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China's balance of emissions embodied in trade: approaches to measurement and allocating international responsibility

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Abstract International trade is characterized not only by the flow of capital and goods, but also by the energy and emissions embodied in goods during their production. This paper investigates the evolving role that Chinese trade is playing in the response to climate change by estimating the scale of emissions embodied in China's current trade pattern and demonstrating the magnitude of the difference between the emissions it produces (some of which are incurred to meet the consumption demands of the rest of the world) and the emissions embodied in the goods it consumes. Estimating China's emissions on a consumption rather than a production basis both lowers its responsibility for carbon-dioxide (CO_2) emissions in 2006 from 5,500 to 3,840mtCO_2 and reduces the growth rate of emissions from an average of 12.5 per cent p.a. to 8.7 per cent p.a. between 2001 and 2006. The analysis indicates that a reliable consumption-based accounting methodology is feasible and could improve our understanding of which actors and states are responsible for emissions. For example, recent emissions reductions by developed countries may lack credibility if production has merely been displaced to countries such as China. Moreover, in the current institutional context, production methodologies encourage leakages through trade that may do more to displace than to reduce emissions. Both equity and efficiency concerns therefore suggest that emissions embodied in trade should receive special attention in the distribution of post-Kyoto abatement burdens.

Key words: balance of emissions embodied in trade (BEET), China, consumption-based accounting, pollution haven effect, processing trade

JEL classification: F18, O53, Q54, Q56

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I. Introduction

Thirty years after its 'opening and reform', China has earned its reputation as the 'factory of the world'. China's rise to become, according to some reports,[1] the largest single emitter of greenhouse gases is closely linked to its economic growth, and particularly the export sector that has driven this growth. Export volumes accounted for 40 per cent of GDP in 2006, with the majority consisting of intermediate or consumption goods destined for developed countries' markets. Under current Kyoto Protocol accounting rules, the emissions associated with these exports are fully attributable to China, since they took place within its territory. As China and other developing exporters watch their emissions increase rapidly relative to the OECD countries, they are beginning to question why they are criticized for such rising emissions by the very consumers whose market demands they are supplying. On top of their historic responsibility for cumulative emissions, a central question for a post-Kyoto framework is whether developed countries should take responsibility for a portion of *current* emissions from developing exporters like China. This is an argument that was raised by senior Chinese officials at the Bali conference in December 2007.[2]

This paper makes three contributions to our understanding of the role of Chinese trade in the response to climate change. First, it estimates the scale of emissions embodied in China's current trade pattern, demonstrating the magnitude of the difference between emissions accounts based on production rather than consumption. In doing so, it extends the range of country studies carried out (for example, Machado *et al*. (2001) on Brazil and Mukhopadhyay (2004) on India) and complements international comparative studies (Ahmad and Wyckoff, 2003; Ward, 2005). We show that China was a net exporter of 1,660 mt of carbon dioxide (CO_2) in 2006, a figure that is growing rapidly. These emissions are incurred to support consumption elsewhere, but establishing specific counterfactuals is difficult since production patterns and energy intensities are endogenous to historical development trajectories. However, for illustration, directly transferring responsibility for emissions from producer to consumer would have raised US CO_2 emissions by 2.6 per cent in 2002.

Second, the paper improves on the methodologies used in previous studies of China, including Wang and Watson (2007), Shui and Harriss (2006), and Li Hong *et al*. (2007), by taking account of total energy intensity in upstream production and changes in energy intensity over time. This illustrates that a reliable consumption-based accounting methodology is feasible and could improve our understanding of emissions responsibility in a post-Kyoto framework. Third, it assesses the economic factors, national policies, and international frameworks that explain the current pattern of emissions in trade. While producers' locational decisions have been influenced by Chinese policies such as a depressed exchange rate and export tax rebates, we argue that complementary policies of deindustrialization in developed countries, trade liberalization, and the failure to harmonize international climate-change policy have also contributed to the emissions surplus.

[1] Netherlands Environmental Assessment Agency (2007); see also IEA (2007).

[2] The issue was first raised on 4 June 2007 by Ma Kai, Director of the National Development and Reform Commission, at a press conference on China's National Programme on Climate Change. It was reiterated at the Bali conference by his deputy, Xie Zhenhua, the head of the Chinese delegation to the 13th Conference of Parties to the United Nations Framework Convention on Climate Change (UN FCCC) Serving as the 3rd Meeting of the Parties to the Kyoto Protocol (COP13/MOP3).

These considerations lead us to conclude that, if Chinese production has merely substituted for production in developed countries, recent emissions reductions in developed countries may lack credibility. Reported Kyoto emissions performance may be a poor guide to the sacrifices that countries are making and the actual environmental impact of their consumption activities. Attributing full responsibility to China (and other developing countries) for historical emissions surpluses may then be unfair according to some normative criteria. Further, the current Kyoto production methodology does not create appropriate incentives for *global* decarbonization, but permits extensive leakages through trade. Consequently, as the distribution of abatement efforts comes to the fore in post-Kyoto negotiations, we stress that close attention should be paid to emissions embodied in trade if future methodologies are to be simultaneously equitable and able to provide effective abatement incentives.

Section I summarizes alternative emissions accounting methodologies and provides a framework for understanding the multiple effects of an expansion of trade on emissions. Section II estimates the emissions embodied in Chinese trade and national emissions on an alternative (consumption accounting) basis. Our methodology and results are also contrasted with previous studies. Section III discusses how we might efficiently and equitably re-assign responsibility for these emissions, evaluating the merits of a consumption basis for emissions accounting.

II. Accounting for greenhouse-gas emissions

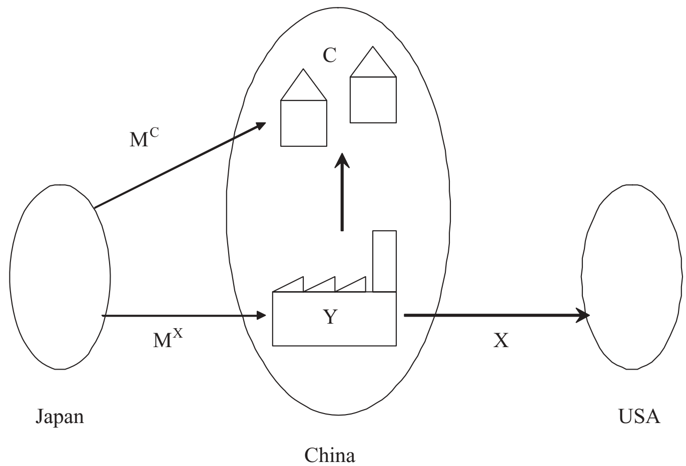
(i) Trade and emissions

There are many links between international trade and emissions, including direct effects from transportation and more subtle links from foreign investment and ownership. In this paper we focus only on the emissions embodied in traded goods themselves. The expansion of international trade has led to a significant divergence between the incidence of production and consumption. Just as countries with a balance-of-trade surplus export more than they import, countries run a surplus on the balance of emissions embodied in trade (BEET) where the emissions involved in producing the goods they consume (including those produced abroad) are less than the emissions from domestic production.

