ENVIRONMENTAL RESEARCH
LETTERS

PAPER • OPEN ACCESS

# The surprisingly small but increasing role of international agricultural trade on the European Union's dependence on mineral phosphorus fertiliser

To cite this article: Thomas Nesme *et al* 2016 *Environ. Res. Lett.* **11** 025003

IOP Publishing Environ. Res. Lett. 11 (2016) 025003 doi:10.1088/1748-9326/11/2/025003

Environmental Research Letters

OPEN ACCESS

RECEIVED
16 October 2015

REVISED
26 January 2016

ACCEPTED FOR PUBLICATION
27 January 2016

PUBLISHED
19 February 2016



PAPER

# The surprisingly small but increasing role of international agricultural trade on the European Union's dependence on mineral phosphorus fertiliser

**Thomas Nesme[1,2,3], Solène Roques[4], Geneviève S Metson[4] and Elena M Bennett[3,4]**

[1] Bordeaux Sciences Agro, Univ. Bordeaux, UMR 1391 ISPA, F-33175 Gradignan Cedex, France
[2] INRA, UMR 1391 ISPA, F-33882 Villenave d'Ornon Cedex, France
[3] McGill School of Environment, McGill University, Montreal, Quebec, Canada
[4] Department of Natural Resource Sciences, McGill University, Sainte Anne de Bellevue, Montreal, Quebec, Canada

**E-mail:** thomas.nesme@agro-bordeaux.fr

**Keywords:** phosphorus, natural resource use, international trade, European Union, environmental footprint

Supplementary material for this article is available online

## Abstract

Phosphorus (P) is subject to global management challenges due to its importance to both food security and water quality. The European Union (EU) has promoted policies to limit fertiliser over-application and protect water quality for more than 20 years, helping to reduce European P use. Over this time period, the EU has, however, become more reliant on imported agricultural products. These imported products require fertiliser to be used in distant countries to grow crops that will ultimately feed European people and livestock. As such, these imports represent a displacement of European P demand, possibly allowing Europe to decrease its apparent P footprint by moving P use to locations outside the EU. We investigated the effect of EU imports on the European P fertiliser footprint to better understand whether the EU's decrease in fertiliser use over time resulted from P demand being 'outsourced' to other countries or whether it truly represented a decline in P demand. To do this, we quantified the 'virtual P flow' defined as the amount of mineral P fertiliser applied to agricultural soils in non-EU countries to support agricultural product imports to the EU. We found that the EU imported a virtual P flow of 0.55 Tg P/yr in 1995 that, surprisingly, decreased to 0.50 Tg P/yr in 2009. These results were contrary to our hypothesis that trade increases would be used to help the EU reduce its domestic P fertiliser use by outsourcing its P footprint abroad. Still, the contribution of virtual P flows to the total P footprint of the EU has increased by 40% from 1995 to 2009 due to a dramatic decrease in domestic P fertiliser use in Europe: in 1995, virtual P was equivalent to 32% of the P used as fertiliser domestically to support domestic consumption but jumped to 53% in 2009. Soybean and palm tree products from South America and South East Asia contributed most to the virtual P flow. These results demonstrate that, although policies in the EU have successfully decreased the domestic dependence on mineral P fertiliser, in order to continue to limit global potential mineral P supply depletion and consequences of P losses to waterways the EU may have to think about its trading partners.

## 1. Introduction

International trade is a key driver of the Anthropocene, especially with regards to agriculture (Steffen *et al* 2015). The volume of agricultural trade in the world increased more than ten-fold from the 1950s to the 2010s (Schmitz *et al* 2012), with the largest increases occurring in the trade of staple commodities such as wheat, maize and rice. For example international flows of wheat increased 42% and rice flows increased 90% between 1992 and 2009 (Puma *et al* 2015). In 2008, biomass trade represented 7.5% of all biomass extracted from ecosystems globally (Krausmann *et al* 2008) and cropland used for exports accounted for 20% of all global cropland area (Kastner *et al* 2014). Currently, 16% of people rely on international trade to meet their demand for staple food and agricultural products (Fader *et al* 2013). All of these agricultural flows contribute to making the planet increasingly inter-connected (MacDonald 2013) and increasingly vulnerable to systemic failures and extreme events (Helbing 2013, Liu *et al* 2013).

The potential environmental consequences of such increases in trade, which decouples the consumption of food from its production, have received growing attention in the last decade. Of particular concern is how developed (and rapidly developing) countries are increasingly consumers of distantly produced food, thereby contributing to the displacement of the unintentional consequences of agriculture to source countries (Frankel and Rose, 2005). For example, by importing products from countries with less-stringent (or even non-existent) environmental regulations, increased trade can 'outsource' environmental degradation from one country or region to another; most often from a more developed country to a less developed one (Davis and Caldeira, 2010, Meyfroidt *et al* 2010). In addition, trade contributes to the increasing distance between consumption practices and their environmental impacts (Cumming *et al* 2014). Such tele-connections have received particular attention for land-use change (Meyfroidt *et al* 2010), fossil energy use and greenhouse gas emissions (West et al 2014), irrigation and groundwater water withdrawal (Dalin *et al* 2014, Marston *et al* 2015) and biodiversity erosion (Hooper *et al* 2012). While these studies showed that trade products generally flow from resource-abundant to resource-scarce countries (Dalin *et al* 2014, Kastner *et al* 2014, Macdonald *et al* 2015), they also confirmed that trade contributes to displacing the environmental burdens related to production activities (Davis and Caldeira 2010, West et al 2014).

