through global supply chains. Indeed, from a global supply chain perspective, the results found that at the top of the chain, a non-trivial volume of EET flows can be attributed to energy products and metal ores, particularly as imports by large industrial centres such as China, Japan, and Korea. Indeed, concerns about the consistency between long-term GHG concentration stabilisation goals and the signing of long-term contracts and trade deals between Australian mining companies and Chinese companies have been raised (The New York Times, 2010). Further down-stream in the supply chain, embodied carbon is traded in various upstream industrial products, such as in the iron and steel sector, primary plastics and non ferrous metals.

Examining cross-country differences EET composition in terms of three supply-chain stages showed that the majority of large emitters import and export similar amounts of embodied carbon via 'midstream' industrial goods such as iron & steel, chemicals, paper & pulp and glass. Some countries have a notable EET surplus through large export volumes of 'upstream' production such as ores and fuel (e.g. Brazil and Australia), whereas others have a notable EET deficit through imports of 'downstream', or consumer goods (e.g. UK and Singapore). It is argued that grouping countries according to patterns of production and consumption may be more relevant in discussions surrounding climate policy and trade, rather than discussing in terms of industrialised vs developing countries, as is often done.

For example, the fact that most large emitting countries have a small net balance of EET at a country level suggests that the role of consumption-based accounting methods may be limited at the country level, for example in the context of multilateral burden sharing agreements. Given the large uncertainties surrounding EET measurement as highlighted in Chapter 2, it is likely that the costs of reaching international agreement on a reasonable range of estimates may far outweigh the gains from incorporating consumption-based metrics into such already politically sensitive decisions. On the other hand, the role of consumption-based accounting methods may be important at the sector level, particularly for key energy-intensive and trade-intensive sectors. This suggests efforts to improve the estimations of EET flows for such sectors is likely to add more value than repeating country-level estimations (as has been the trend in the literature to date), more for the discussions about carbon leakage than about fairness and responsibility.

Relevant constraints to the material balance approach have been highlighted in this chapter. A sensitivity test was conducted using a case study of cement clinker to examine how results vary when using world average emission factors and country specific ones. It showed differences up to around 10%, but typically much smaller. The uncertainty due to this assumption is relatively small, compared to the many other sources of uncertainty in EET estimation. It also shed light on problems with simple methods commonly used in the literature to artificially create country-specific sector level emission factors, as well as analysis (such as carbon leakage assessments) using such data.

One important assumption that was not dealt with in depth in this chapter, is the treatment of imported products in the estimation of EEE, as domestically sourced. This assumption may lead to an overestimation of EEE for countries with substantial volume of intermediate-goods trade such as China (the import content of China's exports in 2005 was around 28% as shown in Figure 1.3 on page 19). The magnitude of the overestimation does not seem large, however, when comparing this study's estimate of China's total EEE with five other studies (Figure 3.20 on page 93). Methods to address this issue outside the use of a regional input-output framework have been put forward and applied (e.g. by Huimin & Ye (2010) in their study on China) but with the large number of countries covered, doing so was beyond the scope of this analysis. Overall, the increasing availability of embodied carbon estimates for more products and regions will improve the robustness of estimates under the approach used in this Nonetheless, this examination demonstrated that there is value in providing product-level embodied carbon flows in bilateral trade. It provides novel insights into the nature of the flows, which was not possible in preceding studies, and country-total estimates are comparable to other studies.

Two new datasets were constructed – product level global bilateral trade in physical quantities and carbon intensities of products. These will be made public upon completion of this thesis and it is hoped that they will contribute towards new research, to complement other EET datasets in the public sphere e.g. Peters et al. (2011b); Davis & Caldeira (2010); Davis et al. (2011).

## 3.7 Appendix

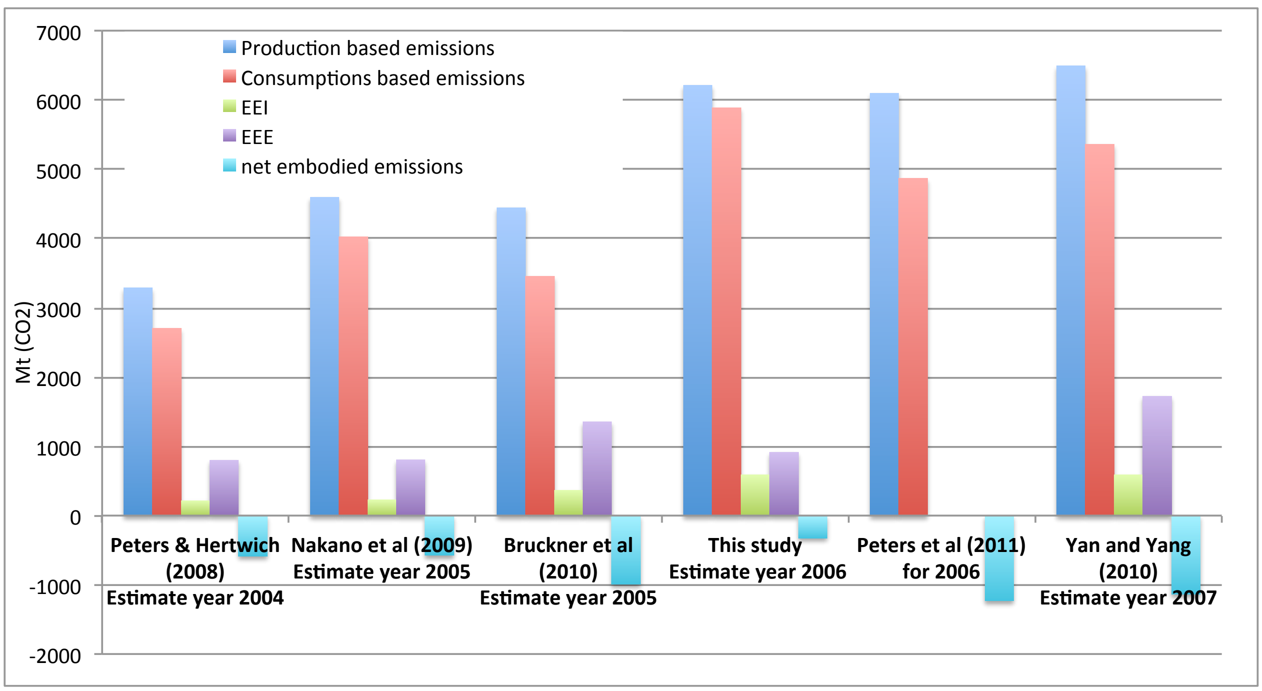
*Table 3.2:* Carbon intensity factors, summary statistics

| | Carbon intensity (kg $CO_2$/kg) |
|---|---|
| Mean | 3.069838 |
| Standard deviation | 4.763699 |
| Median | 2.580637 |
| Minimum | 0 |
| Max | 69.74235 |
| Variance | 22.69283 |
| Skewness | 7.962029 |
| Kurtosis | 91.21711 |
| N | 1026 |

*Table 3.3:* Supply chain stage sector groupings

| | Sectors |
|---|---|
| Food and beverages | meat; dairy; fish; cereals; veg and fruit; sugars; coffee tea cocoa; animal feed; other food; beverages; tobacco |
| Fuel and mined products | metal ore; coal coke; petroleum; gas ; electricity |
| Raw materials | hides skins; oil seeds; crude rubber;cork wood; pulp; textile; crude fertiliser; crude animal material; animal fats; veg. fats; processed fats; leather |
| Upstream industrial products | organic_chemicals; inorganic chemicals; colour dye; fertilisers; plastics primary; cement lime non-metallics; iron steel; nonferrous metals |
| Midstream industrial products | essential oils; plastic non primary; insecticides; rubber manufactures; cork manufactures; textile articles |
| Industrial and transport equipment | power generating machines; industrial machinery; metalworking machinery; general industrial equipment; office machinery; telecom machinery; electrical machinery; road vehicles; non-road transport; power generating machines; industrial machinery; metal working machinery; general industrial equipment; office machinery; telecom machinery; electrical machinery; road vehicles; scientific instruments |
| Final consumer goods | pharmaceutical; paper; metal manufactures; prefab buildings; furniture; travel goods; apparel; foot ware; photo equipment; optical wear; other manufactured goods |

*Figure 3.20:* Estimates of embodied carbon in trade for China across six studies for estimate years 2004 to 2007




Notes: China's production-based emissions for this study is obtained from World Resource Institute (2012).

# Part II

# Carbon leakage effects

# Chapter 4

# Asymmetric industrial energy prices and international trade

## 4.1 Introduction

In recent years, carbon mitigation policies targeting industry sector emissions have proliferated across the world. These include the European Union Emissions Trading Scheme (EU ETS), Australia's Carbon Pricing Mechanism (CPM), New Zealand's ETS, the UK's Climate Change Levy, and British Columbia's carbon tax scheme. There are also many in the pipeline including California's climate programme and pilot trading schemes in some of China's provinces and cities. These policies are intended to provide carbon-intensive sectors with an incentive to reduce their emissions.

As countries adjust to stricter climate change policies at different speeds to reflect national circumstances, one of the recurring debates is around the potential adverse impacts on trade and investment decisions. In a closed economy without trade, a carbon price signal provides incentives for efficiency improvements in production, innovation activity and demand substitution towards lower carbon goods. However, in a free-trade world, taking the lead internationally by strengthening carbon price signals ahead of others can raise concerns about carbon leakage. Multinational companies may also strategically use trade to avoid strong climate controls by replacing the production of 'dirty' goods with imports from production facilities in regions with lax control. Without appropriate measures to address the potential for carbon leakage, implementing ambitious climate change policies could not only undermine efforts to reduce global emissions. It may also have adverse effects in terms of lost jobs, economic output and export revenue.

Politically, concerns around carbon leakage have been paramount in discussions around regulating industry emissions, and are pivotal to the design of carbon pricing policies. The debate is however compounded by the lack of empirical evidence. Among the barriers to finding statistical evidence is the nascent nature of carbon pricing instruments globally and thus the lack of observed data. Where carbon prices have existed, the levels have been low. This is problematic because disentangling the effect of small carbon prices from the multitude of more dominant factors that drive trade and investment decisions – such as exchange rates, transport costs, trade agreements, and relative costs of labour, capital and other input costs – would necessitate very large datasets with many cross-sections and years. Furthermore, it is still early days, to compare the relative stringency of the existing carbon pricing policies in a meaningful way. Definition of the regulatory stringency variable poses another hurdle to empirical investigation.[1] So far, crude measures like a Kyoto Protocol dummy has been used to test the impact of climate policy on trade empirically (e.g. by Aichele & Felbermayr, 2012).

Consequently, quantification of competitiveness impacts and carbon leakage rates have mostly relied on *ex-ante* simulation approaches using general and partial equilibrium modelling (e.g. Babiker, 2005; Gerlagh & Kuik, 2007; Burniaux & Martins, 2000). It has been shown, however, that carbon leakage rates derived from such models are sensitive to model structure and assumptions made, *inter alia*, about technological changes, supply elasticity of fossil fuels or capital mobility (see Dröge (2009) and Zhou et al. (2010) who provide a recent review of this literature). A wide range of leakage rates are reported from these studies, from -14% (suggesting a positive impact of carbon prices on domestic production) to +130% (the leakage effect more than cancels the policy). This highlights the need for *ex-post* analysis to see if observed data provides statistical evidence. The aim of this chapter is to help fill this empirical gap.

This chapter examines detailed sector-level bilateral trade data from the UN COMTRADE database and investigates the impact of changes in relative energy prices between countries on bilateral trade. Because carbon prices work by increasing the effective price of energy for industry, examining the impacts of historic asymmetries in industrial energy prices offer insights into the potential impact of asymmetric carbon prices (Aldy & Pizer, 2011). Econometrically, this chapter uses dynamic panel techniques within a gravity framework. The analysis is applied to extremely rich data – a strongly balanced panel dataset with 51 countries including countries with varying levels of economic development (2550 country pairs), 66 sectors and over 16 years (1996 to 2011). The richness of the data allows the disentangling of the estimates from a range of potential confounding factors through the inclusion of fixed effects.

Analysis of carbon leakage using econometric methods has a key advantage over those using *ex-ante* model simulation approaches, in that it is possible to scrutinise the statistical relationships by subjecting the results to multiple checks. This chapter strives to assesses the robustness of results rigorously. It puts to test whether the results are driven by the underlying theory, by comparing the results across different model specifications. It also asses the sensitivity of the results to underlying assumptions and variable definitions, and conducts tests to try to reduce

[1] Complications arise, for example, in the EU ETS where allowances were allocated for free to most sectors in the first two implementing phases. In the UK Climate Change Levy, the tax revenue is also recycled back to firms via a cut in the rate of the employer's national insurance contribution.

the possibility that the results are driven by other factors. By subjecting estimates to these tests, it aims to understand the degree to which they are statistically robust. In this sense, it can be argued that econometrically derived estimates of carbon leakage are more grounded in statistical evidence, compared with estimates derived from calibrated models. The parameterisation of such models can in turn be improved using the econometric estimations' results.

This chapter presents evidence that a change in the energy price ratio between countries has a statistically significant impact on the volume of trade between the two countries. This result is robust across a range of alternative model specifications and estimators. In terms of magnitudes, the estimates suggest that a one percent increase in the electricity price ratio between the importer and exporter increases exports from the country with a relatively lower energy price to the country with a relatively higher price by around 0.05% to 0.1%. This effect is an average effect across all manufacturing sectors. Further investigation across sectors reveals that the effect is heterogeneous across sectors.

The key novelties of this chapter are as follows. Firstly, it is the first analysis of this kind to be conducted on a panel data of this scale and coverage. Aldy & Pizer (2011)'s study focuses on the US whereas a related study by Gerlagh & Mathys (2011) cover 14 OECD countries. The results in this study can be interpreted in the context of carbon leakage between and within countries both rich and poor. Secondly, whereas the above two studies respectively examine the influence of energy price and energy abundance in levels, this study explicitly tests the influence of the energy price *asymmetry* between trading partners on bilateral exports. Thirdly, it explores the variation of trade impacts across sectors. Finally, this empirical analysis is in line with the recent advances in econometric techniques in the wider empirical trade literature, including the treatment of dynamic effects and zeros in the data.

The structure of this chapter is as follows. Section 4.2 assesses three strands of relevant literature and asks: what is the evidence underpinning the debate around carbon leakage and competitiveness impacts to date?; what are the insights from the wider empirical environment & trade literature?; and what are the recent developments in the gravity model of trade literature? Section 4.3 is on methodology, describing the empirical strategy, the variable definitions, the econometric models and the data. Section 4.4 reports and discusses the results. Section 4.5 explores the sensitivity of the results to certain assumptions and variable definitions. The final section offers conclusions.

## 4.2 Literature

### 4.2.1 Evidence currently underpinning the carbon leakage and competitiveness debates

Thus far, the vast majority of the quantitative examinations of interactions between climate policy and trade flows employ *ex-ante* model simulation strategies,[2] typically using Computable General Equilibrium (CGE) models (e.g. Babiker, 2005; Burniaux & Martins, 2000; Gerlagh & Kuik, 2007; Kuik & Gerlagh, 2003; Paltsev, 2001). This group of studies simulate different emission reduction targets under the Kyoto Protocol and have estimated a wide range of carbon leakage rates. Central estimates are in the range of 5–25% according to Dröge (2009) and 15–30% according to Lanz et al. (2011). However, in some cases, models report leakage rates below zero because of positive emission effects in the model, which are due to the role of technology spillover from mitigation (Barker et al., 2007). Others report leakage rates above 100%, which imply that emission reduction efforts in one region leads to more global GHG emissions rather than less (if production moves to regions with less efficient technology for example).

These studies have identified several channels of carbon leakage, including price effects in energy markets – carbon price induced improvements in energy efficiency in the regulated region reduces energy demand and world energy prices fall, increasing energy demand in other regions (Kuik & Gerlagh, 2003; Burniaux & Martins, 2000). The studies also demonstrate that carbon leakage estimates are sensitive to model setup and assumptions, for example, supply elasticity of fossil fuels, technological change, trade substitution elasticities, returns to scale, market power and capital mobility to name but a few.

Partial equilibrium analysis has been applied in the context of the EU ETS, to examine its potential impacts on trade and investment for heavy industry (e.g. Demailly & Quirion, 2008; Monjon & Quirion, 2009; Demailly & Quirion, 2006; Hourcade et al., 2007). Sectoral differences in carbon leakage rates estimated in these models reflect the differences in parameters such as carbon intensity of production, abatement potential, ability to pass through abatement costs to consumers, as well as different levels of sensitivity to multiple barriers of trade (e.g. product differentiation, service differentiation, transport costs, capacity constraints and import restrictions). Higher carbon leakage rates are estimated for the steel sector which exhibit high product differentiation but also higher abatement potential, relative to the cement sector, which is characterised by homogeneous products but high transport costs relative to value. Above all, the macroeconomic and partial-equilibrium modelling studies highlight the need for empirical analysis in order to better understand the nature and magnitude of these effects.

Several recent contributions have embarked on filling this important empirical shortcoming. Aichele & Felbermayr (2012) and Aichele & Felbermayr (2011) empirically investigate the carbon leakage and competitiveness impacts of legally binding mitigation targets under the Kyoto

[2] See Dröge (2009) and Zhou et al. (2010) for a review of the literature.

Protocol and find statistically significant and large effects. In the former paper, the authors derive a gravity equation for the carbon content of trade and find that the Kyoto commitment effect is associated with a decrease in domestic emissions by 7%, but an increase in the share of imported embodied carbon emissions over domestic emissions by about 14%. The latter paper uses a matching technique (matching bilateral country pairs) and also finds significant and large effects: Kyoto countries' exports are reduced by 13% to 14%. However, the validity of these results have been questioned in the literature, owing to several caveats. In particular, capturing the environmental stringency of regulation using a Kyoto Protocol dummy is crude and limiting. In addition, the reality was that the EU countries were the only ones among the Annex I group to adopt significant climate policies. Yet the authors report that EU membership does not increase $CO_2$ imports when EU and Kyoto membership are included in the regression. This suggests that the changes to $CO_2$ net imports for Annex I countries in their estimations are driven by factors other than the Kyoto Protocol commitments, most likely the effect of China joining the WTO in 2002, which coincided with most Annex I countries' ratification of the Kyoto Protocol (Branger & Quirion, 2013). This explanation is consistent with the fact that most Annex I countries' $CO_2$ net imports are due to trade with China as shown in Chapter 2.

This chapter instead relates to a small recent literature which seeks to empirically examine the relationship between energy price and trade, and use the results to infer the effects of carbon pricing on future trade patterns. This strategy was pioneered by Aldy & Pizer (2011) in their study which focuses on the US. This paper uses historical variation in industrial energy price across states to investigate its effect on sectoral production and consumption. This enables an empirical investigation of the impact of carbon pricing on US industrial supply and demand, despite the absence of carbon pricing in the US historically. It finds that an increase in energy prices in the US following the introduction of a 15$/ton carbon tax would induce a domestic production decline of between 3 and 4 percent among energy-intensive sectors and a roughly 1 percent increase in imports. The authors also find evidence that responses to energy prices are bigger for industries with higher energy intensity.