However, there are a number of differences between trade in goods and in emissions. First, the two do not always coincide—a country running a trade deficit could nevertheless have an emissions surplus if its exports embodied more CO_2 per unit of value than its imports. Second, there are equilibrating forces at work in goods markets to ensure countries cannot remain net goods importers or exporters in the long run (even if recent experience suggests imbalances can be prolonged and large). By contrast, these adjustments are impotent in the emissions trade as there are no international mechanisms to enforce settlement of 'loaned' emissions. Additionally, intertemporal balancing of trade accounts need not imply balancing of emissions accounts, since technological advances imply future production will be less carbon-intensive. So a country may be a net emissions importer without ever having to be a net emissions exporter.[3] Third, while there are well-developed accounting systems for valuing

[3] Since an emissions deficit in one country is a surplus for another country, this cannot imply a reduction in global emissions, but does affect the distribution between countries.

Figure 1: Components of emissions embodied in trading relationships



the level of trade, measuring the emissions embodied in goods along a global value chain is still a nascent discipline.

(ii) The components of different emissions accounting bases

If trade in emissions does not coincide with trade in goods, it is important to understand how traded emissions can be estimated. Figure 1 illustrates the components of emissions embodied in trading relationships. On a production basis—the prevailing Kyoto methodology—emissions are attributed to countries on the basis of territory; all emissions from China's *domestic* production, labelled *Y,* are included. Domestic production includes goods exported for foreign consumption, *X*, and the emissions associated with their production are the emissions embodied in exports. Symmetrically, imported goods M^C, though consumed domestically, embody emissions from production processes that take place abroad.

In evaluating emissions on a *consumption* basis, we mean the emissions embodied in the complete production process of goods consumed by an entity, regardless of the geographical location of production. As Figure 1 illustrates, in moving from the production to the consumption account, it is therefore necessary to subtract emissions embodied in exports and attribute them to recipient countries (in this example, the USA, as the largest export partner), while adding emissions embodied in imports (in this example, Japan, as the largest import partner).

The principal complication illustrated in Figure 1 is that some imports, M^X, may be inputs to domestic production of goods that are subsequently exported. We describe this as the 'processing trade' and attribute the emissions embodied in these imports (and any additional emissions embodied in their processing for export) to the country consuming the final exports (the USA in this case). Hence, imports for consumption must be included in the consumption account, but imports for the processing trade must be excluded.

When dealing with many sectors of the economy, estimation of both accounting bases must combine data from input–output tables with emissions-intensity data. We explain the measures algebraically below.

In goods terms, the output vector Y_i of any sector i can either be used as an input to another sector j, forming the matrix $Y_{ij,}$ or, for final use, forming the vector $Z_{i,}$ which includes consumption, investment, and exports. Final use of all goods, excluding imports, is then represented by Z. This allows us to write sectoral domestic output as the vector $Y_i = \sum_{j=1}^{n} Y_{ij} + Z_i = \sum_{j=1}^{n} a_{ij} \cdot Y_i + Z_i$ where the matrix $a_{ij} = \frac{Y_{ij}}{Y_i}$ is the direct use coefficient. The Leontief Matrix, A, of a_{ij} represents the economy-wide production function. Total domestic output is then given by the scalar $Y = (I - A)^{-1} Z$, where $(I - A)^{-1}$ is the Leontief inverse matrix.[4]

We define the *direct* unit emissions intensity of production processes within a sector as the vector $S_i = \frac{E_i}{Y_i}$ (where E_i represents aggregate sectoral emissions). The Leontief inverse matrix can then be used to construct the *total* unit emissions intensity vector $\hat{S} = S \cdot (I - A)^{-1}$, taking into account embodied emissions in the upstream value chain.

On a production basis, emissions are measured as $E^P = \hat{S} \cdot Z$: total emissions intensity per unit of output multiplied by output for final use. Note that E^P includes emissions from production for export but excludes emissions embodied in imports.

There are two complications in extending this model to the consumption accounting basis. First, goods exports $X = \sum_1^n \sum_1^G X_{ig}$ and imports $M = \sum_1^n \sum_1^G M_{ig}$ for each sector i are assessed over G countries.[5] The gross emissions embodied in exports are given by the scalar $E_X = \hat{S} \cdot X$. However, to get an estimate of the exported emissions *from domestic production* it would be necessary to subtract imported goods that make up the processing trade. This would be achieved using the import coefficient matrix $N_i = M_i/(Z_i + M_i - X_i)$ to obtain the vector $\hat{S}' = S \cdot (I - (I - N)A)^{-1}$, which we term the *total domestic* unit emissions intensity. Thus, our estimate of exported emissions from domestic production would be the scalar $E_{X'} = X \cdot \hat{S}'$. However, in the absence of sectoral level data on the break-down between imports used for the processing trade and the proportion of export value that this accounts for, we use the gross measure E_X. While this is a limitation of the analysis, inducing over-estimation of exported emissions, the magnitude of the error is limited by the concentration of exports in sectors such as textiles that are only partially dependent on the processing trade. Additionally, the bias is counteracted by the re-importation of some goods into China, which may be wrongly incorporated at foreign rather than Chinese emissions intensity.

Second, since imports arrive from many countries with varying emissions intensities of production, an accurate estimate of imported emissions would be the scalar $\hat{M} = \sum_1^G \sum_1^n S_{ig} \cdot M_{ig}$. However, sectoral-level emissions-intensity data for every trade partner are not readily available. Some studies of ecological footprints, for example Li Hong *et al.* (2007), have made the simplifying 'import substitution' assumption that the emissions intensity of foreign production is equivalent to domestic production, such that $\hat{M}^* = \sum_1^n S_i \cdot M_i$. This approach fails to capture potentially important national differences in both the energy intensity of foreign production and the carbon intensity of energy consumption. The compromise we adopt here is to assume that the national average emissions intensity is representative of that country's exported goods, so $\hat{M}' = \sum_1^G S_g \cdot M_g$. Imported emissions are then represented by the scalar $E_{M'} = M' \cdot \hat{S}$. The limitation of this approach is that bilateral trade is often concentrated in particular sectors which may be more or less intensive than the national average. Specialization according to comparative advantage would reduce the risks of divergence, but in practice such specialization is incomplete.

[4] I represents the identity matrix.

[5] Note that M here includes all imports, whether for domestic consumption or the processing trade.

Drawing these arguments together, emissions measured on a consumption basis can be expressed as the scalar $E^C = E^P - E_X + E_{M'}$. The difference between the production and consumption estimates, the scalar $E^B = E^P - E^C$, represents the BEET.[6] This is our estimate of emissions that take place in Chinese territory but are *not* attributable to Chinese consumption. Equivalently, it is a measure of emissions attributable to foreign consumption.

(iii) Decomposing changes in the BEET

Whether a country has a BEET in deficit or surplus depends on whether the goods it consumes embody more or less emissions than the goods it produces. To understand the economic *causes* of any imbalance, we extend Copeland and Taylor's (1994) decomposition of changes in production emissions to more than one polluting sector. Produced emissions by sector are represented by the vector $E_i^P = \hat{S}_i \cdot Z_i$ and both vectors S_i and Z_i can be rewritten to reflect economy-wide values weighted by the share of the sector in emissions, $\hat{s}_i$, and output, z_i, such that $E_i^P = (\hat{S} \cdot \frac{\hat{S}_i}{\hat{S}}) \cdot (Z \cdot \frac{Z_i}{Z}) = (\hat{S} \cdot \hat{s}_i) \cdot (Z \cdot z_i)$. We rewrite $\hat{s}_i \cdot z_i = d_i$, which is the weighted average share of output of a sector, where the weights are the relative emissions of the sector. The vector d_i serves as an index of the concentration of the economy in relatively emissions-intensive or low-emissions activities. Then $E_i^P = \hat{S} \cdot d_i \cdot Z$ for a sector, and $E^P = \hat{S} \cdot d \cdot Z$ for the economy. This can be approximated in differential form as $\Delta E^P = \Delta\hat{S} + \Delta d + \Delta Z$.

