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International Trade and Global Greenhouse Gas Emissions: Could Shifting the Location of Production Bring GHG benefits?

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1 INTRODUCTION

For centuries, countries have produced goods for sale in other countries. Trade was once dominated by highly valued (and highly priced) commodities such as precious metals, high-quality textiles, tea, and spices, but now includes a huge variety of goods for general consumption, from housewares, electronics and clothing, to vehicles and construction materials. While the world has seen "globalization" before (Rodrik 2011), the current expansion is quantitatively and qualitatively different. In recent decades, trade has become a foundation of the world economy – exports now represent nearly a third of global GDP, more than double the share of just 30 years ago (World Bank 2011).

Many economists see trade as a significant source of economic growth and improved standards of living. In a prevailing view, trade enables each country to specialize in producing those goods for which it has a "comparative advantage". Thus trade can provide new revenues to producers, and lower prices to consumers, increasing incomes and purchasing power in both the producing and consuming countries (Irwin and Terviö 2002; Frankel and Romer 1999). Because of these benefits, the United Nations considers access to world markets as a critical step in the development of poorest countries (United Nations 2010). Trade liberalization is also a key part of the "Washington consensus" that has dominated thinking about development (Gore 2000), even as it has since become clear that "free trade" must be embedded in a web of regulations and institutions if it is to improve general welfare (Rodrik 2011).

Analysts have studied whether growth in trade leads to an environmental externality – an increase in global greenhouse gas (GHG) emissions. If increasing trade leads to greater economic activity, more goods will be produced, and GHG emissions will likely increase (Tamiotti et al. 2009). This has been called the "scale effect". However, increasing trade could also reduce GHG emissions, if countries that expand production of goods for export invest in newer, lower-carbon technologies or processes (the "technique effect"), reducing the GHG emissions intensity of producing these goods. Within a country, trade activity may also change the relative balance of activity in different sectors (the "composition effect"), resulting in an increase or decrease in that country's emissions.

The scale effect is the subject of considerable analysis (and debate) among economists. Understanding it requires assessing whether increased trade does, indeed, increase global economic activity; most studies have found that it does, and in that way also contributes to increases in global GHG emissions (Tamiotti et al. 2009). In this paper, we focus on the implications of the *composition* and *technique* effects, for which research results are less clear. Specifically, we assess whether trading more with some countries – those best-positioned to expand low-GHG production – could help reduce global GHG emissions, or at least help counteract the scale effect.

Our paper thus explores the relative average GHG intensity of production of selected goods in different world regions and the potential for regions to access low-GHG fuels and feedstocks needed to expand low-GHG production.[1] While a complete analysis of shifting trade patterns would assess the economic implications, including the scale effect, our simplified approach allows us to gauge what conditions might enable countries to be future low-GHG producers.

We begin by looking at the emissions embodied in trade (Section 2), based on a multiregional input-output model, to help identify significant trade flows for further analysis. Section 3 then examines differences in GHG-intensity among regions for some of the categories identified, while Section 4 asks whether and how shifting the location of steel production could reduce global GHGs. Section 5 assesses a range of national and international policies that could be used to shift trade patterns. Section 6 summarizes the results and identifies areas for further research.

1 This goal is different than for much of the existing analysis of changing trade patterns, which has focused on the (unintended) shifting of production activity between regions due to differences in carbon costs and the potential that these shifts could undermine the goals of the climate policies through emissions leakage. See, for example, the Carbon Trust's work on international carbon flows, http://www.carbontrust.com/resources/reports/advice/international-carbon-flows, as well as European Commission (2009), U.S. EPA et al.(2009), and Dröge et al. (2009).

2 GHG EMISSIONS EMBODIED IN TRADE

As trade has grown – overall and as a share of global economic output – so have the emissions associated with it. One recent analysis found that the emissions embodied in traded goods and services had increased from 4.3 Gt CO_2 in 1990, or 20% of global emissions, to 7.8 Gt CO_2 in 2008, or 28% of global CO_2 (Peters et al. 2011).

There are two prevailing methods for quantifying emissions associated with trade (Peters 2008; Peters et al. 2011). In one method, emissions are attributed to individual trade flows between pairs of countries or regions, regardless of whether the good or material is a final or an intermediate product. This method has been termed *emissions embodied in bilateral trade* in the literature, or "EEBT". The second method attributes all emissions to final products purchased by consumers, and includes all the emissions associated with producing a given product, regardless of where the emissions (including for intermediate products) were released. The second method relies on *multi-regional input-output modelling*, and so has been termed the "MRIO" approach.

To help understand the difference, consider, for example, a car made in Japan, using Chinese steel, and sold in the United States. The EEBT method would attribute the emissions in Japan to trade of cars with the U.S., and the emissions in China, to trade of steel with Japan. Under the MRIO method, all the emissions would be attributed to imports of cars into the U.S. Neither method is optimal for all contexts. MRIO can be more useful if the focus is on understanding the full life-cycle impacts of consumption of particular products, whereas EEBT can be more useful if the focus is on specific country pairs or on relatively homogenous, GHG-intensive, highly traded materials such as steel or aluminium.

Table 1 shows GHG emissions associated with consumption and production of goods and services in 2004, based on analysis by the authors using the MRIO

Table 1: Emissions associated with production and consumption of goods and services, by world region, 2004 (million tonnes CO_2e)

	Producing region											
Consuming Region	**North America**	**Europe**	**Japan**	**Oceania**	**Russia**	**China**	**India**	**Other Asia**	**Africa**	**South America**	**Subtotal: Traded Emissions**	**Total**
North America	6,246	218	58	40	74	553	56	290	107	170	1,566	**7,812**
Europe	286	5,030	48	39	305	488	76	347	263	186	2,038	**7,067**
Japan	87	47	969	39	21	234	10	161	32	23	654	**1,624**
Oceania	19	18	6	349	3	43	4	33	7	4	137	**486**
Russia	7	86	2	1	1,166	21	2	11	5	19	155	**1,321**
China	49	43	27	16	29	4,524	13	129	26	28	360	**4,885**
India	9	16	2	11	7	23	1,510	40	25	5	139	**1,648**
Other Asia	113	125	47	46	43	287	57	2,775	89	58	865	**3,639**
Africa	29	72	5	6	18	55	17	74	1,860	36	310	**2,170**
South America	61	41	4	4	13	46	5	30	15	1,430	219	**1,649**
Subtotal: Traded Emissions	660	665	199	201	513	1,751	241	1,116	568	529	6,442	
Total	**6,906**	**5,695**	**1,168**	**550**	**1,679**	**6,276**	**1,750**	**3,890**	**2,428**	**1,959**		**32,301**

Source: Authors' analysis, using EUREAPA approach (Hertwich and Peters 2010). Note: This table reports embodied emissions, and excludes emissions associated with final (e.g., household) consumption of fuels, e.g. for home heating or vehicle use. Global emissions in 2004 were 37 Gt CO_2e (World Resources Institute 2011).

approach of the EUREAPA model.[2] By this accounting, the emissions associated with inter-regional trade total about 6.4 GtCO_2e in 2004.[3]

The table shows a large share of the emissions embodied in trade (almost half) goes from developing to industrialized countries. The four largest inter-regional flows are from China to North America (553 Mt CO_2e), China to Europe (488 Mt CO_2e), Other Asia to Europe (347 Mt CO_2e), and Other Asia to North America (290 Mt CO_2e). North America, Europe and Japan all have considerably more emissions associated with imports than with exports.

Table 2 shows the emissions associated with different categories of final goods and services, also based on an analysis with the EUREAPA MRIO model (Hertwich and Peters 2010). These types of goods and services, such as food, electronics, or international transport (air travel) are purchased directly by end consumers. The igures in Table 2 represent the full, embodied, or "life cycle" emissions associated with these final products, including emissions associated with raw materials and intermediate products. For example, this table includes all the emissions associated with vehicles purchased in the U.S. and made in Japan, including emissions associated with production of raw and component materials, regardless of where produced (e.g., steel from China). This table does not include, however, emissions associated with vehicles purchased in the U.S. and made in the U.S.

Table 2: Emissions associated with consumption of internationally traded final products, by type of good or service, 2004 (million tonnes CO_2e)

Product category	2004 emissions
Food and agriculture	620
Machinery and equipment	502
Clothing and textiles	489
Electronics	403
Plastic / rubber products	301
Vehicles and parts	289
Other products	563
Services	287
Transport	285
Total traded as final products	**3,739**

Source: Authors' analysis, using EUREAPA (Hertwich and Peters 2010).

Emissions associated with trade of materials, such as steel or aluminium, may also be significant, but are not itemized in Table 2 because they are not final products themselves and are instead included within the other categories (e.g., steel used in vehicles).[4] In Table 3 we estimate the GHG emissions associated with trade in the top five energy-consuming material categories, using physical trade statistics (UN Statistics Division 2011) and estimates of emissions intensity drawn largely from the International Energy Agency (IEA 2007).[5]

As can be seen by comparing Tables 2 and 3, the emissions embodied in some materials can approach or exceed the levels of certain types of final products. For example, an estimated 600 million t CO_2e were associated with steel traded internationally in 2004, on par with the emissions associated with all traded food and agricultural commodities (620 million t CO_2e).

These findings suggest several categories of final products and raw materials that are good candidates for exploring ways to reduce GHGs associated with trade. For example, the final products *food and agriculture*, *clothing and textiles*, *electronics*, and machinery and equipment, as well as the raw materials *steel* and *chemicals* are each responsible for about 1% of global GHG emissions in 2004. The following section explores differences in the GHG intensity of some of these products and materials to explore whether shifting where they are made could reduce global GHG emissions.

2 See https://www.eureapa.net, as well as Hertwich and Peters (2010).

3 The total would be somewhat higher, about 8.4 GtCO_2e in 2004, if the table measured all flows between individual countries and did not combine some regions – e.g., trade among European countries (Davis and Caldeira 2010).

