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# Globalization of agricultural pollution due to international trade

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## Abstract

Almost 90 % of freshwater resources consumed globally are used to produce plant and animal commodities. Water scarce countries can balance their water needs by importing food from other countries. This process, known as virtual water transfer, represents the externalization of water use. The volume and geographic reach of virtual water transfers is increasing, but little is known about how these transfers redistribute the environmental costs of agricultural production. The grey water footprint quantifies the environmental costs of virtual water transfers. The grey water footprint is calculated as the amount of water necessary to reduce the concentrations of fertilizers and pesticides released in streams and aquifers to the allowed standards. We reconstructed the global network of virtual grey water transfers for the period 1986–2010 based on global trade data and grey water footprints for 309 commodities. We tracked changes in the structure of the grey water transfer network with network and inequality statistics. Pollution is increasing and is becoming more strongly concentrated in only a handful of countries. The global external grey water footprint, the pollution created by countries outside of their borders, increased 136% during the period. The extent of externalization of pollution is highly unequal between countries and most of this inequality is due to differences in social development status. Our results demonstrate a growing globalization of pollution due to virtual water transfers.

## 1 Introduction

About 90 % of freshwater used by humans is allocated to agricultural production (Rost et al., 2008; Baron et al., 2002; Hoekstra and Chapagain, 2008). Due to the tight link between water and food, water crises are typically characterized by lack of sufficient freshwater resources for food production rather than drinking water (Falkenmark and Rockstrom, 2006). With the populations of some countries already exceeding their carrying capacity (defined by water resources), isolation from the international community

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may induce water and subsequent food crises (Seekell, 2011; Suweis et al., 2013). However, these countries can adjust their local water needs and carrying capacity through the import of plant and animal commodities from other regions. This trade allows countries to “virtually” use water to produce commodities in other countries (Allan, 1998). This volume of water used during the overall process of production of a commodity is defined as a water footprint (Allan, 1998). In this manner, when a commodity is traded between two countries, the water resources associated with that good are transferred “virtually” from the exporter to the importer (Chapagain and Hoekstra, 2008; Hoekstra and Chapagain, 2008).

Recent analyses of virtual water transfers have revealed both positive and negative impacts. Virtual water transfers favor regional mitigation of water scarcity and food security in overpopulated countries, thereby preventing malnourishment and water wars (Allan, 1998). Virtual water transfers also increase global water-use efficiency because commodities can be produced in regions where production water-use efficiency is high, then consumed in regions where the production water-use efficiency is low (Chapagain et al., 2006; Konar et al., 2012). However, virtual water transfers are not directly related to water scarcity and empirical analyses find that wealthy countries have disproportionate access to external water resources relative to less wealthy countries (Seekell et al., 2011). Further, virtual water transfers reduce long-term societal resilience to drought, raising important questions of about the sustainability of the global trade system from a water resources perspective (D’Odorico et al., 2010).

Virtual water transfers disconnect populations from the resources they use by creating geographic separation between production and consumption (Carr et al., 2012a, 2013; D’Odorico et al., 2012). A consequence of this is consumers might not directly be affected by the environmental degradation resulting from the non-local production of the (food) commodities (e.g., soil erosion and pollution from fertilizers and pesticides). In this manner, highly developed countries may be able to externalize pollution disproportionately relative to less developed countries. However, little is known about the

global extent and long-term changes in the externalization of pollution through virtual water transfer (Mekonnen and Hoekstra, 2011).

Here, we reconstructed the global network of agricultural pollution based on international trade records and commodity/nation specific grey water footprints for the period 1986–2010. The grey water footprint, a metric of pollution, is the hypothetical volume of freshwater needed to dilute the pollution caused by the production of commodities (e.g., Hoekstra and Chapagain, 2008; Mekonnen and Hoekstra, 2011). Countries that import grey water externalize their pollution to countries that export grey water (Fig. 1), while countries that export grey water accumulate pollution within their borders (Fig. 1). Patterns of grey water trade relative to previously described patterns in blue (surface water and ground water) and green (soil water) water transfer (Carr et al., 2012a) were explored. We also examined the contributions of trade to inequality in pollution, in terms of grey water use, and evaluated the role of social-development status in influencing that inequality. We found that the externalization of pollution is increasing with the burden of the pollution being concentrated to a handful of countries.