In a similar vein, Gerlagh & Mathys (2011) use a country specific energy abundance measure to proxy for marginal energy costs, and investigate its impact on net exports using a panel of 14 high income (OECD) countries over 28 years. The authors find that (i) there is high correlation between energy abundance and price; and (ii) energy abundant countries have a high level of energy embodied in exports relative to imports. These results therefore provide indirect support to the existence of a carbon leakage effect. With respect to these studies, this chapter uses a much wider dataset, covering 60 sectors in 51 countries over 21 years.

### 4.2.2 Insights from the wider empirical literature on environment & trade

The handful of *ex-post* analyses examining the impact of carbon policy on global trade patterns form part of a wealth of empirical studies investigating trade impacts of environmental policy more generally. These are gathered under the heading of the so-called Pollution Haven Effect (PHE). The argument goes, that changes in environmental regulation influence the distribution of polluting industries between countries, because pollution abatement costs affect the firm's decisions on output, trade and investment.[3] Reviews of this literature have shown that the empirical evidence to support this hypothesis is mixed (Copeland & Taylor, 2003; Jaffe et al., 1995; Jeppesen et al., 2002, provide a detailed review).

The environment & trade literature offers important insights into a number of potential pitfalls with the empirical investigation of carbon leakage effects. Firstly, it highlights the importance of using panel data rather than cross-sectional data. Panel data analysis is preferable because fixed effects can be used to control for unobserved heterogeneity at the country-pair level, such as historical and structural factors. It can also better deal with endogenous variables (for example, the level of trade may also impact environmental regulation stringency), and persistence in trade levels over time. van Beers & van den Bergh (1997) using a cross-sectional study of 21 OECD countries, find some evidence that the proxy variable for environmental policy impacts bilateral trade. However, Harris et al. (2002) later extends this analysis using panel data models and finds that the stringency variable is no longer significant when using fixed importer, exporter and time effects. The empirical strategy used to model bilateral trade flows in this chapter draws from recent developments in the gravity model of trade literature (applied in a variety of contexts) and this is reviewed in the Section 4.2.3 below.

Secondly, the lack of good measures of regulatory stringency is problematic. Proxy measures may suffer from measurement error – for example, van Beers & van den Bergh (1997) construct their own indicator based mainly on energy intensity and recycling rates, whereas Grether & de Melo (2004) use the difference in GDP per capita as a proxy. Membership of international environmental agreements may be a crude or poor measure of policy stringency. For studies focusing on the US (e.g. Henderson, 1996; Greenstone, 2002; Ederington et al., 2005), a popular choice has been to use regulatory compliance cost as a share of value added in order to proxy for regulatory stringency, typically using data from the pollution abatement cost expenditures (PACE) (e.g. Levinson & Taylor, 2008). There are several known issues with this variable, including an endogeneity problem.[4] A key strength of using historic energy price, as used herein, is that it circumvents a number of these problems (Aldy & Pizer, 2011).

Thirdly, in a meta-analysis of 11 studies, Jeppesen et al. (2002) finds that the smaller the geographical coverage of the study, the larger the estimated effect of environmental regulation.

[3] The PHE can be distinguished from PHH (pollution haven hypothesis) which postulates that differences in environmental regulation is the most important determinant of industry location and that these differences will result in a relocation of pollution intensive industries out of regions with stringent environmental regulation, and lead to specialisation in these industries in regions with lax environmental regulation. Theoretical models of this hypothesis were formulated by Baumol & Oates (1988) and others. A related strand of literature investigates whether (liberalisation of) trade is good or bad for the environment (e.g. Taylor & Copeland, 2004; Antweiler et al., 2001).

[4] The measure of abatement cost expenditure uses production level as the denominator, however, production level can indirectly impact abatement cost via its impact on turn over, scale economies and technology, which in turn impact industries' ability to meet pollution control.

Lastly, the same meta-analysis cautions against aggregating industries that are in fact heterogeneous, because pooling dissimilar industries can lead to aggregation bias in the results, leading to overestimate the true effect for some sectors, and underestimate the effect for others.

In sum, a good proxy variable for environmental stringency, a panel dataset with reasonably wide country coverage and disaggregation to account for sector heterogeneity are the key ingredients for an empirical analysis of the impact of environmental regulation on trade patterns.

Additionally, related to the environment and trade literature is the body of literature which examines the influence of environmental regulation and firm location. This in turn relates to broader new economic geography (NEG) literature on the influence of factor endowments and geographical considerations on location of production. Models of industry location are based on a number of theories, including industry agglomeration (Ellison & Glaeser, 1999) as well as the interaction between industry and country characteristics (Midelfart-Knarvik et al., 2000). Although the review of this literature is beyond the scope of this sub-section, it is worth noting that several studies in this literature also give support to the PHE (e.g. Mulatu et al. (2009); Kahn & Mansur (2013)).

### 4.2.3 Developments in the gravity model of trade literature

This study uses a dynamic panel within a gravity model framework, controlling for fixed effects, and draws heavily on the developments in the gravity trade literature. The gravity model of trade was first applied by Tinbergen (1962) and Pöyhönen (1963), and in its original form specifies that bilateral trade flows are determined by the economic size of, and the distance between two countries. The underlying principle is that trade relations are influenced by the size and richness of the trading partners, as well as transportation costs. Population and income are used as proxies for demand and supply, and distance proxies for transport costs.[5] This model, in its various formulations, has been widely used as the 'workhorse' of empirical trade studies. A survey by Oguledo & Macphee (1994) found that 49 explanatory variables had been explored empirically in this literature.

There are several different theoretical foundations to the model, with earlier gravity equations derived from models that assume product differentiation (Anderson, 1979) and monopolistic competition (Bergstrand, 1985). Later contributions incorporate economies of scale with production differentiation (Helpman, 1987; Bergstrand, 1990) to reflect developments in trade theory (e.g. Krugman, 1980). These mark a move away from classical Heckecher-Ohlin-Samuelson (HOS) frameworks that focus on factor endowment differentials, that poorly explain trade patterns, towards New Trade Theory (NTT) models that accommodate for intra-industry trade (ITT) and offer explanations of both trade structure and volume. Key determinants in the resulting formulation of the gravity model include relative factor endowment differences, overall bilateral country size, similarity in country size and trade costs (Baltagi et al., 2003).

[5] Gravity models have also been successfully applied to flows including migration and foreign direct investment.

Further important theoretical contributions include Anderson & Van Wincoop (2003) which introduces substitutability between trade with different trading partners by relaxing the constant elasticity of substitution (CES) assumption. This is done, in order to account for multilateral trade resistance factors (or the border effect) which can be represented using time-variant bilateral fixed effects within a panel data framework (Baldwin & Taglioni, 2006; Baier & Bergstrand, 2007). Furthermore, Helpman et al. (2008) proposes a theoretical model that rationalises zero trade flows using a model with firm heterogeneity and a correction for the probability of countries to trade, by introducing the notion that the decision to export is not independent of the volume of exports.

In addition to theoretical developments, a large number of empirical applications in the literature on international trade have also contributed to improving the performance of bilateral trade models.[6] Importantly, papers have demonstrated the sensitivity of estimation results to the model specification. The choice of econometric technique can severely affect the magnitude of the coefficients, although there is usually agreement across models in the sign of the parameters for most gravity variables (Gómez-Herrera, 2013).

## 4.3 Empirical analysis: strategy, econometric model, data and descriptive statistics

### 4.3.1 General approach

The objective of this study is to examine the relationship between historic asymmetries in industrial energy prices and bilateral trade patterns. In particular, I test the hypothesis that when the importer's energy price is higher than its trading partner, a larger energy price gap is associated with greater import flows. The empirical approach adopted to test the empirical predictions is based on the gravity model, as stated above.

The results can then be used to infer the effects of potential asymmetries in carbon price on future trade patterns. The key idea is that the impact of a carbon price on a firm is a function of its energy use, because the level of carbon emissions are largely attributable to energy combustion in production (although in some processes, there are non-energy related emissions also such as process emissions in chemical and cement production). Indeed, high correlation between electricity and carbon prices have been found in Europe (Sijm et al., 2006). The main advantage of using energy prices is that, while experience with carbon prices is still limited globally, historic data on industrial energy prices exists for many countries and many years.

To measure the difference in energy price between two trading partners, I define $epgap_{ijt}$ as the difference in the logs of energy prices, or in other words as a log of the ratio of energy prices:

[6] Some early contributions include Ghosh (1976) and Matyas (1997).

$$epgap_{ijt} = ln(EP_{it}) - ln(EP_{jt})$$

where $EP_{it}$ and $EP_{jt}$ are the real industrial energy price respectively in importer $i$ and exporter $j$ at time $t$. A positive value of $epgap_{ijt}$ implies that the importer $i$ has a higher industrial energy price than the exporter. Note that the energy price data availability is such that the *epgap* varies by country, but not by sector.[7]

In a basic regression framework, I regress the value of trade on the electricity price gap to establish whether there is a link:

$$lntrade_{ijst} = \beta_0 + \beta_1 epgap_{ijt} + X_{ijt} + \varepsilon_{ijst} \tag{4.1}$$

where $lntrade_{ijst}$ is the log-value of annual imports by country $i$ from country $j$ for sector $s$ at time $t$, $X_{ijst}$ is a vector of control variables and $\varepsilon_{ijst}$ is the error term.

The choice of control variables is derived from recent advances in the gravity literature. First, I control for overall bilateral economic size, relative economic size (similarity of GDP) as well as differences in relative factor endowments (similarity of capital-labour ratios) (Baltagi et al., 2003; Wang et al., 2010; Egger, 2000). These three variables are specified as follows:

$$gdpt = ln(GDP_{it} + GDP_{jt})$$

$$gdpsim_{ijt} = ln\left[1 - \left(\frac{GDP_{it}}{GDP_{it} + GDP_{jt}}\right)^2 - \left(\frac{GDP_{jt}}{GDP_{it} + GDP_{jt}}\right)^2\right]$$

$$rfac_{ijt} = \left|ln\left(\frac{GDP_{it}}{CAPITA_{it}}\right) - ln\left(\frac{GDP_{jt}}{CAPITA_{jt}}\right)\right|$$

Overall bilateral economic size, $gdpt$, reflects the fact that the volume of exports should be higher, the bigger the overall market size. $gdpsim_{ijt}$ measures the similarity in the levels of GDP in the trading partners, hence captures the relative size of the two trading partners. Before the log-linear transformation, this variable can take the value between 0 and 0.5. A higher value indicates that the two trading partners are similar in size (GDP), with 0.5 indicating equal country size. Theory predicts that the higher this value, the greater the expected share of inter-industry trade (Egger, 2000). $rfac_{ijt}$ measures the similarity in capital-labour ratios, or in other words, the relative factor endowments. A value of 0 represents equal factor endowments proportion. Bergstrand (1990) illustrates empirically using the gravity model that bilateral trade between high income countries is positively related to similarity in relative factor endowments (reflecting similarity in preferences).

[7] This issue is discussed further in Section 4.3.3.2.

The gravity model also explains trade as a function of invariant country pair-specific determinants such as distance, common language, common borders, common currency, colonial ties. Since there is considerable sector variation in trade intensities, when the trade data is disaggregated at sector level, using country-pair-sector fixed effects can usefully control for time invariant sector characteristics such as the global market structure and the nature of the traded good (e.g. homogeneous commodities are traded more intensively). In addition, time dummies control for common macroeconomic shocks, such as the sharp fall in global trade volumes following the financial crisis in 2008. The range of fixed effects used in this analysis substantially reduce possible bias from omitted variables. At the same time, the specification of the fixed effects has been shown to have a large impact in gravity model estimations, and sensitivity will be examined in Section 4.5. Adding these extra terms in equation 4.1, the basic gravity relationship is modelled as:

$$lntrade_{ijst} = \beta_0 + \beta_1 epgap_{ijt} + \beta_2 gdpt_{ijt} + \beta_3 gdpsim_{ijt} + \beta_4 rfac_{ijt} + \omega_{ijs} + \alpha_t + \varepsilon_{ijst} \tag{4.2}$$

where $\omega_{ijs}$ are the country-pair-time fixed effects and $\alpha_t$ are the time dummies. The primary objective of the study is to estimate the coefficient $\beta_1$. The hypothesis here is that trade is larger when the energy price gap is positive (exporter has the lower energy price) and larger, hence a positive sign is expected for the coefficient $\beta_1$.

The gravity literature recommends applying a number of different specifications to the same data to account for fixed effects, dynamics and the distribution of the dependent variable (Gómez-Herrera, 2013; Head & Mayer, 2013). This chapter will follow this recommendation and use different estimators – OLS, Arellano-Bond GMM, Poisson maximum likelihood (PML), negative binomial and zero inflated negative binomial. The next subsection presents and discusses the underlying issues and methodological constraints, to assess which specification is theoretically most appropriate.

### 4.3.2 Econometric issues

Two key issues that arise when estimating the gravity equation are the persistence of trade over time and the presence of zeros in the dependent variable data.

#### 4.3.2.1 Dynamics

As will be seen in Section 4.3.3, the descriptive features of the sectoral trade data display strong persistence. Therefore it is important to account for trade in past periods, by including one or several lags of the dependent variable. The dynamic specification for a linear model is:

$$lntrade_{ijst} = \sum_{p=1}^{n} \lambda_p lntrade_{ijs(t-p)} + \beta_1 epgap_{ijt} + \beta_2 gdpt_{ijt} + \beta_3 gdpsim_{ijt} + \beta_4 rfac_{ijt} + \omega_{ijs} + \alpha_t + \varepsilon_{ijst} \tag{4.3}$$

where lagged dependent variables enter as $\sum_{p=1}^{n} \lambda_p lntrade_{ijs(t-p)}$, where, $p$ is the number of lags. This log-linearised model (estimated for example by OLS) has several limitations. Lagged dependent variables are correlated with the fixed effect $\omega_{ijs}$ and this can lead to a bias in the estimate of the coefficient on the lagged dependent variables as well as the other regressors that are correlated with the lagged dependent variable. This bias, however, decrease with the length of the panel (Arellano & Honoré, 2001; Nickell, 1981). OLS can be inefficient and biased where the data is heteroskedastic (Santos Silva & Tenreyro, 2006) as is often the case with bilateral trade data. To deal with these issues, a popular estimator is the Arellano-Bond difference GMM (used for example by Carrere, 2006; Baier & Bergstrand, 2007; Olivero & Yotov, 2010; Martínez-Zarzoso et al., 2009). In the first-difference transformation (Equation 4.4), constant and individual effects are removed, reducing serial correlation:

$$\Delta lntrade_{ijt} = \beta_1 \Delta epgap_{ijt} + \beta_2 \Delta gdpt_{ijt} + \beta_3 \Delta gdpsim_{ijt} + \beta_4 \Delta rfac_{ijt} + \sum_{p=1}^{n} \lambda_p \Delta lntrade_{ijs(t-p)} + \alpha_t + \Delta\varepsilon_{ijst} \tag{4.4}$$

The differenced equation is treated as a system of equations for each year $t$. Parameters are estimated by using lagged values of the dependent variable as instruments of the endogenous variables as well as the strictly exogenous regressors. It is particularly suited to panel data with a short time dimension and large cross-section, as is the case in this analysis. A common problem with the GMM estimation is that when the series is highly persistent (as is the case with trade), the instruments are very weak predictors and this can leads to a small sample bias. However, weak instruments tests conducted showed that the results presented in Section 4.4 below are robust to this.

#### 4.3.2.2 Presence of zeros

A common problem with trade data is the presence of zeros. With linear models such as OLS and Arellano-Bond GMM, two options can be used to deal with this issue: using $lntrade_{ijst} = ln(trade_{ijs} + 1)$ as the dependent variable, and running the estimations on the sub-sample of strictly positive observations (unbalanced panel). An alternative approach is to use non-linear models – studies have shown that using such models initially developed for count data analysis can be successfully applied to continuous variables (Wooldridge, 2010). Since the value of trade between two countries in any period is a non-negative integer, it is natural to model the conditional mean as a log-link function of explanatory factors and estimate using the PML model. Cluster robust standard errors are used to obtain correct inference: [8]

$$trade_{ijst} = exp(\beta_1 epgap_{ijt} + \lambda_p \sum_{p=1}^{n} trade_{ijs(t-p)} + \beta_2 gdpt_{ijt} + \beta_3 gdpsim_{ijt} + \beta_4 rfac_{ijt} + X_{ijs} + T_t) + u_{ijst} \tag{4.5}$$

where $X_{ijs}$ are country-pair-sector fixed effects, and $T_t$ are time trends. Westerlund & Wilhelmsson (2011) conducts Monte Carlo simulations using panel data and argues that the PML fixed effect estimator eliminates the problem of zero trade, while controlling for heterogeneity. Another approach developed to address the issue of zeros is a two-step estimation technique. As outlined by Helpman et al. (2008), bilateral trade can be viewed as a result of two processes – one that generates zero counts and another that generates positive counts. Practically, this can be implemented using "inflated" models, which essentially has the effect of increasing the variance of the model and increasing the predicted probability of observing zero trade values. The zero inflated negative binomial model, for example, assigns a probability $p$ to generating zero counts, and a probability $(1-p)$ to generate a negative binomial distribution, where $0 \leq p \leq 1$. The probability $p$ is determined by a vector of bilateral characteristics such that $p_{ij} = F(\eta_{ij}\psi)$ where $\eta_{ij}$ is a vector of bilateral characteristics that predict the probability of zero, and $\psi$ are vector parameters to be estimated. A zero inflation approach will indeed be applied in this chapter, which uses data containing a non-trivial share of zeros at 25%.

#### 4.3.2.3 The combination of the issues of dynamics and the presence of zeros

Accounting for dynamic effects in the framework of non-linear models therefore raises further difficulties. While addressing the issue of zeros, the strict exogeneity assumption is violated in equation 4.5 with the inclusion of the lagged dependent variables and fixed effects. A popular technique used to circumvent this problem is to use a pre-sample mean of the dependent variable as a proxy for the time invariant unobserved heterogeneity of export behaviour in place of the true fixed effect (Blundell et al., 2002, 1999). Applications to data on environmental regulation variables include Jug & Mirza (2005) and Egger et al. (2011).[9] Implementing the latter, a pre-sample mean of bilateral exports can be specified as:

$$\overline{trade}_{ijs} = \frac{1}{TP} \sum_{r=0}^{TP-1} trade_{ijsr}$$

[8] The fixed effect PML estimator is available in standard statistical software packages.