This shows that a change in production emissions can be decomposed into (i) a technique effect from changes in the emissions intensity of production; (ii) a composition effect, reflecting the share of 'dirty' versus 'clean' sectors in total output; and (iii) a scale effect from the growth or contraction of the economy. Each effect assumes all other factors are held constant; for instance, if emissions intensity and the size of the economy are stable, production emissions may still increase if 'dirty' sectors—those that have higher emissions intensities than average—are expanding, while 'clean' sectors are contracting.

The same decomposition can be applied to consumption emissions. The three components remain, but their interpretation now refers to changes in the technique, composition, and scale of *consumed* goods. We argue, therefore, that the BEET will evolve to reflect the *differences* between production and consumption in a country along these three effects.

(i) *Technique effects*: Progress made in reducing emissions intensity in domestic industry may differ from other countries. A rise in the BEET may therefore reflect faster technical progress in abatement by import partners relative to domestic production.

(ii) *Composition effects*: If domestic production is shifting towards emissions-intensive sectors, while consumption goods maintain a relatively stable emissions intensity, the BEET will grow. Trade facilitates this decoupling of production and consumption, and comparative advantage suggests that specialization is likely to push an economy towards concentration in particular sectors that may be above or below the emissions intensity of consumption.

(iii) *Scale effects*: The BEET will be growing where the scale of production is increasing faster than the scale of consumption. This situation represents a growing balance-of-trade surplus. So while trade imbalances in the goods and emissions contexts need

[6] See Muradian *et al*. (2002).

not coincide, a trade surplus in goods makes it more likely that a country will have a surplus BEET.

A surplus BEET must be offset by a deficit elsewhere, and the above effects therefore describe the *distribution* of emissions between countries. However, the literature has also identified potential reasons for aggregate changes in emissions arising from trade. First, if trade shifts out the global production possibility frontier, the increased economic activity may have a 'global scale effect', boosting both production and emissions. Offsetting this expansion of emissions there may be an income effect that increases the demand for low-emissions production. This depends on an endogenous policy response and the turning point of the relationship, labelled the Environmental Kuznets Curve (EKC), varies significantly between countries and pollutants, and also depending on the 'deep' determinants of trade, such as factor endowments and distance between markets (Brock and Taylor, 2004).

Second, composition effects need not be zero-sum; while trade could lead to specialization by one country in emissions-intensive production, it could also allow each country to specialize in the goods it produces most efficiently. As Hayami and Nakamura (2002) found for trade between Japan and Canada, 'global composition effects' may reduce both countries' emissions, with Japan exporting manufactured goods it produces at very low energy intensity and Canada exporting energy-intensive products using energy from hydroelectric power with a very low carbon intensity. Such efficiency gains are more likely where emissions are appropriately and universally priced, so that specialization takes into account a country's carbon efficiency. Where there are asymmetries between countries' environmental policies, the 'pollution haven effect' may arise. In this case, the location of dirty industries is determined by lax environmental policy and not just comparative advantage. Not only could this concentrate dirty industry in particular countries and increase their BEET; it could also produce an aggregate increase in emissions by undermining the global composition effect. In practice, variation in labour costs, political risk, and the stage of industrialization (Pan, 2008) are likely to be overwhelming influences on locational decisions, but we merely wish to argue that trade plays an ambiguous role in shaping both the distribution and total level of emissions, particularly where environmental policies are not harmonized.

Third, trade facilitates a diffusion of technology that can produce a 'global technique effect' as best practices diffuse, and may even spur greater technological progress.

The net impact of these effects is ambiguous; free trade is neither inherently good nor bad for the environment, and changes in a country's BEET depend on patterns of trade that are shaped by comparative advantage and national economic policies.

III. China's emissions embodied in trade

(i) Data sources

The key data for estimating China's emissions using an input–output methodology are the *Input–Output Tables of China in 2002* (NBS, 2006). We use energy consumption data from the *China Statistical Yearbooks* (NBS, various years) and carbon intensity data from the International Energy Agency (IEA) and the World Resources Institute (WRI). In the absence of comprehensive data on energy sources, we assume that the carbon intensity of energy use for exports is the same as for domestic production and the same across sectors. Data on trade in

Table 1: Selected energy-intensity measures by sectors in China, 2002

	Direct energy intensity (tce/10,000 yuan)	**Total energy intensity (tce/10,000 yuan)**
Farming, forestry, animal husbandry, fishery	0.23	0.80
Manufacture of textiles	0.33	1.54
Extraction of petroleum and natural gas	1.38	1.90
Smelting and pressing of ferrous metals	1.71	3.45
Raw chemical material and chemical products	1.38	3.07
Electronics and communications equipment	0.06	1.17
Average energy intensity in all 43 sectors	**0.42**	**1.08**
Average energy intensity in 37 traded sectors	**0.57**	**1.13**

goods are sourced from the UN Commodity Trade Statistics. Matching data sources required the classification of 122 sectors in the input–output tables and the Standard International Trade Classification (SITC) into the 37 traded sectors in the energy consumption statistics.

(ii) Energy-intensity calculations

Following the methodology in section I, Table 1 details the direct (S) and total ($\hat{S}$) energy intensities for a selection of sectors and across the economy in 2002, the same year for which we have comprehensive input–output tables. It illustrates the increase from direct to total energy intensity as upstream activities' embodied energy is included. The differences vary by sector, with primary-sector activities, such as the extraction of petroleum, showing little variation and downstream sectors with long production chains, such as electronics and communications equipment, showing much greater variation.

The national average energy intensity derived by this method across all 43 sectors is 1.08 tonnes of coal equivalent (tce)/10,000 yuan RMB, which is identical to official estimates.[7] Reflecting the higher energy intensity of traded sectors over non-traded services, the average energy intensity in the 37 traded sectors is 1.13 tce/10,000 yuan.

Our assumption that the carbon intensity of energy use is the same across all sectors means the same pattern characterizes carbon intensity. The value for Chinese carbon intensity per unit of energy we adopt is the 2002 figure of 2.13 tonnes of CO_2 per tonne of coal equivalent.[8] This figure, and all the estimates we present are based on CO_2 emissions alone and not a broader measure of greenhouse gases.

(iii) Emissions embodied in exports

Calculating the emissions embodied in exports is simply a case of combining the total emissions intensity with the value of exports in each sector. Here, for consistency with the input–output tables, we provide the estimates only for 2002; time-series estimates are discussed in section II (v). In 2002, China's exports totalled $326 billion, embodied energy of

[7] *China Statistical Yearbook 2006* (NBS, various years). Note that there are nevertheless sectoral differences between our estimates and reported figures because we estimate intensity per unit of final demand while official statistics are based on unit of value added.

