Global impacts of the meat trade on in-stream organic river pollution
The importance of spatially distributed hydrological conditions

Wen, Yingrong; Schoups, Gerrit; Van De Giesen, Nick

DOI
10.1088/1748-9326/aa94f6

Publication date
2018

Document Version
Final published version

Published in
Environmental Research Letters

Citation (APA)
Wen, Y., Schoups, G., & Van De Giesen, N. (2018). Global impacts of the meat trade on in-stream organic river pollution: The importance of spatially distributed hydrological conditions. *Environmental Research Letters*, *13*(1), Article 014013. https://doi.org/10.1088/1748-9326/aa94f6

Environmental Research Letters

Global impacts of the meat trade on in-stream organic river pollution: the importance of spatially distributed hydrological conditions

To cite this article: Yingrong Wen *et al* 2018 *Environ. Res. Lett.* **13** 014013

IOP Publishing Environ. Res. Lett. 13 (2018) 014013 https://doi.org/10.1088/1748-9326/aa94f6

Environmental Research Letters

OPEN ACCESS

RECEIVED
5 April 2017

REVISED
4 October 2017

ACCEPTED FOR PUBLICATION
20 October 2017

PUBLISHED
11 January 2018

LETTER

Global impacts of the meat trade on in-stream organic river pollution: the importance of spatially distributed hydrological conditions

Yingrong Wen[1,2], **Gerrit Schoups**[1] **and Nick van de Giesen**[1]

[1] Department of Water Management, Delft University of Technology, Stevinweg 1, 2628CN, Delft, The Netherlands
[2] Author to whom any correspondence should be addressed.

E-mail: y.wen@tudelft.nl

Keywords: organic river pollution, international meat trade, spatial distribution, hydrological conditions, water management

Supplementary material for this article is available online

Abstract

In many regions of the world, intensive livestock farming has become a significant source of organic river pollution. As the international meat trade is growing rapidly, the environmental impacts of meat production within one country can occur either domestically or internationally. The goal of this paper is to quantify the impacts of the international meat trade on global organic river pollution at multiple scales (national, regional and gridded). Using the biological oxygen demand (BOD) as an overall indicator of organic river pollution, we compute the spatially distributed organic pollution in global river networks with and without a meat trade, where the without-trade scenario assumes that meat imports are replaced by local production. Our analysis reveals a reduction in the livestock population and production of organic pollutants at the global scale as a result of the international meat trade. However, the actual environmental impact of trade, as quantified by in-stream BOD concentrations, is negative; i.e. we find a slight increase in polluted river segments. More importantly, our results show large spatial variability in local (grid-scale) impacts that do not correlate with local changes in BOD loading, which illustrates: (1) the significance of accounting for the spatial heterogeneity of hydrological processes along river networks, and (2) the limited value of looking at country-level or global averages when estimating the actual impacts of trade on the environment.

1. Introduction

Livestock is a key provider of food, income, employment and nutrients to humans [1]. Already common in developed countries, intensification of livestock farming is growing rapidly throughout developing nations [2]. Besides improving productivity and efficiency, intensification creates additional positive impacts, such as increased profitability and investment in livestock farms and the development of regulations for livestock systems [3]. However, negative impacts on the environment cannot be overlooked, as intensive livestock farming is a major source of global organic river pollution [4, 5]. The discharge of intensive farm effluents rich in organic pollutants contributes to a reduction of river biodiversity and disruption of aquatic ecosystems through oxygen depletion [6, 7]. These polluted effluents also contain pathogens that threaten human health by causing a variety of diseases, including diarrhea [8].

The production of meat products has increased globally in the past half century [9], a trend that is likely to continue due to a projected doubling of both demand for meat and trade in meat in the coming decades, mostly in the developing world, as a result of population growth, urbanization and increased income [10, 11]. As trade in livestock products plays an increasingly significant role in the global food supply, it is crucial to better understand and quantify the environmental impacts of the meat trade and, specifically, the effects on organic pollution of rivers. The overall impact of trade on the environment is considered beneficial if movement of production to exporting countries generates less pollution than domestic production [12]. Due to spatial heterogeneity of farming practices, the availability of natural resources and climate, environmental impacts and costs occur over a range of scales (from national, to regional, to local). Such complexity should ideally be accounted for in any impact analysis.

The impacts of trade in livestock products on alteration of global nitrogen and phosphorus cycles are of increasing importance [13, 14], and have been the subject of several studies. In 2014 O'Bannon *et al* [15] quantified leaching and runoff of nitrogen fertilizers applied to animal feed crops using the concept of 'grey water footprint' (GWF), which is defined as the volume of water needed to dilute river pollutants to comply with existing water quality standards [16]. However, the study did not account for waste produced by livestock, nor did it estimate whether pollution is avoided by trade. Other studies have provided country-level estimates of the effect of international trade in meat products on nitrogen cycling in several major trading countries using the MEAT model, which is a partial equilibrium model that traces nitrogen inputs to different stages of meat production and nitrogen-use efficiency [17, 18]. These studies concluded that current trade contributes to concentrated impacts of nitrogen pollution in meat exporting countries, but did not assess the net environmental benefit of trade.

Footprint studies have highlighted that pollution associated with agricultural export trade often occurs in developing countries [15, 16, 19]. However, empirical work by economists on the role of trade in improving environmental quality has found that trade has an overall positive impact on the environment, because trade liberalization improves the efficiency of natural resource use and lowers peak levels of environmental degradation in developing economies [20, 21]. Economists have introduced the environmental Kuznets curve (EKC) theory, which states that environmental quality deteriorates faster than income at initial stages of economic development and subsequently improves at higher income levels [22]. Income level is used as an overall indicator for all changes, including trade, technological innovation, regulations etc, that accompany economic development [23]. Improvement of environmental quality can be achieved by either (i) importing goods whose production creates pollution (e.g. meat) or (ii) strengthening environmental regulations and investing in local pollution control (e.g. treatment plants). On the other hand, trade may lead to increased pollution in exporting countries. The impact of trade liberalization on the environment also depends on available water and land resources in a country [24]. Understanding the effects of, for example, trade and technological innovation, on the environment under different economic conditions is essential for building better environmental policies and to mitigate and turn around negative impacts.

Important shortcomings of previous studies, using either footprint (GWF) or economic-based approaches, are the lack of consideration for within-country spatial heterogeneity of the impacts of pollution and reliance on pollutant production or pollutant loadings rather than concentrations in streams. Indeed, in-stream concentrations are arguably more direct indicators of pollution since they include the effects of dilution by natural runoff and degradation by micro-organisms. These effects are significant for evaluating the real risk that humans would face due to changes in trade and climate.

In this paper we develop and apply a new method for quantifying the impacts of the international meat trade on organic river pollution by computing spatially distributed organic pollution in global river networks with and without a meat trade, where the without-trade scenario assumes that meat imports are replaced by local production.