[9] Alternatively, applying a quasi-differencing transformation (generalised method of moments estimator) has been suggested (Wooldridge, 1997).

where $TP$ is the number of pre-sample observations. This country-pair and sector specific pre-sample export variable, provides an attractive way to control for unobserved heterogeneity of export patterns, because unlike the true fixed effect, it is exogenous of the lagged dependent variables. The non-linear model with pre-sample mean estimator can be estimated in its multiplicative form directly:

$$\begin{aligned} trade_{ijst} &= exp(\lambda_p \sum_{p=1}^{n} trade_{ijs(t-p)} + \beta_1 epgap_{ijt} + \beta_2 gdpt_{ijt} + \beta_3 gdpsim_{ijt} + \beta_4 rfac_{ijt} \\ &\quad + \gamma D_{ij} + \delta dist_{ij} + \ T_t + \phi \overline{trade}_{ijspresamp}) + u_{ijst} \end{aligned} \tag{4.6}$$

Because the pre-sample mean estimator may fail to capture every aspect of time-invariant country-pair heterogeneity, the equation also includes standard gravity variables: $\gamma D_{ijt}$ is a vector of geographical dummies (contiguity, common official language and common currency), and $dist_{ij}$ is the log of geographical distance. A dummy variable that takes the value of one if the pre-sample export mean is equal to zero is also included.

The pre-sample mean estimator is best suited for data with a relatively stable mean. Yet in the case of trade, there is a tendency for the mean to increase over time hence the pre-sample mean is not a good proxy of the true fixed effect. It is therefore argued that the PML model specified by Equation 4.5 which is equivalent to a model with individual specific constants, may be a more appropriate model. With a 16 year panel, the bias introduced by violating the strict exogeneity assumption is likely to be small.

Another consideration is the existence of over-dispersion in the data – this is not surprising in trade data and was confirmed by two tests for the null hypothesis that the mean equals the variance – Cameron and Trivedi's regression based approach and Greene's test using Lagrange Multiplier. The PML model in its pure form assumes that the data are equi-dispersed i.e. there is equality of conditional mean and variance, therefore $u_{ijt}$ has an expected value of one. Several solutions have been proposed to overcome this problem including the negative binomial model (Helpman, 1987) and the Poisson pseudo-maximum likelihood (PPML) estimator developed by Santos Silva & Tenreyro (2006). The latter approach relaxes the assumption of equality between the conditional variance and the mean, but rather performs optimally when the two are proportional hence can be applied to over-dispersed data.[10] However, the fixed effect PML estimator which is available in standard statistical software packages now relaxes the equi-dispersion assumption hence consistent with the PPML, although the latter is computationally more efficient.

A further point to note about the empirical model's structure is that it yields common elasticities

[10] The authors argue that this is the most natural procedure without any further information on the pattern of heteroskedasticity. They show that the model performs well with cross-sectional data, and can provide a consistent estimator of bilateral trade in gravity models, even when the data has many zeros.

with respect to energy prices for imports and exports. This is because by applying the estimation models to bilateral data which is symmetric, any one observation of bilateral trade can be consider an import and an export at the same time.[11] This is important because the possibility that that some individual trade flows for a particular country may face a larger or smaller impact than the average cannot be ruled out.

### 4.3.3 Data

One of the key strength of this study is the richness of the dataset gathered for the analysis. This dataset records bilateral trade between 51 countries (both high, middle and low income) [12] and 66 sectors for the period 1996 to 2011. The data is disaggregated at 2-digit SITC sector resolution and covers all traded sector categories.[13]

#### 4.3.3.1 Bilateral trade

Sectoral bilateral trade data in US dollars (US$) at current prices for the dependent variable was extracted from UN COMTRADE, via World Integrated Trade Solution (WITS). The bilateral trade data in the sample covers between 70% to 90% of world trade obtained from the WTO Statistics Database (World Trade Organisation, 2012), depending on the year, as shown in Figure 4.1. Here, trade expressed in nominal value terms is used unlike Chapter 3 where trade data in physical quantities is used, because the data in monetary terms is more complete across the 16 year time period. Using trade data in physical quantities is strongly advantageous for quantifying volumes of trade embodied carbon in Chapter 3, as it leads to better understanding of the relative size of trade flows across countries, and avoids problems regarding exchange rates and price assumptions. It is less important in this chapter, which examines the 'within' variation in bilateral trade over time, for a specific sector and country pair.

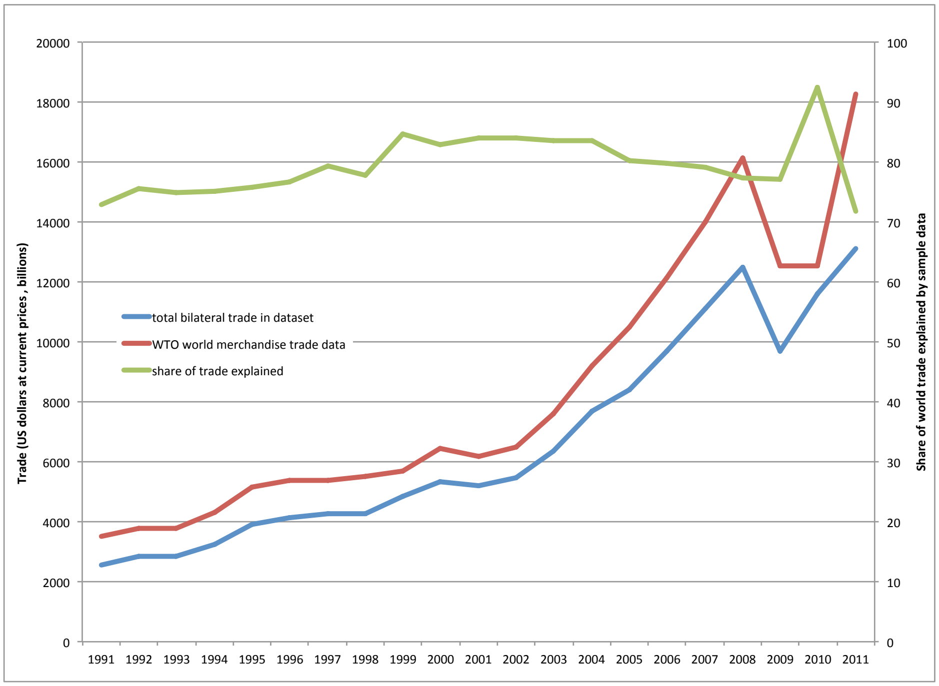
Although the trade data is available at a more granular level, the chosen level of sector aggregation reflects the optimal trade-off between several considerations. More sector disaggregation can

[11] Estimate elasticities separately for imports and exports is of course possible for a subset of countries, but this is not the objective of this study.

[12] Australia, Austria, Belgium, Brazil, Bulgaria, Canada, Chile, Chinese Taipei (Taiwan), Croatia, Cyprus, Czech Republic, Denmark, Estonia, Finland, France, Germany, Greece, Hungary, India, Indonesia, Ireland, Israel, Italy, Japan, Kazakhstan, South Korea, Latvia, Lithuania, Luxembourg, Malta, Mexico, Netherlands, New Zealand, Norway, People's Republic of China, Poland, Portugal, Romania, Russian Foundation, Singapore, Slovak Republic, Slovenia, South Africa, Spain, Sweden, Switzerland, Thailand, Turkey, United Kingdom, United States and Venezuela.

[13] Meat, dairy product, fish, cereals, vegetables & fruit, sugars, coffee&tea&cocoa, animal feed, other food, beverages, tobacco, hides&skins, oil seeds, crude rubber, cork&wood, pulp, textile, crude fertiliser, metal ore, crude animal material, coal&coke, petroleum, gas, electricity, animal fats, vegetable fats, processed fats, organic chemicals, inorganic chemicals, colour dye, pharmaceutical, essential oils, chemical fertilisers, primary plastics, non-primary plastics, insecticides, leather, rubber manufacturers, cork manufacturers, paper, textile articles, cement&lime&non-metallics, iron&steel, non-ferrous metals, metal manufacturers, power generating machinery, industrial machinery, metal working machinery, electrical machinery, road vehicles, non-road transport vehicles, prefabricated buildings, furniture, travel goods, apparel, footwear, scientific instruments, photo equipment & optical wear, other manufactured goods, non-gold coins, gold coins.

*Figure 4.1:* The share of world trade covered by the sample bilateral trade data in this study




Source: Author based on data from WTO Statistics Database (2012)

be advantageous particularly for heterogeneous sectors, enabling control for sub-sector specific characteristics. However, moving to the three or four-digit level substantially increases the number of zero values in the dependent variable and results in a very skewed distribution. At 66 sector level the share of zeros is below 25%, which is manageable for the estimation techniques used. It is important, nonetheless, to check whether the estimation results are sensitive to sector aggregation levels, and this is explored in Section 4.5.

With regards to the missing 25% of observations, it is possible that these represent a genuine zero i.e. no trade in sector $s$ for the country-pair $ij$ for a particular year. It is also possible that zeros represent missing data, or rounding down errors. Unfortunately this distinction cannot be made in COMTRADE data, which includes only positive values, and there is limited consensus in the literature about the share of zeros and missing values.[14] Zeros can be ignored so long as they are randomly distributed, however, if this is not the case, it will introduce a selection bias. There are reasons to believe that zero values of the dependent variable largely reflect genuine zeros (no trade for these products). The value is obtained in this dataset by taking the maximum of two mirroring statistics, which means both the importer and exporter individually recorded zero trade for that product. Also, the panel is relatively recent, compared to the previous studies which consider missing values from a much longer panel and the sample covers major trading

[14] Whereas Gleditsch (2002) states that around 80% are zeros (no trade) and the rest are missing values, on the other hand, Feenstra et al. (2005) finds around 60% to be zeros. Bernasconi (2009) develops a method which distinguishes between reporting and non-reporting countries, and assumes that missing values for reporting countries are zeros. Using this assumption, she finds that only 30% of missing values represent no-trade.

nations.

#### 4.3.3.2 Energy prices

The energy price series are obtained from the IEA Energy price and taxes information (IEA, 2012a). As described in IEA (2012b), the series for Energy end-use prices in US$ gives information on total prices (including taxes) for industrial users by fuel type (electricity, coal etc). The raw data is recorded in national currency per physical unit and submitted to the IEA Secretariat by national country administrations. This is transformed by the IEA to US$/unit using the exchange rates to the US$ from the OECD Main Economic Indicators. Annual data are twelve month averages. Industry prices and taxes are the average of amounts paid for the industrial and manufacturing sectors. This include transport costs to the industrial consumer, taxes that have to be paid by the industrial consumer, and reflect prices actually paid (net of rebates).

When converted to US$/unit, the differences between country end-use prices for industry and the world market price is attributed, not to the exchange rate differentials but other pricing parameters. These include transport costs, transformation costs, energy abundance, costs of non-tradable energy sources, market structure and national and local taxes.

Of the various fuel price series available, the electricity price is used to proxy for the industrial energy price. Electricity is by far the most complete price series available in the database (70% in the sample used here). The problem of missing data is particularly serious for coal and gas (around or less than 50% of data), and for countries important in trade (e.g. China's gas, Germany's coal). Oil price series are also very incomplete for all three types of oil – light fuel oil, low sulphur oil and high sulphur oil. Therefore using any of the other fuel price series to proxy for industrial energy price would severely restrict the sample size. Ideally if the prices were more complete, it would be possible to combine the fuel price data with sector energy use data to calculate a country-sector specific energy price index. For example, for sectors that rely heavily on coal, the coal price can be weighted more than for other fuel types. Given the current data availability, using electricity appears to offer a reasonable option for a number of reasons. In many key industrial regions such as the US and Europe, electricity costs represents the majority of energy expenditure for the majority of manufacturing sectors (Aldy & Pizer, 2011; Eurostat, 2012). On a global scale, in 2004, while the amounts of coal, gas, oil and electricity used in industry are similar, electricity accounted for the highest share of industry $CO_2$ emissions at 40%, followed by coal 25% (IEA, 2007b). Electricity price is also positively correlated with the available prices for oil, gas and coal. A simple panel fixed effect regression (by country) shows that the change in yearly oil, gas and coal prices have a positive and statistically significant effect on the change in electricity prices over time. Therefore it is argued that industrial electricity price provides a reasonable proxy for industrial energy price.

Several obstacles were encountered when using the total energy price expressed in US$/unit to construct the dataset. First of all, it reflects nominal prices rather than real (i.e. it does not

*Figure 4.2:* Top 20 total exports by bilateral trading route in 2008, in sample data (US$ Billions)

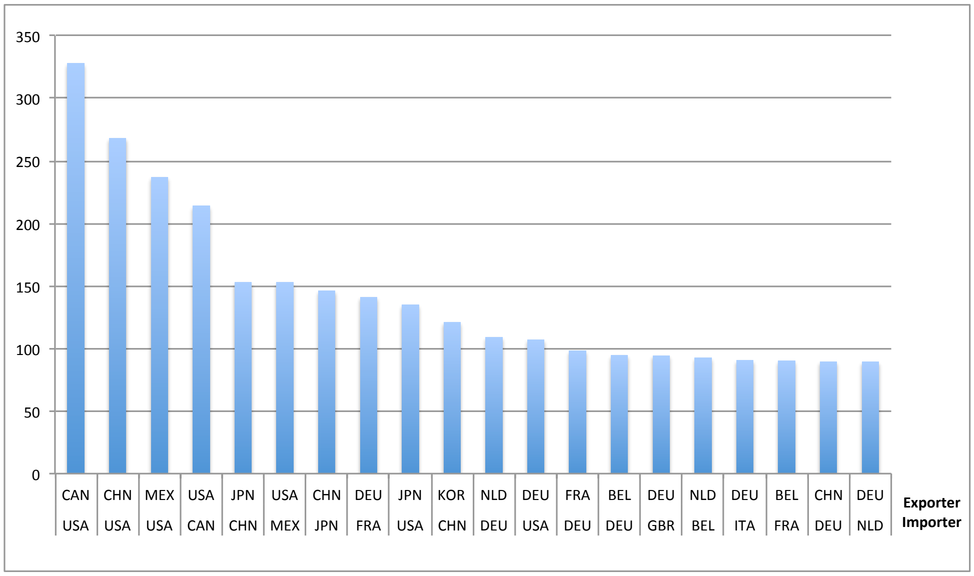
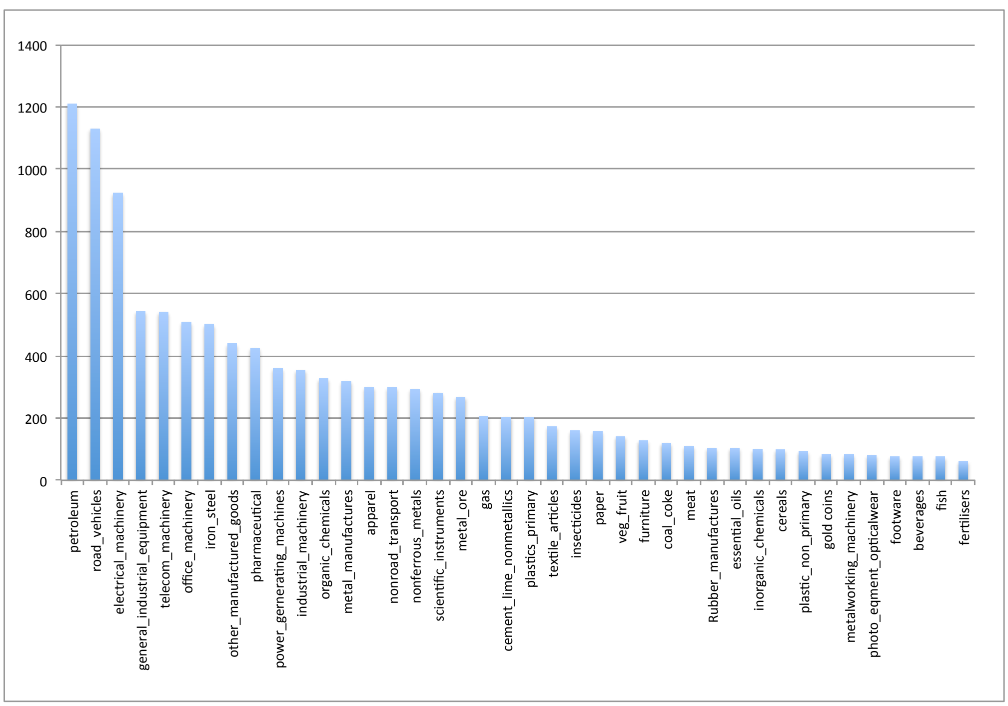
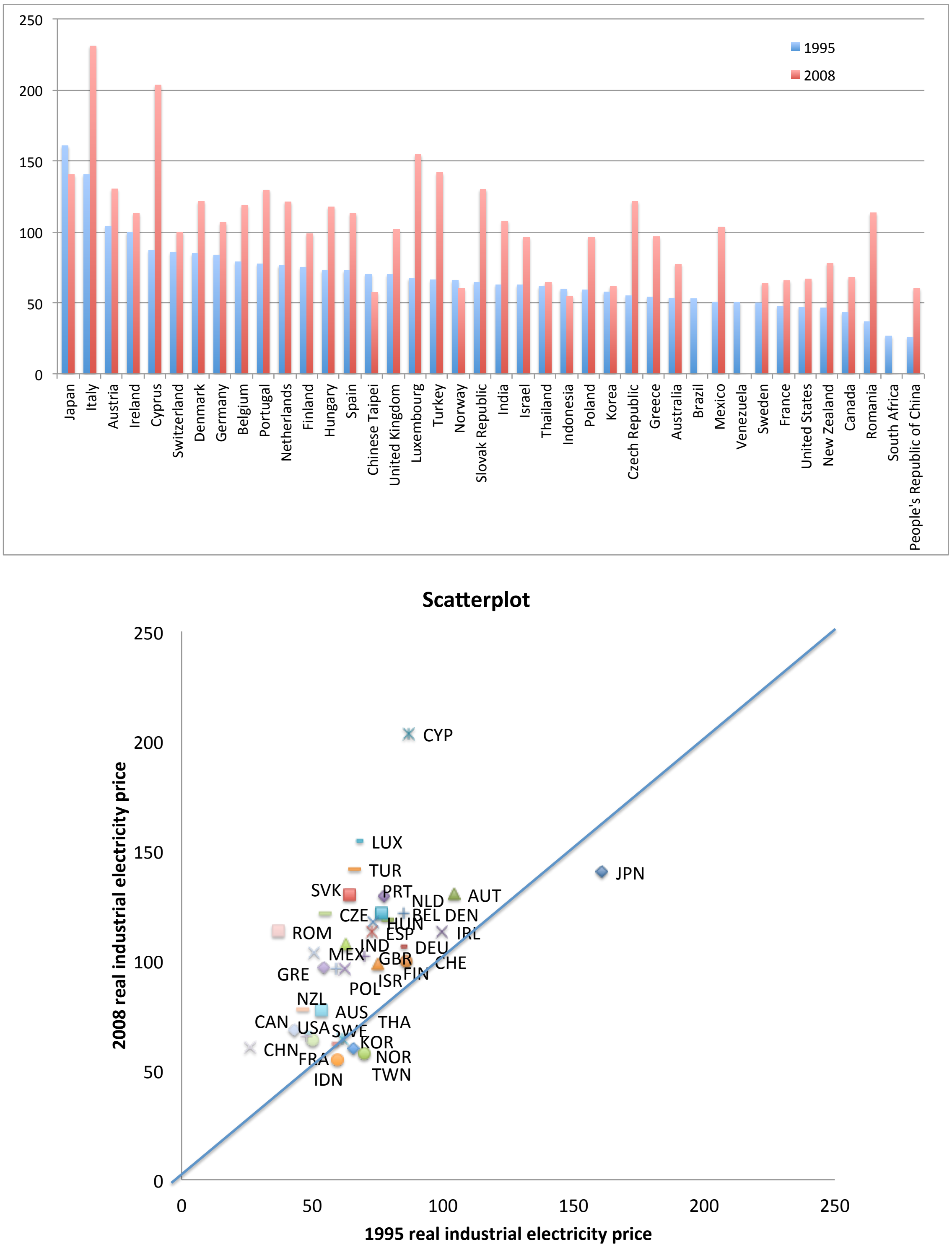



*Figure 4.3:* Top 30 total exports by sector in 2008, in sample data (US$ Billions)




Source: Author based on data obtained from COMTRADE

*Figure 4.4:* Cross-country differences in total electricity prices (including tax) for industry (in real prices, US$/MWh), years 1995 and 2008




Source: Author based on data obtained from IEA (2012a) Note: 1995-2008 is the time span that covers the highest number of countries included in the data set. For example China and India only go up to 2008.

account for inflation). This is problematic because different countries have different inflation rates (energy specific) during the period which this analysis covers.[15] To address this, energy price series are deflated using a fuel specific real price index available from the same IEA database, for IEA/OECD countries only. These are compiled by the national statistical services. In addition to producer price indices and consumer price indices, sub-indices for energy products are sometimes available. For non-OECD countries or for OECD countries where the fuel specific real price index is not available, I apply the World Bank Manufactures Unit Value (15) Index (World Bank, 2012). The MUV is a composite index of prices for manufactured exports from the fifteen major developed and emerging economies to low- and middle-income economies, valued in US$.