[8] This conversion rate is based on CAIT (Climate Analysis Indicators Tool, an information and analysis tool on global climate change developed by the WRI, cait.wri.org). The conversion factor between toe and tce is approximately 1:1.43.

Table 2: Emissions embodied in exports, 2002

	Export volumes (%: sector value of exports/total exports)	Emissions embodied in exports (%: sector total emissions/total export emissions)
Manufacture of textiles	17.41	13.41
Smelting and pressing of ferrous metals	1.02	2.32
Electronics and communications equipment	11.80	12.52
Raw chemical material and chemical products	3.53	7.13

Table 3: Recipients of exported emissions, 2002

Recipient	Chinese export volumes (%: country value of exports/total exports)	Emissions embodied in Chinese exports (%: country total emissions/total export emissions)
USA	21.49	20.64
HK (SAR)	17.96	17.81
Japan	14.88	14.12
South Korea	4.76	4.97
Germany	3.49	3.41
Netherlands	2.80	2.82
UK	2.48	2.50
Australia	1.41	1.44
Canada	1.32	1.33
Russia	1.08	0.95
Total for top 10	71.66	70.00

410 mtce, and embodied emissions of 880 mt CO_2 (million tonnes of CO_2). Domestic energy embodied in exports therefore amounts to approximately 28 per cent of total Chinese primary energy consumption and exported emissions around 24 per cent of production emissions.

Table 2 shows the sectoral share of exports in volume and total emissions. As expected, emissions-intensive sectors, such as raw chemical materials, are responsible for a larger proportion of emissions embodied in exports than their share in the value of exports. High-value but low-emissions sectors, such as electronics and communications equipment, are responsible for a smaller proportion of exported emissions. Interestingly, this sector has even less responsibility when imports are removed, reflecting the importance of the processing trade to cleaner sectors. ‘Dirty’ sectors, such as smelting and pressing of ferrous metals, rely predominantly on domestic inputs of raw materials.

The top ten recipients of these exported emissions, who account for over 70 per cent of total exports, are listed in Table 3. In most cases, embodied emissions correspond closely to export volumes. However, minor differences can arise owing to the structure of a country’s imports; South Korea’s imports of around 4.97 per cent of exported emissions are higher than its share of export volumes because it receives around 16.9 per cent of Chinese exports of emissions-intensive non-ferrous metals. The USA is the largest importer of both Chinese goods and emissions, closely followed by Hong Kong SAR (Special Administrative Region),

Table 4: Imported goods, energy, and emissions, 2002

	Imports volume (%)	Energy embodied in imports (%)	Emissions embodied in imports (%)
Japan	18.11	5.27	5.28
Taiwan, China	12.89	10.74	11.52
South Korea	9.67	9.08	8.62
USA	8.98	5.43	5.82
Germany	5.92	2.80	2.93
China (reimport)	5.07	11.49	14.21
HK (SAR)	3.62	0.92	0.90
Malaysia	3.15	4.61	4.51
Russia	2.85	16.26	15.78
Indonesia	1.52	3.62	2.97
Total for top 10	71.78	70.22	72.54

although in the latter case one would expect the vast majority of goods to be re-exported.[9] This illustrates the importance of adopting a global approach to any assessment of trade in emissions, since if these re-exports from Hong Kong were destined for consumption in the USA, then the energy ultimately embodied in Sino-US trade would be even greater than the estimate made here.

(iv) Emissions embodied in imports

It is straightforward to calculate imported emissions according to the simplistic import-substitution approach, where the energy intensity of imports is assumed to be the energy intensity of domestic production. Our estimate on this basis of 440 mtce of imported embodied energy is slightly higher than the exported value of 410 mtce, suggesting China is a net *importer* of embodied energy. We illustrate below that simply by accounting for differences in the average energy intensity of trade partners, this story is dramatically reversed.

Estimates of total energy intensity in each import partner are taken from primary energy intensity per unit of GDP.[10] We examine the top 32 import partners (those exporting over $1 billion to China in 2001), which account for 93.4 per cent of total imported value. Assuming that these 32 partners also accounted for 93.4 per cent of imported energy and emissions, we infer that the total energy embodied in imports is around 170 mtce. On the basis of emissions embodied in traded goods, China is then a net energy *exporter* of 240 mtce. Total imported emissions are estimated at 257 mt CO_2.

Table 4 illustrates the share of import volumes, energy, and emissions for the top ten import partners. The degree of variation is striking compared to the export partners. One cause is the greater diversity in emissions intensity for import partners, while all exported energy is produced at the same (Chinese) emissions intensity. Additionally, as theories of comparative advantage and specialization suggest, it is likely that while the structure of Chinese exports is relatively constant across export partners, the structure of its *imports* and hence their emissions content, is likely to vary much more sharply. Accordingly, while Japan is the

[9] Hong Kong operates an independent trading system, but since April 2003 has been a party to the UN Framework Convention on Climate Change (FCCC) as a part of China.

[10] We attribute this energy intensity to the full value of imported goods, assuming away any role for the processing trade in the country of origin. These linkages are hard to trace for single-country studies, but would emerge naturally from a comprehensive study that combined the input–output tables of all countries.

Table 5: China's balances of trade with key partners, 2002

	Balance of trade in:		
	Goods ($ billion)	**Embodied energy (mtce)**	**Embodied emissions (mt CO_2)**
Australia	−1.26	3.31	7.86
Russia	−4.88	−25.69	−32.35
Canada	0.67	3.21	9.05
HK (SAR)	47.8	70.90	153.10
South Korea	−13.04	5.11	21.13
UK	4.72	9.46	20.63
Netherlands	7.54	10.95	23.90
Japan	−5.03	48.94	109.65
Germany	−6.09	9.16	22.19
USA	43.48	75.24	165.14
Total	31.01	241.74	623.02

largest import partner by volumes, the energy and emissions embodied in its goods represent a much smaller share of imports owing to both the relatively 'clean' nature of the imported goods and the relatively low emissions intensity with which these were produced. By contrast, Russia accounts for less than 3 per cent of the value of imports, but the concentration of these in raw materials and the high emissions intensity of their production make it the largest source of imported energy and emissions.

(v) The balance of trade in emissions

It was noted earlier that the import-substitution approach implies China is a net importer of energy. However, subtracting our methodology's estimate of imported energy of 170mtce from the exported estimate of 410mtce suggests that China is a net *exporter* of some 240mtce of energy, around 16 per cent of its total energy consumption. The same is true of the balance of emissions: subtracting total imported emissions of 257 mt CO_2 from total exported emissions of 880 mt CO_2 suggests China was a net exporter of approximately 623 mt CO_2, about 19 per cent of its production emissions in 2002. Net exports to the USA alone account for 165.1mtCO_2, about 5 per cent of China's reported production emissions in 2002. Attributing these emissions to the USA would have increased US emissions by 2.6 per cent in 2002.