2. Method

We quantify the resulting changes in biological oxygen demand (BOD) emissions neglecting any subsequent effects of trade restriction, such as changes in the efficiency of meat production and wastewater treatment. We then combine global gridded data on hydrology and country-based meat trade data to estimate the impacts on global patterns of BOD loadings into rivers and in-stream BOD concentrations along rivers. Finally, we compare the environmental impacts of the with- and without-trade scenarios using EKC theory.

The novelty of this paper is threefold: (i) it evaluates the impacts of trade in terms of freshwater dissolved pollutant (BOD) concentrations, as opposed to GWF or pollutant release into the environment, which are incomplete and indirect measures of freshwater pollution; (ii) it considers spatial heterogeneity of land and water resources, as opposed to country-level assessments reported in the economics and EKC literature; and (iii) it focuses on organic river pollution as opposed to nitrogen and phosphorus emissions from fertilizer and pesticide use that have been the focus of GWF studies.

2.1. Country-level BOD production from intensive livestock farming

Our analysis focuses on intensive livestock farming and trade in pig, chicken and cattle meat (including buffalo), which together make up almost 93% of total global meat production, with more than half of pig and poultry meat coming from intensive (industrialized) farming systems [25, 26]. Given this focus, we do not account for BOD production in extensive farming systems, i.e. low-density animal farms, BOD production from animal feed crop farming and BOD production during meat processing. In our calculation, BOD loadings from intensive farms were considered as point sources and treated via wastewater treatment plans before entering rivers. Previous studies on pollution from livestock farming areas suggest that BOD pollution from extensive systems only becomes important

Table 1. Data provided by the FAO for estimating model inputs [9, 27].

Parameter	Symbol/value	Units
Livestock animal population raised in intensive farming systems	P_{trade}	Head
Yield of meat production quantity and slaughtered animals[a]	Y	Ton head^{-1}
Quantity of meat produced	Q_{pr}	Ton
Quantity of imported meat[a]	Q_{im}	Ton
Quantity of exported meat[a]	Q_{ex}	Ton
BOD production rate of buffalo and cattle	4×10^{-4}	Ton BOD/head day^{-1}
BOD production rate of pig	2.3×10^{-4}	Ton BOD/head day^{-1}
BOD production rate of chicken	8.3×10^{-6}	Ton BOD/head day^{-1}

[a] Data in trade and yield of pig and chicken meat are defined as meat with bone in, with nine-item code 1035 and 1058; cattle and buffalo meat include meat with and without bone, with combined item code 2731.

under high-rainfall conditions [7]. During dry periods, BOD from extensive systems is diffused in the soil, which acts as a buffer zone, and does not end up in streams [7]. Seasonal or short-term effects from extensive systems are of less interest in our research, which considers a steady-state, temporally averaged situation.

Since trade data are available by country, we first evaluate the effect of trade on changes in livestock population and BOD production in intensive farming systems at the country level. When there is no meat trade, local meat consumption in a country will need to be completely satisfied by local meat production. Hence, in such a scenario, and assuming consumption is constant, a net importer (exporter) will see a corresponding increase (decrease) in local animal and meat production. This leads to the following relation for each animal type (pig, chicken, cattle), country and year:

$$P_{\text{notrade}} = \max\left(0, P_{\text{trade}} + \frac{Q_{\text{im}} - Q_{\text{ex}}}{Y}\right). \tag{1}$$

Here, P_{trade} and $P_{\text{no trade}}$ are the number of livestock animals (in units of head) slaughtered for meat in intensive farming systems with and without trade, respectively. Q_{im} and Q_{ex} are quantities (ton) of meat imported, respectively exported [9], and Y is meat yield per slaughtered animal (ton head^{-1}). This relation assumes that traded meat originates from intensive farming systems, a reasonable assumption since intensification of a single commodity typically leads to better market access [27]. An exception occurs for large net exporters (negative $Q_{\text{im}} - Q_{\text{ex}}$) with a significant portion of export originating from extensive farming systems (small P_{trade}), as is the case for beef in Argentina for example [9]. In such situations, the max( ) operator in the equation ensures that the computed number of livestock animals in intensive farms without trade is zero rather than negative.

Next, livestock numbers are multiplied by BOD production rates (ton BOD/head day^{-1}) to obtain annual country-level BOD production (ton) from livestock farming. BOD production rates vary by animal type, age, diet and other factors: here, we use average values based on livestock manure production and characterization, while global data on P_{trade}, Q_{im}, Q_{ex} and Y for the year 2000 were obtained from the FAO (table 1). Tropical livestock units (TLU) are used for livestock biomass; one TLU is equivalent to 250 kg, where one cattle or buffalo is equivalent to 1 TLU, a pig is equivalent to 0.3 and a chicken is equivalent to 0.01 [27].

2.2 Gridded BOD loading to river networks

Estimation of the BOD loading to river networks is based on the spatial distribution of river networks, urban population, intensive livestock systems and wastewater treatment. Gridded river networks were derived from a global drainage direction map (DDM 30) with a spatial resolution of 0.5° [28]. In our calculation, we identified rivers as grid cells with runoff exceeding 3 mm year^{-1} [29].

Wastewater discharge from domestic sewage and intensive livestock farms is considered as the main source of organic pollutant loads into rivers in our calculation [5]. The gridded urban population and livestock numbers were multiplied by BOD production rates to obtain gridded BOD loading into global river networks. Country-average urban daily BOD production rates (g person day^{-1}) are available from the US EPA [30]. These numbers were applied to the global gridded map of urban population in the year 2000 to estimate urban BOD production [31, 32]. Mean BOD production rates by livestock are based on manure production and characterization, namely 400, 233 and 8.3 g stock day^{-1} for daily BOD production of cattle (including buffalo), pig and chicken, respectively [5].

Gridded data of livestock numbers for intensive farming systems were obtained from the FAO for the year 2005, and adjusted to match national estimates for reference year 2000 [9]. For chicken and pig, these were available at a spatial resolution of 3′ [27]. For cattle/buffalo production systems, gridded data of total livestock numbers were converted to corresponding maps of intensive livestock numbers by using threshold animal densities which varied by region [27]. The calculation is implemented on a 0.5° grid while livestock maps are available at a finer resolution, i.e. 5 km, and only main rivers are accounted for in a 0.5° grid cell. Thus, urban regions and intensive farming areas that are within a distance of 5 km from a major river were included in the calculation.

Since some of these effluents are treated before being discharged, the final numbers were multiplied by wastewater treatment fractions and efficiencies.