Having transformed the energy prices into real energy prices, there remains a problem of missing values (no electricity price data). Missing values are imputed using country specific indices: electricity real price index, total energy real price index, producer price index, electricity wholesale price index and electricity consumer price index. The average of the five imputed prices is taken. This reduces missing values for energy price from 31% to 20% of observations.

#### 4.3.3.3 Other data

GDP and population data are obtained from IMF World Economic Outlook (IMF, 2012). For Taiwan, GDP data was obtained from Taiwan national statistics (National Statistics of Republic of China (Taiwan), 2012). GDP data are available in US$ in current prices. These are converted into real prices using the GDP deflator index, which is also available from the IMF database. Because the latter has different base years for different countries, I adjust the deflator index, such that the 2005 =100 for all countries.

Additionally, standard gravity model variables obtained from the Gravity Dataset provided by CEPII are used for estimations using the pre-sample mean technique (CEPII, 2012). Data on labour and capital prices were obtained from United Nations Industrial Development Organization (2011) and extracted from ESDS, in order to conduct robustness checks on the influence of other bilateral time variant determinants of trade.

### 4.3.4 Descriptive statistics

Exports (in value terms) on an aggregate level rose steadily during the decade from 1991 ($3,515 billion), doubling to $6,494 billion in 2002 according to the WTO data as shown in Figure 4.1. It then increased at a faster rate until disrupted by the financial crisis and subsequent economic

[15]The IEA database also gives energy prices in terms of US$/unit using PPP. This measure addresses concerns that, because of wide fluctuations in exchange rates, international price comparisons using exchange rates do not capture relative purchasing power in each currency. Hence they convert the national currency prices using average purchasing power parities. Controlling for relative purchasing power is different from controlling for inflation and not immediately relevant for the purpose here.

*Table 4.1:* Descriptive statistics

| | $trade^s_{ijt}$ | $lnrade^s_{ijt}$ | $epgap_{ijt}$ | $gdpt_{ijt}$ | $gdpsim_{ijt}$ | $rfac_{ijt}$ | $dist_{ij}$ |
|---|---|---|---|---|---|---|---|
| *overall* | | | | | | | |
| mean | 49244.91 | 5.620122 | 0 | 6.791015 | -1.654019 | 1.186787 | 5915.786 |
| std. dev. | 476347.4 | 4.0764 | .5275219 | 1.219263 | 1.057392 | .950066 | 4695.793 |
| variance | 2.27e+11 | 16.61703 | .2782794 | 1.486601 | 1.118078 | .9026254 | 2.21e+07 |
| min | 0 | 0 | -2.229854 | 2.454198 | -7.178076 | .0000267 | 160.9283 |
| max | 8.60e+07 | 18.26955 | 2.229854 | 9.887039 | -.6931472 | 5.033718 | 19539.48 |
| median | 430.2025 | 6.066578 | -.0006437 | 6.804666 | -1.281719 | .9628906 | 5406.409 |
| N | 2512992 | 2512992 | 1724148 | 2041484 | 2041484 | 2041484 | 2512992 |
| *between* | | | | | | | |
| std. dev. | 424265.9 | 3.728867 | .5036073 | 1.192057 | 1.052191 | .9002405 | 4695.808 |
| min | 0 | 0 | -1.96002 | 2.930351 | -6.937026 | .0203294 | 160.9283 |
| max | 5.11e+07 | 17.7345 | 1.96002 | 9.698107 | -.6937601 | 4.62633 | 19539.48 |
| *within* | | | | | | | |
| std. dev. | 216578.9 | 1.647018 | .1943792 | .2854588 | .2186315 | .2970659 | 0 |
| min | -2.59e+07 | -8.897687 | -1.218422 | 5.948761 | -4.070698 | -.9104433 | 5915.786 |
| max | 5.21e+07 | 18.24153 | 1.2128 | 8.816855 | .6480542 \| | 4.041534 | 5915.786 |

recession in 2008 (dropping from \$16,140 billion to \$12,542 billion between 2008 and 2009), when world exports fell sharply. Since 2009, aggregate exports have been on an upward trend again, reaching \$18,255 billion in 2011.

At the country-pair-sector level, there is considerable variation in exports as shown in the first column of Table 4.1. With a mean of \$42 million, exports range from zero trade and up to \$86 billion (Canadian exports of petroleum to the US in 2011). As shown in Figures 4.2 and 4.3, more variation comes from the sector heterogeneity (in trade intensity and value) than from the bilateral-pair heterogeneity.

In terms of the industrial electricity prices, there is considerable variation across countries as shown in Figure 4.4. In 1995, prices were below 50US\$/MWh in China, South Africa, Romania, Canada, New Zealand and the USA (no tax included), but three times higher in Italy and Japan. A general increase in prices over the next decade can be observed by comparing the two graphs, as well as a widening of the range of prices. In 2008, real industrial electricity prices remained low (below 60US\$/MWh) for Kazakhstan, Russia, Indonesia, Taiwan, Norway, China, Korea and Sweden, whereas users in Italy and Cyprus faced prices exceeding 200US\$/MWh. Notice that in some countries, real electricity prices for industry went down over the period, such as in Japan and in Norway.

The descriptive statistics in Table 4.1 shows that when expressed as the difference of the logs (i.e. the log ratio) the electricity price gap's mean is zero because of the symmetrical nature of the data – the energy price gap between US and UK is expressed as a negative value when considering UK exports to the US, and as a positive value of the same magnitude when considering US exports to the UK. Notably, the within-group standard deviation of the electricity price ratio is high. That is to say that the historical fluctuations in the energy price gap have been considerable, driven by underlying factors such as energy taxes, energy supply and demand.

## 4.4 Results

### 4.4.1 All sectors

Implementing the estimation techniques described in Section 4.3.2, Table 4.2 reports the parameter estimates from five different models employed, considering all 66 sectors. The dependent variable is the natural logarithm of trade for the OLS and the Arellano-Bond GMM estimations, and trade in levels for the Poisson maximum likelihood, negative binomial and the zero inflated negative binomial estimations. Unobserved heterogeneity is controlled for using fixed effects for country-pair-sector for models (1) to (3) and using the pre-sample mean scaling estimator for models (4) to (5). For the latter group of models, the data for the years 1991 to 1995 is used for the pre-sample mean estimator. All models assess variation in trade and energy prices during the years 1996 to 2011.

The main result from table 4.2 is that the electricity price gap is statistically significant and has a positive effect on bilateral trade. In other words, trade tends to develop between countries with different energy prices. This result holds across the five model estimations. More specifically, a one percent increase in the electricity price ratio (when the importer has a higher energy price) is associated with a change in bilateral import levels by 0.05 to 0.17%.

In addition, across the models, the coefficients for the lagged dependent variables always exhibit a parameter estimate which is significantly different from zero. This suggests there is strong 'think-back' or 'stickiness' in the level of sectoral trade between two countries, and highlights the importance of using a dynamic panel estimator as. This finding is consistent with the recent literature (Olivero & Yotov, 2010, 2012).

I now discuss the results in greater detail. The first column refers to the results based on the baseline OLS model with fixed effects at the country-pair-sector level. I use robust standard errors clustered at the country pair level, given the data (electricity price gap does not vary by sector). Two lags of the dependent variable are included, to allow comparison with other estimations in the table. In addition to year dummies, linear time trends are included for the importer and exporter. Although not as flexible as importer-time and exporter-time dummies, time trends offer a way to capture time variant characteristics of the trading countries, while keeping the estimations computationally manageable.

The key explanatory variable $epgap_{ijt}$ has a positive and significant coefficient estimate of 0.08.[16] The control variables are significant and the signs are consistent with expectation. Increase

[16] For the purpose of comparison, the coefficient on the epgap variable when running the OLS estimation without

*Table 4.2:* Results for all sectors

| | (1) | (2) | (3) | (4) | (5) |
|---|---|---|---|---|---|
| | OLS | Arellano-Bond GMM | PML, fe | Negative binomial | Zero inflated NB |
| Electricity price gap | 0.08*** | 0.11*** | 0.05*** | 0.17*** | 0.15*** |
| | (0.01) | (0.01) | (0.01) | (0.06) | (0.05) |
| **Control variables** | | | | | |
| Relative fact. endow. | -0.05*** | -0.01 | -0.03** | -0.04*** | -0.05*** |
| | (0.01) | (0.00) | (0.02) | (0.02) | (0.02) |
| GDP total | 0.32*** | 0.62*** | 0.53*** | 0.36*** | 0.27*** |
| | (0.02) | (0.01) | (0.02) | (0.08) | (0.08) |
| GDP similarity | 0.20*** | 0.58*** | 0.29*** | 0.27*** | 0.23*** |
| | (0.02) | (0.01) | (0.03) | (0.04) | (0.04) |
| **Gravity variables** | | | | | |
| distance | | | | -0.00*** | -0.00*** |
| | | | | (0.00) | (0.00) |
| contiguity | | | | 0.60*** | 0.60*** |
| | | | | (0.06) | (0.06) |
| common offical language | | | | 0.21*** | 0.18*** |
| | | | | (0.06) | (0.04) |
| common currency | | | | 0.21*** | 0.19*** |
| | | | | (0.06) | (0.05) |
| **Lagged dep. vars.** | | | | | |
| trade_ij(t-1) | 0.38*** | 0.16*** | 0.32*** | 0.37*** | 0.35*** |
| | (0.00) | (0.01) | (0.02) | (0.01) | (0.01) |
| trade_ij(t-2) | 0.08*** | 0.00** | 0.01*** | 0.21*** | 0.20*** |
| | (0.00) | (0.00) | 0.01** | (0.01) | (0.01) |
| trade_ij(t-3) | | 0.00*** | (0.00) | | |
| | | (0.00) | | | |
| **Pre-sample info.** | | | | | |
| trade presample mean | | | | 0.02 | 0.00 |
| | | | | (0.02) | (0.02) |
| zero presamp mean dum | | | | -0.21 | -0.26 |
| | | | | (0.31) | (0.38) |
| **Inflation equation** | | | | | |
| trade presample mean | | | | | -0.49*** |
| zero presamp mean dum | | | | | -1.63*** |
| distance | | | | | -0.00* |
| contiguity | | | | | -0.65*** |
| common offical language | | | | | 0.13 |
| common currency | | | | | -0.08** |
| | | | | | -0.07 |
| Countrypair sector effects | Yes | Yes | Yes | pre-samp | pre-samp |
| Country time trend | Yes | No | Yes | Yes | Yes |
| Year effects | Yes | Yes | Yes | Yes | Yes |
| Observation number | 1706402 | 1745418 | 1706402 | 1706402 | 1706402 |
| Endogeneity test (p-value) | | <1% | | | |
| Identification test (p-value) | | <1% | | | |

Robust Standard errors in parentheses: * p<.05, ** p<.01, ***p<0.001

in total economic mass increases bilateral exports, and the similarity in GDP also tends to increase trade. The positive coefficient on the latter suggests existence of intra-industry trade. The coefficient on $rfac_{ijt}$ is negative, suggesting that the model is consistent with the Linder hypothesis – that bilateral trade is negatively related to differences in relative factor endowments.

The second column reports estimates based on the differenced Arellano-Bond GMM estimator (one-step estimation with cluster-robust standard errors), which gives a positive and statistically significant coefficient estimate of 0.11. Here, the first-differenced lagged dependent variable is instrumented by its own lags in levels.[17] Residual based tests were also conducted to test whether the explanatory variables are endogenous. The null hypothesis of endogeneity was rejected at the 0.1% significance level, therefore, this suggests that the assumption that these variables are exogenous holds. The estimation results again find that the energy price gap variable is positive and significant. In this differenced specification, it is not possible to account for importer and exporter linear time trends. The Arellano-Bond estimator results clearly show that there is considerable persistence in trade over time, as all six lagged explanatory variables are found to be statistically significant. The optimal lag structure (six lags) has been selected on the basis of the autocorrelation test. The coefficient on the first lag is large, and the subsequent ones are smaller.[18] When six lags are specified, the Arellano-Bond test statistic of second order serial correlation (AR(2)) showed value of $z = -0.36$, with an associated $p$-value of 0.721. Hence the null hypothesis of the absence of serial correlation can no longer be rejected.[19] The Sargan and Hansen tests for over-identification of restrictions is also conducted to test the null hypothesis that the instrumental variables are uncorrelated with the residuals. The hypothesis is not rejected, thus the possibility of orthogonality between the instruments and the residuals as required by the GMM cannot be rejected. Additionally, the weak instruments test (first-stage F-statistic and apply the rule of thumb developed by Staiger & Stock (1997)) indicated the risk is low. Although this model addresses the issue of dynamics, it does not deal well with the issue of zeros in the dependent variable.

The Poisson maximum likelihood model estimation is shown in column (3). The coefficient on the key explanatory variable $epgap_{ijt}$ is positive at 0.05 and statistically significant at conventional levels.[20] The coefficients on the control variables are in line with other models.

Turning now to the negative binomial model and the zero-inflated negative binomial model results in column (4) and (5), the coefficient on the *epgap* is positive and significant and the

the lagged dependent variables is 0.10***(0.01). That the inclusion of the lagged dependent variables leads to a downward bias is consistent with theory, which predicts that the bias is negative if the coefficient is positive and smaller than one.

[17] As a comparison, a System GMM estimator was also estimated. The results were similar.

[18] Only three lags are shown in the table in the interest of space. Additionally, it was tested whether the electricity price gap is endogenous, using residual based tests. Lagged values of the price gap were used as instruments. The null hypotheses of exogeneity at the 10% significance level was not rejected. Therefore, it suggests that there is no need to account for endogeneity bias for this variable.

[19] The test for AR(1) process in first differences rejects the null hypothesis, as expected.

[20] Again for the purpose of comparison, the coefficient on the epgap variable when running this estimation without the lagged dependent variables is 0.06***(0.00).

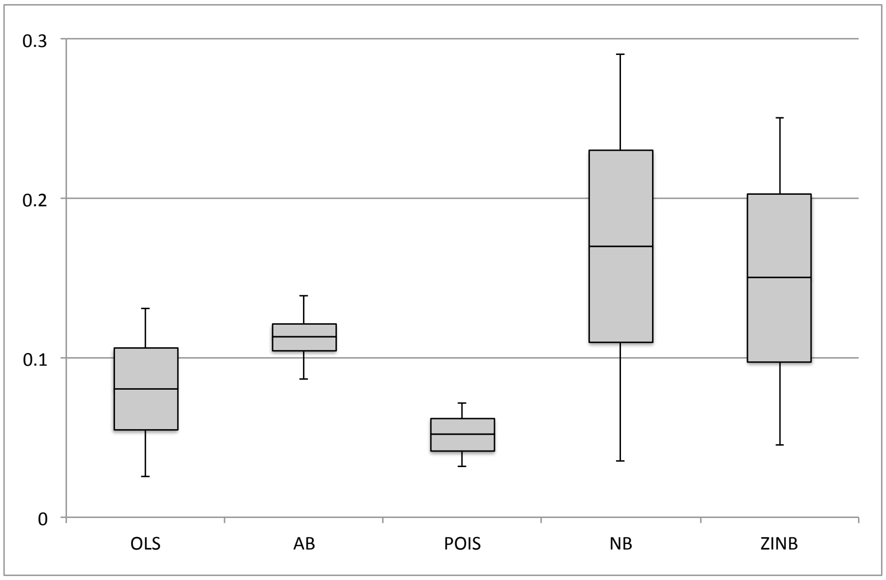
magnitude of the effect is similar, with relatively high values of 0.17 and 0.15 respectively. For both columns, the pre-sample mean scalar is positive and but not significant. This is to be expected for reasons discussed in Section 4.3.2.3 (trade tends to increase over time), and suggests that the PML model with true fixed effects performs better. However, the control variables have expected signs, and may be controlling well for the time-invariant country pair characteristics. Distance is negative and significant effect (but with a zero coefficient). Overall economic size, similarity and contiguity has a positive and statistically significant effect as is expected. Two lags of the dependent variable are significant.

Comparing the two columns' results, the zero-inflated negative binomial model (column 5) includes a probit equation for zero trade. As described in Section 4.3.2.2, a predicted probability of generating a zero count is estimated using a vector of gravity variables and the pre-sample mean estimator. As shown in the table, all explanatory variables included in the probit equation are statistically significant. The rationale behind the zero inflation model is that the zeros in the dependent variable may be non-random. Companies may first choose whether or not to trade, and subsequently decide on how much to trade. The first decision, may be characterised by the existence of a fixed cost of trade which should be higher than the potential benefit of trade, if there is zero trade observed. As only non-zero trade flows are observed for a subset of country-pair-sectors who decided to trade in the first place, this suggests that for this subset, their average potential gains from trade are higher than the average. Estimating the coefficient for the effect of electricity price gap on exports based only on observed trade flows then leads to an overestimate (assuming that there is a constant effect of a price gap on trade flows). Comparing columns (4) and (5), the coefficient on *epgap* is indeed lower for the zero-inflated negative binomial model, which includes an adjustment of the conditional mean, by giving more weight to zero observations. This suggests that there is evidence of a selection bias that can to some extent be addressed by the inclusion of a zero-inflation equation.

Figure 4.5 shows the *epgap* coefficient estimate (middle bar), one standard deviation in each direction (the box) and the top and bottom bars indicate the upper and lower limits of the 95% confidence intervals for each of the seven models. The large standard deviation on *epgap* under the negative binomial and zero inflated negative binomial models may be attributable to the poor quality of the pre-sample mean estimator, as explained above.