## 2 Methods

### 2.1 Data calculation of grey water footprints

We reconstructed international grey water transfers for commodities for the period 1986–2010 using United Nations trade records (FAOSTAT, faostat.fao.org) and commodity specific estimates of grey water footprints based on nitrogen fertilizer use from Mekonnen and Hoekstra (2010a, b). Commodities used, and political boundary changes during the study period follow Carr et al. (2013).

The total grey water footprint of a country ( $GW_{FT}$ ) was calculated as the sum of the grey water production ( $GW_P$ ) and the net trade of grey water ( $GW_{NT}$ ) which is equal to the import-export balance between grey water imports ( $GW_I$ ) and grey water exports ( $GW_E$ ).

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$$GWF_T = GW_P + GW_{NT} = GW_P + GW_I - GW_E \quad (1)$$

In this manner, the  $GWF_T$  is comprised of both internal (production – export) and external (import) components. Due to inconsistencies between the trade and production data sets relative to re-exportation of commodities, roughly 9 countries per year (<0.05 % of the global grey water footprint) had negative total grey water footprints. These nodes and their associated links were removed from the analysis.

## 2.2 Statistical analysis

Complex network analysis was used to analyze variability in the structure of the grey water trade networks (e.g. Konar et al., 2011; Carr et al., 2012b). Each country defines a node in the network with each trade connection specifying a directed link between nodes (Carr et al., 2012b, 2013). We limited our analysis to the grey water export network as this quantifies the “pollution damage” countries “self inflict” by supplying commodities to other nodes, with network statistics for the import network provided for comparison (Table 1).

For each year, the degree of each node, calculated as the number of active grey water export links, and the global network degree, the sum of all countries’ degrees, were used to describe the level of connectivity in the grey water trade network. Similarly for each year, the individual export strength of each node, the sum of the grey water exports for that country, were calculated along with the total strength of the export network. The relationship between nodal strength and degree was explored by fitting a power-law equation

$$s = ck^a, \quad (2)$$

where  $s$  is node strength,  $k$  is node degree, and  $c$  and  $a$  are fit values (Konar et al., 2011). The value of the exponent describes the structure of grey water exports in terms of clustering and dispersion. Exponents greater than unity indicate clustering of grey

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water trade in the network because, in this case, not only would nodes with higher degrees have more export links but each link would also carry on average a higher volume of virtual water.

Virtual water transfers may exacerbate or reduce inequalities in water use (Carr et al., 2012a). Inequalities in pollution could be associated with ethical concerns, especially if differences in the burden of pollution are driven by social-development factors (Carr et al., 2012a). To this end, we calculated inequality statistics to understand variability in total grey water footprints following Seekell et al. (2011). Inequality was quantified with the Gini coefficient ranging from zero (indicating perfect equality) to one (indicating perfect inequality). Gini coefficients were then decomposed to identify the relative contributions of internal and external grey water use to inequality (Yao, 1999; Seekell et al., 2011). Large contributions of external use indicate that trade is mostly responsible for inequality, whereas large contributions of internal use indicate that within-country consumption (primarily controlled by geographic factors) accounts for most of the international variability in grey water use. Further, the contributions of social development status to inequality were explored by grouping countries together based on social development status (defined here by using the Human Development Index, HDI) into three classes ( $0 \leq \text{HDI} \leq 0.5$ ;  $0.5 < \text{HDI} \leq 0.8$ ;  $0.8 < \text{HDI} \leq 1$ ) and decomposing the Gini coefficient based on these groups (Seekell et al., 2011). The results of the decomposition are measures of the proportional contributions of within- and between-class inequality to the overall measure of inequality. Prior analyses of water footprints found that social development status contributes differently to internal and external water use (see Seekell et al., 2011). As such, this analysis was performed for the years of the study period with both HDI data as well as internal and external grey water footprints. Any countries for which HDI data were not available in a given year were removed from the groupings for that year.