The impact of agricultural commodity trade on a scarce essential fertiliser and potential pollutant—phosphorus (P)—has received less attention. Extensive mineral P fertiliser application to agricultural land has increased yields during the last decades (van der Velde *et al* 2013). However, mineral P fertiliser production is dependent on increasingly scarce and geopolitically concentrated phosphate mine resources since the majority of the resource is found in only three countries: Morocco, China, and the USA (Cordell *et al* 2009, Van Vuuren *et al* 2010). Our dependence on this essential but non-renewable resource is of great concern for food security, especially for poor urban and rural populations with lower purchasing power and highly P-deficient soils such as in Sub-Saharan Africa (Obersteiner *et al* 2013). In addition, P losses to water bodies through runoff and erosion from agricultural soils, inadequate management of animal manure, and insufficient treatment of wastewater and human excreta can cause aquatic eutrophication (Carpenter *et al* 1998, Schindler *et al* 2008). European (e.g. the Baltic Sea,) and North American (e.g. the Great Lakes and the Gulf of Mexico) waterways have experienced the high costs of such pollution including toxic drinking water and loss of fishery and ecosystem resources (Diaz and Rosenberg 2008). Developing and implementing governance, technologies, and practices that increase the efficiency of P in the food system is essential to ensuring food security and water quality for all (Cordell *et al* 2012). As such, in this study we consider P footprints, defined here as the amount of mineral P fertiliser required to produce agricultural products imported for consumption in the European Union (EU) minus the fertiliser used domestically to export agricultural products abroad, as this measure embodies part of the potential impact on resource depletion and hampered water quality that can be associated with increased mined P fertiliser use.

To address these P-related issues, the EU has developed a set of policies to limit aquatic eutrophication (McDowell *et al* 2016), including both agricultural practices and technologies targeting improved wastewater treatment and banning P-rich detergents (Van Drecht *et al* 2009). Regulations were initiated in the 1990s (e.g. through the Nitrate directive, which indirectly affects P) and were reinforced in the early 2000s (e.g. through the Water Framework Directive, which provides an overall objective of 'good status' for all water bodies by 2015) to limit P losses from urban wastewaters and from agricultural soils, in part through limiting P fertiliser use. These regulations provided a wide range of tools targeted at mineral fertiliser and manure application that operate both at the farm and the catchment scales. Taking advantage of the legacy of past P fertilisation and improvements in fertilisation decision knowledge and tools, these regulations have contributed to improving water quality in European inland and coastal ecosystems without reducing agricultural production (Herzog *et al* 2006, Dubois 2009). For instance, the orthophosphate concentration in European rivers has decreased by >50% between 1992 and 2012 (http://www.eea.europa.eu/data-and-maps/indicators/nutrients-in-freshwater/ds_resolveuid/JCIQ2VOFK9, accessed 12 October 2015). These regulations contributed to a reduction in domestic mineral P fertiliser use: EU mineral P fertiliser consumption has decreased by 42% between 1995 and 2009, with average mineral P fertilisation dropping from 9.2 to 5.4 kg P/ha/yr, while crop acreage has remained similar (figure 1). As such, it appears that these measures have helped the EU to decrease losses of nutrients to waterways and to inadvertently be less dependent on rock phosphate, limiting the EU's contribution to the depletion of this non-renewable resource.

However, the EU's imports of agricultural products as both food and feed, and thus of the P they physically contain, have increased by 18% over the same period (figure 1). This increase has been mostly

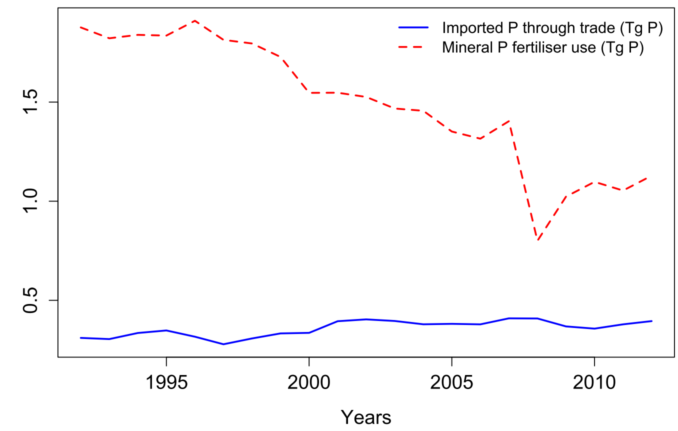


**Figure 1.** Changes in imported P through agricultural product imports and domestic mineral P fertiliser use in the EU from 1992 to 2012. Imported P refers to the amount of P that is physically embedded in products imported to the EU. It has been calculated by multiplying the amount of ~300 crop and animal products imported into the EU by their respective P content. The data on agricultural product imports in the EU are corrected from intra-community trade. Note that imported P through trade has increased by 18% during the study period.

driven by demand for animal feed to supply intensive livestock production in the EU (Spiertz 2010, de Visser *et al* 2014). Even if the physical P inflow they represent is much smaller than domestic P fertiliser use (figure 1), these imports represent an indirect contribution to global rock phosphate depletion, because imported products were grown using mineral P fertilisers in exporting countries. However, the magnitude of such indirect contribution is unknown, and this gap in knowledge impairs the proper estimation of the EU 'phosphorus footprint' (i.e. the total amount of P fertilisers required to produce agricultural products imported for consumption in the EU minus the fertiliser used domestically to export agricultural products abroad). Therefore, the question exists whether increase in trade would be used to help the EU reduce its domestic P fertiliser use by outsourcing its P fertiliser demand abroad.

In this paper, we assess how domestic mineral P fertiliser use (hereafter called 'real P flow') and mineral P fertiliser use for the production of imported food/feed products (hereafter called 'virtual P flow') have changed over time (1995 and 2009) to better estimate the real EU's contribution to mineral P use and potential P-related pollution, in other words the EU P footprint.