Comparing the 95% confidence intervals for the *epgap* coefficient estimates across the five models in Figure 4.5 enables some conclusions to be drawn. The results collectively support the hypothesis that energy price asymmetry plays a role in bilateral trade flows. However, it also shows that the results are sensitive to model specification and the magnitude of the effect is estimated with limited precision. The estimates by PML and Arellano-Bond have a relatively narrow 95% confidence interval, but their intervals do not overlap. The PML is preferred over the Arellano-Bond specification from the perspective that it is able to control for importer and exporter specific time variant characteristics, by using linear time trends. This is shown to be important in Section 4.5. However, the PML represents a lower-end estimate. Across all five

*Figure 4.5:* EPGAP coefficient estimates 95% confidence intervals across 5 models




Note: The *epgap* coefficient estimate across the seven models are represented by the middle bar of the box. The boxes represent one standard deviation in each direction and the top and bottom bars indicate the upper and lower limits of the 95% confidence intervals.

models, there is some overlap in the estimates' 95% confidence intervals are in the area of 0.05 and 0.15, suggesting the true effect is more likely to lie in this range.

### 4.4.2 Sector level

The importance of sector heterogeneity in the trade impacts of carbon pricing has been explored in partial equilibrium modelling for Europe's heavy industry, as well as in econometric analysis for the EU production sectors (Section 4.2.1). This section examines whether similar evidence can be found for a wider geographical scope.

Unfortunately sector level industrial energy price data is available for few of the 51 countries covered in this study. In the absence of this data, one way to investigate this issue is to assume that the average sector energy intensity in the US (or other countries for which data is available) is representative, and apply to all countries. Another strategy involves interacting sector dummies with the energy price variable $epgap_{ijt}$. This approach is adopted in this section. On one hand, this avoids making the assumption that sector energy intensities are uniform across countries. On the other, it assumes that the same energy price level is faced by all sectors in one country. There is by definition measurement error, because in reality sectors may face different prices within a country e.g. they may have long-term contracts with energy suppliers, self-generation or other energy subsidies. The assumption of a uniform energy price level across all sectors in one country becomes more problematic, as the level of sector disaggregation increases.

As a start, the 66 sectors are aggregated into 17 sector groups – the categorisation is described in

Table 4.5 in the Appendix. The sector-group dummies $SGROUP$ are then interacted with the country-pair specific *epgap* variable such that in the non-linear form:

$$\begin{aligned} trade_{ijst} \quad = \quad & exp(\lambda_p \sum_{p=1}^{n} trade_{ijs(t-p)} + \beta_1 epgap_{ijt} + \beta_2 gdpt_{ijt} + \beta_3 gdpsim_{ijt} + \beta_4 rfac_{ijt} \\ & +\psi SGROUP + \sum_{k=1}^{n=g} \gamma_k (SGROUP * epgap_{ijt}) + \ X_{ijt} + \ T_t) + \varepsilon_{ijst} \qquad (4.7) \end{aligned}$$

The Poisson maximum likelihood estimator is used to estimate the above, as the preferred model. Post estimation tests are conducted to evaluate whether the slope of the coefficients differ from one another.[21] The p-values for the conducted test for each sector group's interaction were checked, and it was found that the p-value for the test is below 0.1 (hence where the hypothesis is not rejected at the 10% level) for over 30%. Hence it indicates that the effect of the energy price gap is heterogeneous across different sector groups.

Because the PML is a non-linear model, the interaction terms are difficult to interpret. Thus I run this estimator for the 17 sector groups separately. The results reported in Table 4.3 give support to the notion that impacts of the energy price gap on trade are heterogeneous across sectors. The coefficient is positive and statistically significant at conventional levels for half the sectors – mining, fuel, iron & steel, pulp & paper, chemicals, semi-manufacturing, transport equipment, textile & clothing and finally, the "other manufacturing" sector group. Many of these are energy intensive sectors, but some non-energy intensive industries also experienced larger impacts (transport equipment and textiles). It is also interesting to note that the coefficient is not statistically different from zero for several energy-sectors such as fertilisers, non-ferrous metals (aluminium) and cement & lime. This suggests an interesting phenomenon: trade in the latter energy-intensive industries may be more resilient to higher energy prices than less energy-intensive industries that compete with large volumes of net imports such as textiles.

An attempt was also made to examine the issue of heterogeneity further, by repeating the above at the 66 sector level. At this granular level, observation numbers are significantly reduced to between 15,000 and 30,000 observations. Only for a very small number of shares, an effect statistically different from zero was found for the energy price gap variable, although the control variables and lagged dependent variables had the expected signs. Table 4.4 reports the sectors for which the coefficient on the energy price gap variable is statistically significant at least at the 10% level.

It is important to bear in mind that the magnitude of coefficients are imprecisely estimated – they are sensitive to model specification. The further disaggregation of sector-groups also magnifies both problems of measurement error (the *epgap* is averaged over the country pair

[21] Whether the levels differ cannot be estimated because the model examines within group variation.

*Table 4.3:* Results for estimation by 17 sector groups using Poisson maximum likelihood

| | Food | Raw materials | Mining | Fertilisers | Fuel | Non-ferrous metals | Iron & Steel | Cement, lime and glass | Pulp and paper | Chemicals | Pharma-ceutical | Plastic | Semi-manufac-turing | Machinery | Transport equipment | Textile and clothing | Other Manufac-turing |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Electricity price gap** | 0.02 | 0.02 | 0.10* | 0.03 | 0.18*** | 0.08 | 0.08** | 0.01 | 0.05** | 0.05* | 0.05 | 0.01 | 0.04* | -0.02 | 0.13*** | 0.05** | 0.05* |
| | (0.02) | (0.03) | (0.05) | (0.03) | (0.04) | (0.05) | (0.04) | (0.05) | (0.02) | (0.03) | (0.04) | (0.02) | (0.02) | (0.02) | (0.04) | (0.02 | (0.03) |
| ***Control variables*** | | | | | | | | | | | | | | | | | |
| **Relative fact. endow.** | 0.06 | -0.03 | 0.06 | -0.03 | 0.01 | 0.09 | 0.02 | 0.20** | 0.01 | -0.03 | -0.00 | -0.01 | 0.04 | 0.04 | 0.07 | 0.08*** | 0.03 |
| | (0.04) | (0.04) | (0.06) | (0.05) | (0.06) | (0.07) | (0.05) | (0.10) | (0.04) | (0.04) | (0.07) | (0.03) | (0.03) | (0.02) | (0.06) | (0.03) | (0.05) |
| **GDP total** | 0.56*** | 0.45*** | 0.32*** | 0.43*** | 0.21** | 0.25*** | 0.30*** | 0.41*** | 0.31*** | 0.55*** | 0.54*** | 0.41*** | 0.54*** | 0.56*** | 0.79*** | 0.34*** | 0.77*** |
| | (0.05) | (0.09) | (0.10) | (0.07) | (0.09) | (0.07) | (0.09) | (0.09) | (0.07) | (0.06) | (0.12) | (0.06) | (0.08) | (0.04) | (0.08) | (0.07) | (0.07) |
| **GDP similarity** | 0.37*** | 0.21*** | -0.01 | 0.27*** | 0.23*** | 0.26** | 0.25** | 0.08 | 0.25*** | 0.24*** | 0.25** | 0.32*** | 0.30*** | 0.33*** | 0.59*** | 0.11** | 0.21** |
| | (0.05) | (0.07) | (0.10) | (0.11) | (0.08) | (0.11) | (0.10) | (0.10) | (0.09) | (0.06) | (0.11) | (0.05) | (0.05) | (0.04) | (0.09) | (0.05) | (0.08) |
| ***Lagged dep. vars.*** | | | | | | | | | | | | | | | | | |
| **Trade_ij(t-1)** | 0.32*** | 0.37*** | 0.31*** | 0.21*** | 0.25*** | 0.23*** | 0.18*** | 0.12*** | 0.20*** | 0.19*** | 0.24*** | 0.14*** | 0.35*** | 0.36*** | 0.17*** | 0.27*** | 0.22*** |
| | (0.02) | (0.02) | (0.03) | (0.02) | (0.03) | (0.03) | (0.02) | (0.01) | (0.03) | (0.02) | (0.03) | (0.01) | (0.05) | (0.03) | (0.02) | (0.04) | (0.03) |
| **Trade_ij(t-2)** | 0.01** | 0.01** | -0.01 | 0.00 | 0.02** | -0.02*** | -0.01*** | -0.01*** | 0.00* | 0.01** | 0.0271** | -0.01*** | 0.01*** | 0.00 | -0.00 | 0.01*** | 0.00 |
| | (0.01) | (0.01) | (0.01) | (0.00) | (0.01) | (0.01) | (0.003) | (0.00) | (0.00) | (0.00) | (0.01) | (0.00) | (0.00) | (0.00) | (0.00) | (0.00) | (0.00) |
| **Country-pair sect. FE** | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes |
| **Country trends** | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes |
| **Year FE** | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes | Yes |
| **Observations** | 389493 | 194623 | 26446 | 79365 | 73055 | 27987 | 28108 | 28242 | 51341 | 111306 | 27950 | 55543 | 82964 | 225404 | 55959 | 139697 | 84391 |

*** p<.05, ** p<.01, ***p<0.001. Robust standard errors clustered at country-pair in parentheses.**

*Table 4.4:* Sectors with *epgap* coefficient statistically different from zero, estimation by 66 sectors using PML

| **Sector** | **Energy price gap** | |
|---|---|---|
| **insecticides** | **0.05*** | **(0.03)** |
| **metal manufacture** | **0.04**** | **(0.02)** |
| **metal ore** | **0.08*** | **(0.05)** |
| **other manufacturing** | **0.05*** | **(0.03)** |
| **paper** | **0.04***** | **(0.02)** |
| **petroleum** | **0.14***** | **(0.05)** |
| **pharaceutical** | **0.07*** | **(0.04)** |
| **road vehicles** | **0.08**** | **(0.03)** |
| **sugars** | **0.11**** | **(0.04)** |
| **telecom machinery** | **0.11**** | **(0.05)** |

* p<.05, ** p<.01, ***p<0.001. Robust standard errors clustered at country-pair in parentheses.

and not differentiated by sector) and reduced sample size. Nonetheless, the analysis in this subsection provides some evidence that the magnitude of this effect varies across sectors. As should be expected, it suggests that impacts are more pronounced for relatively energy intensive sectors.

### 4.4.3 How does an energy price gap relate to a carbon price gap?

Interpreting a one percent change in the energy price gap variable is not intuitive. To aid interpretation of the results, I use an example of the UK and South Korea in 2007. To calculate this, it is necessary to take account of the different carbon intensity of electricity in each country. In 2007, South Korea had an average carbon intensity of 0.4558 $tCO_2$/MWh whereas the UK's intensity was slightly higher at an average of 0.4882 $tCO_2$/MWh. The real electricity price in the UK in 2007 was 101.40US$/MWh and 65.44US$/MWh in South Korea. Recall that the energy price gap is defined as $epgap_{ijt} = ln(EP_{it}) - ln(EP_{jt})$ hence $epgap_{ijt} = ln\left(\frac{EP_{it}}{EP_{jt}}\right)$. The energy price gap without carbon pricing is thus 0.44.[22] Implementing a \$30/$tCO_2$ in the UK and \$5/$tCO_2$ in South Korea would instead imply an electricity price gap of 0.54.[23] Thus, a carbon price asymmetry of \$25/$tCO_2$ between the UK and South Korea roughly equates to a 10% change in the *epgap*.

This example is illustrative, however. A crude method is used to translate the impact of a carbon price on industrial electricity prices, by simply taking the respective country's average electricity carbon intensity and multiplying by the carbon price. In reality, this is a function of multiple factors including the carbon intensity of the marginal electricity technology in the system, as

[22] $ln(101.4) - ln(65.4) = 0.43855083$
[23] $ln(101.4 + (30 * 0.4882)) - ln(65.4 + (5 * 0.4558)) = .53921072$

well as the short-term demand and supply elasticities in response to the introduction of the carbon price.[24] If anything, this simplifying assumption is likely to overstate the impact on electricity price because the electricity prices faced by industries consists of many components including wholesale electricity prices and taxes.

It is also worth noting that the variance of the *epgap* observed historically is large – a 10% change in the electricity price gap variable discussed here is small relative to one standard deviation in this variable, as shown in the descriptive statistics (Table 4.1). That is to say that the historical fluctuations in the energy price gap has been considerable, due not to climate policies, but to underlying factors (energy taxes, energy supply and demand, etc.). Therefore, from the perspective of ordinary fluctuations in the electricity price gap, the 10% change that has been discussed here represents around a half of one standard deviation (within variation), hence, nothing out of the ordinary. It would be reasonable to expect that even when exposed to international competition, firms will be able to absorb carbon price differentials (of the kind of magnitudes discussed here), at least in the short run. Hence the additional asymmetries that may be induced by carbon price differentials will be smaller in comparison to underlying variations.

## 4.5 Robustness checks

### Aggregation bias

The analysis in Section 4.4.2 demonstrated that the results in general hold when the panel is aggregated at different sector resolution. This suggests the possibility that results for these estimations can be sensitive to the level of sector aggregation in the data. To test sensitivity, the panel was aggregated at different sector resolution – at the 266 sectors (3 digit SITC sectors), 10 sectors (3 digit SITC sectors), and up to the aggregated country level. It was not possible to fit any model to the data at the original 900+ product level, due to the large share of zeros and the distribution of the dependent variable observations with positive values, which becomes extremely skewed towards the left (many small observations). Because the energy price gap is at country-pair level, it is natural to run the regressions at the country-pair level. If the results do not hold, it is possible that statistically significant results obtained in the sector-level analysis may be driven by artificially inflated observation numbers. The country level estimation, suffers from the problem of a much reduced sample size. However, the energy price gap coefficient is statistically significant from zero across five mohels hence provides support to the main empirical results presented in this chapter. Similarly the results of the regressions at the 266 and 10 sector level gave support – the coefficients on the $epgap_{ijt}$ were statistically significant and similar in magnitude, particularly for the PML model.

[24] The impact of carbon pricing on electricity price is usually calculated using macro economic models and CGE models. Given the absence of modelling analysis results, however, the alternative simpler method was applied.

### Sensitivity to fixed effects specification

Baltagi et al. (2003) experiment with eight different fixed effects models and show how the specification of the fixed effects impacts estimations of the gravity model. He argues for the importance of controlling for a full interaction of the importer, exporter and time dimensions to analyse bilateral trade flows – that is, including importer specific time effects, exporter specific time effects, as well as importer-exporter fixed effects. This method ensures that all characteristics that could possibly be correlated with annual bilateral exports between the two countries, other than those that are time-variant and bilateral specific, have been accounted for, hence acts as an additional check on potential omitted variable bias. Due to issues with the practical implementation of the full interaction of importer, exporter and time dummies in this analysis (the large number of countries implies large number of dummy variables), linear time trends (importer and exporter specific) were included instead in the regressions. Two tests are possible, to assess the sensitivity of the results to the inclusion of the time trends.

First I compare the results to a specification excluding the time trends. In this case, the estimation included only the time invariant country-pair-sector fixed effects, and a time dummy. Across all five models, the coefficient on the energy price gap was statistically significant, but smaller than the estimations with the linear time trends (e.g. the OLS coefficient is reduced from 0.08*** (0.01) to 0.04**(0.02) and the PML is reduced from 0.05** (0.01) to 0.04*** (0.01)). This suggest that besides time invariant factors, time-variant exporter and importer specific factors such as cultural and institutional change and business cycles also matter. Moreover, it suggests that the variation that is taken out by the importer-time and exporter-time linear trends represent factors which have the net effect of reducing bilateral exports. The results indirectly support the argument made that controlling for all interaction effects in gravity models is important and that ignoring them can lead to biased estimates and incorrect inferences (Baltagi et al., 2003; Kellenberg & Levinson, 2011). In this analysis, all models except the Arellano-Bond difference GMM took included linear time trends for the importer and exporter.

In addition, including importer-time and exporter-time specific fixed effects is theoretically possible, and computationally helped by the Pseudo Poisson Maximum Likelihood (PPML) estimator (Santos Silva & Tenreyro, 2006). This would represent the most demanding specification, controlling for all characteristics that could possibly be correlated with bilateral trade flows between two countries, other than those that are time-variant and bilateral specific. Importantly, this should control for any other unobserved country specific influences on their energy prices. However, comparing the results from this was not possible as the model did not converge for the dataset used.

### Other idiosyncratic factors

It is reasonable to believe that the estimated effect of the energy price gap is in fact capturing variations in other time-variant factors, the labour and capital price differentials. The FDI and industry location literature, as well as the gravity model of trade literature have examined the role of labour price and capital price differentials in international trade patterns, although the evidence is mixed (Baltagi et al., 2007). It is possible that such variations (which are also country-pair-time specific) are inadequately addressed by the combination of the fixed effects. Unfortunately the data on capital prices (country level), and labour prices (country sector level) had many missing values. Due to the gaps in the data, over 79% of observations were dropped. Nonetheless using the available data, after converting to real price and constructing gap variables for each, these were included directly into the estimations to test that there is indeed no omitted variable biased here. OLS, PML and Arellano-Bond GMM equations were used (both with and without the energy price gap). For both variables, across the three models, the coefficient was not statistically different from zero, and the coefficients on the energy price gap were similar to the reference results. The signs of the control variables were as expected.

### Specification of the energy price gap

To test the possibility that the estimated effects are sensitive to the specification of the energy price gap variable. Estimations with several alternative specifications of the energy price gap were carried out. The estimations were repeated using the absolute difference, only the positive values then only the negative values (to check asymmetry), the absolute difference over the sum of the energy prices (following Kellenberg & Levinson (2011)), and also by constructing the reverse 'similarity' variable (following Costantini & Mazzanti (2012)). The hypothesis that a larger gap is associated with more bilateral exports stood up to these tests in all models except the OLS. A negative relationship was found between the similarity in energy prices between two trading partners and their bilateral trade.

### Dynamic lag selection

Dynamic estimations raise the question as to how many lags of the dependent variable to include. There is a trade-off involved with lag length selection – using too few lags can decrease accuracy because information is lost, but adding lags can also increase estimation uncertainty. Sensitivity testing to the inclusion of only two lags in the non-linear models was conducted by including different numbers of lags of the dependent variable. The results demonstrate that the marginal value of including additional lags is such that two lags represents a good choice.

### Other concerns

To ensure that results are not driven by observations with extreme values, the models were also run using a variation of transformations of the dependent variable. Observations with large values of the dependent variable (top 1%) are discarded under 'trimming' and capped under Winsoring. The outlier issue is of concern particularly with least square models, and also because of the known quality issues with the COMTRADE trade data described in Section 4.3.3. It was found that trimming and Winsoring of the dependent variable does not change the results significantly.