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### 3 Results

#### 3.1 Summary results

In 1986, the United States was the largest exporter of grey water (approximately 23 billion  $m^3$ ). Thus, in the perspective here, the United States was accumulating the most agricultural pollution due to virtual water transfers in comparison to other countries. Similarly in 1986, Japan was the largest importer of grey water (approximately 9.3 billion  $m^3$ ), externalizing the most agricultural pollution through virtual water transfers. While the United States was the largest exporter and Japan the largest importer of grey water in 1986, China had the largest total grey water footprint (166 billion  $m^3$ ), equal to roughly 21 % of the global grey water footprint (793 billion  $m^3$ ) for that year. China's average per capita GWF (152  $m^3$ ), however, was smaller than many countries in the Western Hemisphere (Fig. 2). As such China generated the most agricultural pollution in 1986, but not on a per capita basis.

The global grey water footprint increased substantially over the record (1.26 trillion  $m^3$  in 2010), but the identities of major grey water importers and exporters showed little change (Fig. 3). In 2010, China maintained the largest grey water footprint (386 billion  $m^3$ ) equal to 31 % of the global grey water footprint. However, China's per capita grey water footprint was substantially higher in 2010 (293  $m^3$ ) than 1986, but was still lower than western countries (Fig. 2). The United States remained the largest exporter of grey water (41 billion  $m^3$ ) in 2010, with Germany and China importing the largest volumes of grey water (18 and 15 billion  $m^3$ , respectively).

#### 3.2 Network analysis results

Global grey water trade intensified substantially over the 25 yr record, both in terms of the number of active trade links and the overall strength of the nodes (Fig. 4). There were 205 active trading nodes (countries) and 8644 active trading links in 1986. By 2010, the number of active trading links increased by 70 % to 14 669, but the number

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of active nodes remained roughly the same (208). The total strength of the network (i.e., the sum of all fluxes in the network) increased by 136 % (Fig. 4). The average strength per trade link also increased over this time period from 11.7 million m<sup>3</sup> to 16.3 million m<sup>3</sup>. These results indicate that the global grey water trade network became more connected during the period 1986–2010 (Fig. 4). These changes parallel previously reported long-term trends in total (green + blue) virtual water trade networks (Carr et al., 2012a). This study found a 92 % increase in the number of active trading links and a doubling of network strength, with only a small change in number of active nodes (Carr et al., 2012a).

For the grey water export network, the scaling exponent for the strength-degree power-law equation fluctuated about a mean of 2.52 (standard deviation = 0.11). The scaling exponent for the import network had a mean of 2.70 (standard deviation = 0.10). There was no discernible trend in the exponents; the scaling exponent for the import network is consistently higher than the scaling exponent for the export network. The high exponents (i.e. > 1) show that the average strength of links connected to countries with a large number of trade partners is greater than for countries with fewer trade partners. This finding indicates that the global grey water trade network is clustered in few very well connected countries (“hubs”), and its grey water flows are not spread out amongst many nations with fewer links (Konar et al., 2012). The higher exponent for imports than exports means that as countries increase their number of trade partners, on average, they begin to externalize pollution more quickly than they subject themselves to internal production pollution (cf. Konar et al., 2012).

### 3.3 Inequality analysis results

The Gini coefficient for the total grey water footprint varied between 0.5 and 0.55 (mean = 0.51), with a slight decrease during the 25 yr period due to a decline in inequality in the internal grey water footprints (Table 1). These values are twice the previously reported values (0.226) for total (blue + green) water footprints (Seekell et al., 2011). External grey water footprints were highly unequal (mean Gini coefficient = 0.74), but



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only accounted for 14.1 % of the inequality in the total grey water footprint because external water footprints are small compared internal water footprints and internal grey water footprints had lower inequality (mean Gini coefficient = 0.5). Internal grey water footprints accounted for an average of 85.9 % of the inequality in the total grey water footprint. In other words, agricultural commodities produced and consumed within a country account for most inequality in agricultural pollution between countries.