## 2. Material and methods

### 2.1. Real and virtual P flow calculations

We determined the value of 'virtual' and 'real' P flows for the EU27, including its oversea territories[5] for 1995 and 2009. These years were selected to cover the period during which the EU set up policies to limit domestic water eutrophication risks, and because they corresponded to the first and last years for which we had comprehensive data availability (e.g., about detailed imports, yields, and fertilisation rate; see below). Estimates of both flows were focused on mineral P fertiliser use and did not account for P supplements used in animal feed.

The virtual P flow corresponded to the amount of mineral P fertiliser that was used in source countries to grow agricultural products for export to the EU (Matsubae *et al* 2011, Schaffartzik *et al* 2015). It is also sometimes referred as upstream or embodied fertiliser use to support agricultural export. Following (MacDonald *et al* 2012a), virtual P flow was determined by the (equation (1):

$$\text{Virtual } P \text{ flow} = \sum_{i=1,j=1}^{n,m} Q_{i,j} \times \varepsilon_i \times \frac{1}{\text{yield}_{i,j}} \times \text{ferti}_{i,j}, \quad (1)$$

where:

$Q_{i,j}$ represents the amount in metric tons of the product $i$ that was imported from country $j$ into the EU (where $i = 1, 2 \ldots n$ are the different crop products and $j = 1, 2 \ldots m$ are the source countries that were considered in this study). We considered a limited set of products that together represented the large majority of trade flows to the EU. To do so, we ranked all the products entering the EU according to their direct P inflow value (i.e. according to the amount of P they physically contain). Starting from those products that, due to the amount of product imported and its P content, bring in the most P, and moving towards those that import the least, we developed a list of products that together contributed 95% of the total direct P inflow (i.e. excluding very small trade flows). In total, we considered 30 imported products in 1995 and 27

[5] The EU27 corresponds to the following list of countries: Austria, Belgium, Bulgaria, Cyprus, Czech Republic, Denmark, Estonia, Finland, France, Germany, Greece, Hungary, Ireland, Italy, Latvia, Lithuania, Luxembourg, Malta, The Netherlands, Poland, Portugal, Romania, Slovakia, Slovenia, Spain, Sweden and the United Kingdom.

imported products in 2009 (table S1); noting that no animal product was included in 1995 and only one animal product (chicken meat) contributed in 2009. However, even in 2009, chicken meat contributed only marginally (0.2% of total direct P inflow through trade) and was thus not included in our calculations. The $Q_{i,j}$ data were extracted from the FAOSTAT trade module which provides annual data on total import and export of ~300 agricultural products for each FAO country ($n = 203$) from 1961 to 2013 (http://faostat3.fao.org/download/T/*/E, accessed 16 September 2015).

$\varepsilon_i$ represents the coefficient of conversion from processed products into raw commodities ($\varepsilon = 1$ for raw imported commodities and $\varepsilon > 1$ for processed products; $\varepsilon$ is unitless and is product specific). This set of coefficients helps to reconstruct commodity trees (i.e. to identify how a given raw commodity is broken down into processed products and by-products and vice-versa) and to convert data from process equivalent to raw commodity equivalent. The multiplication of $Q$ by $\varepsilon$ helped to express imported raw and processed products (e.g., wheat grain and bread) into a single raw commodity unit (e.g., wheat grain). Applying these coefficients helped to reduce the number of raw commodities considered in this study down to 22 commodities for both 1995 and 2009 (table S1). The $\varepsilon_i$ data were given by the FAO Statistics Division (http://www.fao.org/economic/the-statistics-division-ess/methodology/methodology-systems/technical-conversion-factors-for-agricultural-commodities/en/, accessed 16 September 2015).

$Yield_{i,j}$, represents the yield of the product $i$ in the country $j$ (in tons of product per ha). The multiplication by the inverse of the yield was needed to convert imported raw commodities into the agricultural area that was used to produce these commodities. The data on yields were extracted using FAOSTAT production module which provides annual data on crop yields for each FAO country and for each of the considered commodities from 1961 to 2013 (http://faostat3.fao.org/download/Q/QC/E, accessed 16 September 2015).

$Ferti_{i,j}$ represents the amount of mineral P fertiliser that was applied on crop $i$ in the country $j$ (in kg of P per ha of cropland). The data on fertiliser use were extracted using reports from the Fertilizer Industry Association (IFA), the International Fertilizer Development Center (IFDC) and the FAO (FAO *et al* 1994, FAO *et al* 1996, FAO *et al* 1999, FAO 2006, IFA 2009, IFA 2013). These reports provide expert-based data on the amount of mineral P used as fertiliser for major crops and major agricultural countries. As some data were sometimes missing for some crops, some countries or some years, we used 5 year windows, centred on the year 1995 and 2009. This helped to correct for possible inter-annual variability in fertiliser use and to avoid missing data for some specific years. Additional details about P fertilisation rates are available in SI. The full dataset of imported commodities, crop yields and fertilisation rates in source countries is available in table S2.

The real P flow corresponded to the amount of mineral P fertiliser that was used on the EU's domestic agricultural area in 1995 and 2009, including that used to support domestic crop and grass production, which was both consumed domestically and exported. Since our study aimed to estimate only the amount of P needed to support EU food and feed consumption, we removed the amount of mineral fertiliser needed to produce exported agricultural products. To do so, we used an equation similar to (equation (1) with $Q_{ij}$, yield$_{ij}$ and ferti$_{ij}$ representing the amount of agricultural products exported out of the EU, the average yield of the exported commodities at the EU scale and the average crop fertilisation rate at the EU scale, respectively. The corresponding data were extracted from FAOSTAT for $Q_{ij}$ and yield$_{ij}$ and from IFA, FAO and IFDC reports for ferti$_{ij}$. Animal products represented a significant fraction of the total P exported through trade from the EU (18% and 14% of total P exported through trade in 1995 and 2009, respectively). We thus accounted for the P needed to produce these animal products in our P footprint approach and provide detailed calculations in SI.