Particularly for the linear models (OLS and Arellano-Bond), there is concern that the issue of zeros in the dependent variable is inadequately addressed, by simply adding one before taking the natural log. Estimations using an unbalanced panel is conducted as a test, excluding the observations to those strictly with positive values. When excluding the zero observations, it was not possible to specify the first differenced Arellano-Bond estimator with a lag structure such that the conditions of absence of serial correlation and over-riding identification are met. This may be explained by the fact that the first-difference transform magnifies gaps in unbalanced panels, and this issue may be serious when 25% of the observations have a value of zero. For all other models, the size of the coefficients on the energy price gap is usually smaller than when using a balanced panel, as one would expect, but overall the results are similar. This suggests that the issue of zeros has been adequately addressed in this analysis.

Overall, I probed the robustness of the estimates to determine the sensitivity of the results but I find little evidence contradicting the basic conclusions of this paper. The results presented in Table 4.2 remain robust. The industrial energy price gap effect on bilateral trade is positive and significant at the 1 percent level.

## 4.6 Conclusions

As countries strengthen carbon pricing policies at different speeds, there is considerable interest around the potential trade impacts particularly for the energy intensive trade-exposed sectors. This paper measured the response of bilateral trade to differences in industrial energy prices, which historically vary across countries. Using a 16 year panel dataset that includes 51 countries and 66 sectors (covering 80% of global merchandise trade) and a gravity model framework, I estimate the effect of energy price asymmetry on trade. The coverage and detailed disaggregation of the data used also goes beyond previous work.

I find evidence that trade tends to develop between countries with different energy prices. In terms of magnitudes, the estimated effect implies that a 10% increase in the electricity price ratio between the importer and exporter (which by example can be represented by a carbon price gap of \$25/t$CO_2$ in between the UK and South Korea) increases bilateral trade levels by around 0.5% to 1.5%. However, this is an average effect across 66 sectors and when the sectors are examined separately, the effect of the energy price gap is statistically different from zero for some sectors but not others. This suggests that the link between trade and energy price exists for only a few sectors of the economy.

The hypothesis was tested using a number of different models, both linear and non-linear. While I argued that theoretically, the Poisson Maximum Likelihood model is the preferred model, nonetheless, the characteristics of the panel dataset used suggests trade-offs between each model specification. Indeed the portfolio-approach is recommended in the gravity model of trade literature. In the chosen model specifications, I account for the persistence in trade (using the latest pre-sample mean estimator technique), issue of zero-values in bilateral trade data, and different specifications of non-linear models used for panel data, as well as the sensitivity of panel gravity estimations to different fixed effects.

The small, positive and statistically significant effect of the energy price gap on trade holds across the five models. This suggests that the existence of the effect is robust across model specification, and not driven by the underlying assumptions of the models. The sensitivity of the results to other potential sources of bias was further assessed by subjecting it to rigorous testing. The results were robust to potential aggregation bias, fixed effects specification, dynamic lag selection, the specification of the energy price gap variable, as well as the possibility that the energy price gap is capturing the effect of other time variant factors. Nonetheless, a key finding was that the magnitude of the effect is difficult to establish with precision with the data and methods used here. The estimated effect is thus expressed as a range, where there is overlap in the 95% confidence intervals of the five specifications – the area of 0.05 and 0.15.

The findings in this chapter suggest that the concerns around short-term impacts on carbon leakage and competitiveness are not entirely ungrounded, but that such concerns have been overstated. For most sectors, international differences in energy or carbon prices do not influence trade patterns, hence leakage concerns need not dictate the design of carbon mitigation policy instruments. In sectors where I find evidence that trade tends to develop between countries with energy price differences is problematic, it is important to note that this is problematic, only if the difference in energy prices results from different carbon prices. If the difference arises from asymmetric carbon prices, this could then lead to carbon leakage (either positive or negative). Otherwise, the development of trade is good news.

The elasticities obtained in this study can be interpreted in a broader geographical context compared to previous studies which examined only industrialised countries. This analysis used bilateral trade data covering more countries, which represent economies with different levels of development. This is important, because carbon pricing policies are being implemented across the Annex-I and non-Annex I divide, and carbon leakage is no longer a rich nation's problem. For example, carbon leakage concerns have been raised following China's pledge to achieve its GDP energy intensity reduction targets largely through changes in sectoral composition of GDP

(Tekes, 2011). The estimations from this study would predict that changes of production do not imply large changes in trade patterns, at least in the short-term.

One of the key limitations to this analysis is the lack of sector-level disaggregation in the electricity price information, given the large number of countries in the data sample. While the detailed trade data allows controlling for sector and country-pair fixed effects, unfortunately, sectoral level variation in the energy price data was not available. As more detailed data become available, incorporating this variation into the analysis will likely enable more robust estimations at sector level.

## 4.7 Appendix

*Table 4.5:* 17 Sectors groups – sector grouping

| | | 2-digit SITC (Rev. 3) sectors included |
|---|---|---|
| 0 | Food and beverages | meat; dairy; fish; cereals; veg and fruit; sugars; coffee tea cocoa; animal feed; other food; beverages; tobacco |
| 1 | Raw materials | hides skins; oil seeds; crude rubber;cork wood; crude animal material; animal_fats; veg. fats; processed fats; |
| 2 | Mining (ores) | metal ore; |
| 3 | Fertilisers | crude fertiliser; fertilisers; insecticides; |
| 4 | Fuel | coal coke; petroleum; gas ; electricity |
| 5 | Non ferrous metal | nonferrous metals |
| 6 | Iron & steel | iron steel; metal manufactures |
| 7 | Cement, lime and non-metallics | cement; lime ; glass |
| 8 | Pulp & paper | pulp; paper |
| 9 | Chemicals | organic chemicals; inorganic chemicals; colour dye; essential oils |
| 10 | Pharmaceutical | pharmaceutical |
| 11 | Plastic | plastics primary; plastic non primary |
| 12 | Semi-manufacturing | leather manufactures; rubber manufactures; cork manufactures; |
| 13 | Machine manufacturing | power generating machines; industrial machinery; metalworking machinery; general industrial equipment; office machinery; telecom machinery; electrical machinery; power generating machines; industrial machinery; metal working machinery; general industrial equipment; office machinery; telecom machinery; electrical machinery;scientific instruments; photo equipment; optical wear |
| 14 | Transport equipment | road vehicles; non-road transport |
| 15 | Textile and clothing | textile; textile articles; travel goods; apparel; foot ware |
| 16 | Other manufacturing | prefab buildings; furniture; other manufactured goods |

# Chapter 5

# Net embodied carbon effects from carbon pricing policies

## 5.1 Introduction

Chapter 3 sought to improve upon estimates of embodied carbon in trade by conducting a detailed quantification of global embodied carbon in bilateral trade at the product level, using the material balance approach. Chapter 4 aimed to provide econometric estimates of the impacts of industrial energy prices on trade flows using observed data. Combining the findings from this thesis, this short chapter assess the magnitude of the energy price effect on trade, as well as potential net impacts of carbon pricing policies in terms of net embodied carbon.[1] The results are used to infer the effects of potential carbon leakage.

Any unilateral carbon pricing measure, invariably raises concerns about carbon leakage. In the EU ETS, large volumes of free allowances have been allocated to energy intensive and trade exposed (EITE) sectors to address carbon leakage. Similar proposals have appeared in climate legislation in (and proposals for) the US, Japan and elsewhere (Heilmayr & Bradbury, 2011; Ellerman et al., 2010). In Australia's Carbon Pricing Mechanism (CPM), the majority of emissions allowances are allocated by auction from the start of the flexible price phase (CDC Climate Reserach, 2012), but free emission allowances are also set aside for industries which fall under the EITE category.

Before one can answer whether or not these measures are necessary, one must ask whether the unilateral carbon price is likely to result in substantial leakage in the first place. In general, quantitative assessments of the potential trade impacts from unilateral policy to date have been conducted using *ex-ante* partial and general equilibrium models, but they predict a wide range

[1] This is defined as embodied emissions in imports minus embodied emissions in exports. It is also termed the Balance of Embodied Emissions in Trade (BEET).

of outcomes – central carbon leakage rates fall between 5% and 30%. As reviewed in Section 4.2.1, as of yet, empirical analysis of the trade impacts of carbon pricing policy remain few.

This chapter aims to apply the results from this thesis to evaluate the range of the magnitude of potential leakage. Three steps are involved.

1. Illustrate the magnitude of the effect of energy price on trade, using the examples of UK imports from South Korea, French imports from and Indonesia and finally by generalising these examples.
2. Simulate the near-term impact of carbon prices on bilateral import and export levels, using the estimation results from Chapter 4. The simulation is conducted for Australia's CPM and a hypothetical unilateral carbon price in the US.
3. Convert the near-term effects on trade to embodied carbon terms, for carbon price on trade for Australia's CPM .

For data reasons, the embodied carbon flows quantified in this thesis is based on world average emission factors (WAEF) and consequently cannot directly be used to quantify carbon leakage effects as explained in Section 3.1. However, as will be demonstrated, the results from this research can be combined to provide insights useful for the the policy debates around carbon leakage.

Applying the results from previous chapters, analysis in this chapter finds that historically, energy price differences explain a very small share of the variation in bilateral trade (less than 0.1%). In addition, using Australia's new carbon pricing policy as an example, a small net effect of the impacts of CPM on embodied emissions in exports (EEE) and imports (EEI) is found, in the order of 0.9% of Australia's domestic annual $CO_2$ emissions.

## 5.2 The magnitude of the energy price gap effect

### 5.2.1 An illustration using an example for UK's imports from South Korea, and France's imports from Indonesia

The analysis in Chapter 4, found the energy price gap effect on bilateral trade is positive and significant. This then raises the question as to how much of the overall variation in sectoral bilateral trade is explained by energy price differences. Two simple examples are used to illustrate the magnitude of the effect.

During the period 2003 to 2004, South Korea's real industrial electricity price increased from 44 to 49 US$/MWh. Over the same period, UK's real electricity prices rose from 73 to 84 US$/MWh. Recall that the energy price gap is defined as $epgap_{ijt} = ln(EP_{it}) - ln(EP_{jt})$ which can also be expressed as $epgap_{ijt} = ln\left(\frac{EP_{it}}{EP_{jt}}\right)$. In other words, the energy price gap between two countries is defined as the difference of the natural logs of the importer's industrial electricity price and the exporter's. In the case of UK and South Korea, the implied change over this period reflects roughly a 3% increase in the $epgap$ .[2]

Chapter 4 found that a 1% increase in the energy price gap was associated with a 0.05% to 0.15% increase in bilateral trade. These results thus predicts that a 3% change in the price gap between the UK and South Korea will increase South Korea's exports to the UK by 0.15% to 0.45%. What was actually observed over this period was a 40% growth in South Korea's exports to the UK. The electricity price gap is therefore explaining between 0.4 to 1% of the change in trade volumes. Other explanatory factors, such as underlying trends in transport costs, globalisation and supply chain integration, population growth and economic growth cycles are likely to play important roles in explaining this variation in trade over time.

In another example, between 2007 and 2008, Indonesia's real industrial electricity price decreased from 58 to 55 US$/MWh, while France's increased from 58 to 66 US$/MWh. This reflects roughly an 8% increase[3] in the energy price gap, as electricity prices rise for French industry. Thus by applying the range of the estimated effect, France's imports are predicted to increase by 0.4% and 1.2%. During this period, French imports from Indonesia increased by 21%.

### 5.2.2 A generalised analysis of the variance

Generalising the above two examples, this section tries to evaluate the share of variation in bilateral trade which is explained by the energy price gap in general. The following evaluation approach adopted from Wooldridge (2010) is chosen for transparency and simplicity.[4] The variation in the variables $epgap_{ijt}$ and $trade_{ijt}$ are compared in the following way:

- $var1 = \frac{var(\beta_1 epgap_{ijt})}{var(trade_{ijt})}$
- $var2 = \frac{var(\beta_1 epgap_{ijt} - \beta_1 \overline{epgap_{ijt}})}{var(trade_{ijt} - \overline{trade_{ijt}})}$

The first compares the overall variation and the second looks at the de-meaned variation. The variables are expressed in natural logarithms as defined in Section 4.3.1. The indicators are applied to data for all sectors (at the country pair level), and the evaluation is also conducted at the sector-level. Results are presented for a select number of sectors in Table 5.1. The estimation results from Section 4.4 are used for the beta coefficients. There is some variation across sectors. For the iron and steel sector, the energy price difference can explain around 0.2% of the variation

[2] $(ln(84/49) - ln(73/44)) = 0.0327$.

[3] $(ln(66/58) - ln(58/55)) = 0.0761$.

[4] Other common checks such as the partial R-square are not suitable for this data and estimation method (panel data with fixed effects).

*Table 5.1:* Analysis of the variance

| | β1 | Var1 | Var2 |
|---|---|---|---|
| Country pair level | 0.05*** - 0.15*** | 0.00% - 0.03% | 0.01% - 0.09% |
| Iron and steel | 0.08*** | 0.01% | 0.18% |
| Semi-manufacturing | 0.04* | 0.00% | 0.01% |
| Food | 0.02 | 0.00% | 0.00% |
| Fuel | 0.18*** | 0.04% | 0.06% |
| Transport equipment | 0.13*** | 0.03% | 0.04% |
| Textiles | 0.05** | 0.00% | 0.01% |
| Paper & pulp | 0.05** | 0.00% | 0.01% |

in trade according to the second indicator. For Fuel and Transport equipment, the price gap explains around 0.05%, However, across the entire sample, the variation in the energy price gap variable explains a very small share of the variation in trade, typically well below 0.1%.

As noted in Section 4.4.3, it is worth recalling that the historical variation in the electricity price gap has been considerable (as shown in the descriptive statistics in Table 4.1). Few meaningful carbon prices were in place during the time-period covered in the data. Thus energy price gap variations have been driven by factors other than climate policies. Therefore, from the perspective of ordinary fluctuations, additional changes to the energy price gap that may be caused by carbon pricing are unlikely to explain a much further part of the variation in bilateral trade.

## 5.3 The trade impacts of carbon pricing policies

What does the energy price gap effect translate to, in terms of carbon price differences? In Section 4.4.3, an example was used to illustrate how a one percent change in the energy price gap variable may translate in carbon price terms. This section takes this one step further. It takes two examples of unilateral carbon price policies and predicts the impacts on imports and exports using simple simulations. The simulations are conducted on the Australian CPM and a hypothetical carbon prices in the USA.

The simulation involves a few simple steps. First, I predict the level of trade in the absence of the carbon price, using the same models as in Section 4.4.1 and the observed energy price gap. Second, I generate a new energy price gap variable which takes account of the unilateral carbon price. To do so, the expected impact of the carbon price on electricity price is taken from the literature. Then I predict the level of trade using the new energy price gap, using the coefficient estimates from the original model. The difference between the two levels of trade gives the predicted carbon price impact. This simulation is run separately for imports and exports, and using four models for comparison – Arellano-Bond GMM, PML, Negative binomial and Zero

*Table 5.2:* Predicted impact of unilateral carbon price in Australia (A\$23/t$CO_2$ ) on Australian imports and exports across four model specifications

| | Impact on imports | Impact on exports |
|---|---|---|
| Arellano-Bond | 1.29% | -1.27% |
| PML | 0.54% | -0.56% |
| Negative Binomial | 1.61% | -1.55% |
| Zero inflated NB | 1.34% | -1.21% |

Note: The models are: differenced Arellano Bond GMM (one-step estimation with cluster-robust standard errors); Poisson maximum likelihood model, negative binomial model and zero inflated negative binomial model. The PML is estimated in levels, includes two lags of the dependent variable, country-pair-sector specific fixed effects, country time trends and time dummies. The latter two are also estimated in levels, and include two lags of the dependent variable, country-pair-sector specific fixed effects (using the pre-sample mean estimator), country time trends and time dummies. (Equation 4.6).

inflated negative binomial.

### 5.3.1 Australia's Carbon Pricing Mechanism

Based on the Australian Treasury's modelling of the CPM (The Treasury, 2012), implementation of the A\$23/t$CO_2$ carbon tax, is estimated to raise retail prices of electricity by 10%. Unfortunately, no estimates of the impact on industrial electricity prices are reported in the study. However, since the mark-up on the retail price is higher, industrial prices are, if anything, more insulated from the CPM. 10% is therefore a maximum estimate of the impact on the industrial energy price. Note that this analysis assumes zero free allowance allocation, thus the Australian sectors face the full impact of carbon pricing.

Table 5.2 presents the predicted impacts of the CPM on Australia's imports and exports, using the Arellano-Bond, PML, negative binomial and zero inflated negative binomial equations. Between the four models, Australian imports are predicted to increase, on the order of 0.56% to 1.61%. Exports are predicted to decline by 0.54% to 1.55%. As noted in Section 4.3.2.3, the empirical model's structure is such that it yields common elasticities for imports and exports. Hence the asymmetry between import and export impacts are driven by the differences in sector compositions of Australia's imports and exports. The net effect depends on the size of imports and exports.

Decomposing the estimates from the PML specification by sector, Table 5.3 reports the sector-specific trade impacts. The method used to account for sector heterogeneity are described in Section 4.4.2. The results indicate that when looking at the sector level, some are predicted to experience outside of the range for the country-level. Among those with the largest impact are some basic industries and energy intensive industries such as fuel, iron & steel, mining. Some non-energy intensive sectors also experienced larger impacts where trade volumes are high, for example transport equipment, and textiles.

*Table 5.3:* Predicted impact of unilateral carbon price in Australia (A$23/t$CO_2$) on Australian imports and exports, by sector group, using PML

| | Impact on imports | Impact on exports |
|---|---|---|
| Food | (0.38%) | (-0.37%) |
| Raw materials | (0.39%) | (-0.38%) |
| Mining | 0.91% | -0.89% |
| Fertilisers | (0.38%) | (-0.37%) |
| Fuel | 1.75% | -1.72% |
| Non ferrous metals | (0.88%) | (0.89%) |
| Iron & steel | 0.97% | -0.89% |
| Cement, lime, glass | (0.35%) | (-0.33%) |
| Paper & pulp | 0.67% | -0.62% |
| Chemicals | 0.68% | -0.67% |
| Pharmaceuticals | (0.60%) | (-0.55%) |
| Plastic | (0.28%) | (-0.26%) |
| Machinery | (0.42%) | (-0.40%) |
| Transport equipment | 1.38%) | -1.34% |
| Textile | (0.83%) | (-0.87%) |
| Other manufact. | (0.35%) | (-0.34%) |

Note: Brackets indicate the underlying beta coefficient is not significantly different from zero. The PML is used here, estimated in levels, includes two lags of the dependent variable, country-pair-sector specific fixed effects, country time trends and time dummies. The latter two are also estimated in levels, and include two lags of the dependent variable, country-pair-sector specific fixed effects (using the pre-sample mean estimator), country time trends and time dummies. (Equation 4.6).