In 1986, differences in social development status accounted for 49 % of inequality in the total grey water footprint. This decreased substantially over time and by 2010 differences in social development status only accounted for 26 % of inequality in the total grey water footprint. This change is mainly due to changes in internal grey water footprint inequality. Differences in social development status accounted for 40 % of inequality in internal footprints in 1986 but this decreased to only 13 % in 2010. In contrast, social development status accounts for the largest portion of inequality in the external grey water footprint (60 %, on average). The most developed nations had, on average, the highest external grey water footprints.

#### 4 Discussion

The externalization of pollution due to agricultural production is pervasive globally and increased considerably during the period 1986–2010. This is a general result of the intensification of international trade and is consistent with previous reports of blue and green virtual water transfers during the period (e.g., Dalin et al., 2012; Carr et al., 2012b, 2013). As a consequence, grey water transfers will likely increase with projected increases in virtual water transfers (Suweis et al., 2013).

Previous analyses of grey water footprints are limited, and our analysis represents the first description of the structure and dynamics of the grey water transfer network, as well as the first measurements of inequality in grey water use. These grey water footprints proved a conservative estimate of pollution as they were estimated based off of nitrogen pollution (Mekonnen and Hoekstra, 2010a, b) and do not account for

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5 other pollutants. For example, while nitrogen pollution is important for coastal systems, phosphorus is typically considered the key nutrient in degrading of inland surface water resources and is not accounted for in this analysis (Carpenter and Bennett, 2011). The grey water footprint concept can be extended to incorporate these other types  
10 of degradation, but development of methods for doing this are just beginning (Liu et al., 2012). A more serious shortcoming of the grey water concept is that it does not explicitly account for the adverse environmental effects of flow alteration for agriculture. Nonetheless, the results here are useful as a first step in illustrating the extent and structure of the externalization of pollution. The general results in terms of the behavior  
15 of intensification of grey water transfers and externalization of pollution are unlikely to change significantly by inclusion of additional pollutants.

There was substantial between-class inequality in the external grey water footprint. This indicates that social development status contributes to inequality in the externalization of pollution. Specifically, highly developed countries have the highest per capita  
20 externalization of pollution and the least developed countries have the lowest. This is a consequence of the ability of wealthy (highly developed) countries to participate in international trade (Seekell et al., 2011). Less wealthy countries have a restricted ability to trade for commodities and hence a restricted ability to externalize their pollution. The differences in social development status likely form the basis for the different  
25 structures of the grey water import and export networks. Our scaling analyses revealed that more highly connected countries externalize pollution at a greater rate than they accumulate pollution. More highly developed and connected countries likely increase their imports of luxury animal products, which are associated with higher grey water footprints. Hence the differences in network structure are likely due to differences in the consumptive patterns associated with changes in wealth (cf. Carr et al., 2013).

Our results prompt ethical and social questions. For instance, is it okay for more highly developed countries to disproportionately externalize their pollution? In terms of externalizing pollution the answer is in part context dependent, because water resources are unequal in distribution naturally, and because some regions are likely to

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have a greater capacity to buffer pollution than others (Seekell, 2011; Liu et al., 2012; Ridoutt and Huang, 2012). Hence, while measures of inequality are typically made relative to hypothetical cases of complete equality, it is not clear if an equal distribution is a necessary goal in terms of environmental justice.

5 The role of social development status in influencing potential ethical concerns is also complex. Differences in development status could lead to concerns that less developed countries are unfairly targeted with pollution. However, virtual water transfers are generally not an explicit consideration in trade decisions and this suggests that grey water is also not directly considered (Allan, 1998). Further, if the situation were reversed,  
10 would there still be an ethical concern related to development status? We cannot resolve these questions with our quantitative analysis, but we believe that these types of questions are important to global water management and food security efforts.

In conclusion, the water resource degradation associated with the production, and trade of, agricultural commodities has been conservatively estimated by using the concept of a grey water footprint in order to examine effects of globalization and inequality on pollution. As population and demand for food has increased globally, so has the pollution due to agricultural food production. Moreover, trade provides a mechanism that, while allowing for virtual access to the water resources of a trade partner, directly leads to the externalization of pollution. Therefore, because consumers are not completely  
15 affected by the environmental impacts of their choices, they may not feel the need to voluntarily adopt environmentally responsible consumer behaviors. In this sense the externalization of pollution through grey water trade may lead to a loss of environmental stewardship. Importantly, differences in social development status are the source of most of the inequality in the externalization of pollution, raising potential ethical concerns. These ethical questions cannot be answered with quantitative analysis alone,  
20 and development of a social and philosophical understanding of water issues at the global scales stands to enrich our collective understanding of global water resource issues.