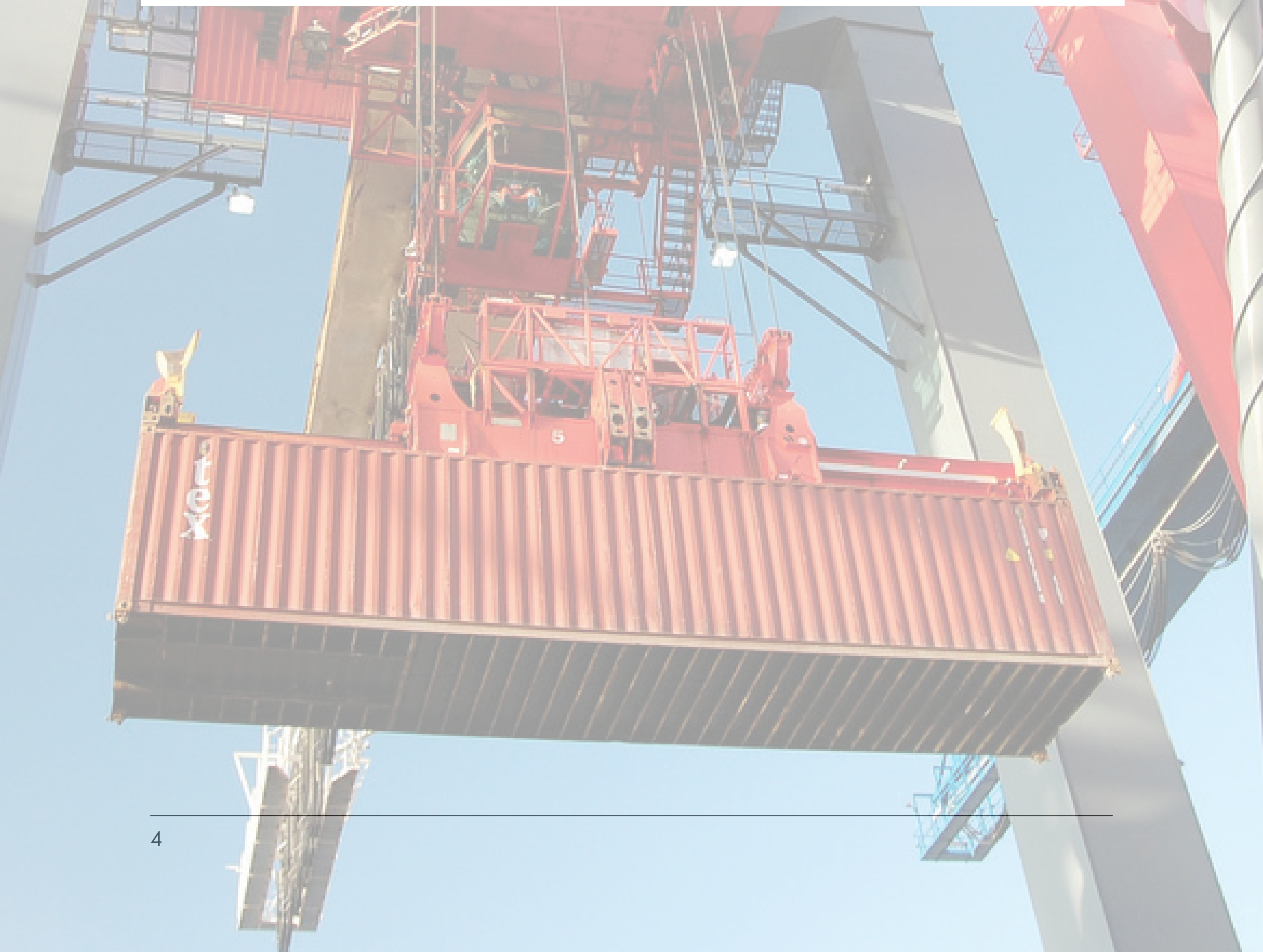
4 Input-output models, such as the one used to generate the figures in Table 2, are not best suited to estimate emissions associated with individual materials because of the coarse resolution of most input-output data, which are insufficient to distinguish specific materials such as steel, cement, or aluminium. For example, the most widely used global input-output model, GTAP, includes cement in the category *mineral products* and aluminium in the category *nonferrous metals* (Peters et al. 2011).

5 Data are presented for 2004, to be consistent with the input-output results in Table 1 and Table 2.

Table 3: Estimated trade of selected raw materials and associated emissions, 2004

Sector	**Material**	**2004 exports, million tonnes (UN Statistics Division 2011)**	**Average emissions intensity of production, tCO_2e/tonne (IEA 2007)**	**Emissions associated with producing traded materials, 2004, million tonnes CO_2e**
Iron and steel	Iron	16	<2	<30
	Crude steel	323	2	600
Primary aluminium		18	12	200
Cement	Clinker	43	0.9	40
	Cement	87	0.7	60
Paper	Pulp	44	0.5	90
	Paper	131		
Chemicals	Organic and inorganic	242	1.6	400

Source: Authors' analysis, using export statistics from UN Comtrade (UN Statistics Division 2011) and average emissions intensities from IEA (2007), except averages for clinker and cement are from the Cement Sustainability Initiative (2009), and statistics for aluminium are from the Carbon Trust (2011a).



3 POTENTIAL FOR SHIFTS IN TRADE TO REDUCE GLOBAL GHGS

If a significant fraction of global production of a commodity moves from countries with high GHG emissions per unit to countries with low unit emissions, global GHG emissions should decrease, assuming constant technology and production volume.[6] Accordingly, in this section, we explore how the GHG intensity of production varies among regions.

Changing the location of production could also affect GHG emissions indirectly. For example, if trade itself leads to economic growth (the "scale effect"), it could lead to increased consumption and production could lead to increased related GHG emissions (Tamiotti et al. 2009). Also, if increased production in one region uses resources (e.g., energy, labour, or capital) that would otherwise have been used in another industry, the changes could affect net GHG emissions (Strømman et al. 2009). While the latter is an important area for further research, we do not pursue it here; models to assess these interactions are highly complicated and still in their infancy (ibid.).

FACTORS AFFECTING GHG INTENSITY OF PRODUCTION

The GHG intensity of production indicates the GHG emissions released in producing a unit (e.g., one tonne) of a given product. Several factors affect it, including:

- **Technologies and processes** used, such as the balance of labour versus technology used to make a product, or the specific technology used (e.g., whether steel is made in a blast furnace or electric arc furnace);
- **Efficiency of operation** of those technologies and processes, such as how well equipment or furnaces are operated and optimized;
- **GHG intensity of energy**, such as whether any electricity used is produced from fossil fuels or from renewable sources;
- **GHG intensity of feedstocks**, including all the above factors for the production of feedstocks and/or component parts, such as fabric for clothing; and
- **Transportation distances and modes**, such as how far the product was transported by ship, train, truck and airplane.

Rarely is complete information available to compare the GHG intensity of production across countries or between facilities, much less to compare these individual factors. Still, enough information is available to compare average emissions intensity of production for some goods and materials. Those initial, limited comparisons can help develop methods and identify products that may warrant further research.

GHG INTENSITIES OF CONSUMER PRODUCTS

Figure 1 displays estimates of the GHG intensity of clothing production by country, based on the same MRIO model (Hertwich and Peters 2010) used to produce the summary of emissions embodied in trade presented in Section 2.[7] By these estimates, the median (50th percentile) GHG intensity of clothing production globally is just under 20 kg CO_2e per kg of product, similar to what other studies focused on clothing have found (Steinberger et al. 2009; Girod and de Haan 2010; Carbon Trust 2011b). However, Figure 1 also shows the wide range of GHG intensities for clothing production in different countries: from less than 10 kg CO_2e per kg, to about 60.

Uncertainty in these results, as with most MRIO models, can be large. At the level of a country's entire imports or exports, uncertainties in MRIO models have been found to be on the order of 7% for imports to the United Kingdom (Lenzen et al. 2010). For exports of a single product (in this case, clothing), uncertainties could be much larger. Along with actual differences due to production technologies and input fuels, these figures may reflect variations in the products made or materials used (Erickson et al. 2011), as well as underlying data errors in the model, such as trade data or input-output tables themselves, or systematic errors such as price conversion or differences in sector definitions across countries (Lenzen et al. 2010) .

6 Shifts within countries could also potentially reduce GHG emissions, but the GTAP data we use for the analysis are available only at the level of countries and world regions.

7 GHG intensities in kg CO_2e / $ from the MRIO model were adjusted to be per kilogram of clothing based on trade data from the UN's Comtrade database (UN Statistics Division 2011).