Under Australia's Clean Energy Future legislation, additional assistance is provided for "strategic sectors", such as coal, steel and LNG. The Steel Transformation Plan provides, for example, assistance of A$300 million over four years, to encourage low-carbon investment and R&D, environmental and productivity improvements, as well as skills training in the sector (Australian Government, 2011). In addition, under the Jobs and Competitiveness Program, "strongly EIET" industries including many listed in Table 5.3 will receive 94.5% of their historic emissions multiplied by the average sectoral emission benchmark, and "moderately EITE" sectors will receive 66%.[5] This analysis suggests that the choice of assisted sectors appears well targeted, at least for iron & steel and coal. However, whether the levels of compensation are well chosen is unclear. If firms are over-compensated, this may be due to the lack of evidence on carbon leakage effects, as well as by other factors such as contribution to employment or successful lobbying.

### 5.3.2 A unilateral carbon price in the US of $15/tCO_2$

Aldy & Pizer (2011) simulate the effect of a policy scenario whereby the US implements a $15/tCO_2$ carbon tax unilaterally, and assesses its impact on industrial supply and demand (and the resulting competitiveness effect) using 1985-1994 data, assuming energy prices remain stable in all other countries. The authors assume that such a carbon price leads to an 8% increase in industrial sector electricity price, based on results of the US Energy Information Administration's modelling of the potential impact of the Lieberman-Warner Climate Security Act 2007 (US Energy Information Administration, 2008), and find a positive but small effect. Specifically, their results find that a unilateral US$15/tCO_2$ carbon price is associated with a 1.4% decline in domestic supply, and about two thirds of this is due to a reduction of domestic demand. Therefore, only one third of the decrease in domestic output is due to the rise in net imports, hence implying a very small trade impact – an indirectly estimated impact on net trade (the difference between domestic supply and demand) which they term the 'competitiveness effect'.

Using this assumption of 8% impact on electricity price, and the same method applied to the Australian CPM, I simulate the impact on imports and exports for hypothetical unilateral carbon price policy in the US. The results are presented in Table 5.4. Under all four models, the change in the *epgap* leads to an increase in bilateral imports to the US, and a decrease in bilateral exports. Impact on increased imports are estimated to be in the range of 0.27 to1.28%, whereas the impact on reduced exports is between 0.28% and 1.17%. The effect on imports and exports is asymmetrical because for each country the US trades with, the share of imports to that country relative to total imports, is different from the share of exports to that country relative to total exports.

[5] Both declining at 1.3% each year.

*Table 5.4:* Predicted impact of unilateral carbon price in the US ($15/tCO_2$ ) on US imports and exports across four models

| | Impact on imports | Impact on exports |
|---|---|---|
| Arellano-Bond | 0.85% | -0.84% |
| PML | 0.27% | -0.28% |
| Negative Binomial | 1.28% | -1.17% |
| Zero inflated NB | 1.01% | -0.95% |

Note: The models are: differenced Arellano Bond GMM (one-step estimation with cluster-robust standard errors); Poisson maximum likelihood model, negative binomial model and zero inflated negative binomial model. The PML is estimated in levels, includes two lags of the dependent variable, country-pair-sector specific fixed effects, country time trends and time dummies. The latter two are also estimated in levels, and include two lags of the dependent variable, country-pair-sector specific fixed effects (using the pre-sample mean estimator), country time trends and time dummies. (Equation 4.6).

The magnitudes of the effects for Australia are higher than those estimated for the US (Table 5.4 in Section 5.3.2) because of the higher carbon price (A$23/tCO_2$ approximately equates to a US$24/tCO_2$ carbon price), and also because of differences in sector composition of the US and Australian imports and exports.[6]

Whilst this is not a direct comparison – the sample period used here (1996 to 2011) does not overlap with the period used by Aldy & Pizer (2011) (1985-1994) and the outcome variables are different – the simulation results are supportive of one another. In sum, short-term trade impacts from a proposed unilateral US carbon tax is predicted to be small.

## 5.4 Embodied carbon impacts of Australia's Carbon Pricing Mechanism

The predicted change in Australia's imports and exports due to the CPM found in Section 5.3.1 can be translated into embodied carbon emission, in order to understand the impact of the policy on net EET trade. To do so, I multiply the predicted trade impacts by the embodied carbon estimates presented in Chapter 3.

The product level embodied carbon in bilateral trade is aggregated to calculate Australia's embodied carbon in total imports and exports in 2006. EEI and EEE are found to be 91Mt and 161Mt $CO_2$ respectively. The negative balance in embodied carbon is around 70Mt $CO_2$ or 18% of Australia's production-based emissions. Figure 5.2 in the Appendix presents Australia's EEE and EEI by trading partner. Australia's EET embodied in trade with China, the US, Japan, South Korea and Malaysia collectively accounts for around 50% of the total, for both exports and imports.

[6] How the different model specifications lead to differences in the estimated impact of the energy price gap on trade is explained in Section 4.4.1.

*Table 5.5:* The impact of CPM on Australia's embodied carbon trade, simulation results at the country level for 2006

| | EEI without tax | EEI with tax | EEE without tax | EEE with tax | net EET without tax | net EET with tax |
|---|---|---|---|---|---|---|
| Total (Mt CO2) | 91.43 | 92.01 | 161.36 | 160.30 | -69.92 | -67.86 |
| Change (Mt CO2) | | 0.58 | | -1.06 | | 2.06 |
| net embodied carbon | | | | | | 0.53% |

Note: Underlying trade impacts are obtained from the PML estimator. Carbon leakage here is defined as the share of net imported over domestic emissions.

Tables 5.5 presents the results of the net EET effect, using trade impacts estimated by the PML specification at the 16 sector level (coefficients are reported in Table 4.3). It shows the levels of Australia's EEI and EEE and net EET (EEI minus EEE) for 2006, with and without the CPM. Table 5.5 uses trade impacts calculated at the country level and shows that the CPM increases the carbon embodied in Australia's imports by 0.58Mt of $CO_2$, and decreases their exports by 1.06Mt $CO_2$. This gives an increase of net carbon imports of 2.06Mt $CO_2$. This in turn translates into a ratio of net carbon imports over domestic emissions by about 0.53%. Since the PML specification coefficients represent a lower bound estimate of the energy price effect on trade (see Figure 4.5), this exercise was repeated using the Arellano Bond model and the net EET effect was found to be around 0.86%. The results suggest the CPM is likely to increase Australia's net imports of embodied carbon in the area of 0.5% to 0.9%.

## 5.5 Net embodied carbon effect vs carbon leakage effect

How does the net EET effect relate to the carbon leakage effect? In Section 5.4, the measurement of net embodied carbon effect of the CPM was based on EET estimates from Chapter 3 calculated using world average emission factors. How the net EET effect relates to carbon leakage is determined by the degree to which Australia and its trading partners' emission factors diverge from the world average. A serious attempt to translate the 0.86% impact of the CPM on Australia's net EET into carbon leakage rates require further data. In particular, country and sector specific emissions intensity data is necessary not only for Australia but also its key trading partners, but high quality data with good coverage is difficult to obtain as discussed in Section 3.5. Although this is beyond the scope of this analysis, in this specific case, the good news is that the matter is somewhat simplified by the fact that Australia has a large net export balance (Table 5.5). This means that what matters more is understanding how Australian sectors' emission intensities deviate from the average.

It does not seem implausible to assume that the emission intensities of Australia's industry sectors do not diverge significantly from the world average. Australia has an abundant coal supply and is the world's ninth-largest coal consumer and records higher than average levels of carbon emissions in per capita and per GDP terms, relative to the world average (World Resource Institute, 2012). 70% of electricity is coal-fire powered. However such indicators may not give a good indication of relative emission factors for Australia's manufacturing sectors. Industrial process emissions intensity is lower than the world average, when measure per unit of GDP. The same applied to energy emissions intensity in manufacturing in per $GDP terms (World Resource Institute, 2012). Indeed, many key mainland industrial production centres, rely more on gas than coal (IEA, 2011b). The cement case study in Section 3.5 also showed that Australia's emission factor is close to the world average. Thus, the net EET effect estimated in this chapter may be in the right ballpark of Australia's carbon leakage effect.

## 5.6 Robustness and key assumptions

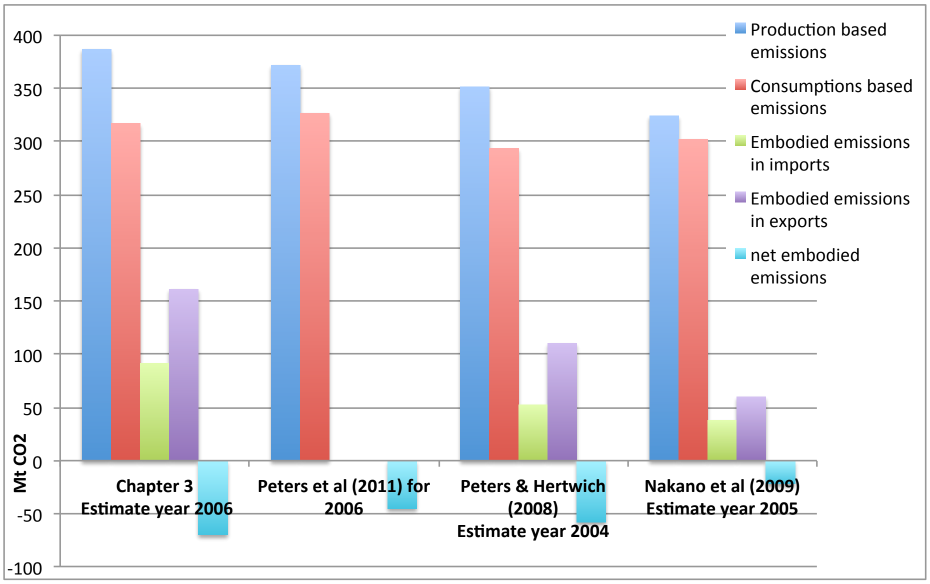
Firstly, the estimates of the net embodied carbon effect from carbon pricing policies in this chapter rely on the quality of the underlying EET estimates. As was discussed in Chapters 2 and 3, the variations in the choice of methodology, assumptions and parameter choices within studies as well as underlying data all give rise to varying estimates of embodied carbon. As has been empasised, product level EET estimation required using world average emission factors for products. Hence over- or under-estimations of EET can occur if Australian production technologies deviate significantly from the world average.

Figure 5.1, plots Chapter 3's estimates of Australia's embodied carbon against estimates from three other studies and shows they are comparable. The estimate years for these studies are between 2004 and 2006, due to availability of estimates for Australia. As the Figure shows, estimates of Australia's EEE and EEI in Chapter 3 are slightly higher relative to other studies (purple bar on the right side of Figure 5.1). Comparing the production- and consumption-based emissions with the study for the same year shows they differ by less than 5%. It is more difficult to compare the differences in terms of EEE and EEI as the estimate years differ and the time trend suggested by the comparison is ambiguous (estimated 2005 emissions are lower than 2004) and not aligned with the increasing trend globally. Overall, the net embodied carbon effect estimated here appears robust to the embodied carbon estimates used. If anything, using embodied carbon data from other studies would find a smaller net embodied carbon impact.

Secondly, the net embodied carbon effect estimates also rely on the quality of the underlying elasticities estimated using a regression framework. As shown in Section 4.4.1, the magnitudes of the effect of the energy price gap on imports and exports are imprecisely estimated. However, using a feasible range of coefficient estimated from different models enables us to also understand a feasible range of the net EET effect.

Thirdly, this analysis assumes that a 1% change in trade flows in value terms corresponds to a 1% change in EET flows. This is because the impact of asymmetric energy prices on trade

*Figure 5.1:* Embodied carbon in Australia's trade - A comparison of results from three studies for years 2004 to 2006.




Notes: Australia's production-based emissions for this study is obtained from United Nations Framework Convention on Climate Change (2012). Peters et al. (2011) does not explicitly report EEE and EEI estimates.

was estimated using bilateral trade data in value terms. Flows of embodied carbon in trade, are measured in tonnes of carbon dioxide.

A one-to-one proportionality is assumed when translating the estimated change in trade levels (in value terms) due to the CPM, to changes in embodied carbon. If additional assumptions are made to allow for price and exchange rate adjustments, this would imply a smaller coefficient of proportionality, hence a smaller net embodied carbon effect than when using the 1:1 proportionality assumption.

Fourthly, as noted, this analysis does not account for the fact that compensation was paid to EITE sectors. The details of the "additional assistance" has not been released publicly, hence they are difficult to address. In terms of the allocation of free allowances to these sectors, this analysis makes the key assumption that the opportunity cost of freely allocated allowances provides the same carbon signal to the sectors, in order to incentivise greater emissions reduction. There is some evidence in the literature against this assumption from the early phases of the EU ETS (e.g. Lecourt et al., 2013). However, it is unclear whether the EU experience can be generalised to different ETS schemes.

Finally, it is also worth noting that the predicted change in Australia's imports and exports resulting from the carbon price under the CPM is based on the estimated impact of asymmetric industrial electricity prices on trade flows for 51 countries, and is not specific to Australia's trade. This is problematic if factors driving Australia's trade substantially differ from those that characterise the average trade for the set of 51 countries. Given Australia's sustained high terms of trade in mining exports (The Treasury, 2012), a more detailed sector-specific analysis for Australia may be warranted.

In sum, the additional assumptions made are reasonable. While there is room for further investigation and improvements, there is no serious contradictions that underly the basic results – that the magnitude of the effects of energy price differences on trade is small.

## 5.7 Conclusion and discussion

The objective of this chapter is to contribute to the understanding of carbon leakage impacts of climate policy. It does so by combining a number of empirical findings from this thesis. In Chapter 3, embodied carbon were quantified using the material balance approach, for bilateral trade between 195 countries and 970 traded products. In Chapter 4, the statistical relationship between asymmetric industrial energy prices and bilateral trade flows was examined using dynamic panel data methods within a gravity framework. These two strands of empirical analyses were needed to estimate net embodied carbon impacts from carbon pricing.

As a start, the magnitude of the effects estimated in Chapter 4 are illustrated using the examples of UK's imports from South Korea and France' imports from Indonesia. These examples were then generalised across the entire sample and the variation in the energy price gap variable was found to explain a small share of the variation in trade. Although larger for some sectors than others, it can still be said that less than 0.1% of the overall variation in bilateral trade is explained by energy prices.

Next, simulations using the estimation models from Chapter 4 were conducted to assess the near-term trade impacts of carbon pricing policies and were assessed for Australia's CPM and a hypothetical carbon price scenarios in the US. The simulations predicted that the change in energy price induced by a carbon price tends to increase imports and reduce exports. Sector level simulations for Australia's CPM showed that impacts on imports and exports vary across sectors. However, evidence of substantial trade impacts is not found for any sector.

Finally, using the example of Australia's CPM, the near term effects on trade was translated to embodied carbon terms using EET estimate from Chapter 3. This analysis finds a small net embodied carbon effect as a result of the introduction of the A\$23/t$CO_2$ carbon price, in the order of magnitude of 0.5-% to 0.9% relative to Australia's domestic annual $CO_2$ emissions. Although this cannot be interpreted directly as a carbon leakage effect, the true carbon leakage effect is unlikely to be significantly higher. Thus suggests the CPM would imply around a very small percentage shift of carbon emissions overseas.

The results obtained in this ahapter represents a middle-of-the-range estimate, compared with the previous modelling analysis conducted for the CPM, which have found mixed results. Siriwardana et al. (2011) found a 0.32% positive net impact on trade (terms of trade), Meng et al. (2013) found a negative impact on exports (3.8% to 6.4%), and finally the analysis by The Treasury (2012) found a small impact on trade.[7]

How robust are these results? The quality of the EET measurement and the estimates of the effect of energy price on trade from previous chapters largely determine the reliability of the results in this. Furthermore, a number of assumptions are required in order to translate the estimated trade impact of historic industrial energy price asymmetry into predictions about embodied carbon, as discussed in Section 5.6. However, Furthermore, these need to be weighted against all of the assumptions that go into the complex model based estimates.

Australia is currently in the midst of a heated debate on carbon leakage and competitiveness impacts. The CPM implemented as part of the Clean Energy Future legislation is now in operation and due to transfer to an ETS system in 2015. The quantified impacts obtained in this work can provide empirically grounded assessments of the policy to inform the surrounding discourse. The approach presented in this paper complements the quantitative assessments provided by modelling analysis.

Based on the findings in the chapter, policies to ’prevent’ carbon leakage may be justified only for a few sectors, and are likely to be required in small measures. For many of the sectors, such policies may have only a limited impact because little leakage is expected in the first place. Indeed, many have pointed out the economic drawbacks of free allocation (Hepburn et al., 2006; Sterner & Muller, 2008), and the damage to emissions trading’s credibility with polluters receiving large windfall profits (Sandbag, 2011). In Australia, critics of free allocation point to forgone tax revenues which could have been hypothecated towards low-carbon investments (Denniss, 2012). In addition, questions have been raised as to whether the free allocation of emission allowances could place Australia in breach of its obligations under the World Trade Organization (WTO) agreements (Haywood, 2011). It appears there is both empirical and political currency in providing data driven, econometrically based and transparent estimates of trade and embodied carbon impacts.

[7] See the Literature Review in the Appendix, on the existing literature on the CPM trade impacts.

## 5.8 Appendix

### 5.8.1 Literature

Carbon leakage effects in the Australian context have been thus far examined using general equilibrium models. Meng et al. (2013) uses a static computable general equilibrium model with an environmentally extended social accounting matrix, and finds that a A\$23/t$CO_2$ carbon tax reduces Australian exports by 3.8% in the case with no free allowance allocation, and by 6.4% in the case with compensation. This is explained in terms of the rise in commodity prices due to the carbon tax which makes exports less attractive. This price effect is magnified with industry assistance (free allowance allocation) because it increases domestic demand, which further increases commodity prices. Although the assumptions are not made clear, the fact that industry assistance is found to increase domestic demand implies that the paper is modelling a scenario with over-compensation for industry, and this needs to be considered when interpreting and comparing the results.

Siriwardana et al. (2011) uses the same static computable general equilibrium model as Meng et al. (2013) (i.e. ORANI-G) but without the aforementioned extension. The authors find a 0.32% positive effect on Australia's terms of trade, i.e. higher net exports[8] from a A\$23/t$CO_2$carbon tax without compensation. In this model, the carbon tax leads to inflation, and this appreciates the real exchange rate (The Australian dollar becomes relatively weak), driving down the price of exports in domestic currency terms.

The Australian Treasury's modelling analysis does not explicitly report predicted trade impacts, nor does it compare scenarios with and without EITE sector compensation – the specified levels of compensation under the Jobs and Competitiveness Program is implicit within the scenarios reported. However, the analysis of the impacts on industry output finds that changes to trade volumes are likely to be small: "At the broad sectoral level, structural changes due to carbon pricing are much smaller than the effects of ongoing changes in the terms of trade or tastes... Sectors will grow at similar rates with or without carbon pricing." (The Treasury, 2012, Section 5.4.1) "Modelling results show this transitional assistance will support output in emission-intensive industries. Output remains as high as, or higher than, it would be in the... scenario without domestic carbon pricing." (The Treasury, 2012, Box 5.5).