### 2.2. Assumptions, omissions, and uncertainty

While our methods and analysis allow us to determine the contribution of virtual P flows to the EU, they do not encompass all of the EU's dependence on imported P. For example, in addition to chicken meat in 2009, we excluded three additional imported crop products due to very ambiguous composition (i.e., crude materials, prepared food, and feed and meal), noting that these four products represented less than 3% of total, direct P import through trade into the EU. This omission did translate into a slightly conservative underestimation of the EU P footprint.

It is also important to note that there is an inherent amount of uncertainty related to mineral P fertiliser application rates on crops because farming practices are generally highly variable across time and space (Nesme *et al* 2005, Yunju *et al* 2012). This uncertainty is difficult to estimate but it may be limited in this study by the fact that our dataset integrated possible changes in fertilisation rates through time. As such, we avoided using outdated data on farming practices which is often a severe limitation of virtual resource flow calculations (Yang and Suh 2015). There is also uncertainty about P fertiliser application rates to specific crops when they are used in rotation. Farmers generally make their decisions about fertilisation over the duration of a whole crop rotation sequence by accounting for carry-over effects of applied fertilisers on past crops (Haileslassie *et al* 2007). Our annual crop fertilisation averages cannot explicitly account for such nuances (and how these numbers may be under-

**Table 1.** 'Virtual' and 'real' P flows in 1995 and 2009.

| | 1995 | 2009 |
|---|---|---|
| 'Virtual' P flow to the EU (applied to imports to the EU) (Tg P/yr) | 0.55 | 0.50 |
| 'Real' P flow (domestic use in the EU) (Tg P/yr) | 1.87 | 1.09 |
| 'Virtual' P outflow from the EU (applied to exports from the EU) (Tg P/yr) | 0.18 | 0.14 |
| *In crop products* | 0.12 | 0.09 |
| *In animal products* | 0.06 | 0.05 |
| EU P footprint (virtual + real P flows − virtual P outflow) (Tg P/yr) | 2.24 | 1.45 |
| Virtual P flow/(virtual + real P flows − virtual P outflow) (%) | 25 | 34 |

or over-estimates). This uncertainty is reduced however for source regions that export all the crop products in a particular rotation to the EU.

## 3. Results and discussion

### 3.1. A decreasing virtual P flow but of increasing importance to the EU

The EU agricultural P footprint (i.e. the sum of P fertilisers required to produce 95% of agricultural products imported for consumption in the EU plus the P fertiliser used to support domestic crop production minus the fertiliser used domestically to export agricultural products abroad) declined from 2.24 in 1995 to 1.45 Tg P in 2009. As expected, the real P flow (i.e. domestic fertiliser use) dropped dramatically from 1.87 to 1.09 Tg of P (a 42% decline). Surprisingly, we found that 'virtual' P flow to the EU27 amounted to 0.55 Tg P in 1995 and decreased moderately to 0.50 Tg P in 2009 (table 1).

These results were contrary to our hypothesis that trade increases would be used to help the EU reduce its real P flow by outsourcing its P footprint abroad. Still, the contribution of virtual P flows to the total P footprint of the EU has increased by 40% from 1995 to 2009 (table 1). In 1995, virtual P was equivalent to 32% of the P used as fertiliser domestically to support domestic consumption but jumped to 53% in 2009. This increase was clearly due to the sharp reduction in domestic fertiliser use during the study period, but still marks the increasing relative importance of trade in the EU's P footprint (table 1 and figure 1). High virtual P flows have also been observed for other countries that are either large agricultural producers such as the USA (MacDonald *et al* 2012a) or large food importers such as Japan (Matsubae *et al* 2011). However, to our knowledge, our study is the first one to provide any insight into changes in the P footprint of a given country or region through time.

We identified three potential drivers for this observed decrease in total virtual P flows: (1) a reduction in fertilisation rates in exporting countries, (2) changes in the imported items to less P intensive crops, and (3) import shifts to more P efficient countries. Our results showed that a large part of the reduction in the total P imported to Europe is likely due to an overall reduction in mineral P fertilisation rates at the global scale: applying 1995 fertilisation rates to commodities imported in 2009 translated to a virtual P inflow of 0.60 Tg P into the EU which is 19% higher than the virtual P inflow calculated with actual 2009 fertilisation rates. For instance, for soybean, one of the major crops imported to Europe (see below), the average fertilisation rates have declined by 70% (from 23 to <10 kg P/ha/yr) in the USA, one of the major exporters to the EU, over the study period. This overall reduction in P fertiliser use may result from past fertilisation practices that have contributed to build-up soil P status. Such legacy P effect is particularly important in world regions that received massive amounts of mineral P fertilisers in the 1980s or 1990s such as North America (Sattari *et al* 2012). As such, decreases in fertilisation rates in source countries helped to reduce the EU P footprint. Changes in the amount of imported products could also contribute, e.g. through the dramatic reduction in the import of some specific products such as copra, cassava or cottonseed in the EU (see table S1). Shifts to source countries with more P fertilisation efficient practices may also have contributed. Our results indicated that virtual P was mostly imported from the Americas, and to a much lesser extent from South-East Asia (figure 2) but with a clear shift in virtual flows from North to South America between 1995 and 2009: taken together, Brazil and Argentina represented only 27% of virtual P inflow into the EU in 1995 while they represented more than 60% of virtual P inflow into the EU in 2009.