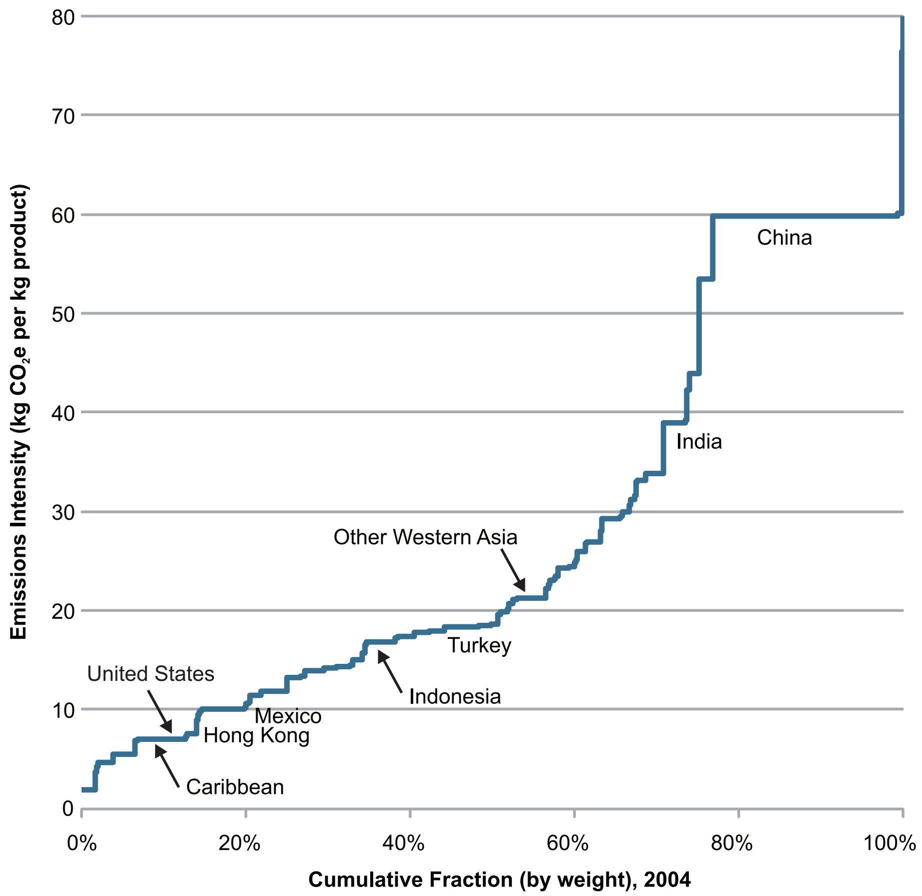


Figure 1: Estimated average GHG intensity of clothing production, by country, 2004

Source: Authors' analysis, using EUREAPA (Hertwich and Peters 2010).

Emissions intensities are ideally presented in terms of a "functional unit", a measure of the functions that the goods or service provides (Finnveden et al. 2009). Figure 1 presents emissions intensities in terms of CO_2e per kilogram of product, since the mass of clothing is more closely related to a garment's function as a body covering than is its monetary value. (For more specific items of clothing, the choice could be even easier, if associated data were available: for example, CO_2e per t-shirt.) Determining functional units can be highly subjective and subject to data limitations. Choosing a functional unit is particularly difficult for broad categories of products, such as clothing or "plastic / rubber products", where the range of products can be highly varied and function is not necessarily related to either its mass or its value.

Figure 2 compares the GHG intensity of four other categories of products, across countries. Due to data limitations and the difficulty of choosing functional units for these categories of products, we present these results in terms of CO_2e per dollar value of product.[8]

Together, the results in Figure 1 and Figure 2 show a wide variation across countries in GHG intensities for the products analyzed. The ratio of high (e.g., 80th percentile) GHG intensity to low (e.g., 20th percentile) GHG intensity varies from over five (clothing) to about two (vehicles and electronics). These findings suggest that shifting production from high-intensity to lower-intensity regions could reduce GHG emissions associated with manufacturing these goods. Relative to total emissions for the commodity, the opportunity may be greatest where the spread of GHG intensities is widest, such as for clothing.

8 Monetary values in GTAP, which underlie the MRIO model used here, are calculated using market exchange rates. We labelled the top five countries (in terms of trade value) or the top countries that collectively represent 50% of global trade value, whichever was greater.

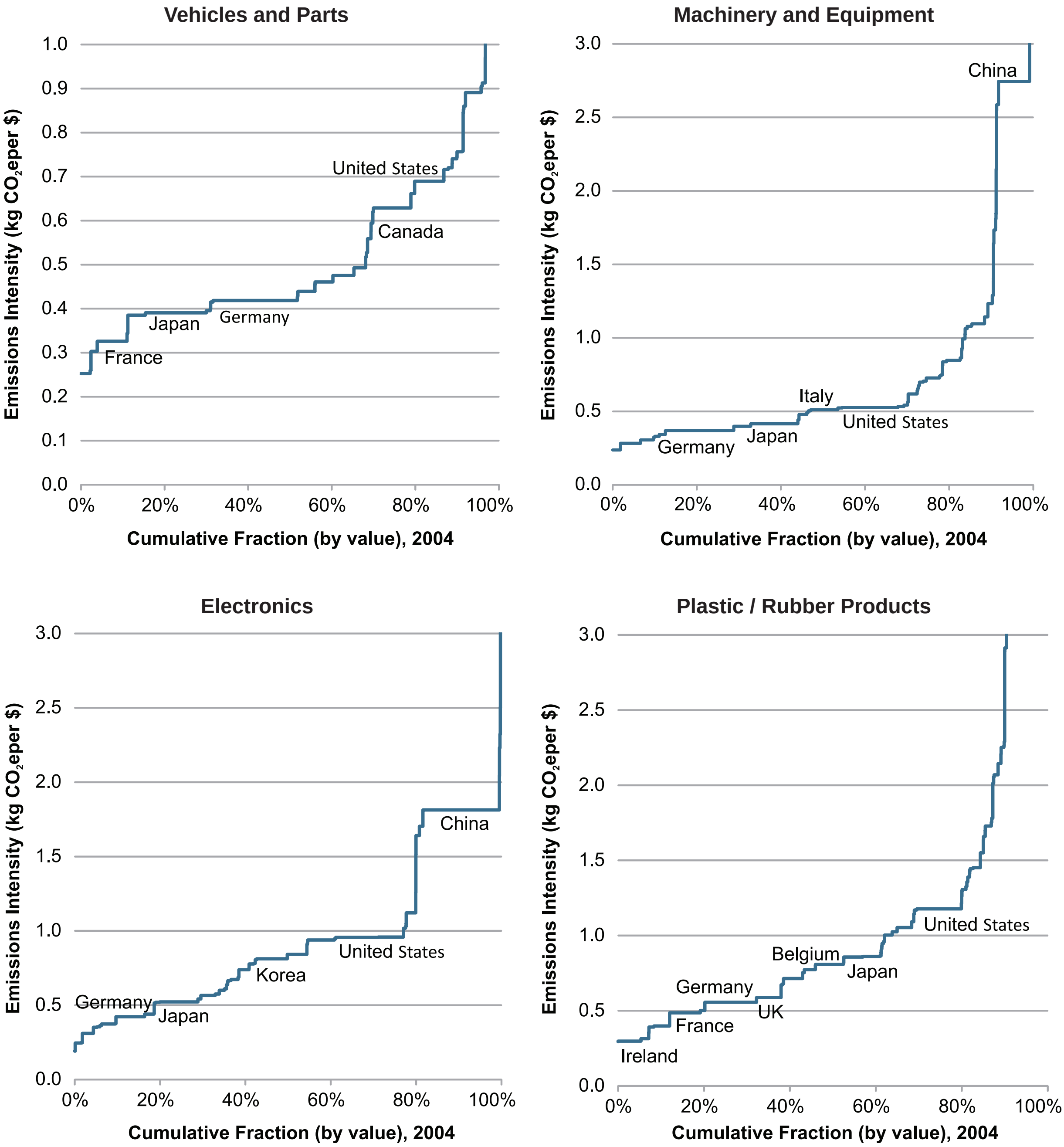


Figure 2: GHG intensity curves for selected products, 2004

Source: Authors' analysis, using EUREAPA (Hertwich and Peters 2010).

It should be noted that because Figure 2 presents emissions intensity on a per-dollar basis, these results could be skewed by differences in prices and values between regions. Even if GHG emissions per product (e.g., for a computer, or a "machinery" item, such as a refrigerator) were the same, the emissions per dollar would vary if prices differed among regions. The lower per-dollar GHG emissions intensity of Germany, France, and Japan could be explained, at least in part, by the fact that the goods they produce are more expensive.

Furthermore, the findings above compare *average* GHG intensities, but a more meaningful calculation would consider changes in energy sources and other factors of production at the *margin* (Erickson et al. 2011). Shifting production from one region to another would result in reduced demands for energy in one region and increases in another, and the energy sources added or subtracted may be more or less GHG-intensive than the countries' *average* energy mixes. For example, consider a country with substantial, but largely tapped,

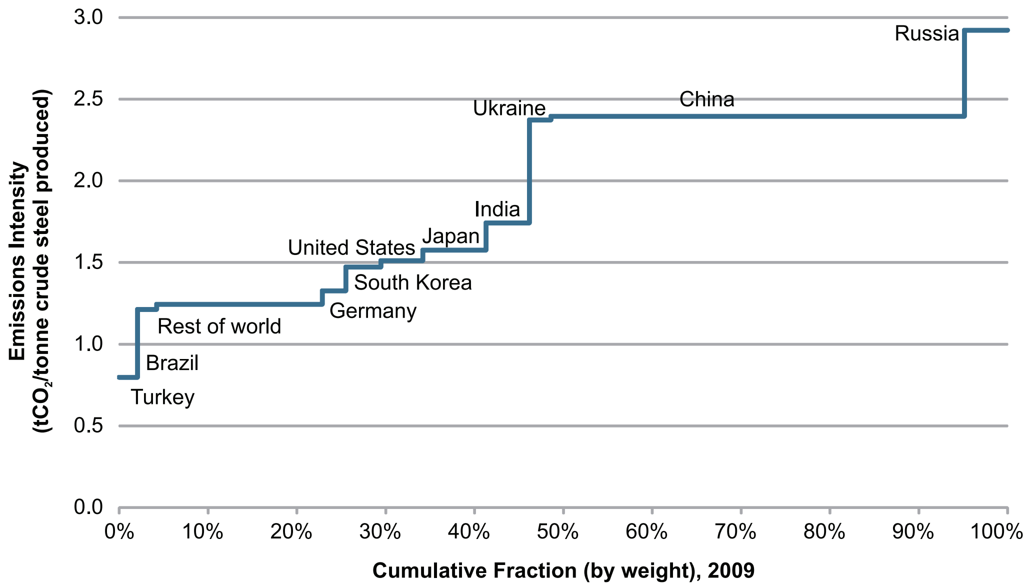


Figure 3: Estimated average CO_2 intensity of steel production, by country, 2009

Source: Authors' analysis based on energy statistics from the IEA (2011b; 2010) and steel production statistics from worldsteel (2011)

