Trade and the role of non-food commodities for global eutrophication

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**Summary**

The oversupply of nutrients (phosphorous and nitrogen) in fresh and marine water bodies presents a serious ecosystem threat due to impacts on water quality through eutrophication. With agriculture characterized as a primary driver of eutrophication, the role of food consumption and trade has been the focus of recent phosphorus and nitrogen impact studies. However, the environmental impacts associated with non-food commodities are significant and yet to be characterized. We link a spatially-explicit treatment of phosphorous and nitrogen eutrophication potentials to a multi-regional input-output approach to characterize the importance of overall consumption for marine and freshwater eutrophication across 44 countries and 5 rest-of-world regions over the years 2000-2011. We find clothing, goods for shelter, services and other manufactured products account for 35% of global marine eutrophication and 38% of the global freshwater eutrophication footprints in 2011 up from 31% and 33% respectively in 2000. Relative to food consumption, non-food consumption is also significantly more income elastic and shaped by trade. Thus, as economies develop, this points to the need for trade agreements and policies to consider the displacement of ecosystem impacts.

**Introduction**

Current farming practices and the high rates of fertilizer use increase nitrogen (N) and phosphorus (P) mobilization into marine and freshwater systems.1–3 Excess nutrients in water bodies have severe environmental consequences, as they trigger eutrophication which degrades water quality, reduces biodiversity and creates aquatic dead zones.4,5 Industrial animal farming further contributes to eutrophication associated with i) the requirement of intensively farmed feed crops and ii) the buildup of manure which, without sufficient management practices, can leach or runoff into aquatic systems.2,6–9 Eutrophication is a pressing environmental problem worldwide; over 400 ecological ‘dead zones’ caused by eutrophication have been reported, spreading over 245,000 km2 and leading to the loss of over 300,000 metric tons of carbon in biomass.10 In fact, N and P biogeochemical flows have exceeded the levels considered safe for avoiding environmental catastrophe, with 150 Tg N/yr (boundary for avoiding the high-risk zone: 62 Tg N/yr) and 22 Tg P/yr (boundary: 11-100 Tg P/yr) in 2009, thereby highlighting the urgency of addressing this challenge.11

With agriculture characterized as a primary driver of eutrophication, the role of food consumption has been the focus of recent N and P environmental studies.8,12–16 In particular, meat consumption has been identified as a major driver of eutrophication; red meat, for example, has over 50 times the eutrophication potential of cereals.12,16 Increasing food trade has exacerbated this trend by i) reducing the prices of food15, ii) increasing global access to animal feed and, thus, animal products12 and iii) allowing countries to displace emissions by consuming food without bearing the environmental consequences of producing it.12–14,17–21

However, as countries develop, the share of food expenditure relative to total GDP decreases and is instead directed towards services and secondary/tertiary goods that can also depend on agriculture in their supply-chains, e.g. textiles, clothing and furniture.22–24 Therefore, with wealth, the environmental impacts associated with the consumption of non-food commodities becomes increasingly important.25 In addition, the supply chains of these goods are increasingly complex, often involving trade and emissions through a number of countries before reaching the final consumer.26 This displaces emissions, which makes it particularly difficult to address pollution when the relevant actors are spread across several countries.21

In summary, when developing eutrophication mitigation strategies, a sole focus on food consumption can leave a large number of potentially important drivers unaccounted for that will increase with wealth and disperse with globalization. This necessitates a consumption-based accounting perspective because it i) accounts for all impacts (both domestic and foreign) due to a given country’s consumption and ii) can, thus, initiate and motivate both demand- and production-side domestic and international policy development to target environmental issues. For these reasons, here, we for the first time characterize the changing role of trade and consumption in driving marine and freshwater eutrophication (hereafter referred to as ME and FE expressed in million tonnes N equivalents (Mt N eq.) and kilotonnes P equivalents (kt P eq.), respectively). This is based on the assumption that ME is N limited and FE is P limited and, therefore, only considers N as relevant for ME and P for FE. We use a spatially-explicit impact assessment method based on ReCiPe27, calculating country-specific eutrophication footprints using multi-regional input-output (MRIO) analysis, where the entirety of eutrophication impacts occurring along global supply chains are attributed to the final goods and services consumed by citizens of a specific country. We analyze changes over time and aim to understand the role of food and non-food consumption, trade and wealth in driving eutrophication impacts.