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**Table 1.** Gini coefficients for total, internal, and external grey water footprints (GWF).  $G_a$  is the proportion of inequality due to differences within social development classes.  $G_b$  is the proportion of inequality due to differences between social development classes.  $G_a$  and  $G_b$  do not sum to unity because some inequality is caused by overlap between classes (Seekell et al., 2011).

Year	Total GWF			Internal GWF			External GWR		
	Gini	$G_a$ (%)	$G_b$ (%)	Gini	$G_a$ (%)	$G_b$ (%)	Gini	$G_a$ (%)	$G_b$ (%)
1986	0.545	34.1	49.4	0.592	36.2	39.7	0.780	19.0	67.2
1990	0.530	35.9	44.0	0.461	43.3	44.1	0.780	18.8	68.4
2000	0.528	33.2	27.9	0.514	35.0	23.6	0.743	24.4	64.7
2005	0.517	44.2	23.6	0.503	47.3	12.6	0.730	28.9	56.3
2006	0.503	45.9	20.4	0.489	48.6	07.5	0.721	29.4	56.2
2007	0.498	43.2	26.1	0.475	42.3	14.5	0.716	29.0	55.4
2008	0.505	42.8	26.1	0.492	36.7	16.2	0.728	28.0	58.7
2009	0.506	43.1	25.4	0.488	45.9	15.0	0.714	28.2	58.2
2010	0.495	45.9	26.4	0.474	45.4	13.0	0.726	31.0	57.6

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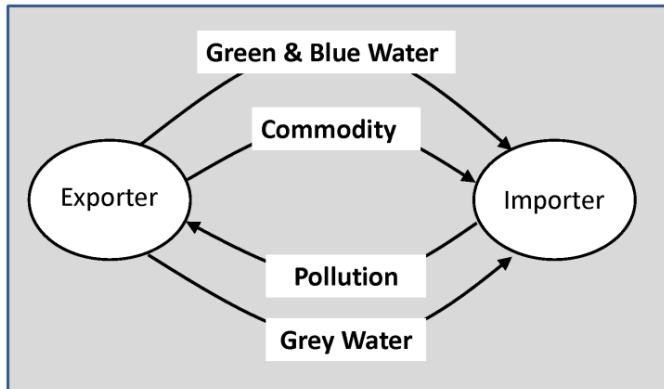


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**Fig. 1.** Green and blue water are virtual water sourced from soil and surface water sources, respectively. Virtual water goes from exporters to importers in the form of commodities, but the pollution due to production stays in the exporting country. Hence exporters accumulate the pollution of importers and importers externalize their pollution at the expense of the exporters.

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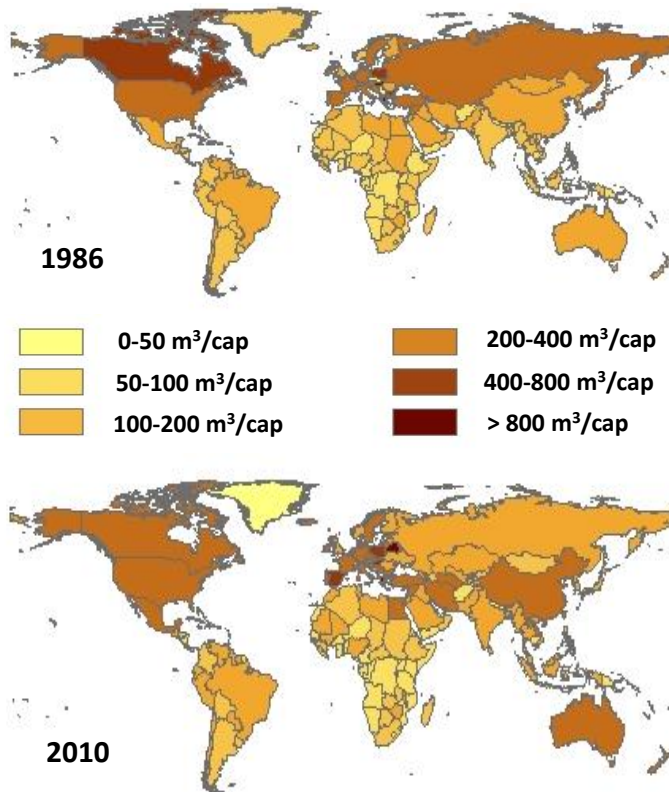