Interestingly, the virtual P inflow to the EU27 was driven by a limited set of commodities: soybeans, palm kernel, copra, coffee beans, and cottonseed accounted for 69% and 78% of the total virtual P inflow in 1995 and 2009, respectively (figure 3). Those crops were either imported in large quantities by the EU (e.g. soybeans which is largely used as animal feed in the EU) or intensively fertilised in source countries (e.g., palm tree plantations). Although it is a protein crop largely used as concentrate feed in livestock production in the EU, soybean production has strongly declined in European croplands since the Blair House agreement under the GATT umbrella in 1992. Soybean production is now facing a large yield- and profitability-gap compared to most cereal crops in Europe due to a lack of technological and research investments (de Visser *et al* 2014), making the EU largely dependent from imports from the Americas. Together these five imported crops represent leverage points (West et al 2010) that could be targeted if the EU P footprint had to be reduced.

Finally, although the total virtual P flow to the EU has decreased, the fact that virtual P flows represented up to one third of the EU total P footprint illustrates how the embeddedness of the EU in the global market affects the global P cycle. More specifically, our results

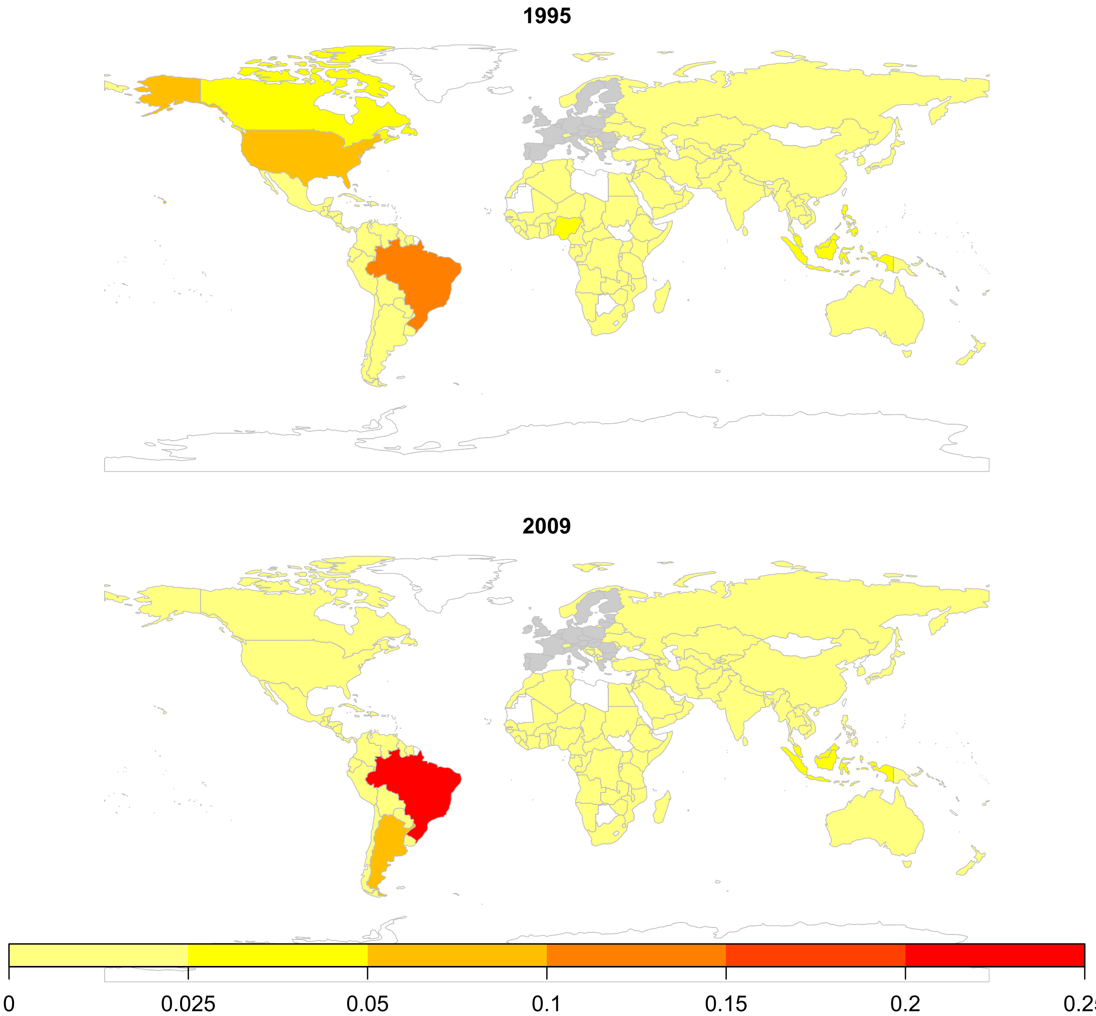


**Figure 2.** Virtual P inflow from the different source countries in 1995 (top) and 2009 (bottom) in Tg P/yr. EU27 countries appear in grey. Countries in white do not export significant amounts of virtual P to the EU.

demonstrated that the EU reduced its overall P footprint but that this reduction was largely due to a decrease in domestic fertiliser use and to a lesser extent to the decrease in outsourced P fertiliser demand, resulting in a greater contribution of virtual P flow to the EU overall P footprint.