Treatment fractions of urban domestic wastewater were derived from percentages of the population connected to different treatment types and percentages of the population living in cities in the year 2000. Country-scale or downscaled data for domestic wastewater treatment systems were used [33–38]. Intensive livestock farming is considered as a manufacturing activity and wastewater treatment fractions vary by region [39]. The overall values for wastewater treatment efficiencies were estimated as a weighted fraction of zero (no treatment), 25% (primary treatment), 85% (secondary treatment) and 99% (higher treatment) [40]. Due to a lack of available data, all livestock farming treatment is considered as a manufacturing activity and assumed to have 85% efficiency [41]. The assumption was based on the following considerations. (i) Manufacturing activity is subject to secondary or higher treatment, at least in Europe [41]. (ii) In Asia, wastewater from large livestock farms is either diluted for irrigation or processed through (an) aerobic treatment plants, which roughly corresponds to secondary treatment levels [42, 43]. (iii) Another important input into the calculation relates to the wastewater treatment efficiency numbers for livestock farming used in the analysis, as these directly affect how much BOD load enters the river network. For historical conditions, these numbers were based on wastewater treatment data from manufacturing activity [39], and were shown to adequately reproduce historically observed BOD concentrations. Fortunately, regions with sparse data (Africa, South America) also have low rates of wastewater treatment (from 6%–20%) [5], so that computed results are relatively insensitive to assumed efficiencies. For example, the number of polluted grid cells in Africa and South America increased by just 2% when assuming only primary instead of secondary treatment in these regions.

The above data are for the current existing situation with trade. To obtain the corresponding gridded numbers for the no trade scenario, we took country-level computed values of $P_{\text{no trade}}$ (see the previous section) and downscaled them to individual grid cells using the same spatial distribution as for the current situation. Our computed results are affected by various assumptions. Many of these have been previously validated by an elaborate comparison with observed BOD concentrations reported in Wen *et al* [5]. However, assumptions related to the hypothetical no trade scenario cannot be validated in this manner. An important assumption relates to the spatial distribution of livestock farming. Specifically, our computed results rely on the assumption that the spatial distribution of livestock production within each country does not change between the two scenarios (trade/no trade). Is that a good assumption? Locations of intensive livestock farming depend on many characteristics, including land cover, human population distribution and accessibility to markets [27]. If these factors do not change then it seems likely that major shifts in the spatial distribution of livestock farming should not be expected either, unless increases in livestock production due to trade restriction are such that livestock densities become unrealistically large, thereby requiring an expansion of the intensive livestock area. Supplementary figure S4 available at stacks.iop.org/ERL/13/014013/mmedia shows the frequency distribution of livestock density values in the two scenarios: while densities increase for the no trade scenarios, especially for chicken, the numbers are within the range of the current situation, as also shown by intensive livestock density maps from the FAO (figures S1 and S2).

2.3. Gridded BOD concentrations in river networks

As a final step, in-stream BOD concentrations along river networks at a spatial resolution of 0.5° were calculated as a function of the accumulation of BOD loading, from both urban and intensive livestock areas, wastewater treatment, transportation, dilution and natural degradation.

Estimation of in-stream BOD concentrations is based on a local mass balance that relates upstream and downstream river segments [5, 44]. Assuming stream and wastewater discharge are at steady state, the instantaneous mixing of all upstream flows in one river segment drains into the downstream segment with simultaneous first-order natural degradation. The instantaneous mixing BOD concentration C_i in river segment i (mg l^{-1}) is calculated as

$$C_i = \frac{\sum C_{\text{up}} Q_{\text{up}} \mathrm{e}^{-k(T)t_{\text{up}}} + E_{w,i}}{Q_i} \tag{2}$$

where Σ indicates a summation of degraded BOD concentrations from upstream segments 'up', corrected for in-stream biodegradation affected by travel time t_{up} (day), Q_i is river discharge in segment i (l day^{-1}), k is the first-order degradation rate coefficient, which depends on temperature T and is assumed to have a constant value of 0.35 day^{-1} for in-stream BOD degradation, and $E_{w,i}$ is BOD load generated in the current river segment i (mg day^{-1}).

While the model, due to its global extent, necessarily omits important local processes, such as industrial pollution and eutrophication, for which consistent global data are not available, comparison between simulated and observed in-stream BOD concentrations in Wen *et al* [5] shows a 94% match across BOD concentration categories, which suggests that, globally, the model accounts for the main first-order processes of BOD pollution. We also emphasize that the model considers temporally averaged steady-state conditions and thus is not meant to represent short-term or even seasonal dynamics.

Resulting in-stream BOD concentrations were computed for the trade and no trade scenarios using the respective BOD loading data, as described in the previous section, and with all other model inputs identical, thus allowing quantification of the impacts on river water quality due to international meat trade. The

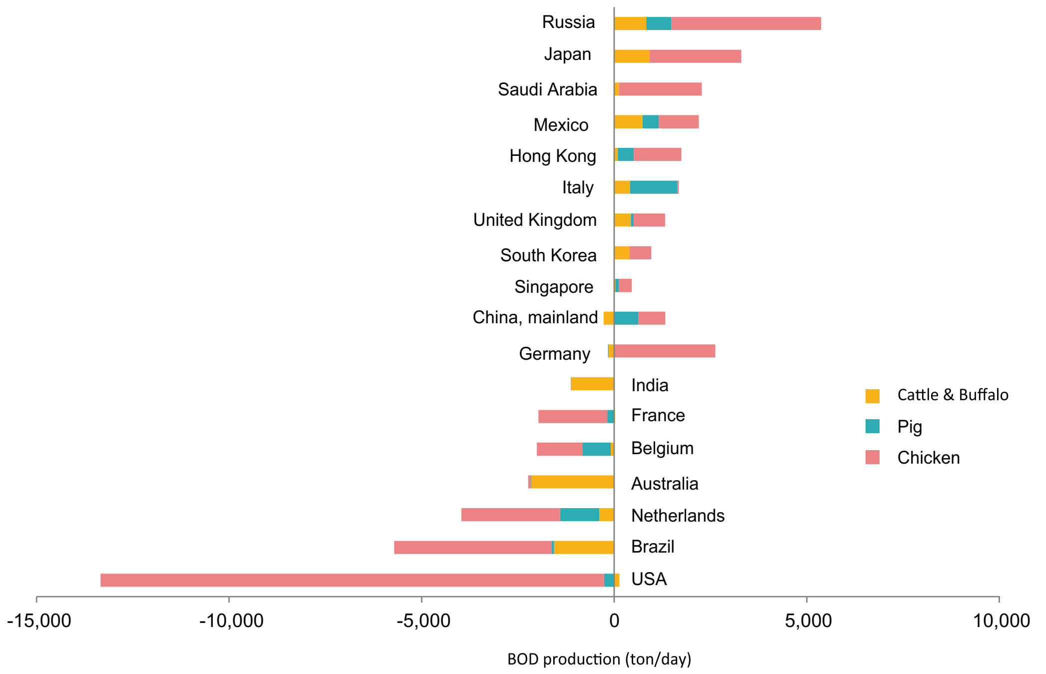

Figure 1. Country-based changes in BOD production from intensive livestock farming if all meat consumption were produced locally (no trade). Net importers (exporters) experience an increase (decrease) in BOD production.