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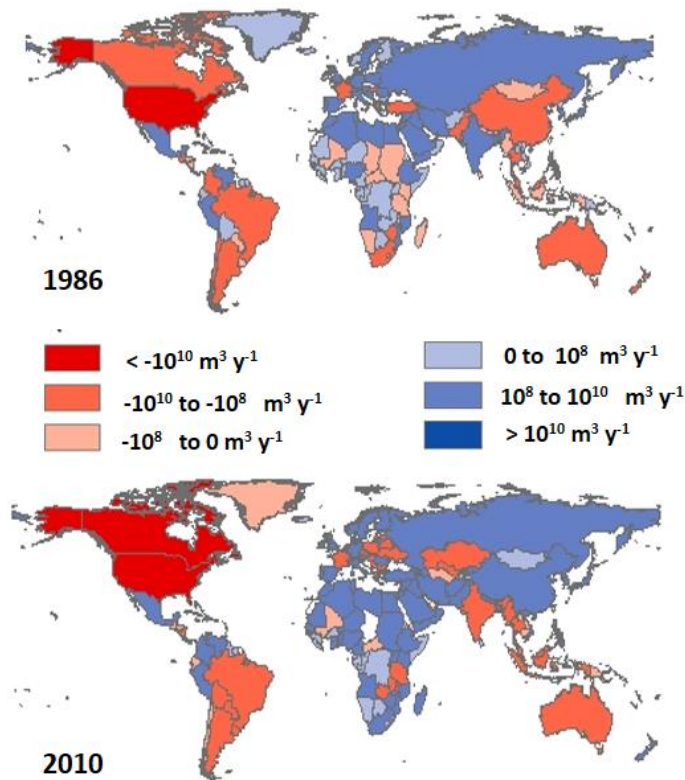
## Grey water trade

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**Fig. 2.** National grey water footprints (per capita) in 1986 and 2010.

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**Fig. 3.** Net grey water trade (imports-exports in  $\text{m}^3$ ) in 1986 and 2010. The red countries are grey water net exporters, meaning that they are burdened with agricultural pollution from virtual water that was subsequently transferred to other countries. The blue countries are grey water net-importers, meaning that they externalize agricultural pollution by importing virtual water from other countries. The identities of large net importers and exporters was relatively static over the 25 yr record.

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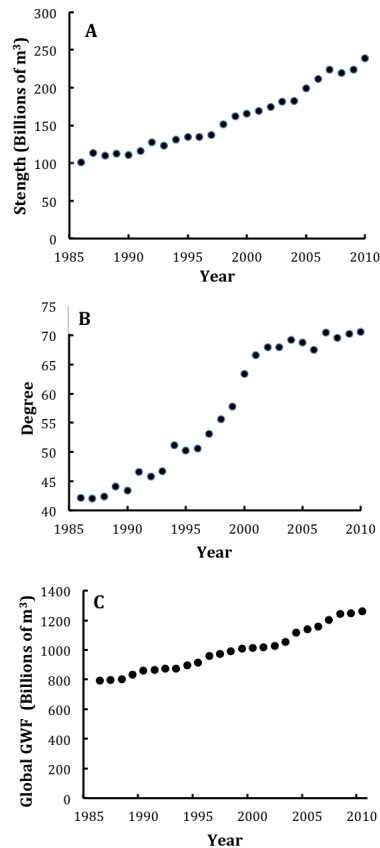
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**Fig. 4.** (A) The total strength ( $\text{m}^3$ ) of the global grey water trade increased over the 25 yr period. (B) The average nodal degree in the network increased over the 25 yr period. (C) The global total grey water footprint increased over the 25 yr period.

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