Table 4: Production, energy, and CO_2 intensity of crude steel production (2009)

Country	Production (million tonnes)	CO_2 (million tonnes)	CO_2 Intensity (t CO_2/t)	Energy Intensity (MJ/t)	CO_2 Intensity of Energy (kg CO_2/GJ)	% of Production by Electric Arc	CO_2 Intensity of Electricity (t CO_2/MWh)
China	577	1,374	2.4	28	86	10	0.74
Japan	88	138	1.6	22	73	22	0.41
India[a]	60	105	1.7	19	94	60	0.95
Russia	60	175	2.9	52	56	27	0.32
United States	58	88	1.5	25	61	62	0.51
South Korea	49	72	1.5	22	68	43	0.50
Germany	33	43	1.3	20	68	35	0.43
Ukraine	30	71	2.4	33	72	4	0.37
Brazil	31	32	1.0	23	45	21	0.06
Turkey	25	20	0.8	12	64	70	0.48
ROW[b]	221	286	1.3	21	61	59	N/A
World	1,231	2,404	2.0	26	75	29	0.50

Source: Authors' analysis, based on energy statistics from the IEA (2011b; 2010) and steel production statistics from worldsteel (2011).

a Emissions and energy are underestimated, because India's energy statistics did not include electricity.

b Emissions and energy for rest of world (ROW) are likely underestimated, because several smaller producers may have included energy use for making iron and steel in a broader industrial sector.

hydroelectric potential. New electricity production in that country may come from other resources (e.g., coal or natural gas), which would be much more emissions-intensive than the prior average, and hydro-dependent, electricity supply. We consider likely fuel sources for new electricity production later, in Section 4.

GHG INTENSITIES OF RAW MATERIALS: STEEL

As displayed in Table 3, about 600 Mt CO_2e are associated with internationally traded steel, on par with many of the finished products discussed above. The GHG intensity of producing steel also varies by country, from nearly 3 t CO_2e / t steel to less than 1. Figure 3 presents the GHG intensity across countries, as for the products shown in Figure 1 and Figure 2.

While Figure 3 provides a useful picture of the range of GHG intensities and production levels across countries, it doesn't explain why the variations exist. GHG intensity can be broken down into two factors: the energy intensity of production (i.e., energy per tonne) and the carbon intensity of that energy (i.e., tonnes of CO_2 per unit of energy). Exploring these metrics, as shown in Table 4, can help explain why some countries make lower or higher-GHG steel. Results in Table 4 are based on an analysis of energy statistics from the IEA (2011b; 2010) and steel production statistics from worldsteel (2011)

As shown in Table 4 and Figure 3, of the top 10 steel-producing countries, the two with the lowest GHG intensity are Brazil and Turkey. Table 4 shows that the main reason Brazil produces relatively low-GHG steel is that its energy is much less GHG-intensive (45 kg CO_2/GJ) than the world average. This is due to the country's high reliance on charcoal (considered CO_2-free in these IEA statistics, though it may not actually be low-GHG when considering forestry and production practices) instead of coke for making iron, as well as its relative abundance of low-GHG hydropower. Brazil's plants are also less energy-intensive than the world average. Turkey's low-GHG steel can be explained in large part by its low energy intensity, which itself is a consequence of its heavy reliance on electric arc furnaces to make steel from scrap.

Assuming that the average emissions intensities in Table 4 remained unchanged, then shifting production from high-intensity to low-intensity regions might help reduce global GHGs. However, in several cases, the average GHG intensities may not remain the same. For example, increased production in the lowest GHG-intensity producer, Turkey, could be met either through virgin steel – increasing the GHG intensity – or recycled scrap – lowering or maintaining the GHG intensity. Because the supply of scrap globally is limited (by rates of capital turnover), increasing steel-making from scrap in Turkey could pull scrap from steel-making on other areas, leading to no net emissions benefit.

For the GHG intensity of steel production to continue to be as low as in some of the lowest GHG-producers (such as Turkey or Brazil), new sources of key feedstock and energy resources – scrap steel, low-GHG fuels (e.g., charcoal, which can, though not necessarily, be low-GHG), and renewable electricity (for operating electric arc furnaces) would need to be available.

Exploring trends in GHG intensity over time

Table 4 and Figure 3 presented GHG intensities for 2009, the most recent year available at the time of this writing. Additional insights may be gleaned by looking at global or regional trends over time. Figure 4 shows a gentle decline in global GHG intensity of steel production over the last decade and a half, dropping from a peak of 2.2 t CO_2 per tonne of crude steel in 1995 to a relatively stable level of 1.8 t CO_2 per tonne between 2001 and 2008, an 18% decline.[9] The decline in most countries was partly offset by dramatic growth in steel production in GHG-intensive China, even as China itself decreased its own GHG intensity by a proportionally larger amount.

Over the period 1992 to 2008, process choice and the carbon intensity of energy (fuel choice) have fluctuated globally, but contributed relatively little to the overall change in GHG intensity.[10] Instead, a shift to more energy-efficient technologies made the biggest impact, especially in China, which accounted for 11% of global steel production in 2002 and 47% in 2009. Figure 6 shows the changes in drivers of GHG intensity in Chinese steel. China's closure of inefficient open hearth furnaces and rapid investment in high-efficiency blast furnaces with waste gas and heat recovery has

9 It is possible that the peak in 1995 in Figure 4 is (at least in part) an artifact of changes in the underlying IEA data, as reporting practices for China and transition economies (countries of non-OECD Europe and Eurasia) changed during the economic reforms of the 1990s (IEA 2012a). Indeed, the countries that display the peak in 1995 include China, Russia and Ukraine. Without these countries, the CO_2 intensity of steel production in 1995 was about the same (2% less) than in 1992.

10 Our analysis of GHG intensity of electricity used by the steel industry is based on national average GHG intensities of electricity from IEA data (IEA 2011b). It is possible that the GHG intensity of the electricity used by steel plants differs from this average.

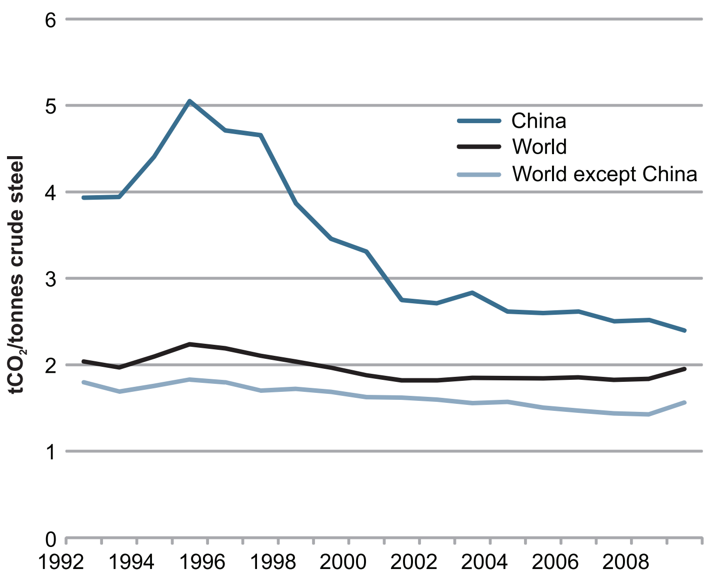


Figure 4: Estimated average CO_2 intensity of global steel production, 1992-2009

Source: Authors' analysis, based on energy statistics from the IEA (2011b; 2010) and steel production statistics from worldsteel (2011)

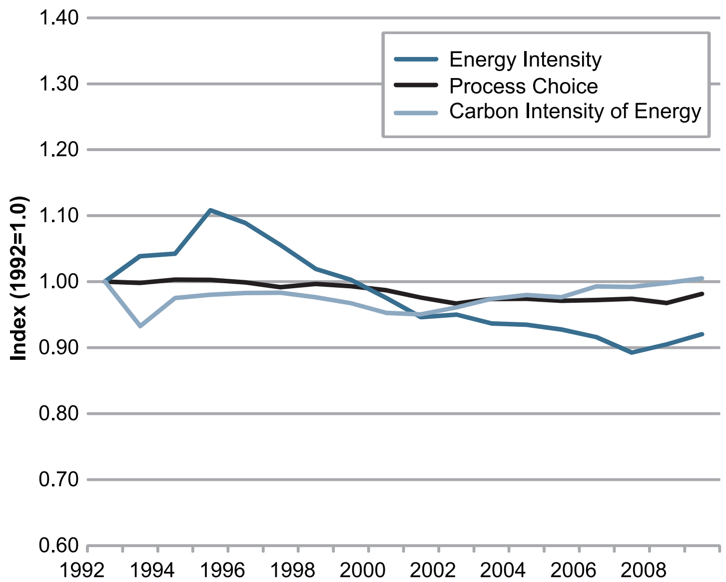


Figure 5: Changes in three drivers of global steel CO_2 intensity, 1992-2009: process choice, energy intensity, and carbon intensity

Source: Authors' analysis, based on energy statistics from the IEA (2011b; 2010) and steel production statistics from worldsteel (2011)

rapidly decreased the energy intensity of Chinese steel production since the mid-1990s (Oda et al. 2007).

Together, these trends have led the GHG intensity of steel production in China to fall by roughly half even as total production grew by 700%. In China, this growth in production has been driven largely by domestic demand. In other cases, however, it has been driven by export demand, as in Ukraine in recent years and Brazil in the late 1980s (Kim and Worrell 2002).