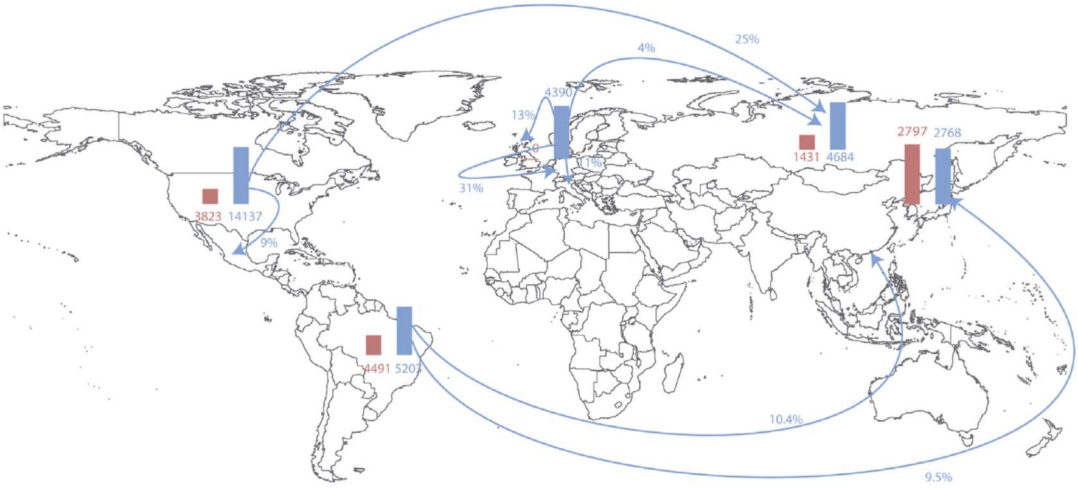

Figure 2. BOD production rates (in ton day^{-1}) associated with intensive livestock farming in selected countries. Red bars indicate local BOD production rates for meeting domestic meat consumption, while blue bars show virtual BOD production rates associated with meat trade, as indicated by blue arrows (with percentages showing shares of total exports going to different countries).

relatively coarse spatial resolution of the calculations (0.5° grid) was locally refined to a 5′ grid for Singapore and Hong Kong due to their small land area yet large population and meat demand [47–52].

3. Result

3.1. Global impact of meat trade on BOD production

Figure 1 highlights countries where BOD production increases or decreases due to hypothetical restriction of the meat trade. Net importing regions, including Russia, Japan and Saudi Arabia, would experience net increases in BOD production to meet their domestic demands. This includes countries with relatively small land areas compared with their meat demand, such as Singapore and Hong Kong. For large exporters, such as the USA, Brazil, the Netherlands and Belgium, the opposite applies. The large decrease in BOD production in the USA due to chicken farming, mostly for export, especially stands out. For countries with both significant imports and exports, like China and Germany, the impacts are mixed. In China, although domestic livestock production is large, additional imports are needed to meet its large demand for pig and chicken meat.

Tracing the flows of BOD production rates associated with trade, figure 2 shows changes in BOD

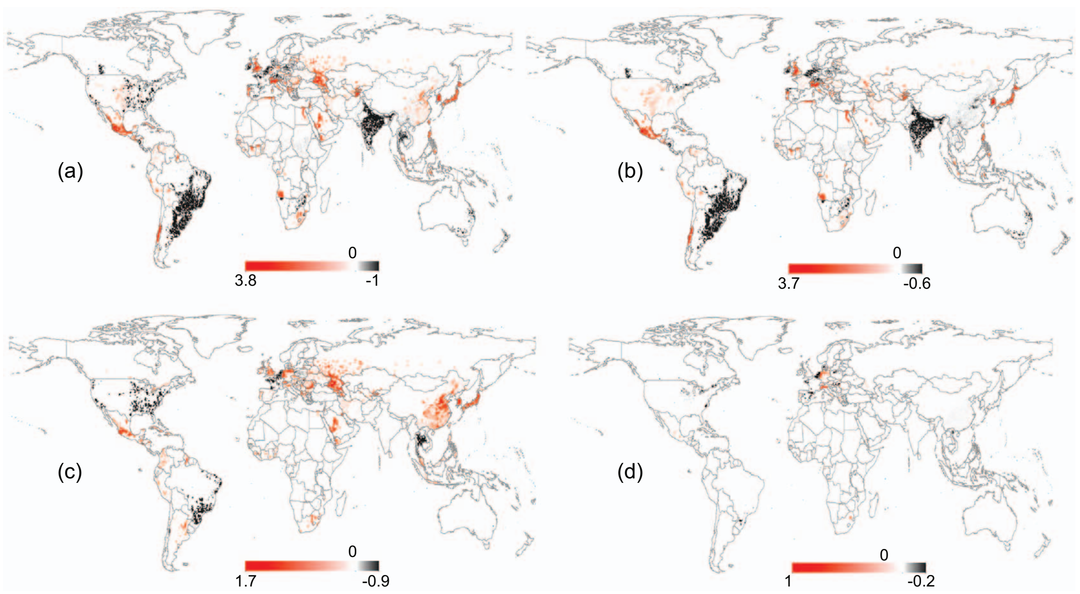

Figure 3. Changes in BOD loading to global river networks from intensive livestock farming systems (ton × 10^8 day^{-1} per grid cell) if all meat consumption were produced locally (no trade): (*a*) accounting for all intensive livestock farming systems; (*b*) intensive cattle and buffalo farms only; (*c*) intensive chicken farms only; (*d*) intensive pig farms only.

production for several bilateral trade relationships. For example, local meat demand in the Netherlands can be completely met by extensive livestock farms, which means that, in the absence of trade, current intensive BOD production in the country would disappear and instead would move to the main importers of Dutch meat, i.e. Germany (31%), the United Kingdom (13%) and Italy (11%). Both Brazil and the USA have large domestic BOD production rates, yet even larger exports (Brazil: 10% to Hong Kong, 9.5% to Japan; USA: 25% to Russia, 9% to Mexico). Russia imports large amounts of meat, corresponding to virtual BOD imports at a rate of 4684 ton day^{-1}, compared with domestic BOD production equal to about one-third of this value.

3.2. Global impact of meat trade on BOD loading to river networks

Figure 3 presents changes in BOD loading from intensive livestock farming into rivers between the trade and no trade scenarios. It is assumed for each country that changes in livestock production occur uniformly over all grid cells that currently have livestock farms. As such, country-level changes translate into changes in livestock production along rivers, which are relevant for our calculation. These maps highlight the spatial heterogeneity of potential impacts on river systems, resulting from spatial patterns of intensive livestock farming systems (figures S1 and S2) superimposed on changes in intensive livestock populations when trade is removed. BOD loading typically increases in net importing countries with high proportions of intensive farming, for example Russia, South Korea, Japan and Mexico, resulting in significant growth of livestock populations along major rivers. Major meat exporting countries like Brazil, Argentina, India, Belgium, the Netherlands and Australia show the opposite trend, suggesting potential benefits for river health by terminating meat trade.

Changes in the USA, Germany and China are mixed due to their dual roles as both importers and exporters of meat, combined with the uneven distribution of intensive livestock farms in these countries. For the USA, increases in BOD loading dominate in the middle and western parts of the nation, home to intensive cattle farms (figures S1 and S2) [9], which increase their meat and BOD production in the absence of trade, since the USA is a large importer of cattle meat with 16% of global cattle meat trade. As a large exporter of chicken and pig meat, mid-eastern parts of the USA experience a decrease of organic pollution threats in the absence of trade.