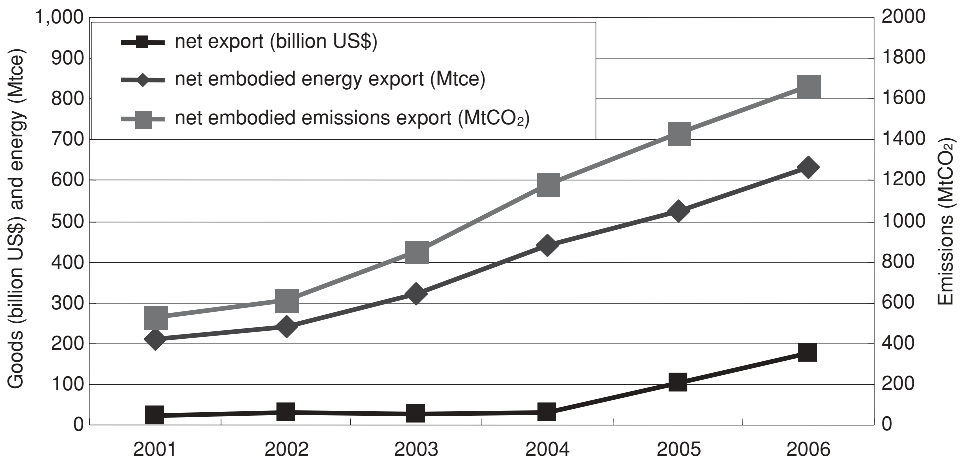
Table 5 illustrates the geographical distribution of these flows of goods, energy, and emissions. In all but one case, China is running an energy and emissions surplus. With Russia, its deficit reflects the import of high emissions-intensity raw materials and the export of comparatively 'cleaner' goods at lower emissions intensity.

(vi) The balance of trade in emissions over time

Input–output tables are estimated only every 5 years, so to conduct a time-series analysis we assume that changes in national energy intensity apply equally to all sectors. It is also necessary to make adjustments for exchange-rate movements over time, although this is simplified by the pegging of the yuan against the dollar until 2005. The methodology for assessing imports is unchanged when looking at the time-series data.

Figure 2 illustrates the balance of trade in goods, embodied energy, and emissions (the BEET) between 2001 and 2006. All are in surplus and rising rapidly, with emissions trends closely tracking changes in embodied energy.

Figure 2: China's balances of trade



To a large extent, growth in exports has driven both the trade surplus and associated energy and emissions surpluses. In 2006, exports reached $969 billion, a 27 per cent increase on the previous year, while imports stood at $792 billion, a 20 per cent increase on 2005. The share of exports in GDP has grown from 24.4 per cent in 2001 to about 40 per cent in 2007. However, the BEET rose substantially in 2001–4 when the trade balance was stable, which leads us to believe that composition and technique effects, rather than just scale effects, are important.

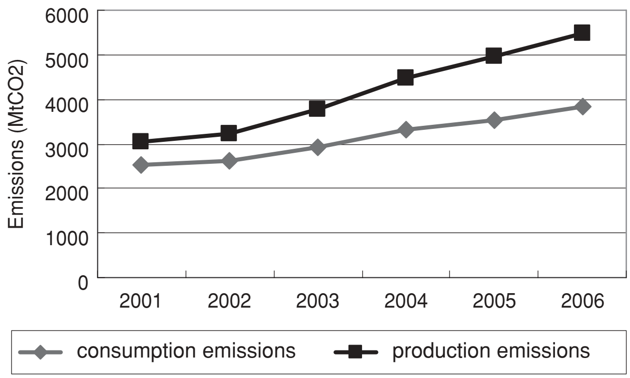
(vii) Emissions on a consumption basis

In this section we compare China's emissions on a *production* basis and a *consumption* basis. The current Kyoto figures reflect the production basis, and are obtained from the World Resources Institute and estimates for 2005/6 from the Netherlands Environmental Assessment Agency. Our methodology allows us to estimate emissions on a consumption basis by subtracting the BEET, $E^C = E^P - E^B$.[11] The results are displayed in Figure 3. For 2006, produced emissions were around 5,500 mt CO_2. Subtracting the 1,660 mt CO_2 BEET surplus implies consumption emissions of 3840 mt CO_2, some 30 per cent lower.[12] Just as importantly, the difference has grown over time, suggesting that if we are even partially interested in consumption measures of emissions responsibility, the production accounting method is becoming increasingly misleading. From 2001 to 2006, production emissions have increased from 3,050 mt CO_2 to 5,500 mt CO_2, indicating that 47 per cent of the growth in production emissions between 2001 and 2006 is due to the

[11] As noted earlier, there are many components to a consumption account, of which the emissions embodied in trade estimated here are only one. Others include transportation and tourism.

[12] According to WRI CAIT, the emission factor in 2006 is 0.86tC/toe, larger than the figure of 0.83tC/toe in 2002. Using WRI emissions factor, total emission of CO_2 from fossil-fuel combustion is estimated at 5,500 mt CO_2, which is lower than the figure 6,200 mt given in the study by the Netherlands Environmental Assessment Agency (2007). Please note our figure does not include emission from industrial processes, such as cement production and methane.

Figure 3: China's emissions on different accounting bases



increase in the BEET, with the remaining 53 per cent reflecting increased levels of Chinese consumption.

(viii) Comparison with other estimates

Wang and Watson (2007) conducted a similar analysis and estimate the net export of emissions from China in 2004 at around 1,109 mt CO_2, above our estimate of 748 mt CO_2. This is surprising, since Wang and Watson only undertook an analysis of direct energy intensity in traded goods, rather than the total energy intensity (including upstream inputs) considered here.

Ahmad and Wyckoff's (2003) comparative study estimated China's trade emissions surplus at around 12 per cent of production emissions in 1995. Given the growth in export volumes since that time, our estimate of 19 per cent in 2001 and 30 per cent in 2006 is not inconsistent with this estimate. The study incorporated detailed data, including Chinese input–output tables and country-specific emissions-intensity figures. However, its time-series estimates assumed unchanged energy technologies.

Shui and Harriss (2006) focused on USA–China trade. Their methodology adjusts for national differences in the fuel mix of energy production, but assumes the energy intensity of Chinese production is the same as in the USA, omitting the influence of different technology levels. Given that Chinese energy intensities are, in practice, higher, we would expect this method to provide underestimates. However, they estimate gross exports to the USA at 449 mt CO_2 in 2002, which is much larger than our own estimate of 167 mt CO_2. The principal reason is that the authors use a purchasing power parity (PPP) adjustment to capture the fact that 'the same dollar value of a US product and a Chinese export in the same/similar category can represent different quantities of merchandise produced in each country'. So the use of PPP exchange rates goes some way to capturing the higher energy use per traded dollar of output in China—simply because more goods must be produced in China for \$1m worth of exports than for \$1m worth of US production. The adjustment is necessary because US energy intensity is clearly a poor proxy for Chinese energy intensity under a very different industrial structure. Yet, it is a blunt adjustment, since PPP measures capture differences in the *prices* of non-tradable inputs, which need not be directly related to differences in energy used to produce a particular dollar value of exports. Our use of direct energy-intensity figures is therefore preferable and avoids the need for (notoriously unreliable) PPP adjustments.

Li Hong *et al*. (2007) provide results which contradict our own, suggesting that China has consistently been a net *importer* of embodied energy since 2000. While their data rely on earlier input–output figures, from 1997, and they assume constant energy intensity across time, the crucial difference is the adoption of the import-substitution approach to assess the energy intensity of imports. The approach is chosen because the authors' objective is to assess the ecological footprint of China and the impact of trade on energy use, rather than to quantify real energy flows.