Figure 5 and Figure 6 indicate that improvements in energy intensity and process choice have been the dominant drivers of declining GHG intensity of steel globally and in China. By contrast, the GHG intensity of energy appears to have changed very little

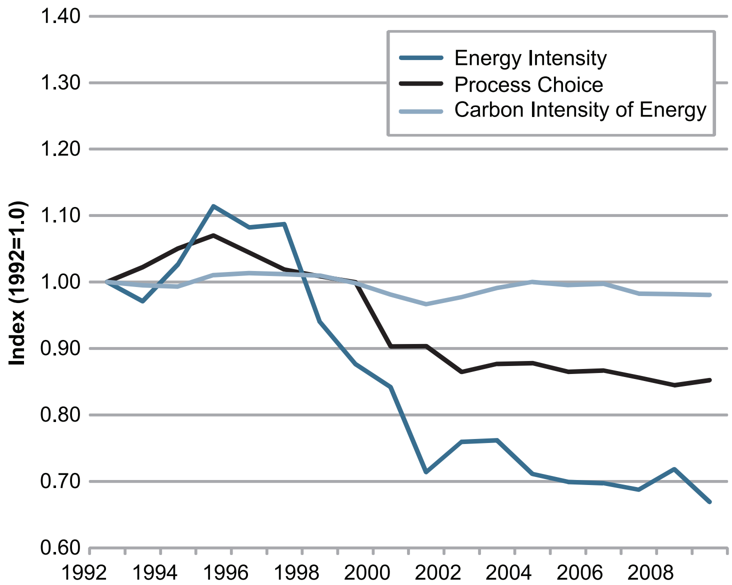


Figure 6: Changes in drivers of Chinese steel GHG intensity, 1992-2009: process choice, energy intensity, and carbon intensity

Source: Authors' analysis, based on energy statistics from the IEA (2011b; 2010) and steel production statistics from worldsteel (2011).

over the last couple of decades.13 This highlights an opportunity to develop a low-GHG steel industry using lower-GHG energy. Choices in the steel industry about future energy supplies may significantly influence the GHG intensity of future steel production.

SUMMARY OF FINDINGS ON GHG INTENSITY AMONG REGIONS

World regions produce goods (e.g., food, electronics, and vehicles) and raw materials (e.g., steel) with a wide range of technologies, processes, feedstocks, and associated GHG emissions. As described in Section 2, the GHG intensity of production can vary among regions by a factor of five or more. This alone would suggest that shifting production from more-intensive regions to less-intensive regions could reduce global GHG emissions, as long as the improvement in emissions intensity were greater than any increases in emissions associated with transportation (e.g., over farther distances). However, many other factors affect whether a region can expand production with low GHG intensity. Arguably, any region could install the latest, most energy-efficient technologies and processes. However, some may be better positioned to expand use of low-GHG feedstocks or energy sources. If increasing production in these regions displaces production in lower-GHG regions, it could help reduce global GHG emissions. For example, if some regions had greater opportunities to expand renewable electricity generation (and access to the necessary raw materials), they could be well-positioned to be future centres of low-GHG production of electricity-intensive materials, such as aluminium or steel from electric arc furnaces. The following section explores this question for steel, and whether certain world regions may be better positioned to significantly expand low-GHG steel production.

4 ROLE OF LOCATION IN FUTURE LOW-GHG STEEL PRODUCTION

As discussed in the previous section, the steel industry has gradually improved its GHG intensity over the past two decades, mostly by improving energy intensity. Improvements in the GHG intensity of the energy used (e.g., electricity mix, or fuel choice) have been, by contrast, modest. In this section, we explore whether some regions may be well positioned to be significant future producers of low-GHG steel by conducting a simple analysis of factor availability.

To better understand possible paths for the steel industry, it is useful to understand the three primary routes for producing crude steel. They are:

- **Basic oxygen furnace (BOF),** in which oxygen is blown through molten pig iron (made in a blast furnace from iron ore and a carbon source, such as coke), oxidizing the carbon, silicon, and phosphorus in the pig iron to produce steel;

- **Electric arc furnace using direct reduced iron (DRI-EAF)**, in which iron is first reduced "directly" by reacting it in the presence of carbon monoxide and other gases, after which it is melted using a high-temperature electric arc; or

- **Electric arc furnace using scrap (Scrap-EAF),** which melts scrap steel using a high-temperature electric arc.

As shown in Table 5, the GHG intensity of these three primary routes can vary by a factor of 10, from as little as 0.3 t CO_2e per tonne of steel – the Scrap-EAF route if powered by renewable electricity – to greater than 3 t CO_2e – DRI-EAF with iron-making fuelled by coal and steel-making powered by coal-based electricity.

Unsurprisingly, the lowest-GHG way to make steel is to use scrap steel in an electric arc furnace (EAF), because new crude steel can be made directly from other steel without first needing to reduce iron.[11] For this route, variations in GHG intensity depend primarily on the electricity source, whether high-GHG fossil fuels or low-GHG renewables.

When steel is made in an EAF from direct reduced iron (EAF-DRI), variability arises from the GHG intensity of the direct reduced iron (DRI) production – whether natural gas or coal is used[12] – as well as the electricity source for the EAF. In a BOF, the greatest variability in GHG intensity is in iron-making and depends strongly on whether low-GHG charcoal or coke/coal is used as a fuel and carbon source.

In summary, the choice of fuels and feedstocks in each of the three main steel-making routes can significantly influence GHG emissions: availability of low-GHG charcoal for iron in the blast furnace of the BOF routes, natural gas for DRI production, and low-GHG electricity for steel-making in the two EAF routes. For these reasons, the IEA has stated that gas-based DRI and charcoal-based BOF are important components of a future low-GHG steel industry (IEA 2012b). Situating iron and steel production where these resources are most available may help realize the potential for low-GHG fuels and feedstocks to contribute to a future low-GHG steel industry. For example, as shown in Table 6, locating future BOF steel production in areas with a high availability of biomass for charcoal and low coal availability could reduce the GHG intensity by up to 2 tonnes of CO_2e per tonne of steel relative to a location with low biomass and high coal availability.

The possible GHG benefits displayed in Table 6 depend, in part, on whether low-GHG charcoal can be made sustainably from biomass. Research on sources of low-GHG biomass is still developing, but forest harvest residues, waste biomass from agro-industrial production, or from biomass grown on abandoned agricultural lands planted with perennials are likely to be lower-GHG (Fargione et al. 2008; McKechnie et al. 2011), while sources that rely on dedicated crops (e.g., whole trees) may be higher GHG. Their net emissions would depend heavily on the sustainable management of land and water resources in the presence of competing interests such as agriculture and food production, other bioenergy demands, forest products,

11 Because of this, the three routes are mainly distinguished by whether they produce virgin steel (from iron ore) or recycle scrap steel. To pass from iron ore to steel, iron must be "reduced", a chemical process that is always accompanied by an oxidation process. Carbon (either elemental carbon or carbon monoxide) is the main reducing agent, which is converted to carbon dioxide after oxidation. Virgin steel production therefore always produces carbon dioxide as an essential part of the process, separate from any carbon dioxide emissions from fuel combustion.

12 We consider natural gas as the alternative to coal in DRI production based on discussion in IEA (2012b), but DRI production could also use biomass instead of natural gas (Buergler and Di Donato 2009), in which case GHG savings could be even greater.

Table 5: GHG intensity of steel-making by route and process (tonnes CO_2e per tonne steel) [a]

Process Step	BOF	DRI-EAF	Scrap-EAF
Mining and treatment of raw materials (e.g., making coke, sinter, pellets)	0.4-0.8		N/A
Iron-making	0-2	0.7-1.1	0
Steel-making		0.1-1.9	0.1-0.6
Casting, rolling	0.2-0.5		
Total	**0.4-2.8**	**1.0-3.5**	**0.3-1.1**

Source: Authors' analysis, using data from IEA (2007) and Worrell et al. (2008)

a Low-end values for iron- and steel-making in a BOF assume low-GHG charcoal fuel. High-end values assume the current average BOF intensity of China (values could be higher for individual, inefficient plants). Low-end values for both EAF routes assume CO_2-free electricity; high-end values assume coal-based electricity..

Table 6: Potential GHG impacts of shifting location of major steel production pathways

Route	From region with:	To region with:	Potential range of GHG impacts
BOF	Low charcoal availability; high coal availability	High charcoal availability; low coal availability	Up to 2.0 t CO_2 / tonne crude steel
DRI-EAF	Low natural gas and renewable electricity availability; high coal availability	High availability of renewable electricity and natural gas; low coal availability	Up to 2.2 t CO_2 / tonne crude steel

and ecosystem preservation (Lattimore et al. 2009; Chum et al. 2011). Not all of these biomass sources may be equally suitable for charcoal production, however, nor be available within economic distance to current or potential steel mills. Furthermore, given limited quantities of sustainable biomass, it is possible that greater GHG benefit would be realized by using these sources to displace other fossil fuels in other sectors, such as coal in electricity generation.[13]

The total potential GHG abatement from such shifts in production would depend on how much steel is being made in areas with high-GHG energy supplies and how much could really be shifted to areas with plentiful charcoal and low-GHG electricity. Detailed data to do so are not readily available. Other factors may also affect abatement potential, such as limits on the size of blast furnaces that can use charcoal (IEA 2007).

A useful proxy for how much steel production could be shifted to low-GHG regions is the rate of *growth* in the steel industry. In the years immediately preceding the global recession, production of BOF steel grew by about 70 million tonnes per year, with nearly all of that growth in China, which is rich in coal but not in biomass for charcoal. Annual expansion of DRI-EAF steel averaged about 4 million tonnes per year, about three-quarters of it in India, which also has substantial coal resources but limited biomass for charcoal. Moving 70 million tonnes of BOF steel production and 4 million tonnes of DRI-EAF steel production to areas

13 Relatively few studies exist that analyze the comparative GHG emissions among alternative fates of biomass (Lee et al. 2010). Among the factors to consider in comparing the benefits of biomass for power generation versus charcoal production for iron production would be the relative efficiencies, fuels displaced, and methane production rates of charcoal production (Kammen and Lew 2005).

with substantial low-GHG biomass and renewable electricity could reduce global GHG by about 150 million tonnes per year, over 5% of the steel industry's current GHG emissions. Relocating production of greater quantities of steel, of course, (e.g., several years' worth of new production) would increase the total GHG abatement potential.