3.3. Global impact of meat trade on river BOD concentrations

In this section we evaluate the impact on river BOD concentrations due to trade-induced changes in BOD loading. The calculation accounts for upstream–downstream effects as pollutants are transported through the river network with spatially varying degrees of dilution by natural runoff and natural degradation by micro-organisms [5]. The results are computed and presented in figure 4 on a global 0.5° grid map, i.e. approximately 60 km. Each colored 'point' in figure 4 presents a grid cell with a calculated change in river BOD concentration due to the removal of trade. These changes are largely congruent with corresponding changes in BOD loading from intensive farms (figure 3). Regions with a large increase in BOD loading, for example the United Kingdom, Italy, Mexico, South Korea and Japan, exhibit remarkable deterioration of

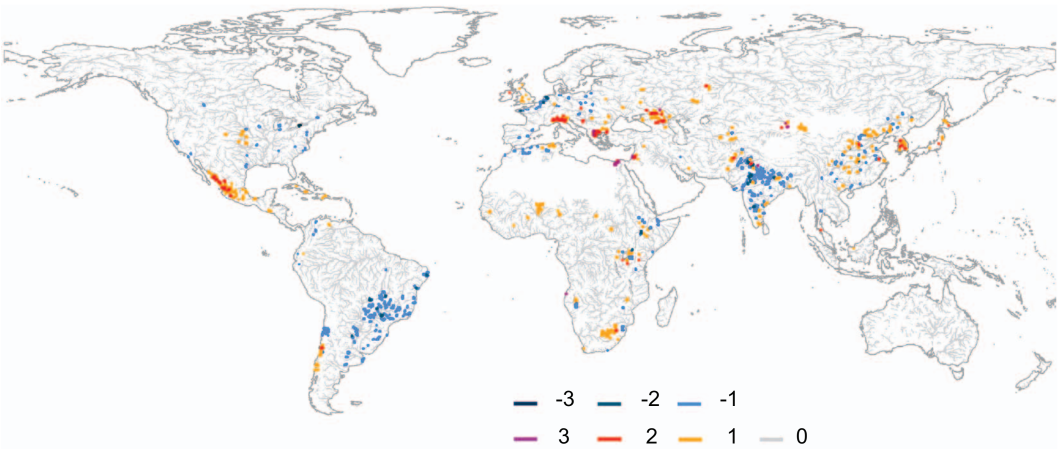

Figure 4. Changes in organic river pollution in the absence of meat trade, i.e. when all animals are raised locally in the country where they are consumed. Positive (negative) numbers indicate an increase (decrease) in BOD pollution level on the following categorical scale: BOD < 5 mg l^{-1}, $5 <$ BOD < 10, $10 <$ BOD < 30, BOD > 30 mg l^{-1}. For example, a value of +2 (–2) means pollution levels worsened (improved) by two categories.

Table 2. Global impacts of meat trade on BOD production, loading to major rivers, polluted river segments and affected human population.

	Trade	No trade	Change (%)
Intensive cattle and buffalo population ($\times 10^7$ head)	4.3	5.1	18
Intensive pig population ($\times 10^8$ head)	5.1	5.2	0.4
Intensive chicken population ($\times 10^{10}$ head)	1.2	1.3	5.5
Total BOD production from intensive livestock farms ($\times 10^5$ ton day^{-1})	2.4	2.5	3.9
Total BOD loading into major rivers from intensive livestock farms ($\times 10^5$ ton day^{-1})	1.09	1.11	2.5
Polluted river length (km)[a]	115928	110166	–2.3
Population affected by organic pollution (billion)[b]	1.14	1.09	–3.9

[a] Sum of all river segments with BOD > 5 mg l^{-1}.

[b] Sum of all people living in grid cells with polluted river segments.

water quality, while regions with a decrease in BOD loading, for example India, the Netherlands, Brazil and Belgium, experience improvement in river water quality. However, increases in BOD loading do not always translate into more pollution, as is evident for rivers in eastern Australia, New Zealand and the Philippines, where river cleaning capacities, consisting of dilution and natural degradation, are sufficiently high to assimilate increased loading without a significant effect on river BOD concentrations [29].

Figure S3 further highlights the poor correlation ($R^2 = 0.019$) between local BOD loading and river BOD concentrations, clearly showing the importance of quantifying actual pollution levels as opposed to local pollution loading, by taking into account the intricate spatial patterns of pollutant loading, upstream–downstream transport, dilution and natural degradation along river networks.

While the results in figure 4 clearly reveal a large spatial variation of the impacts of trade between and within countries, it is also instructive to look at overall global effects of trade. Table 2 summarizes various measures at the global scale. First of all, even though total meat consumption is constant between the two scenarios, international meat trade leads to a lower total livestock population and consequently lower total global BOD production. Table 2 further shows that actual total BOD loading into major rivers, in both scenarios, is less than half of total BOD production due to wastewater treatment and exclusion of BOD generated far from rivers, with slightly less overall BOD loading due to trade, although the difference from the no trade scenario is smaller than for BOD production. Interestingly, the effect of trade on actual river pollution and numbers of people affected is exactly the opposite. Although the differences are relatively small, trade overall results in more pollution relative to no trade. This again underscores the importance of considering actual impacts rather than changes in pressures.

Figure 5 presents a further breakdown of these results for major meat trading countries. In line with the numbers reported in figure 1, population fractions affected by polluted rivers increase (decrease) for meat importing (exporting) countries when trade is removed. However, the results in figure 5 are also affected by the spatial distribution of water resources, livestock farms and urban centers. In the early stages of industrialization, large-scale intensive farms emerge close to cities driven by a series of factors such as expanding labor demands, quicker access to urban demand centers and availability of food preservation [2]. This proximity of intensive farms to cities in meat exporting regions with insufficient technology, such as southern Brazil and the Ganges river basin in India,

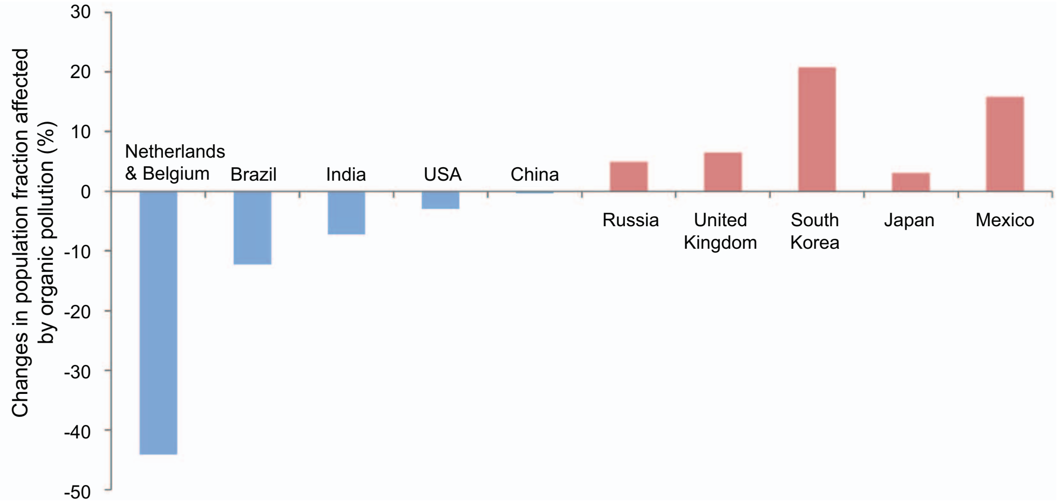

Figure 5. Changes in population fractions affected by organic river pollution in major meat trading nations.