(ix) Decomposing changes in the BEET

In section I we showed that changes in the BEET can be decomposed into scale, composition, and technique effects.

Scale effects are unambiguous. Chinese nominal GDP has grown at an average annual rate of 13.7 per cent p.a.—10 per cent in real terms—between 2001 and 2006. So it is not surprising that both production and consumption emissions show an upward trend. Moreover, the BEET has risen because production growth has outpaced consumption growth. This is directly reflected in the growing balance-of-trade surplus, which has risen from $22 billion in 2001 to $177 billion in 2006. For 2007 there has been a further jump to $262 billion, 48 per cent higher than the previous year.

Composition effects are harder to detect, but the data suggest that there has been a gradual change in the sectoral composition of exports, most strikingly away from textile and clothing, which made up 22.69 per cent of exports in 1998 and only 13 per cent in 2006, towards electronics and communications, which has risen from 6.05 per cent of exports in 1998 to 12.76 per cent in 2006. Given the greater energy intensity of textiles manufacture, this is significant. However, in the most intensive sectors, such as ferrous metals, there has been a gradual increase in exports, from 2 to 4 per cent, which has offset this trend. Rosen and Houser (2007) document an economy-wide shift from light to heavy industry. We are not aware of any estimates of composition effects arising from emerging consumerism, although Rosen and Houser note that any trend towards carbon-intensive activities, such as vehicle ownership, is extremely recent and limited to wealthier coastal provinces.

Technique effects are assessed at the national level, and it is apparent that energy efficiency has contributed significantly to a reduction in energy-intensity figures. IEA data show that world total primary energy supply per unit of GDP has only decreased slightly from 0.365 kgoe/$US (kg of oil equivalent, 2000 prices) in 1990 to 0.315 kgoe/$US in 2005, while for China the fall has been from 1.941 kgoe/$US in 1990 to 0.908 kgoe/$US in 2005.[13] Had it not outpaced world efficiency improvements, China's BEET would have been even higher. However, more recently the trends may have become adverse. China's energy intensity has, in fact, risen from 0.844 kgoe/$US in 2002 while world trends have been stable. Garnaut *et al*. (2008, this issue) revise IEA projections upwards on the basis that this trend is likely to persist. Moreover, the carbon intensity of energy use has been rising, from 3.03 t CO_2 per tonne of oil equivalent (toe) in 2000 to 3.23 t CO_2/toe in 2004.[14] As Garnaut *et al*. explain,

[13] If purchasing power parity (PPP) is used, the figures for China would be close to world averages. For example, in 2005 the world average was 0.209 kgoe/$US PPP and 0.219 kgoe/$US PPP.

[14] WRI CAIT.

this reflects a growing reliance on coal-fired electricity generation and could have placed the BEET on an upward trajectory.

(x) Has Chinese trade caused an aggregate increase in emissions?

Our estimate of the growth of consumption emissions illustrates that, even abstracting from its export role, China's rapid economic growth has not been decoupled from CO_2 emissions. There is no evidence that China is anywhere near the downward-sloping part of the EKC. Yet, a 'global scale effect' is unclear because the counterfactual of Chinese growth in the absence of trade cannot be assessed.

There is some evidence of a global composition effect having increased aggregate emissions. The pure relocation of production from developed countries to China has increased emissions because Chinese heavy industry has, in static terms, a 20–40 per cent higher energy intensity than its OECD counterparts (Wan, 2006). Accordingly, Shui and Harriss (2006) find that emissions avoided in the USA owing to imported Chinese goods were 314 mt CO_2 in 2002, while China incurred significantly higher emissions of 449 mt CO_2 in the process of exporting to the USA. So even if only a fraction of industry relocates, carbon leakage to higher-intensity production locations may rapidly cancel out any reductions achieved in developed countries.

Finally, the evidence does not permit us to separate a 'global technique effect' from domestic efficiency improvements. We have seen that global efficiency gains have been limited in recent years, but it is plausible that the gains have been concentrated in countries such as China.

A formal evaluation of aggregate effects is more difficult because (i) it is unclear how to specify the relevant counterfactual and (ii) we do not know whether China could have raised living standards so sharply without opening to trade. If China had never undertaken its 'opening and reform', production may have been substituted either in developed countries, affecting both the mix of goods produced and national emissions intensity, or in other developing countries with potentially higher emissions intensities. In short, the scale, composition, and techniques of the world economy have all been endogenous to China's trade openness, making an assessment of its aggregate emissions impact difficult and uninformative. What is of interest is how we determine responsibility for emissions *given* the pattern of trade that has emerged.

IV. Allocating responsibility for emissions

(i) Responsibility for the existing pattern of emissions

The current international framework attributes responsibility for emissions to China on a production basis. However, China is not yet required to make binding emissions reductions and an alternative accounting basis could still be adopted in any post-Kyoto agreement. Given our decomposition of the various effects that have created a BEET surplus, the causes can be traced to the unique role that China has claimed in global trade. China has grown rapidly on the basis of a comparative advantage in relatively emissions-intensive goods consistent with its self-image as the 'factory of the world'. Garnaut *et al*. (2008) anticipate, contrary to the assumptions the IEA has made in its forecasts, further restructuring towards heavy industry in line with China's exceptionally high levels of investment. In turn, this role has been shaped by national and international policies.

At the international level, the absence of any global carbon-pricing framework has permitted trade patterns to develop without regard to environmental comparative advantage. Since appropriate carbon pricing would have encouraged a shift of production to countries with lower emissions intensities, current trade patterns are partly a reflection of global coordination failures.

By contrast, the liberalization of international trade has been a comparatively successful endeavour. Copeland and Taylor (2003) distinguish the 'pollution haven effect', owing to changes in environmental regulation, from the 'pollution haven hypothesis' that trade liberalization encourages relocation to places where production is dirtier. They find little evidence to support the 'pollution haven hypothesis'. However, trade liberalization could also affect the emissions pattern through the more potent force of comparative advantage. Grether *et al*. (2006) find evidence of significant changes in the pollution content of imports owing to differences in factor endowments, though they do not assess CO_2 emissions. This supports our argument that changes in the location of production can have strong distributional impacts on emissions, and may even affect aggregate emissions. However, the promotion of free trade has been a partnership between developed and developing countries, with developed countries sharing directly in the benefits through higher levels of consumption. The problem has not been trade *per se*, but that emissions related to trade have evolved autonomously from the negotiated Kyoto process.

A slightly more compelling argument is that China's *national* policies have artificially boosted its heavy industry and that it should therefore be responsible for the resulting emissions. Rosen and Houser (2007) lay the blame on microeconomic policies, including the granting of tax rebates; 'the abnormalities in costs and capital flows that have promoted energy intensive industry in China have altered the global distribution of production' (Rosen and Houser, 2007, p.35). The economic and environmental stresses of these policies have encouraged the government to repeal them. Since 2004, the government has taken extensive steps to reduce or eliminate these rebates. From July 2007, the tax rebate was specifically lifted from 553 energy- and pollution-intensive goods.[15] Perhaps most significantly, the artificial depression of the Chinese exchange rate has been explicitly motivated by the desire to encourage export-based growth. However, our analysis suggests that, in all these cases, responsibility for the resulting emissions need not automatically transfer to China.