As noted above, a detailed analysis of the potential for such shifts is beyond the scope of this study, but we can provide an initial assessment. Below, we use data from the IEA's *World Energy Outlook 2011* (IEA 2011a) and the U.S. Geological Survey's *International Minerals Yearbook* (U.S. Geological Survey 2009) to explore whether particular world regions may have greater future access to the fuels and feedstocks identified above. Specifically, we assess:

- **GHG intensity of planned new electricity generation**, based on net additions to electricity generation between 2020 and 2030 in the WEO's *new policies* scenario.[14] This factor helps gauge each region's ability to expand the low-carbon electricity supply, which could help lower the GHG intensity of electric arc furnace steel production, either via the DRI-EAF or Scrap-EAF routes.

- **Natural gas availability in industry**, assessed as the WEO's forecast of natural gas demand in industry as a fraction of total industrial fuel demand (excluding electricity and heat) in the new policies scenario in 2030.

- **Biomass availability in industry**, used as a proxy for charcoal availability, and assessed as the WEO's forecast of biomass demand in industry as a fraction of solid industrial fuel demand (sum of coal and biomass) in the new policies scenario in 2030.

- **Iron ore reserves**, assessed as a ratio of current (2011) production, based on data from the U.S. Geological Survey.

The field for specific factors in a region are colour-coded in dark blue if the region is in the best 20th percentile, and light blue for regions in the 40th percentile. While these thresholds are arbitrary, they enable an initial assessment of future factor availability for low-GHG steel production.

The assessment in Table 6 suggests that the following regions may be particularly well suited to future production of low-GHG steel:

- **The Americas (North and South) could be a host for BOF steel using low-GHG biomass**, given relatively abundant biomass resources and considerable iron ore reserves. However, significant further research and analysis would be needed to determine whether biomass can be sourced sustainably and economically and whether use in the iron and steel industry would result in lower GHG emissions than alternative uses (given limited biomass supplies), such as displacing fossil fuels in power generation.

- **The U.S. could be a host to expanded production of EAF steel**, both from DRI and scrap, due to the country's plans for low-GHG electricity, relative abundance of natural gas, and significant iron reserves. Other regions with extensive supplies of natural gas and iron ore could also be centres of DRI iron, though these regions (e.g., Russia) are not currently planning to develop as much low-GHG electricity as the U.S.

Other regions – including individual countries within these aggregated regions – could also have significant access to these resources. For example, if Japan develops low-GHG electricity as assumed by the IEA, it could produce low-GHG steel in electric arc furnaces by importing some of the other key low-GHG resources (e.g., DRI). Interestingly, the region that currently is expected to be by far the world's dominant steel producer – China – does not rate high in any of the factors above.

Note that the indicators presented in Table 7 are subject to significant uncertainties. Most significantly, the assessment relies heavily on energy data that

14 We calculate GHG intensity of the planned net increases in electricity generation for 2020-2030 by dividing increases in CO_2 emissions for power generation by increases in electricity generation over the same period. We do this instead of using an average emission factor, because increasing EAF steel production in a region would require increased electricity generation, so we use this marginal electricity emissions factor calculated from the WEO as a proxy for the emissions intensity of new power generation capacity. This procedure introduces two inaccuracies: one, because it is calculated on a net basis, it does not include the GHG intensity of added fossil-fuel plants that simply replace retired fossil-plants. Second, since IEA includes heat generation in power but not electricity generation, the numerator (CO_2) is for a bigger population than the denominator (MWh). The first factor may underestimate GHG intensity, while the second may overestimate it; we assume the two combined factors do not significantly distort the regional comparisons.

Table 7: Assessment of resource availability for low-GHG steel

	Factors important for EAF steel		Factor important for BOF steel	
	Marginal Electricity Emissions Factor (tCO2/MWh)	Natural Gas Availability in Industry (% of industrial fuels, by mtoe)	Biomass Availability in Industry (% of solid industrial fuels, by mtoe)	Iron Ore Reserves (as ratio of 2011 production)
China	0.64	0.17	-	19
India	0.37	0.07	0.24	29
Rest of Non-OECD Asia	0.29	0.39	0.34	N/A
Japan	0.03	0.19	0.11	N/A
Rest of OECD Asia-Oceania	0.05	0.35	0.47	73
Russia	0.17	0.58	0.06	250
Europe + Other CIS	0.19	0.50	0.45	97
United States	0.05	0.55	0.86	128
Rest of OECD Americas	0.15	0.52	0.87	137
Middle EAst	0.20	0.64	0.36	83
Africa	0.17	0.23	0.75	32
Brazil	0.05	0.22	0.89	74
Rest of Latin America	0.11	0.44	0.65	250

Source: Authors' analysis, using data from IEA (2011a) and the U.S. Geological Survey (2009).

Colour-coding: Light blue – best 40th percentile; dark blue – best 20th percentile

are not specific to the steel industry. More details on planned investments and availability of energy supplies in the steel industry would significantly improve the assessment. Another key consideration in expanding future steel production is the relative cost of these and other factors among regions. A Carbon Trust analysis (2011c) found steel production costs to be roughly equivalent in China, the European Union, and North America, but lower in South America and in economies in transition. This suggests that North America may not be significantly disadvantaged relative to the largest expected centre of demand (China), and that South America may be particularly well positioned.

Shifts of production among world regions could also increase transportation requirements, and hence CO_2 emissions, however. Industry data suggest that increases in steel production in China, for example, have been largely to satisfy domestic demand (worldsteel 2011), so fulfilling Chinese demand for steel by overseas production could increase overall transport requirements. Shipping goods by ocean vessel ranges from 11 (bulk shipping) to 14 (container ship) tonnes CO_2e per million tonne-kilometres (Weber and Matthews 2008). Supposing an upper-range shipping distance of 20,000 km (from Brazil to China) would therefore add roughly 0.2 to 0.3 t CO_2e per tonne of steel. That is roughly 10 to 15% of current average GHG intensity of production and significantly less than the potential differences in GHG intensity of production among higher and lower-intensity regions, about 2 t CO_2 per tonne steel (per Figure 3 and Table 6). This suggests that increased emissions from transportation are worth considering if the potential GHG reductions from shifting production are small (perhaps less than 0.5 t CO_2 per tonne) but less so if they are large (more than 1 t CO_2e per tonne).

While this analysis has focused on the location of production given a fixed level of steel production and scrap availability, we note that in a low-carbon transition, the amount of scrap available might change over time. This possibility is discussed in Box 1.

Box 1: Capital Turnover and Scrap Steel

Replacing capital stock, such as older, inefficient power plants and industrial facilities, is likely to be necessary to achieve deep global GHG reductions. Furthermore, new investments in low-carbon transportation (e.g., rail), building retrofits, and renewable power (e.g., wind farms) all require new capital investments. But replacing and making new capital equipment itself results in further emissions. Indeed, steel and cement production release substantial GHG emissions, and steel in particular is an essential input to buildings, roads, rails, and machines.

Emissions from new investment would therefore partially offset the emissions reductions from the more efficient capital, if other things remain the same. However, they need not remain the same: in the case of steel, retiring existing capital will release a stream of scrap steel, which can be recycled to produce new steel through less carbon-intensive processes than steel production from iron ore. A technical assessment of rapid decarbonization therefore requires an analysis of demand for steel and production of steel scrap under accelerated retirement of capital.

5 HOW POLICIES COULD SHIFT TRADE FLOWS

Previous sections have described how shifting where some goods and materials are produced could help reduce global GHG emissions. This section discusses what policies may be available to steer trade flows and, by extension, production, to world regions with low GHG intensity. It begins with an overview of the various policy options to address emissions associated with trade, then assesses these options according to four criteria: 1) environmental effectiveness, 2) legal feasibility, 3) political feasibility, and 4) administrative feasibility. The section does not discuss cost-effectiveness, as this depends on several factors outside of the producing sectors themselves – such as electricity production and port facilities – that would provide benefits as well as incurring costs. Finally, we discuss trade-offs between different types of policies.

OVERVIEW OF POLICY OPTIONS

Governments can adopt a range of measures to influence the emissions intensity and location of production of traded goods. These include:

- **Quantitative restrictions** on imported goods based on processes and production methods (e.g. country A specifying that it will only import X tonnes of steel produced with fossil fuels from country B). The most extreme case of such a restriction would be an outright import ban.
- **Punitive tariffs** could be set high enough to make imported products less competitive than domestic goods, and are normally implemented in response to unfair or illegal trade practices (Epps and Green 2010, p.212).
- Imposing **anti-dumping duties** on imports as a response to "environmental dumping" or imposing **countervailing duties** as a response to unfair subsidies (Pauwelyn 2007; see also Stiglitz 2006). In both cases, duties are imposed because of the failure to internalize environmental externalities (i.e. the social cost of carbon) in the production process of a good, and the measures increase the costs of imports.
- **Border carbon adjustments** (BCAs). Two broad design options can be distinguished: 1) border tax adjustments (BTAs); and 2) the requirement for importers to surrender allowances at the border (Cosbey 2008). A BTA requires importers to pay a charge equivalent to a tax applied to goods produced domestically, whereas a requirement to surrender allowances is linked to an emissions trading scheme (ETS), and permits goods to enter a country only if a certain amount of emission allowances are purchased that reflect the GHGs emitted during production (Van Asselt and Brewer 2010).
- **Technical regulations or standards** related to the emissions of energy-intensive products applied to both imported and domestic products. Such measures could be targeted at the production process (e.g. a requirement to produce aluminium with renewable energy), thereby directly affecting embedded emissions (Buck and Verheyen 2001).
- **Preferential tariffs and quotas**, such as zero (or low) tariff rates or quotas, to exporting developing countries, which could be linked to the ways in which goods are produced, such as granting preferential treatment to low-carbon imports (McKenzie 2008). Such measures can be framed negatively – i.e. preferential treatment is conditional upon climate action – or as positive incentives – i.e. additional benefits will be granted if (additional) climate action is undertaken (Epps and Green 2010, p.182; see also Callahan and Vasile 2010). Preferential treatment for developing countries is already common in trade policy, notably through the Generalized System of Preferences, but other trade concessions are also conceivable, for instance in regional trade agreements or economic partnership agreements.