### 3.2. Importance of virtual resource flow estimation for public policy design

Although the EU P footprint and virtual P flow have decreased, the fact that virtual P flows amounted to more than half the amount of mineral P fertiliser used domestically to support domestic consumption in the EU (table 1) demonstrates that assessing the contribution of a given country or region to the global rock phosphate depletion on the sole basis of its domestic P fertiliser use misses the whole picture. The contribution of a region to mineral P use should perhaps be based on crop consumption inventories. That is, it should account for both domestic and imported crop production instead of only domestic crop production. The difference between production-based *versus* consumption-based resource use is particularly important for wealthy countries that are strongly involved as importer in international trade. For instance, it has been estimated that >30% of consumption based $CO_2$ emissions in Europe were imported from elsewhere (Davis and Caldeira 2010). The impact of domestic consumption on global resources such as fossil energy, land (Fader *et al* 2013, Macdonald *et al* 2015) and water (Dalin *et al* 2014, Marston *et al* 2015, Vörösmarty *et al* 2015) as well as pollutants emitted during production processes such as $CO_2$, $N_2O$ and $NO_3^-$ leached to water bodies (Galloway *et al* 2007) is staggering but not often considered in domestic environmental policies. Impacts might be particularly large if wealthy countries shift from domestic production where environmental regulations are stringent, such as the Nitrate Directive in the EU or the Clean Water Act in the USA which guide the management of nutrients to protect water quality, to instead importing commodities from countries with limited environmental regulations.

Although our P footprint indicator is strongly focussed on mineral P fertiliser use and its impact on global resource depletion, it also encompasses

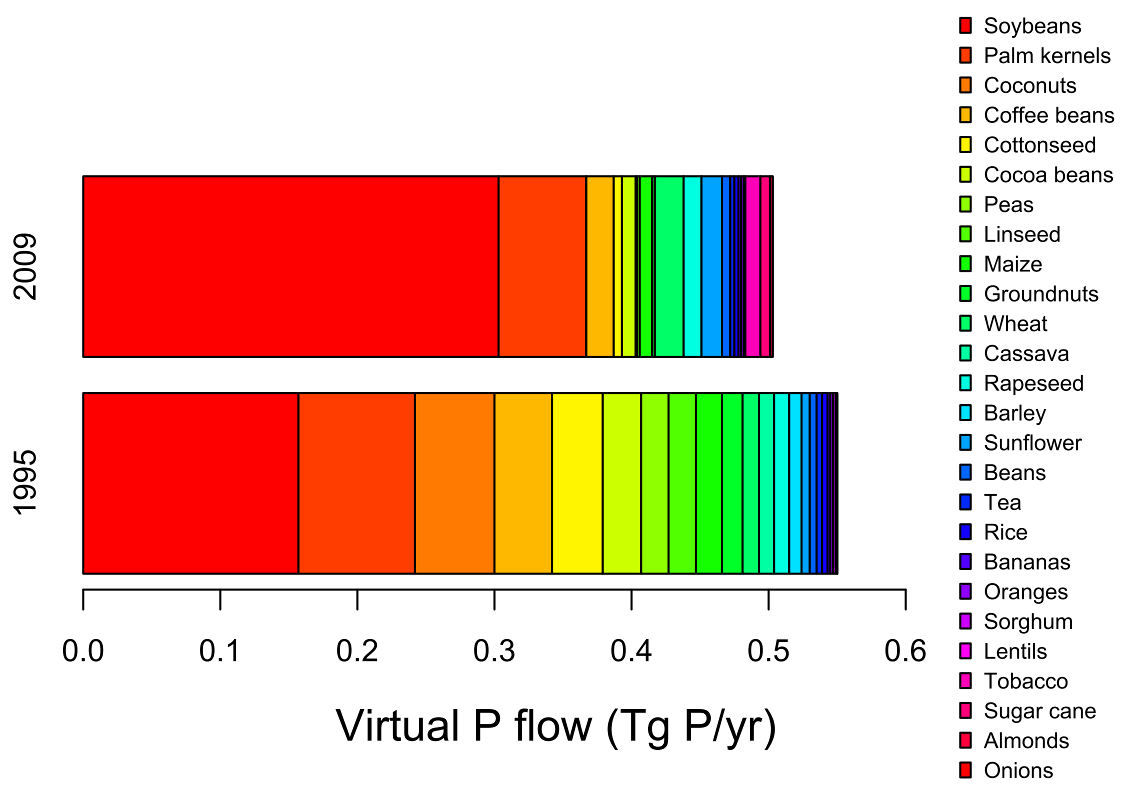


**Figure 3.** Virtual P inflow to the EU27 in 1995 and 2009 according to the imported commodities. Note that copra did not carry any more significant amount of virtual P to the EU in 2009.

potential impacts on hampered water quality since increased fertiliser application rates that accompany export-oriented production could increase the potential for P losses to waterways. However, complex interactions between biophysical (e.g. precipitation, soil type, slope) and human management (e.g. tile drains, timing and placement of fertiliser application) mediate such losses (Withers *et al* 2001, Djodjic *et al* 2005). In addition, P losses to waterways can also result from improper management of P in manure and household sewage sludge as well as from poor accounting of soil P legacy (Toth *et al* 2006, MacDonald *et al* 2012b, Roy *et al* 2014). For instance, the outsourcing of soybean production in Brazil shows us that context matters in assessing potential environmental risks of increased fertiliser application for water quality. Indeed, in addition to contributing to the EU's decrease global P requirements because of its higher P use efficiency (Schipanski and Bennett 2012), increased fertiliser consumption on Brazilian nutrient poor soils (Riskin *et al* 2013a) has not lead to increases in P losses to waterbodies. This is due to the fact that inter-tropical soils are highly weathered and therefore exhibit high P sorption capacities (Riskin *et al* 2013b) that limit P losses to local waterways. In other words considering specific trade partners is key to assessing the full impact of virtual P flows.