Our decomposition of effects from trade underlines the complexity. To focus on bilateral trade with the USA, if the depressed exchange rate has attracted exporting industries and increased the scale of China's economy, or shifted its composition towards dirtier goods, then there would be a corresponding decrease in the scale of US production and a shift in its composition towards cleaner goods. The relocation of production is still meeting the same US consumer demand, and the counterfactual would have been the same pattern of consumption but with emissions produced in the USA.[16]

However, if relocation to China creates *additional* emissions relative to what would have occurred under US production, China may still bear some responsibility. The lower energy efficiency of Chinese industry may have created a global composition effect by replicating

[15] Based on the *Shanghai Securities Daily*, 22 June 2007.

[16] Of course, we are abstracting from transportation emissions which necessarily rise when production is relocated abroad.

US production at higher emissions intensities. Any boost to aggregate economic activity may also have produced a global scale effect that has increased emissions. Yet, the relocation may at the same time have improved the efficiency of Chinese industry through a global technique effect, with potential spillovers to other sectors, even if this trend has recently slowed.

Estimating these effects is difficult, but it may be possible to isolate a global composition effect using our data. Consider the net export of $43 billion of goods, 53 mtce of energy, and 167 mt CO_2 of emissions from China to the USA in 2002. If the same energy had been used in the USA at its domestic *carbon* intensity, emissions would have been 133 mt CO_2. If the same goods had been produced in the USA at its domestic *energy* intensity, emissions would have been 25 mt CO_2. So while the bulk of net exported emissions are attributable to more carbon-intensive and more energy-intensive Chinese production, some emissions would have been unavoidable.[17] While such an analysis is not a good basis on which to attribute responsibility—since the counterfactual is open to dispute and many other effects are omitted—it highlights the significant role that carbon leakage could play in boosting the emissions embodied in trade.

It is not only China's policies which have affected the pattern of emissions. At the same time as China has 'pulled' production within its own borders, Helm *et al*. (2007) argue that developed countries have 'pushed' dirty production abroad by undertaking complementary policies of deindustrialization. The UK's success in meeting its Kyoto targets and sustaining high levels of consumption have been premised on the possibility of displacing production of dirty goods to developing countries such as China. Low savings and budget deficits in the USA have also contributed towards a sustained trade deficit; just as the USA is consuming beyond its current income, it is consuming beyond its Kyoto footprint in emissions terms. In general, as part of a complex but rapid process of globalization, developed countries have been willing partners in the relocation of production and the growth of China's BEET, and they may even have benefited from 'abatement through trade'.

These arguments question the credibility of emissions reductions achieved by developed countries. On the one hand, if reductions have been premised on emissions increases in developing countries as industry has relocated, using the production account to allocate the burden of future emissions reductions may be unfair because 'easy' reductions have already been achieved by developed countries. One of the dimensions of equity referred to by Ashton and Wang (2003) is 'comparability of effort' and seems to rule out precisely this scenario in an equitable climate-change response. Additionally, developing countries may be unable to follow the same strategy when binding reductions are required; locked into their emissions-intensive comparative advantage, abatement may be disproportionately costly. On the other hand, the ability of developed countries to live outside of their carbon budgets by consuming emissions beyond their produced emissions implies that consumption has yet to be decoupled from emissions. Since there is not necessarily an equilibrating force in the BEET, the distributional transfer embodied in the BEET may dwarf offsetting financial transfers, such as the Clean Development Mechanism (CDM).

A related argument is made by Copeland and Taylor (2003) who emphasize that the EKC evidenced in developed countries might be an artefact of their encouraging dirty industry to relocate, rather than of domestic abatement efforts. If this is the case, 'even if an EKC

[17] The exercise is only a hypothetical one; in practice, had the USA produced these goods the structure of its economy would be altered and its energy intensity would be endogenous to the alternative pattern of trade and industrial structure.

exists for rich countries, the newly industrializing countries may not replicate the experience' (Copeland and Taylor, 2003, p. 22); dirty industry must be located somewhere, and if it is located in developing countries then eventual abatement investments will place a much larger burden on their economies. In the context of a global public bad, such as climate change, it is difficult to see how an equitable response can be created in the presence of such a fallacy of composition.

(ii) Responsibility for future emissions

While there is a strong case for giving at least some weight to a consumption basis when assessing historic emissions, accounting bases will have a more important role in shaping our ability to make future reductions. Especially as the cost of abatement rises, the scope for reducing emissions will depend not on their geographical location, but on tackling their *economic* causes. Allocating responsibility to producers or consumers directly affects the incentives for emissions reductions, the distribution of this burden, and its political feasibility.

Incentives and opportunities for abatement

A major advantage of the consumption basis is that it avoids international spillovers arising from trade, including both carbon leakage and 'abatement through trade'. To the extent that these have diluted environmental policy by displacing rather than reducing emissions, consumption accounts can help replace the pollution haven effect with a positive global composition effect that encourages carbon- and energy-efficient production in *each* country, just as it has between Japan and Canada. Additionally, as Peters and Hertwich (2008) stress, a consumption basis would solve allocation problems for international transportation, for which no one is currently responsible, and carbon capture and storage.

However, the value of attributing emissions to producers is that they are physically in control of emissions production and have the most information about feasible abatement opportunities. Just as in the management of risk, responsibility is usually best placed with the agent most able to control the outcome. A consumer basis would leave countries responsible for emissions at all points in a potentially long global value chain, but with no direct control over abatement. Indirect forms of consumer choice can be effective; just as individuals can choose between similar goods on the basis of both price and quality, countries can source their imports from countries with both low prices and low emissions. Indeed, Peters and Hertwich argue that this generates an intrinsic incentive for countries to transfer technology to their import partners, enhancing the CDM. Nevertheless, a natural policy counterpart to consumption accounts is the delegation of emissions responsibility to *individual* consumers in the form of personal carbon budgets (Pan, 2008). The conditions for this to be effective are stringent, with consumers requiring information on the emissions embodied in imported goods if they are to discriminate between foreign producers.

Distribution of burden

We have shown that China's emissions in 2006 would be 30 per cent less if measured by consumption. Even in 1995, Ahmad and Wyckoff (2003) concluded that OECD countries' emissions on a consumption basis were 550 mt CO_2 greater than on a production basis. We have argued that beyond the historic responsibility for emissions that developed countries bear, they may be responsible for a significant proportion of *current* emissions at present attributed to developing countries such as China.

A proliferation of equity concepts makes it difficult to assess what a fair distribution of the burden would require. Yet our analysis is not independent of these concepts; for example, proposals for a regime of 'contraction and convergence' could have very different implications depending on whether *per capita* emissions are assessed on a consumption or a production basis. Footloose global production will cause these indicators to continue to diverge. When the moment comes for China to make binding emissions reductions, the accounting basis used should be consistent with the economic role it plays in the global economy. If China continues to run both a balance-of-trade and a BEET surplus, its role in supporting consumption in developed countries while postponing its own consumption suggests that it would be unfairly penalized by using a production methodology. As Ashton and Wang (2003) stress, 'there are equity grounds for the proposition that those who receive the benefits from the emissions (or "embedded carbon") associated with the production of such goods should carry the cost'. This would be particularly important if current failures in the international environmental architecture reinforced the specialization of certain countries in emissions-intensive sectors; if countries are locked into these trade patterns, a shift to more 'efficient' international policies could entail high and concentrated burdens.