ASSESSING THE OPTIONS

This section evaluates the policy options according to their legal, political and administrative feasibility. Where appropriate, we discuss modifications that could make an option more palatable.

It is beyond the scope of this report to assess the environmental effectiveness (in terms of net global GHG emission reductions) of these options, as effectiveness would depend so strongly on mechanism design, and few quantitative analyses of the impact of the different measures exist. Clearly, to have significant environmental (i.e., emissions) impact, they would need to address significant flows of traded emissions.

Legal feasibility

The legal feasibility of measures is framed largely by two bodies of international law: international trade law (in particular the agreements of the World Trade Organization) and international climate change law (the United Nations Framework Convention on Climate Change and its Kyoto Protocol). We should note upfront that the legality of most measures cannot be fully ascertained. First, the compatibility with WTO or climate law depends largely on the design and application of measures. Second, the final word on WTO compatibility rests with the judicial bodies of the WTO. And third, even in the cases that measures could be found to violate international law, it might be possible to change the law to allow them. However, such changes will be limited by considerations related to political feasibility, as discussed in the next sub-section.

Various provisions of the General Agreement on Tariffs and Trade (GATT)[15] are relevant when considering the adoption of domestic policy measures. First, a country should not discriminate between producers from other member countries and domestic producers – the "national treatment" rule (Article III). This means that products that imported and domestic products that are "like" should be treated as such. The second rule is that a country should not discriminate between its trading partners – the "most-favoured nation" rule (Article I). The purpose of this rule is to ensure that like products are treated alike, irrespective of their origins or destination.

The GATT further stipulates under which conditions countries could adopt quantitative restrictions (Article XI). If these specific provisions are violated, it does not necessarily mean that measures are deemed illegal. The GATT contains two (limited) environmental exceptions, which can justify trade-restrictive measures "necessary to protect human, animal or plant life or health"; or "relating to the conservation of exhaustible natural resources if such measures are made effective in conjunction with restrictions on domestic production or consumption", provided that "such measures are not applied in a manner which would constitute a means of arbitrary or unjustifiable discrimination between countries where the same conditions prevail, or a disguised restriction on international trade" (Article XX).

From a legal point of view, a multilateral solution raises far fewer questions than unilateral trade measures. Unilateral policies that apply equally to all WTO members will pass muster under the most-favoured nation rule, whereas policies targeting only specific WTO members will require saving under Article XX. The requirements developed in WTO case law related to Article XX include showing a clear link between the measure and the environmental objective, meaning that a measure cannot be justified on economic grounds. Furthermore, if multiple measures are possible, for a measure to be "necessary", it must be shown that the less trade-restrictive measure was chosen. The assessment of whether such a measure is available depends on the strength of the link between the measure and the objective, as well as the trade impacts of a measure (Condon 2009, p.913). For a measure to be "made effective in conjunction with restrictions on domestic production or consumption", a country needs to show that there are equivalent domestic measures in place. This does not mean that domestic and imported products must receive identical treatment, but that measures should impose similar restrictions on both (Pauwelyn 2007, p.36). Furthermore, to satisfy the conditions of Article XX, any unilateral measure should only follow after serious efforts to negotiate such a solution; take into account local conditions in other countries; comply with requirements of basic fairness and due process; and not discriminate in ways counter to its environmental objective (Van Asselt et al. 2009).

With these considerations in mind, some of the policy options discussed above are likelier to be deemed WTO-incompatible than others (Van Asselt and Biermann 2007). For instance, quantitative restrictions or punitive tariffs or taxes that are not clearly based on the GHG content of a product are unlikely to be the less trade-restrictive option, when alternatives such as BCAs are available. Moreover, there is only limited flexibility that can be built into such measures, and it will be hard for countries to argue that equivalent measures are in place for domestic industries (Epps and Green 2010, pp.216–218). Therefore, it can be questioned whether such measures are sufficiently related to the goal of climate protection, and should rather be regarded as protectionist tools (Charnovitz 2003).

Preferential treatment for developing countries could be allowed under WTO rules if the conditions of the so-called "enabling clause" are met. Based on WTO jurisprudence, this means that such preferential treatment should effectively meet the development, financial and trade needs of developing countries, and that similarly situated countries should receive identical treatment. It could perhaps be argued that preferential treatment linked to the level of climate mitigation action in each country would help meet such needs, though it is not clear that mitigation

15 The full text of the GATT is available at http://www.wto.org/english/tratop_e/gatt_e/gatt_e.htm.

actions could support all three needs. Moreover, if the needs are explicitly constructed to refer to addressing climate change, there is an argument that it is possible to grant preferential treatment to countries particularly vulnerable to climate change (McKenzie 2008), though it is not clear if or how lower-GHG production in developing countries could contribute to climate change adaptation or resiliency. Perhaps the most promising rationale for preferential treatment is via Existing Generalized System of Preferences (GSP) practice, which shows that preferential treatment can be differentiated among developing countries on the basis of criteria established by the importing country, such as respect for human rights or meeting certain environmental standards (in this case, GHG-intensity of production). As long as these criteria can be considered objective, they are allowed (Pauwelyn 2013).

The legality of BCAs is much disputed,[16] but some general statements can be made about improving the chances of the measure being deemed compatible with WTO law. First, clearly basing a measure on the environmental rationale of minimizing carbon leakage would be more helpful than using the economic rationale of safeguarding the international competitiveness of energy-intensive industries (Pauwelyn 2007). Second, delaying the effective implementation of a BCA would buy some time for international negotiations. Third, a measure is likelier to be considered non-discriminatory if the level of the adjustment corresponds to a product's actual embedded emissions (Tamiotti 2011). Fourth, it is important how the country implementing the measure accounts for climate policies in other countries (Hertel 2011), and whether other countries are involved in the operationalization of the measure (Hufbauer et al. 2009). Finally, if a measure is targeted at specific countries – rather than applied across the board – it is likelier that the measure contributes to a specific environmental rationale and thereby can be saved under the GATT's environmental exception.[17]

In addition to WTO law, certain measures may contravene the rules and principles of the climate treaties. A key principle in this regard is that of "common but differentiated responsibilities and respective capabilities" (CBDRRC), in Article 3.1 of the UNFCCC. Differentiation of financial burdens of climate change mitigation measures between developed countries and developing countries, combined with developed country leadership, is part and parcel of the climate treaties, and also plays an important role in the international trade regime (Charnovitz 2010; Eckersley 2010; Hertel 2011; Ladly 2012; Pauwelyn 2013). For instance, if the United States adopted measures limiting carbon-intensive imports from developing countries (e.g. China), without implementing ambitious climate policies itself, this would be a clear case of violation of the CBDRRC principle.

However, the precise contents of the principle are not cast in stone, and it can be argued that in some cases, trade restrictions may be possible because of a country's aggregate or future emissions (Dröge 2011). Moreover, rather than sticking to the developed/developing country (Annex I/non-Annex I) distinction in the Kyoto Protocol, it may be possible to work with more fine-grained categories of developing countries, reflecting the differences in economic and environmental conditions between, for instance, China and Bangladesh, or Qatar and Burkina Faso. While such differentiation amongst developing countries may be in line with socio-economic realities, it also creates tensions with the most-favoured nation treatment, which requires equal treatment of all WTO members.

Political feasibility

The political feasibility of policy options is inherently difficult to assess, but it can be argued that policies that have the backing of potentially affected countries and/or firms are more politically feasible, as they are viewed as more legitimate. The likely impacts of any of the policy options on major trading partners are therefore an important factor. The potential impacts on trading partners of unilateral trade measures have also sparked fears of trade wars and tit-for-tat retaliation (e.g., Bordoff 2009).

Generally, some of the more severe trade-restrictive measures, such as trade bans or quotas aimed at specific countries, are likely to meet with opposition from affected countries, as such measures leave little room for adjustment to the exporters' national circumstances, and could be seen as imposing the standards of the adopting country on its trading partners (Epps and Green 2010, p.218). For some options, such as BCAs, it is possible to design the measure in such a way that it is only aimed at specific products or countries.

16 For a set of opposing views, see Ismer and Neuhoff (2007), arguing that BCAs can be WTO-compatible, and Quick (2008), arguing that they are likely illegal under WTO law.

17 Note that this is only if a country adopting the measure chooses the strategy to ensure WTO-compatibility through the environmental exception. An alternative strategy, advocated by Ismer and Neuhoff (2007), is ensuring that a measure does not violate the core principles of the GATT, including that of most-favoured nation treatment. A country following this strategy needs to apply its measure across the board by definition.

In the context of emissions embodied in trade, there is a clear North-South dimension, as a large share of the emissions embodied in trade are "exported" from developing countries and "imported" by developed countries (Peters et al. 2012).[18] For instance, if the EU were to adopt BCAs for the sectors that are currently considered at risk of carbon leakage, this would have a significant impact on exports from Egypt, India and South Africa (International Centre for Trade and Sustainable Development 2011; see also Atkinson et al. 2011). In addition, as noted below, administrative costs could be prohibitive for some developing countries (Jensen 2009; Persson 2010). Therefore, any trade measure adopted by developed countries will almost certainly be met with scepticism – or outrage – from developing countries. Developing countries are likely to point to the principle of CBDRRC, and question whether trade measures will be in line with this principle.[19] Another reason why trade measures will be regarded with suspicion is that they open up a possibility for protectionist policies. The effects of protectionism may be felt widely even if policies are targeted at specific products. For instance, BCAs could serve as "blueprints" for similar trade restrictions in other, unrelated policy areas (Horn and Mavroidis 2011; see also Atkinson et al. 2011).