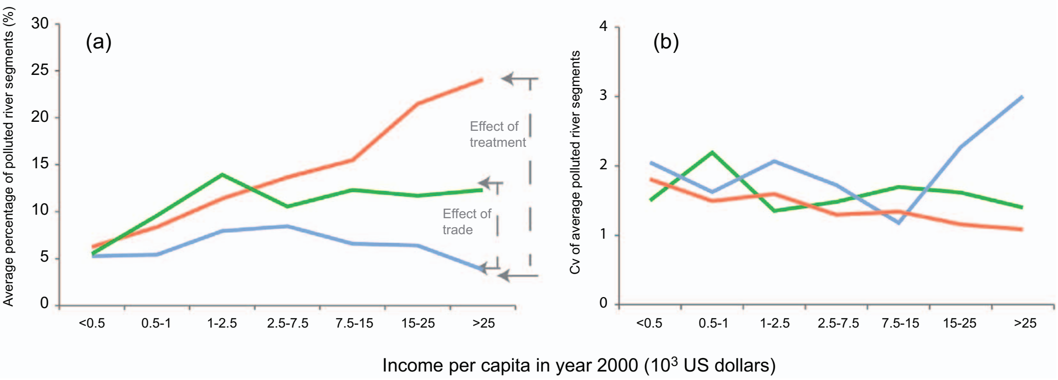

Figure 6. Left: country-wide percentage of polluted river segments (blue, trade and wastewater treatment; green, treatment only; red, trade only) as a function of per capita income in the year 2000 for countries grouped into World Bank income categories [53]. Right: coefficient of variation of environmental impact within each income category; countries within each category are listed in supplementary table S1.

as shown in figure 4, contributes greatly to organic river pollution and affects on populations. Our results show that when the corresponding livestock animals are raised in importing countries with a lower population, such as Russia, higher technology, such as Japan, or less intensive farms along rivers, such as Iran, it leads to a reduction of negative impacts.

3.4. Links between trade, economic growth and river pollution

Figure 6(*a*) presents the computed relation between country-wide average organic river pollution in terms of fraction of polluted river segments and the level of economic development in terms of per capita income for the trade and no trade scenarios. Organic river pollution levels with trade (blue line in figure 6(*a*)) follow an inverted-U shape with respect to economic growth (income), in line with the hypothesis of the EKC. The corresponding curves without trade (green line) and without wastewater treatment (red line) are also shown in figure 6(*a*) for comparison. Hence, the EKC records the combined effects of international trade and technological investment in wastewater treatment for reducing environmental impacts.

For poor nations, differences between the effects of international meat trade and wastewater treatment are relatively small due to low consumption of meat and pollutant loading. Technological investment in wastewater treatment increases as a country's economy grows, hence wastewater treatment has a bigger effect on the EKC than trade in rich countries. Global meat trade also improves environmental impacts in some rich nations via externalization of organic pollution to other economies. However, high standards of food safety and quality in rich nations result in large portions of the meat trade being between developed countries. For instance, approximately 94% of pig meat imported into the United Kingdom is from western and northern Europe [9].

Two important remarks apply to the EKC analysis, however. First, despite the fact that overall there are fewer polluted river segments in the absence of trade (table 2), the no trade curve in figure 6(*a*) lies everywhere above the curve with trade. This contradictory result is due to averaging pollution impacts over large and small countries within each income category. Specifically, regions with very limited land and natural resources, such as Singapore and Hong Kong, are unlikely to produce meat efficiently and largely rely on imports [17]. Given limited water resources, growing livestock locally for expanding meat demands in these small polluted countries has a relatively large influence when averaging over countries in each income category. Second, as shown in figure 6(*b*), variation of pollution impacts within each income category is large, reducing the significance of the effects of trade and wastewater treatment depicted in figure 6(*a*).

4. Discussion and conclusion

In line with previous environmental assessments of global water and land footprints of international trade in agricultural products [16, 54], we find that at the global scale trade is beneficial by reducing livestock populations and pollutant loads. The export of meat from a production-yield efficient region to an inefficient region saves BOD production globally. On average, yield of meat production (variable Y in equation (1)) in exporting countries is higher than in importing countries [9]. Intensification of livestock farming is not only characterized by high animal population density but also by greater input of protein-rich and high-energy animal feeds [9]. Non-ruminants, pigs and chickens, have the advantage of making use of feed concentrates to improve livestock growth rates and yields of meat production. The better feed conversion for ruminants, cattle and buffalo, is limited to developed countries with low grain–meat price ratios [2]. Chicken meat exporters like the USA and Brazil also produce protein-rich animal feeds, such as cereal and soybean, which results in higher yields of meat production (20% in Brazil and 29% in the USA) than their trade partner Russia. Eventually, meat yields may adjust to the new no trade situation [55], but these subsequent effects are not considered here. As agricultural intensification, food consumption and international trade are projected to increase globally, this effect will become more significant in the future.

However, in terms of BOD concentrations, the overall extent of downstream impacts becomes more severe with trade, illustrating the importance of distinguishing between potential impacts of BOD production and loading and actual environmental impacts as BOD concentrations. While pollutants discharge at distinct locations along rivers, impacts extend to downstream populations and ecosystems and depend on the self-cleaning capacities of rivers, local hydrological characteristics and upstream–downstream connections. As a result, it was found that the correlation between computed local BOD loading and in-stream BOD concentrations is poor.

The main message of our paper is that impacts should be assessed in a spatially distributed manner, which is exactly what our approach does. Comparison of impacts by country, by income category or even globally results in a loss of spatial information, underscoring the need to account for within-country spatial patterns when quantifying the effects of international trade on water quality [12]. Global and country-scale aggregation hides the spatial heterogeneity of impacts. For example, even though the globally averaged effect of trade is relatively small, due to canceling of positive impacts in some regions and negative impacts in others, local impacts are substantial, as shown by our global maps. Intensive agricultural systems are generally concentrated in areas with good soil and water resources [56], and thus it is important to account for spatially detailed environmental vulnerability in trade analysis, especially in large and heterogeneous countries such as the USA.