The disconnect between agricultural production and associated resource use and the consumption activities that drive such resource use practices has been facilitated by international trade and makes coordinated interventions to increase sustainability difficult (Hertwich and Peters 2009). As such the fact that many countries are increasingly involved in the global food/feed trade (Fader *et al* 2013, Puma *et al* 2015) calls for the development of robust and flexible indicators and databases that could help quantify virtual resource flows among countries and potential positive and negative effects on local and global resource use. For instance, using such indicators in our study helped to identify a limited set of commodities and countries that could be targeted for the EU to reduce its overall P footprint: prioritising trade relations with Brazil and Argentina, and/or the five crops (soybeans, palm kernel, copra, coffee beans and cottonseeds) that represent the majority of the virtual P flow. Such indicators could then be used in international trade agreements to better share the responsibility of resource depletion and pollution among crop producers and consumers and work towards strategies that promote food security and environmental integrity across regions.

## 4. Conclusion

In a globalised world, international trade is an important part of global resource use patterns, and P is no exception. Because P resources are scarce, and physical and economic access to this essential agricultural resource are un-equal (and inequitable) across the planet, it is especially critical to consider resource management from a global perspective, and not only a domestic one. As such, it is important to understand the EU's role in driving global P use and pollution. Unexpectedly, our results demonstrate that virtual P flows to the EU27 through agricultural trade decreased in total amount from 1995 to 2009, although they did increase as a percentage of the EU's overall mineral P fertiliser footprint. Although the EU has decreased its consumption of mineral P fertilisers domestically and abroad, in order to continue to move towards both local and global P security, the EU will increasingly need to consider how it meets food and feed needs through trade, and the P management practices of its trading partners.

## Acknowledgments

We thank Peder Engstrom and Nathan Mueller (Univ. Minnesota) for providing some data on P fertilisation rate for coffee, cocoa, tea and pea. We also thank Bruno Ringeval, Sylvain Pellerin and Tamara Ben-Ari, as well as two anonymous reviewers for their comments on an earlier version of this manuscript. This work was funded by grants from Bordeaux Sciences Agro (Univ. Bordeaux) and the McGill School of Environment during TN's sabbatical at McGill, as well as by an NSERC Discovery Grant to EMB.

## References

Carpenter S R *et al* 1998 Nonpoint pollution of surface waters with phosphorus and nitrogen *Ecological Appl.* **8** 559–68

Cordell D, Drangert J-O and White S 2009 The story of phosphorus: global food security and food for thought *Glob. Environ. Change* **19** 292–305

Cordell D, Neset T-S S and Prior T 2012 The phosphorus mass balance: identifying 'hotspots' in the food system as a roadmap to phosphorus security *Curr. Opin. Biotechnol.* **23** 839–45

Cumming G S *et al* 2014 Implications of agricultural transitions and urbanization for ecosystem services *Nature* **515** 50–7

Dalin C, Hanasaki N, Qiu H, Mauzerall D L and Rodriguez-Iturbe I 2014 Water resources transfers through Chinese interprovincial and foreign food trade *Proc. Natl Acad. Sci.* **111** 9774–9

Davis S J and Caldeira K 2010 Consumption-based accounting of $CO_2$ emissions *Proc. Natl Acad. Sci. USA* **107** 5687–92

de Visser C L M, Schreuder R and Stoddard F 2014 The EU's dependency on soya bean import for the animal feed industry and potential for EU produced alternatives *Oilseeds Fat Crops Lipids* **21** D407

Diaz R J and Rosenberg R 2008 Spreading dead zones and consequences for marine ecosystems *Science* **321** 926–9

Djodjic F, Bergström L and Grant C 2005 Phosphorus management in balanced agricultural systems *Soil Use Manage.* **21** 94–101

Dubois A 2009 La qualité des rivières s'améliore pour plusieurs polluants, à l'exception des nitrates *Commissariat Général au Développement Durable* (La Défense: Commisariat Général au Développement Durable) p 4 (www.statistiques.developpement-durable.gouv.fr/fileadmin/documents/Produits_editoriaux/Publications/Le_Point_Sur/2009/PointSur18.pdf)

Fader M, Gerten D, Krause M, Lucht W and Cramer W 2013 Spatial decoupling of agricultural production and consumption: quantifying dependences of countries on food imports due to domestic land and water constraints *Environ. Res. Lett.* **8** 014046

FAO 2006 *Fertilizer Use by Crop* (Rome: FAO)

FAO, IFA and IFDC 1994 *Fertilizer Use by Crops* (Rome: FAO, IFA and IFDC)

FAO, IFA and IFDC 1996 *Fertilizer Use by Crop* (Rome: FAO, IFA and IFDC)

FAO, IFA and IFDC 1999 *Fertilizer Use by Crop* (Rome: FAO, IFA and IFDC)

Frankel J A and Rose A K 2005 Is trade good or bad for the environment? Sorting out the causality *Rev. Econ. Stat.* **87** 85–91

Galloway J N *et al* 2007 International trade in meat: the tip of the pork chop *Ambio* **36** 622–9

Haileslassie A, Priess J A, Veldkamp E and Lesschen J P 2007 Nutrient flows and balances at the field and farm scale: exploring effects of land-use strategies and access to resources *Agric. Syst.* **94** 459–70

Helbing D 2013 Globally networked risks and how to respond *Nature* **497** 51–9

Hertwich E G and Peters G P 2009 Carbon footprint of nations: a global, trade-linked analysis *Environ. Sci. Technol.* **43** 6414–20

Herzog F *et al* 2006 Assessing the intensity of temperate European agriculture at the landscape scale *Eur. J. Agronomy* **24** 165–81

Hooper D U *et al* 2012 A global synthesis reveals biodiversity loss as a major driver of ecosystem change *Nature* **486** 105–8

IFA 2009 *Assesment of Fertilizer Use by Crop, 2006/07-2007/08* (Paris: IFA)