These modelling analyses clearly have not been effective in allaying concerns among policy makers about "perceived carbon leakage". This may be due to the very complex nature of the models used [9] and the numerous assumptions they necessitate. For example regarding growth

[8] This is not defined in the paper, and interpreted as the impact on net trade according to convention.

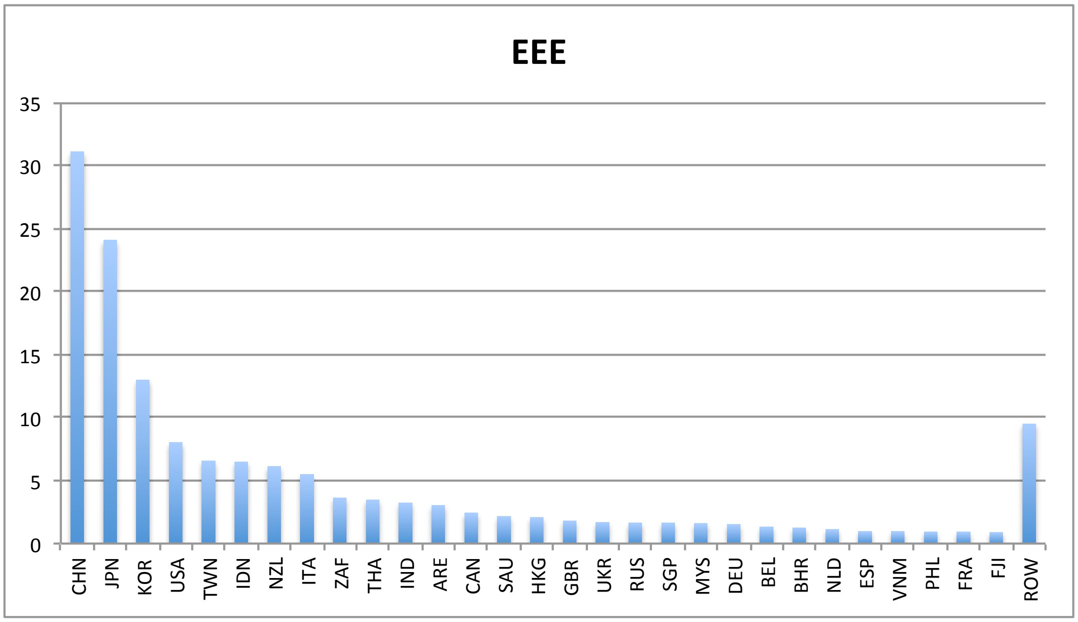
[9] The Australia Treasury model for example consists of a number of models. The GTEM provides the international economic and emissions context for modelling of the Australian economy. Projections of the national, regional and sectoral impact of carbon taxes are obtained from the MMRF model. The model for the assessment of greenhouse-gas-induced climate change (MAGICC) estimates the GHG atmospheric concentration levels. The ROAM model and the SKM MMA model provide detailed bottom-up information on the Australian power sector. Road and Transport is

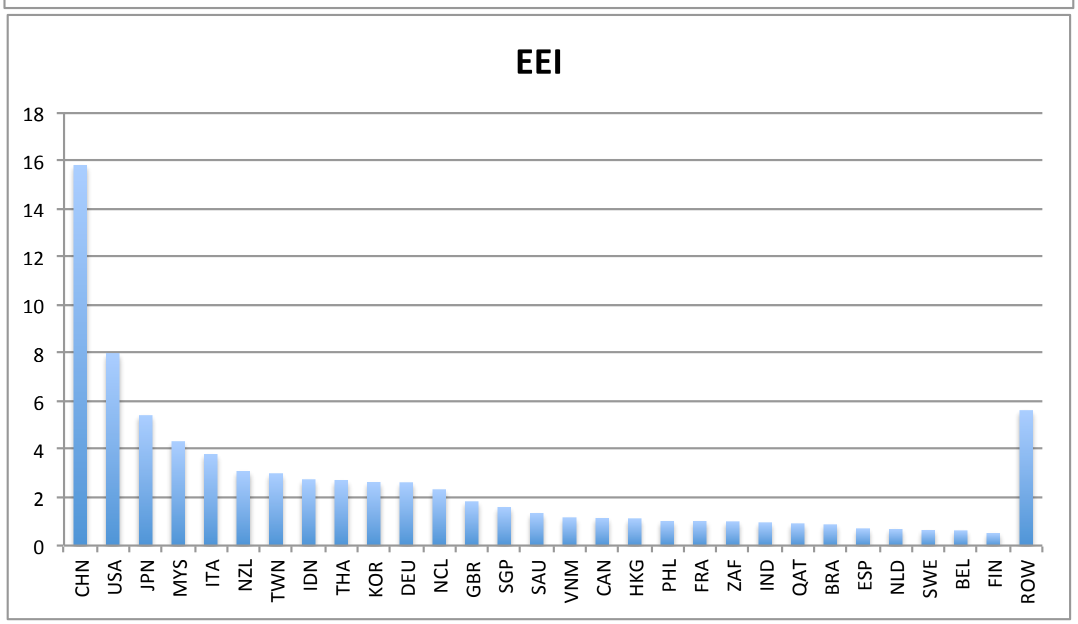
trends, exchange rates, productivity and technology changes, fossil fuel demand elasticities, trade elasticities, specification of sector demand and supply functions, energy efficiency options, nominal wage levels and exchange rates to name but a few. This suggests that there is room to compliment the existing knowledge with analysis using different methods. Although econometric estimations also involve multiple assumptions and require careful interpretation, it also has advantages. For example, observed data is used directly rather than after calibration of the model.

### 5.8.2 Supplementary charts

modelled by the Energy sector model (ESM). Household impacts are simulated using the price revenue incidence simulation model and distribution model (PRISMOD.DIST). When multiple models are used, the accuracy of results is subject to the degree of integration among the models and the way they are integrated (Siriwardana et al., 2011).

*Figure 5.2:* Australia's embodied emissions in exports and imports by trading partner in 2006



# Chapter 6

# Synthesis and conclusions

This thesis conducted two separate but related strands of research: the first on quantifying embodied carbon in trade, and the second strand on the trade impacts of asymmetric climate policy. It was presented as two Parts, each comprising of two research papers:

1. Embodied carbon in trade
    - A survey of the empirical literature
    - Product-level embodied carbon flows in bilateral trade
2. Carbon leakage and competitiveness impacts
    - Asymmetric industrial energy prices and international trade
    - Estimating net embodied emissions impacts from carbon pricing policies

To conclude the thesis, this Chapter briefly synthesises the two strands of research, highlighting the original contributions, key findings, and policy implications. Finally, it offers suggestions for future work.

Both embodied carbon in trade and carbon leakage have received much attention, as was discussed in Chapter 1. The empirical relationships between climate change policy and trade can inform many policy questions, as was discussed in Chapter 2. 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 convention of production-based emissions accounting and inventory. Many have argued that explicitly incorporating consumption-based principles to complement production-based principles can improve fairness of outcomes in terms of the distribution of responsibility across producers and consumers. At a *lower level*, or in terms of detailed policy elements, embodied emissions in trade (EET) flows quantified at the sector level have facilitated discussions around the carbon leakage concerns that surrounds the implementation of unilateral climate change policies. They can provide useful insights for the potential design, functioning and distributional consequences of measures to address these concerns.

This thesis tried to contribute to the evidence base for these debates surrounding climate change policy and trade. Part 1 sought to improve upon estimates of embodied carbon in trade. Part 2 aimed to provide more empirically grounded estimates of trade impacts from climate change policies.

## 6.1 Main results and policy implications

### 6.1.1 Embodied carbon in trade

Previous reviews of the embodied carbon literature have focused on methodology (e.g. Lutter et al., 2008; Wiedmann et al., 2009; Hertwich & Peters, 2010; Liu & Wang, 2009; Wiedmann et al., 2011; Peters & Solli, 2010) but the reported results had yet to be subject to careful comparative evaluation. Chapter 2 sought to fill this gap.

The literature finds large and growing volumes of carbon dioxide emissions embodied in global trade (around 30% of global carbon emissions in 2006). Yet the synthesis in Chapter 2 found that **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, China's EEE in 2005 is found to be in the range of 18% to 49% of their production-based emissions, whereas the estimated EEI for the same year ranged from 8% to 44%. By examining the sources of uncertainty inherent in the models used, it concludes **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 into strategies for $CO_2$ mitigation, for example as a shadow indicator for countries with large net imports of embodied carbon. Moreover, **better quantitative understanding of embodied carbon at the sector or supply chain level can provide useful insights for policy**, such as the potential design, functioning and distributional consequences of measures to address these concerns.

To complement and improve the existing empirical work synthesised in Chapter 2, Chapter 3 **quantified global embodied carbon in bilateral trade at the product level**. This represents a first quantification exercise of global embodied carbon at this level of disaggregation both in terms of countries and sectors. This was done by collecting product carbon intensity factors from multiple data sources and using the material balance approach, whereby bilateral trade flows expressed in physical quantities are multiplied by product pollution intensities. Although it has limitations of its own, this method overcomes a number of key sources of uncertainty in the existing studies.

The granular mapping of embodied carbon revealed a number of new insights which were masked in previous quantifications using aggregated models. It found that **focusing attention on the balance of embodied carbon in trade between Annex I and non-Annex I regions invites simplistic and problematic interpretations of EET** estimates. For example, China's large surplus and the US's large deficit of EET has been highlighted in the literature (e.g. Peters & Hertwich, 2008; Davis & Caldeira, 2010; IEA, 2008) and in the media (Watts, 2009; The Economist, 2011). This study examines EET in *bilateral trade* and shows that other than trade with China, **embodied carbon is focused in regional trade** – for the US, the embodied carbon trade flows with neighbouring countries such as Canada and Mexico are also important. It suggests that **regional harmonisation of climate mitigation policy should be a priority**.

In terms of the distribution of global EET across products, **70% of global EET is attributable to 15% of the 970 products examined.** This suggests that **focusing mitigation efforts and trade-measures on these products would be an effective strategy to address potential carbon leakage, and to decarbonising international supply chains**.

Examining product level bilateral trade in EET revealed **striking differences in the origin and destination of countries' EEI and EEE, as well as the product compositions**. For example, China's carbon imports are typically embodied in primary inputs to industrial production and originate from resource-rich countries. In contrast, China exports embodied carbon via manufactured products such as electronics, apparel, and also up-stream industrial products, to economies such as the US, EU, Japan and South Korea. Evaluating the type of products by which a country imports and exports embodied carbon revealed that some countries (e.g. Brazil, Russia, Australia) export considerably more embodied carbon than they import. Typically the emissions are embodied in exports of upstream industrial products such as ores and basic chemicals. Other countries import more than they export (e.g. US, Singapore and Belgium). Many countries both import and export large volumes of embodied carbon (e.g. China, Japan, EU). It is argued that **grouping countries according to patterns of production and consumption may be more relevant in discussions surrounding climate policy and trade**, rather than discussing in terms of industrialised vs developing countries, as is often done. The new grouping suggests, for example, that advancing consumption-based accounting principles for climate policy is particularly relevant for countries with high levels of net imports of embodied carbon.

Although there are important limitations to quantifying EET using the material balance approach, this examination demonstrated that **there is value in providing product-level embodied carbon flows in bilateral trade**. It provides novel insights into the nature of the flows, which were not possible in preceding studies. The case study on the cement sector underlined the importance of obtaining reliable country specific emission factors for EET estimations, and shed light on **problems with methods commonly used in the literature to artificially create country-specific sector level emission factors.**

**Two new datasets were constructed – product level global bilateral trade in physical quant-**

**ities and carbon intensities of products**. These will be made public upon completion of this thesis and it is hoped that they will contribute towards new research, to complement other EET datasets in the public sphere e.g. Peters et al. (2011b); Davis & Caldeira (2010); Davis et al. (2011).

### 6.1.2 Carbon leakage impacts of unilateral climate policy

The second line of analysis is **quantification of carbon leakage** – how unilateral carbon pricing measures relate to trade patterns. So far, quantitative assessments of carbon leakage have been conducted using *ex-ante* partial and general equilibrium models. However, results are sensitive to model structure and assumptions made, *inter alia*, technological changes, supply elasticity of fossil fuels or capital mobility. A wide range of leakage rates have been reported by these studies, highlighting the need for empirical analysis to improve understanding of these effects and their magnitudes.

There are a number of empirical papers that study carbon leakage in various contexts, yet there is limited agreement in the magnitude of the effect. As indicated in Chapters 4 and 5, this thesis has contributed to this line of empirical research.

The approach taken in Chapter 4 was to examine the response of bilateral trade to industrial energy prices. The **effect of energy price on trade is positive and significant at the 1 percent level across several model specifications** (both linear and non-linear), such that **a one percent increase in the electricity price ratio between the importer and exporter increases bilateral export levels of around 0.05% to 0.15%**. In other words, where the exporter has a lower industrial energy price, a larger energy price gap between the two countries is associated with greater bilateral trade activity. It also found that **differences in industrial energy price explain a small part of the variation in trade flows (<0.1%)**. Whilst a number of empirical studies conduct comparable analysis, this study is the **first to examine trade impacts using bilateral trade between over 50 countries, and using dynamic panel methods within a gravity model framework**.

Simulations were conducted to extrapolate from the results of the econometric analysis, what are the predicted trade impacts associated with a unilateral carbon tax in the US. In line with the simulation in the paper by (Aldy & Pizer, 2011), it was **estimated that a \$15/t$CO_2$ carbon tax (assuming no carbon pricing in other regions) increases US imports by around 1%, and decreases exports by a similar degree**.

Finally, to demonstrate how the empirical research conducted in this thesis can help inform concrete policy debates, the findings from Chapters 3 and 4 were combined to **provide more estimates of net embodied carbon impacts from carbon pricing**. To do so using the example of Australia's Carbon Pricing Mechanism (CPM) involved simulating the impact of the A\$23/t$CO_2$ carbon tax on Australia's bilateral import and export levels using estimation results from Chapter

4, then translating the trade impacts into embodied carbon using quantification from Chapter 3. **A small net embodied carbon effect is found from Australia's Carbon Pricing Mechanism, in the order of magnitude of 0.5% - 0.9% of Australia's domestic $CO_2$ emissions** is found. To the author's knowledge, this is the first estimate of the net embodied carbon effects for Australia, which are derived from a regression framework rather than using equilibrium simulation modelling.

Therefore, the quantified impacts obtained in this work can provide empirical assessments of carbon leakage effects of the CPM. A number of assumptions are required in order to translate the estimated trade impact of historical industrial energy price asymmetry into predictions about the net embodied carbon effect. Nonetheless, robustness tests on these assumptions found that there is no serious contradictions that underly the basic results – that the magnitude of the effects of energy price differences on trade is small. These assumptions also need to be weighted against all of the assumptions that go into estimates based on more complex models. The results obtained represents a middle-of-the-range estimate, compared with the previous modelling analysis conducted for the CPM, which have found mixed results – one study found a 0.32% positive net impact on trade (terms of trade), another found a negative impact on exports (3.8% to 6.4%), and finally the Australian Treasury analysis found a small impact on trade.

The findings from this thesis suggests that **policies to 'prevent' carbon leakage may be justified only for a few sectors, and are likely to be required in small measures**. For many of the sectors, such policies may have only a limited impact because little leakage is expected in the first place. Indeed, many have pointed out the economic drawbacks of free allocation. Therefore, **concerns about impacts on carbon leakage are not entirely unfounded, but have been overstated.**

**This study finds a substantially smaller net embodied carbon effect, compared with the previous empirical analysis**. The study by (Aichele & Felbermayr, 2012) finds a 14%[1] carbon leakage rate for Annex I countries as a result of then Kyoto Protocol commitments. Part of the discrepancy may be explained by the use of world average emission factors in this study. At the same time, the approach presented in Chapter 5 is arguably more empirically grounded because it estimates trade impacts based on real historic data. Aichele & Felbermayr (2012) instead derives the statistical relationship based on estimated embodied carbon flows, measuring of which involves a considerable degree of uncertainty as demonstrated in Chapter 2.

This is application one of many ways in which the empirical research conducted in this thesis can help better inform a variety of policy debates surrounding climate change policy and trade.

## 6.2 Summary of policy implications

- Methodological and data considerations involved in measuring embodied carbon in trade limit the practical application of consumption-based accounting in climate policy in a

[1] Also expressed as a ratio of net carbon imports over domestic emissions

serious way, but it may be a useful shadow indicator for countries with large net imports of embodied carbon.

- Embodied carbon flows are focused in regional trade, hence regional harmonisation of climate mitigation policy should be a priority.
- Focusing mitigation efforts and trade-measures on certain products would be an effective strategy to address potential carbon leakage, and to decarbonise international supply chains – around 70% of global embodied carbon in trade is attributable to 15% of the traded product-categories examined.
- Patterns of production and consumption may be more relevant in discussions surrounding climate policy and trade, because the revealed complexity of EET flows renders the conventional grouping of countries – industrialised countries vs developing countries – inappropriate.
- Most large emitting countries are both large producers and consumers of embodied carbon, and tend to have a small net balance of EET at a country level. This suggests that the role of consumption-based accounting methods may be limited at the country level, for example in the context of multilateral burden sharing agreements.
- The role of consumption-based accounting methods may be important at the sector level, particularly for key energy-intensive and trade-intensive sectors. This suggests efforts to improve the estimations of EET flows for such sectors is likely to add more value to carbon leakage discussions.
- Overall, concerns about impacts on carbon leakage are not entirely unfounded, but have been overstated. Historical differences in industrial energy price explain a small part of the variation in bilateral trade flows.

## 6.3 Future directions

### 6.3.1 Quantifying embodied carbon in trade

The propensity to improve measurement of embodied carbon flows using the material balance methodology will be a function of improved and increased availability of carbon intensity multipliers at the product level. Carbon intensity estimates have been obtained largely from few countries such as Germany, the US, Japan. A considerable share of LCA data are privately owned. Estimates are also rarely available as time-series. Chapter 2 suggests there is a strong case for focusing detailed analysis of embodied carbon in the global production supply chains for some 150 key industrial products – including certain metal product, chemicals, ore and mining product, as well as some down-stream consumer products – which account for 70% of global EET flows.

### 6.3.2 Carbon leakage impacts

This research found evidence of the carbon leakage effect, although small in magnitude. The implications of this finding can be politically sensitive. Some interesting extensions to this work may be to address some of the related concerns about the impacts of climate change regulation on domestic industry sectors, which will continue to be an important theme in political debates surrounding these policies:

- effect of carbon leakage on employment –are the sectors most impacted, labour intensive?
- effect on profits – how will multi-national companies evaluate embodied carbon in supply chain vs value chain?

Another important issue for future research is the trade-off between the impact of unilateral carbon pricing policies on international trade, and on other channels of emissions mitigation for industry, such as innovation and demand substitution. By imposing a price on carbon emissions, climate change policies not only provide incentives for firms to import carbon-intensive inputs/products from countries with lower carbon prices. They also provide incentives to develop new technologies that reduce the emissions intensity of their output. Better understanding, therefore, of the interaction between carbon leakage and induced technological change will be interesting for policy makers, who envisage environmental policies to create leadership in clean industrial production.

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