In contrast to atmospheric pollution, river pollutant concentrations do not mix globally, and thus assessing *local* impacts on human health and river biodiversity is important [6]. That is not to say that local impacts only have local causes. Indeed, our study shows that non-local factors play an important role, namely via upstream–downstream effects, as accounted for here by considering BOD solute transport in river networks, and via trade between countries and its effect on livestock production (virtual pollution).

Averaging trade impacts by income category, as done in our EKC analysis, skews the effect of trade on pollution due to the relatively large influence of small countries in EKC analysis. It also hides a large variation in the impacts of pollution within each income category. Still, the EKC analysis helps to distinguish the effects of different economic variables, namely trade and wastewater treatment, in different economic groups, and influences in small countries can be highlighted that are easily neglected on a map. Rich nations tolerate relatively high levels of organic river pollution threats, but mitigate the negative impacts by importing environmentally damaging products and treating symptoms. The resulting contributions to the downward-sloping portion of the EKC of these combined effects have previously been shown for energy use, water quality, including BOD, and air quality in developed countries [57, 58]. Environmental externalities do not bring environmental benefits for poor exporting countries. Developed countries move the pollution of intensive livestock farming away from human populations by enhancing environmental regulations, shifting farms further away from populated centers and investing in wastewater control. Such actions rely on substantial financial investments that are not within reach of many developing countries. Current affordable solutions for improving environmental impacts include tying livestock densities to availability of the surrounding environment for waste application, i.e. waste reuse [18], improving animal feed-conversion efficiencies [26], and getting compensation from EU importers for raising investment for increasing food safety and farming sanitation in Brazil [59].

The analysis presented here can be extended in several directions. First, while the range of traded meat products is quite diverse, our current study only takes single raw meats into account. An obvious extension is to include other meat products as well, which may alter the magnitude and pattern of in-stream river pollution. Second, our calculations only account for BOD pollution linked to direct discharge of wastewater from cities and from intensive livestock production units. Livestock systems also induce additional sources of pollution, for example via animal feed production and slaughtering of animals [17]. There is also an indirect link to nutrient loading, specifically nitrogen and phosphorus, which may lead, via eutrophication, to increased BOD concentrations in rivers. These dynamics of BOD pollution would further extend the steady-state calculations used here. Pollution during the live animal stage on the other hand enters into rivers as point sources and its loading depends more on technological investment rather than seasonal runoff in our calculation. Seasonal or shorter-term effects on organic pollution from non-point sources, which are affected by precipitation, secondary effects, such as eutrophication, and buffer effects of soil and vegetation, should be accounted for in future studies. A fourth extension would be to also consider pollution of coastal waters and of groundwater, thereby including impacts in countries that play significant roles in the international meat trade but have limited surface water resources, like Saudi Arabia. Last, but not least, an interesting topic for future work is to look for 'optimal' trade scenarios that selectively move livestock production from more to less vulnerable regions with the aim of minimizing global impacts.

As consumption of meat by humans continues to increase due to population growth and changes in diet, reducing the impacts of intensive livestock farming requires globally aware policies for changing meat consumption patterns. One option to curb the current global trajectory of livestock intensification is to more accurately reflect actual environmental costs in meat prices [17], even, or especially, if these costs are incurred in places far away from consumers. Our study provides a new method for estimating shifts in pollution patterns between countries due to changes in consumption habits, trade and environmental policies. Its particular strengths are: (i) quantification of actual concentration in addition to potential loading impacts, and (ii) accounting for spatial heterogeneity of processes and impacts. We believe these two key features should also form the basis of other environmental analyses related to global trade, such as climate change, and changes in water resources and land use.

ORCID iDs

Yingrong Wen https://orcid.org/0000-0002-0723-5481

References

[1] Perry B and Sones K 2007 Poverty reduction through animal health *Science* **315** 333–4

[2] Steinfeld H, Gerber P, Wassenaar T, Castel V, Rosales M and de Haan C 2006 Livestock's long shadow *Technical Report* (Rome: FAO) (www.fao.org/docrep/010/a0701e/a0701e00.HTM)

[3] Garnett T *et al* 2013 Sustainable intensification in agriculture: premises and policies *Science* **341** 33–4

[4] Meybeck M 2003 Global analysis of river systems: from earth system controls to Anthropocene syndromes *Phil. Trans. R. Soc.* B **358** 1935–55

[5] Wen Y, Schoups G and van de Giesen N 2017 Organic pollution of rivers: combined threats of urbanization, livestock farming and global climate change *Sci. Rep.* **7** 43289

[6] Vörösmarty C J *et al* 2010 Global threats to human water security and river biodiversity *Nature* **467** 555–61

[7] Hooda P S, Edwards A C, Anderson H A and Miller A 2000 A review of water quality concerns in livestock farming areas *Sci. Total Environ.* **250** 143–67

[8] Pruss A, Kay D and Bartram J 2002 Estimating the burden of disease from water, sanitation, hygiene at a global level *Environ. Health Perspect.* **110** 537

[9] FAO 2017 FAOSTAT (www.fao.org/faostat/en/#home)

[10] Bruinsma J 2003 *World Agriculture: Towards 2015/2030: an FAO Perspective* (London: Earthscan Publications Ltd)

[11] IAASTD 2009 *Executive summary of the synthesis report of the International Assessment of Agricultural Knowledge, Science and Technology for Development (IAASTD)* (www.sdi.com.my/IAASTD/IAASTD_ExecSummary.pdf)

[12] Dalin C and Rodríguez-Iturbe I 2016 Environmental impacts of food trade via resource use and greenhouse gas emissions *Environ. Res. Lett.* **11** 035012

[13] Lassaletta L, Billen G, Grizzetti B, Garnier J, Leach A M and Galloway J N 2014 Food and feed trade as a driver in the global nitrogen cycle: 50 year trends *Biogeochemistry* **118** 225–41

[14] Schipanski M E and Bennett E M 2012 The influence of agricultural trade and livestock production on the global phosphorus cycle *Ecosystems* **15** 256–68

[15] O'Bannon C, Carr J, Seekell D A and D'Odorico P 2014 Globalization of agricultural pollution due to international trade *Hydrol. Earth Syst. Sci.* **18** 503–10

[16] Mekonnen M M and Hoekstra A Y 2012 A global assessment of the water footprint of farm animal products *Ecosystems* **15** 401–15

[17] Burke M, Oleson K, McCullough E and Gaskell J 2009 A global model tracking water, nitrogen, and land inputs and virtual transfers from industrialized meat production and trade *Environ. Model. Asses.* **14** 179–93

[18] Galloway J N *et al* 2007 International trade in meat: the tip of the pork chop *Ambio* **36** 622–29

[19] Oita A, Malik A, Kanemoto K, Geschke A, Nishijima S and Lenzen M 2016 Substantial nitrogen pollution embedded in international trade *Nat. Geosci.* **9** 111–15

[20] Dasgupta S, Laplante B, Wang H and Wheeler D 2002 Are economists confronting the environmental Kuznets curve? *J. Econ. Perspect.* **16** 147–68

[21] Tisdell C 2001 Globalisation and sustainability: environmental Kuznets curve and the WTO *Ecol. Econ.* **39** 185–96