These arguments question the sovereignty sometimes attributed to the 'polluter pays principle'. This principle has become popular more for its simplicity and advocacy properties than its economic rationale. In a discussion of equity issues related to climate change, Ashton and Wang (2003) use the specific example of trade in carbon-intensive goods to stress the ambiguity of the principle. Where benefits and damage are spread widely, and in complex chains of economic causation, an individual 'polluter' cannot be determined merely by the location of emissions release.

Of course, in theory it is possible to separate out the allocation of emissions responsibility from the financing of abatement efforts. Transfers of technology and finance are likely to play a role in any post-Kyoto agreement; what our analysis shows is that if a production-based methodology is retained, these transfers would have to play a much larger role to compensate for the increased burden that developing countries such as China will face.

Political feasibility

A political barrier to consumption accounts is that countries would become liable for the dirty production techniques of their import partners, and switching options may be limited or costly. Set against this, however, Peters and Hertwich note that the consumption approach would address competitiveness concerns which have been a major barrier to previous international agreements. While the precise effects depend on the method of implementation, if UK consumers were required to make their consumption choices taking into account embodied emissions, UK firms would not be penalized any more than French or Chinese firms, whose goods the consumer can choose between. Indeed, this system creates a competitiveness boost for developed countries since emissions intensities are usually lower and domestic production inevitably minimizes transportation emissions. In this way, environmental performance would become an element of a country's comparative advantage. One issue in the competitiveness debate has been the value of border tariff adjustments to supplement domestic carbon pricing. By extending national responsibility for emissions up the value chain, a consumption methodology provides a more natural basis for countries to impose such adjustments on countries that fail to implement robust carbon pricing.[18] Tariff adjustments may even help overcome the practical difficulties in exercising control over foreign abatement noted earlier.

[18] However, the legality of border tariff adjustments remains unclear. See Deal (2008) for a summary.

Consumption accounting is also likely to enhance the scope to bring developing countries into an effective post-Kyoto framework. According to the IEA, China's production emissions will constitute 30 per cent of global emissions increases until 2030.[19] However, even on a production basis, *per capita* emissions are likely to remain below OECD averages, so participation by developing countries will need every encouragement.[20] A consumption methodology would both extend developed countries' control over emissions growth beyond their own borders and allow developing countries to grow into their responsibilities as their consumption rises.

A persistent challenge is perceived to be the difficulty of measuring emissions on a consumption basis. We have demonstrated that, by using readily available input–output tables and emissions-intensity data, an informative estimate can be produced. Many other national studies are now accumulating, including of India, Brazil, Australia, Vietnam, Thailand, South Korea, Spain, Japan, Finland, Norway, and Italy.[21] Crucially, there are increasing returns to the approach, since sectoral emissions intensities estimated for one country can be used to classify more accurately the emissions embodied in imports received by all other countries. While there are challenges, these are no longer insurmountable and, operating in parallel with production accounts, the improved understanding of emission flows would contribute greatly to our understanding of the drivers of emissions increases and the political economy of emissions reductions. Even if full consumption accounts proved intractable, estimating the BEET might facilitate adjustments to emissions accounts to reflect emissions-trade imbalances, mimicking the equilibrating forces present in the goods trade.

V. Conclusion

Estimating China's emissions on a consumption rather than a production basis both lowers its responsibility for CO_2 emissions in 2006 from 5,500 mt CO_2 to 3,840 mt CO_2 and reduces the growth rate of emissions from an average of 12.5 per cent p.a. to 8.7 per cent p.a. between 2001 and 2006. Emissions growth from China's transition to a consumer society has therefore been significantly slower than real income growth rates of 10 per cent.

China's role as a net exporter of goods has made it responsible under the Kyoto protocol for a large volume of emissions—1,660 mt CO_2—which support consumption abroad, primarily in developed countries. Conversely, developed countries' emissions have been lower than if they had continued to produce these goods domestically; for the USA, 2002 emissions would have been 2.6 per cent higher.

The magnitude of these differences is large and rising because (i) China runs a large and growing balance of trade surplus; (ii) China has a comparative advantage in relatively energy-intensive production (although contradictory trends in low-emissions electronics and

[19] IEA *World Energy Outlook 2006*, Summary, p.3.

[20] Standard *per capita* emissions measures are undertaken on a production basis and so fail fully to reflect equality in emissions *consumption* that they usually aim to express. For example, with a population of 1.3 billion, our analysis suggests that Chinese consumption emissions *per capita* would be 3.5 t CO_2 in 2006, compared to 4.8 t CO_2 on a produced-emissions basis.

[21] Lenzen (1998); Machado *et al*. (2001); IGES (2002); Straumann (2003); Mukhopadhyay (2004); Sánchez-Chóliz and Duarte (2004); Chung (2005); Mongelli *et al*. (2006); Nguyen and Keiichi (2006); Limmeechokchai and Suksuntornsiri (2007); and Mäenpää and Siikavirta (2007).

high-emissions raw materials may alter this); and (iii) China's emissions intensity of production remains high, with efficiency improvements stalling since 2001.

By taking account of total energy intensity in upstream production and changes in energy intensity over time, our analysis shows that consumption accounts are both feasible and informative. A limitation of our approach, and an appropriate starting point for further research, is that the role of the processing trade is not fully accounted for, and the bias may increase as global production becomes increasingly fragmented.

Our analysis is also informative for the energy market. In contrast to the findings of other studies and to popular perceptions, when the energy embodied in traded goods is taken into account, China is a net exporter of energy. This suggests that a more subtle interpretation of China's impact on commodity prices is required, since its energy hunger has been as much to meet global demand as for domestic consumption. None the less, a plateau in energy-intensity reductions will make controlling energy use a long-term challenge.

While appropriate counterfactuals are difficult to specify, it is possible that China's unique role in global trade has boosted global emissions. Yet this has been tightly bound up in the relocation of dirty industry away from developed countries. China's depressed exchange rate and export tax rebates may have played some role in attracting industry, although these policies have recently been diluted. At the same time, policies of deindustrialization in developed countries have pushed dirty industries abroad, while a lack of international coordination has failed to price emissions efficiency into industry's locational decisions. So while China may hold some responsibility for the additional emissions that its production has generated, the bulk of its emissions from trade have merely substituted for developed countries' production and supported their consumption. By allocating the full BEET to China's emissions account, the Kyoto Protocol fails to reflect the complexities of global trade and these distributional concerns. Indeed, reported Kyoto emissions performance may be a poor guide to the sacrifices that countries are making and the actual environmental impact of their consumption activities.

In this issue, Garnaut *et al*. (2008) stress the degree to which stabilization scenarios, even at 550 ppm-CO_2e, will require sharp reductions in the growth rate of emissions from developing countries. We have argued that the current production methodology creates leakages through trade that may do more to displace than to reduce emissions. This both reduces the efficiency of abatement and places a disproportionate burden of responsibility on developing countries. Just as importantly, it could also cast doubt on the credibility of the abatement efforts so far undertaken by developed countries and which have allowed them to sustain growing levels of consumption. At the very least, acknowledgement of countries' emissions embodied in trade could play an important role in bridging the gap between the concerns of developed and developing countries, and encourage the active participation of key players such as China in a post-Kyoto framework.

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