IFA 2013 *Assessment of Fertilizer use by Crop at the Global Level, 2010-2010/11* (Paris: IFA)

Kastner T, Erb K-H and Haberl H 2014 Rapid growth in agricultural trade: effects on global area efficiency and the role of management *Environ. Res. Lett.* **9** 034015

Krausmann F, Erb K-H, Gingrich S, Lauk C and Haberl H 2008 Global patterns of socioeconomic biomass flows in the year 2000: a comprehensive assessment of supply, consumption and constraints *Ecological Econ.* **65** 471–87

Liu J *et al* 2013 Framing sustainability in a telecoupled world *Ecology Soc.* **18** 26

MacDonald G K 2013 Eating on an interconnected planet *Environ. Res. Lett.* **8** 021002

MacDonald G K, Bennett E M and Carpenter S R 2012a Embodied phosphorus and the global connections of United States agriculture *Environ. Res. Lett.* **7** 044024

MacDonald G K, Bennett E M and Taranu Z E 2012b The influence of time, soil characteristics, and land-use history on soil phosphorus legacies: a global meta-analysis *Glob. Change Biol.* **18** 1904–17

Macdonald G K *et al* 2015 Rethinking agricultural trade relationships in an era of globalization *Biosci.* **65** 275–89

Marston L, Konar M, Cai X and Troy T J 2015 Virtual groundwater transfers from overexploited aquifers in the United States *Proc. Natl Acad. Sci.* **112** 8561–6

Matsubae K, Kajiyama J, Hiraki T and Nagasaka T 2011 Virtual phosphorus ore requirement of Japanese economy *Chemosphere* **84** 767–72

McDowell R W *et al* 2016 A review of the policies and implementation of practices to decrease water quality impairment by phosphorus in New Zealand, the UK, and the US *Nutr. Cycl. Agroecosyst.* in press (doi:10.1007/s10705-015-9727-0)

Meyfroidt P, Rudel T K and Lambin E F 2010 Forest transitions, trade, and the global displacement of land use *Proc. Natl Acad. Sci.* **107** 20917–22

Nesme T, Bellon S, Lescourret F, Senoussi R and Habib R 2005 Are agronomic models useful for studying farmers' fertilisation practices? *Agric. Syst.* **83** 297–314

Obersteiner M, Peñuelas J, Ciais P, van der Velde M and Janssens I A 2013 The phosphorus trilemma *Nat. Geosci.* **6** 897–8

Puma M J, Bose S, Chon S Y and Cook B I 2015 Assessing the evolving fragility of the global food system *Environ. Res. Lett.* **10** 024007

Riskin S H *et al* 2013a The fate of phosphorus fertilizer in Amazon soya bean fields *Phil. Trans. R. Soc.* B **368** 20120154
Riskin S H, Porder S, Schipanski M E, Bennett E M and Neill C 2013b Regional differences in phosphorus budgets in intensive soybean agriculture *Biosci.* **63** 49–54
Roy E D, White J R and Seibert M 2014 Societal phosphorus metabolism in future coastal environments: insights from recent trends in Louisiana, USA *Glob. Environ. Change* **28** 1–13
Sattari S Z, Bouwman A F, Giller K E and Van Ittersum M K 2012 Residual soil phosphorus as the missing piece in the global phosphorus crisis puzzle *Proc. Natl Acad. Sci. USA* **109** 6348–53
Schaffartzik A *et al* 2015 Trading land: a review of approaches to accounting for upstream land requirements of traded products *J. Ind. Ecology* **19** 703–14
Schindler D W *et al* 2008 Eutrophication of lakes cannot be controlled by reducing nitrogen input: results of a 37 year whole-ecosystem experiment *Proc. Natl Acad. Sci.* **105** 11254–8
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
Schmitz C *et al* 2012 Trading more food: implications for land use, greenhouse gas emissions, and the food system *Glob. Environ. Change* **22** 189–209
Spiertz J H J 2010 Nitrogen, sustainable agriculture and food security. A review *Agronomy Sustainable Dev.* **30** 43–55
Steffen W, Broadgate W, Deutsch L, Gaffney O and Ludwig C 2015 The trajectory of the Anthropocene: the great acceleration *Anthropocene Rev.* **2** 81–98
Toth J D, Dou Z, Ferguson J D, Galligan D T and Ramberg cf Jr 2006 Nitrogen- versus phosphorus-based dairy manure applications to field crops: nitrate and phosphorus leaching and soil phosphorus accumulation *J. Environ. Qual.* **35** 2302–12
van der Velde M *et al* 2013 Affordable nutrient solutions for improved food security as evidenced by crop trials *PLoS One* **8** e60075
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
Van Vuuren D P, Bouwman A F and Beusen A H W 2010 Phosphorus demand for the 1970–2100 period: a scenario analysis of resource depletion *Glob. Environ. Change* **20** 428–39
Vörösmarty C J, Hoekstra A Y, Bunn E, Conway D and Gupta J 2015 Fresh water goes global *Science* **349** 478–9
West P C *et al* 2010 Trading carbon for food: global comparison of carbon stocks versus crop yields on agricultural land *Proc. Natl Acad. Sci. USA* **107** 19645–8
West P C *et al* 2014 Leverage points for improving global food security and the environment *Science* **345** 325–8
Withers P J A, Edwards A C and Foy R H 2001 Phosphorus cycling in UK agriculture and implications for phosphorus loss from soil *Soil Use Manage.* **17** 139–49
Yang Y and Suh S 2015 Changes in environmental impacts of major crops in the US *Environ. Res. Lett.* **10** 094016
Yunju L *et al* 2012 Fertilizer use patterns in Yunnan Province, China: implications for agricultural and environmental policy *Agric. Syst.* **110** 78–89