[22] Dinda S 2004 Environmental Kuznets curve hypothesis: a survey *Ecol. Econ.* **49** 431–55

[23] Panayotou T 1997 Demystifying the environmental Kuznets curve: turning a black box into a policy tool *Environ. Dev. Econ.* **2** 465–84

[24] Cole M A 2003 Development, trade, and the environment: how robust is the environmental Kuznets curve? *Environ. Dev. Econ.* **8** 557–80

[25] Verheijen L A H M, Wiersema D and Hulshoff L W 1996 Political management of waste from animal product processing *Technical Report* (Wageningen: International Agriculture Center)

[26] Steinfeld H, Wassenaar T and Jutzi S 2006 Livestock production systems in developing countries: status, drivers, trends *Rev. Sci. Tech.* **25** 505–16

[27] Robinson T *et al* 2011 *Global Livestock Production Systems* (Rome: FAO)

[28] Döll P and Lehner B 2002 Validation of a new global 30 m in drainage direction map *J. Hydrol.* **258** 214–31

[29] Fekete B M, Vörösmarty C J and Grabs W 2002 High-resolution fields of global runoff combining observed river discharge and simulated water balances *Glob. Biogeochem. Cycles* **16** 15-1-10

[30] US EPA 1999 Quantification of methane emissions and discussion of nitrous oxide and ammonia from septic tanks, latrines, and stagnant open sewers in the world. *Technical Report* EPA-600-R-99-089

[31] United Nations 2014 *World Urbanization Prospects (2014 Revision)* (New York: United Nations)

[32] Balk D, Yetman G and De Sherbinin A 2010 Construction of gridded population and poverty data sets from different data sources *Proc. European Forum Geostatistics Conf.* pp 12–20

[33] van Drecht G, Bouwman A F, Harrison J and Knoop J M 2009 Global nitrogen and phosphate in urban wastewater for the period 1970–2050 *Glob. Biogeochem. Cycles* **23** GB0A03

[34] UNEP 2000 *International Source Book On Environmentally Sound Technologies for Stormwater Management* (Osaka: UNEP)

[35] Raghupathi U P 2005 *Status of Water Supply, Sanitation and Solid Waste Management in Urban Areas. Research Study Series no. 88* (New Delhi: National Institute of Urban Affairs)

[36] Ministry of Housing and Urban-Rural Development 2010 *China Urban Waste Water Collection and Treatment Status Report 2006–2010*

[37] Marcon G and Philippi A 2010 Analysis of basic sanitation in Brazil and its impact on water resources and health *REGA* **7** 61–70

[39] IBGE 2002 *Pesquisa Nacional de Saneamento Básico 2000* (Instituto Brasileiro de Geografia e Estatística) (https://sidra.ibge.gov.br/pesquisa/pnsb/tabelas)

[39] Flörke M, Kynast E, Bärlund I, Eisner S, Wimmer F and Alcamo J 2013 Domestic and industrial water uses of the past 60 years as a mirror of socio-economic development: a global simulation study *Glob. Environ. Change* **23** 144–56

[40] World Bank Group 2013 *Introduction to Wastewater Treatment Processes* (http://water.worldbank.org/shw-resource-guide/infrastructure/menu-technical-options/wastewater-treatment)

[41] Williams R, Keller V, Voß A, Bärlund I, Malve O, Riihimäki J, Tattari S and Alcamo J 2012 Assessment of current water pollution loads in Europe: estimation of gridded loads for use in global water quality models *Hydrol. Process.* **26** 2395–410

[42] Ju X, Zhang F, Bao X, Römheld V and Roelcke M 2005 Utilization and management of organic wastes in Chinese agriculture: past, present and perspectives *Sci. China C* **48** 965–79

[43] IAEA 2008 *Guidelines for Sustainable Manure Management in Asian Livestock Production Systems* IAEA-TECDOC-1582

[44] Voß A, Alcamo J, Bärlund I, Voß F, Kynast E, Williams R and Malve O 2012 Continental scale modelling of in-stream river water quality: a report on methodology, test runs, and scenario application *Hydrol. Process.* **26** 2370–84

[45] Cox B 2003 A review of dissolved oxygen modelling techniques for lowland rivers *Sci. Total Environ.* **314** 303–34

[46] Texas Water Development Board 1971 Simulation of water quality in streams and canals *Texas Water Development Board Report 128* (Austin, TX: Texas Water Development Board)

[47] WWF HydroSHEDS 2017 (www.hydrosheds.org)

[48] Xu Y, Luo F, Pal A, Gin K Y H and Reinhard M 2011 Occurrence of emerging organic contaminants in a tropical urban catchment in Singapore *Chemosphere* **83** 963–69

[49] Sin S N, Chua H, Lo W and Ng L M 2001 Assessment of heavy metal cations in sediments of Shing Mun River, Hong Kong *Environ. Int.* **26** 297–301

[50] Savage V R, Huang S and Chang T C 2004 The Singapore river thematic zone: sustainable tourism in an urban context *Geogr. J.* **170** 212–25

[51] The Government of the Hong Kong Special Administrative Region 2017 Drainage Service Department sewerage strategy (www.dsd.gov.hk/EN/Sewerage/Sewage_Treatment_Facilities/Effluent_Quality_of_Major_Sewage_Treatment_Works/index.html)

[52] Zhou F, Liu Y and Guo H 2007 Application of multivariate statistical methods to water quality assessment of the watercourses in northwestern New Territories, Hong Kong *Environ. Monit. Assess.* **132** 1–13

[53] World Bank 2015 World Bank country and lending groups (https://datahelpdesk.worldbank.org/knowledgebase/articles/906519-world-bank-country-and-lending-groups)

[54] Fader M, Gerten D, Thammer M, Heinke J, Lotze-Campen H, Lucht W and Cramer W 2011 Internal and external green-blue agricultural water footprints of nations, and related water and land savings through trade *Hydrol. Earth Sys. Sci.* **15** 1641–60

[55] FAO 2014 Russia's restrictions on imports of agricultural and food products: an initial assessment (www.fao.org/3/a-i4055e.pdf)

[56] van Drecht G, Bouwman A F, Boyer E W, Green P and Siebert S 2005 A comparison of global spatial distributions of nitrogen inputs for nonpoint sources and effects on river nitrogen export *Glob. Biogeochem. Cycles* **19** GB4S06

[57] Cole M A 2004 Trade, the pollution haven hypothesis and the environmental Kuznets curve: examining the linkages *Ecol. Econ.* **48** 71–81

[58] Suri V and Chapman D 1998 Economic growth, trade and energy: implications for the environmental Kuznets curve *Ecol. Econ.* **25** 195–208

[59] Souza-Monteiro D M and Caswell J A 2004 The economics of implementing traceability in beef supply chains: trends in major producing and trading countries *Working Paper No. 2004–6* (University of Massachusetts Amherst: Department of Resource Economics) (http://scholarworks.umass.edu/cgi/viewcontent.cgi?article=1209&context=peri_workingpapers)