Trade restrictions such as BCAs could become less politically (as well as legally) charged if they were accompanied by a reimbursement mechanism that leaves the principle of CBDRRC intact (Eckersley 2010). For instance, the proceeds from BCA duties paid on imports from developing countries could be sent to those countries, either directly or through an international fund. Such a mechanism would advance the goals of climate protection and a fair international playing field in energy-intensive industries.

The most advanced proposal in this regard has come from Grubb (2011), who argues that a coupling of BCAs with climate finance may be attractive for all parties concerned. For parties wishing to adopt a BCA, taking away the fear of carbon leakage might remove the pressure of continuing the inefficient system of free allocation of allowances under a domestic ETS, while at the same time forming an incentive to increase the ambition of domestic climate policies. Exporting countries – i.e., those targeted by the measures – should in theory also welcome such changes in the policy of the importing country. More importantly, however, they would likely welcome new climate finance generated by such measures.

The granting of trade preferences (e.g. the lowering of tariffs on certain low-carbon products from a group or sub-group of developing countries) could be considered politically feasible, as this option should lead to significant development benefits by reducing the duties that exporting developing countries otherwise would have to pay, which could amount to several billion U.S. dollars per year (Callahan and Vasile 2010). The feasibility depends on the framing of the measure – if developing countries see this as a conditionality, they may still oppose it. Moreover, the measure may still be challenged by those countries not receiving preferential treatment.

Administrative feasibility

In this section we examine the logistics of implementing the various policy measures discussed above, in terms of administrative burdens in the importing country. A key factor will be whether a measure is related to the carbon content of a product (Moore 2011).

Calculating the embodied carbon in primary goods (e.g. cement, paper, steel) is relatively straightforward, even though these goods represent a relatively small share of emissions embodied in imports. Calculating the emissions embodied in downstream goods (e.g. electronics, cars), by contrast, can be incredibly complex. These goods are often assembled from parts stemming from different countries, which would require complex systems to trace the emissions back through the production process. Moreover, although there is increasing convergence on how to determine the carbon content of broad product categories (Peters et al. 2011), it is not clear which methodology would be chosen (Jensen 2009). Still, the administrative challenges should perhaps not be overstated, as customs offices in developed countries have experience with measuring the production processes of various commodities (Ireland 2010). However, this calculation of the carbon content may also require action on the exporter side, and for developing countries this may pose a much more significant administrative burden (Jensen 2009; Persson 2010).[20]

18 This situation is of course not a given and could change dramatically if, for instance, Canada started exporting oil from the Alberta tar sands to developing countries.

19 A prelude to the reaction of developing countries to climate-motivated trade restrictions can be seen in the case of including aviation in the European Union's ETS (Scott and Rajamani 2012).

20 Izard et al. (2010) suggest an additional method to lower administrative burdens, by only requiring information on the embodied carbon at the stage of export of the final product.

One option would be to use emission standards based on either the carbon content of domestic products or that of imported products (Persson 2010). The former includes the "predominant method of production" standard (Biermann and Brohm 2005; Mattoo et al. 2009). The latter has been advocated by Ismer and Neuhoff (2007), who suggest that the standard should be the "best available technology" in an exporting country. However, the standard could also be set for a specific product category of imports (Persson 2010). Although standards make the administrative tasks less complex, their application across the board may discriminate against individual exporters that produce goods with lower emissions. To address this, a solution would be to allow individual firms to prove that their emissions are lower (Persson 2010).

POLICY OPTIONS – SUMMARY

Several policy options exist to address emissions embodied in trade, including by shifting the location of production. Not all of them are especially feasible, however. For example, outright bans or quantitative restrictions on goods from specific countries would be challenging to demonstrate legally and would likely be politically challenging.

Second, there may be tensions related to the legal feasibility of unilateral trade measures. There is a catch-22, where trade measures that are most consistent with the most-favoured nation treatment rule are also most likely to go against the idea of CBDRRC, while measures that differentiate on the basis of climate action undertaken will most likely violate be viewed as discriminatory under WTO law (Eckersley 2010).

Third, measures that are likeliest to be consistent with WTO law are also likeliest to be difficult to administer. WTO consistency requires that policies be well-targeted, and not unjustifiably or arbitrarily discriminate between domestic and foreign products, or among foreign products. This means, for instance, that policies should not favour domestic low-carbon products over "like" foreign low-carbon products. But to achieve this it is necessary to identify the carbon content of products, and each of the existing options to calculate this will pose significant administrative challenges. Moreover, to enhance chances of WTO legality, it is best to apply measures to all trading partners rather than, for instance, the largest exporters of embodied emissions. However, this again adds to the administrative complexity.

Fourth, and related to the previous point, policies that are well-targeted are also more likely to be administratively complex. Moreover, given that a significant amount of emissions embodied in trade concern final products such as cars and electronics, measures to reduce emissions embodied in these products will also pose administrative challenges for both governments and exporting firms.

Fifth, to make policies affecting traded goods more politically feasible as well as more environmentally effective in the long run, it is important that they address the underlying reasons for emissions embodied in trade. These underlying reasons may be high levels of consumption in the importing country (calling for measures aimed at reducing consumption in the home country) or lower levels of financial and technological capacity in the exporting countries (calling for measures aimed at capacity-building, financial and technological support). Reducing or shifting trade as such would not address these underlying concerns. As Peters and Hertwich (2008, p.1405) note, "the challenge for policy is to ensure that countries that specialize in pollution intensive exports do so with clean technology, rather than moving production elsewhere (assuming production can be relocated) or not taking part in a global climate regime". In this regard, policies that recycle revenues of any trade measure back to developing countries may be particularly promising. This idea could make consumption-based measures that generate revenues more politically feasible (and arguably enhance their compatibility with international law), though it has yet to be tested in practice.

6 SUMMARY

In recent years, analysts have increased their attention on GHG emissions embodied in trade, leading some to suggest that decreasing trade will help reduce emissions. Of course, reducing trade will not reduce emissions if domestic production of otherwise imported goods or materials is equally or more GHG-intensive. Furthermore, decreasing trade could reduce incomes in the developing countries that produce goods and which have, in general, lower standards of living than the countries buying their goods (Erickson et al. 2011).

This analysis has explored whether *increases* in trade, and/or shifting the location of global production, could have GHG benefits. We find that the GHG intensity of producing some of the most significant traded goods and materials (e.g., clothing, electronics, vehicles, steel) may vary by a factor of two (vehicles) to over five (clothing) among countries. This finding suggests that shifting production (and trade) away from the more GHG-intensive regions to the less GHG-intensive could have a corresponding benefit, one that would be only slightly reduced by an increase in GHG emissions associated with longer shipping distances.

The actual benefits of shifting production are much more complicated to assess, however, and depend on several factors, including the ability of countries to invest in new, efficient technologies and expand use of low-GHG energy; the availability of raw materials, the opportunity cost of diverting resources (energy, labour, or capital), and relative prices. Our analysis has focused primarily on whether countries could expand use of low-GHG energy and feedstocks, for which we conducted a simplified analysis for the steel sector using data from the IEA. Access to low-GHG energy and feedstocks varies strongly between regions, suggesting a new opportunity to reduce global GHGs associated with steel production by shifting production to the regions with greater future access to these resources.

In the case study for steel, we argued that global steel production could perhaps be made less GHG-intensive if production were to shift to (or new production located in) countries with access to expanded sources of low-GHG biomass (for charcoal-based iron production in a blast furnace) or to natural gas and low-GHG electricity (for production of steel in an electric arc furnace from direct reduced iron). Based on data from the IEA, the Americas (both North and South) score well on these metrics, though further analysis is needed to analyse the availability, economics, and GHG balances of the various inputs.

There are other limitations to our analysis that we must briefly address. For example, our analysis was performed at the level of countries or groups of countries, but within individual countries or regions, there may be similarly large distinctions in current GHG intensity of production and potential for lower-GHG production. Also, national and regional boundaries don't necessarily limit access to resources. Some countries may be able to import energy and feedstocks to support low-GHG production, so regional differences in potential low-GHG steel production, for example, may not actually be as great as implied by our analysis.

Other analyses have devoted significant attention to charting low-GHG transitions for major industrial sectors, especially by international agencies (IEA 2012b; IEA 2009) but also with support from the industries themselves (Müller and Harnisch 2008). These analyses focus on the technologies, fuels, and feedstocks needed to produce low-GHG materials and goods, but they rarely explore regional differences in access to these factors. Our preliminary analysis suggests that more attention may be warranted on expanding production of some materials or products in particular world regions.

Our assessment also considered policy measures that buyer countries could adopt to steer trade – by giving preference to particular countries or world regions, or by favouring products with lower-GHG intensity. There are multiple options, but the effectiveness and feasibility of these measures is highly uncertain. Based on our review, there is one proposal that would use a mechanism that can likely be WTO-legal to address emissions embodied in significant trade flows: Grubb's (2011) suggestion to return revenues from border carbon adjustments to the exporting countries as climate finance. This proposal deserves further research.

Another promising avenue for further research is to develop much more detailed information and analysis on the costs and availability of key resources for low-GHG production in each region, to help develop more specific low-GHG pathways for specific industries. Such an analysis could be combined with country-specific analyses of the feasibility of different policy options. Given the data challenges, data from industry associations may be helpful, as they likely have greater access to information about possible alternative locations of future production.

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