**Results**

*Food and non-food eutrophication impacts*

In 2011, the overall final demand for non-food products accounted for over one-third of the global ME (12 Mt N eq. out of 33 Mt N eq.) and FE impacts (580 kt P eq. out of 1520 kt P eq.; figure 1 right). This is a 28% increase from 2000 values, with 8.7 Mt N eq. and 420 kt P eq. for ME and FE, respectively. In line with previous findings21,28, agriculture was the dominant production-side driver, accounting for 84% of the total footprints for both ME and FE; however, we find that, from a consumption standpoint in 2011, approximately one-quarter of these agricultural impacts were due to non-food consumption (see SI data file for a list of non-food products).

In total, Asia and Pacific (in particular China and India), US and Germany accounted for 54% of the global ME footprint (33 Mt N eq., figure 1 top left) and Africa, US, China, Brazil and India accounted for 62% of the global FE footprint (1500 kt P eq., figure 1 bottom left). A monte carlo-based sensitivity analysis confirms the robustness of these and all following results (see SI for details). Under a “high uncertainty” scenario (model parameters with assigned relative standard deviations of 30-50%) the relative standard error of the FE and ME footprint was <5% at the global total level, with a mean relative standard error at the country level of 16.9% (1 std. dev. = 6.8%) for FE footprints and 12.4% (1 std. dev. = 4%) for the ME footprints. These reduce to 5.7% and 4.2% for FE and ME respectively in a less conservative scenario.

For ME in 2011, the highest country-level footprint related to the consumption of both domestically produced and imported non-food commodities occurred in China, with 3 Mt N eq. out of a total footprint of 8.6 Mt N eq. (see SI, figure 1, left), doubling China’s 2000 non-food footprint of 1.5 Mt N eq. For FE associated with non-food consumption, the U.S. was the largest country-level contributor (263 kt P eq.) in 2011, which is nearly triple their 2000 values of 97 kt P eq. China had the second highest FE non-food footprint in 2011 (180 kt P eq.), over five times the 2000 values (33 kt P eq., See SI for a full breakdown by sector and region).

In contrast to non-food footprints, the global food footprints only modestly increased from 2000 to 2011 (ME: 19 Mt N eq. to 21 Mt N eq. and FE: 850 kt P eq. to 950 kt P eq.). However, this was not the case for China’s food footprint, which increased by over 25% in the same period, peaking at 5.4 Mt N eq. for ME in 2011. For FE, the highest country-level food footprints were observed in the U.S. with 149 kt P eq. in 2000 and 140 kt P eq. in 2011.

The breakdown of global footprints reveals the relative importance of specific consumption categories for ME and FE and how this has changed over time. In 2000 for ME, plant-based food accounted for 22.8% of the total footprint, 28.7% for animal-based food, 17% for processed food, 2.1% for clothing, 6.2% for shelter, 6.2% for manufactured products, 2.9% for waste services and 13.9% for other services (see SI data file for aggregation key). In 2011, the relative share for plant- and animal-based food declined by 3.2% and 2.7%, respectively, while all other categories increased, especially processed food (from 17% to 19.1%), manufactured products (6.2% to 7.6%) and waste services (2.9% to 4.7%). Compared to ME, the FE consumption shares had a different trend: plant-based food accounted for 6%, 50.9% for animal-based food, 10.3% for prepared food, 2.5% for clothing, 4.4% for shelter, 4.3% for manufactured products, 6.6% for waste services and 15% for other services in 2000. In 2011, both plant-based and animal-based food decreased to 5% and 46.9%, respectively, while processed food stayed the same and all other categories increased, especially shelter (4.4% to 5.2%) and waste services (6.6% to 9%).

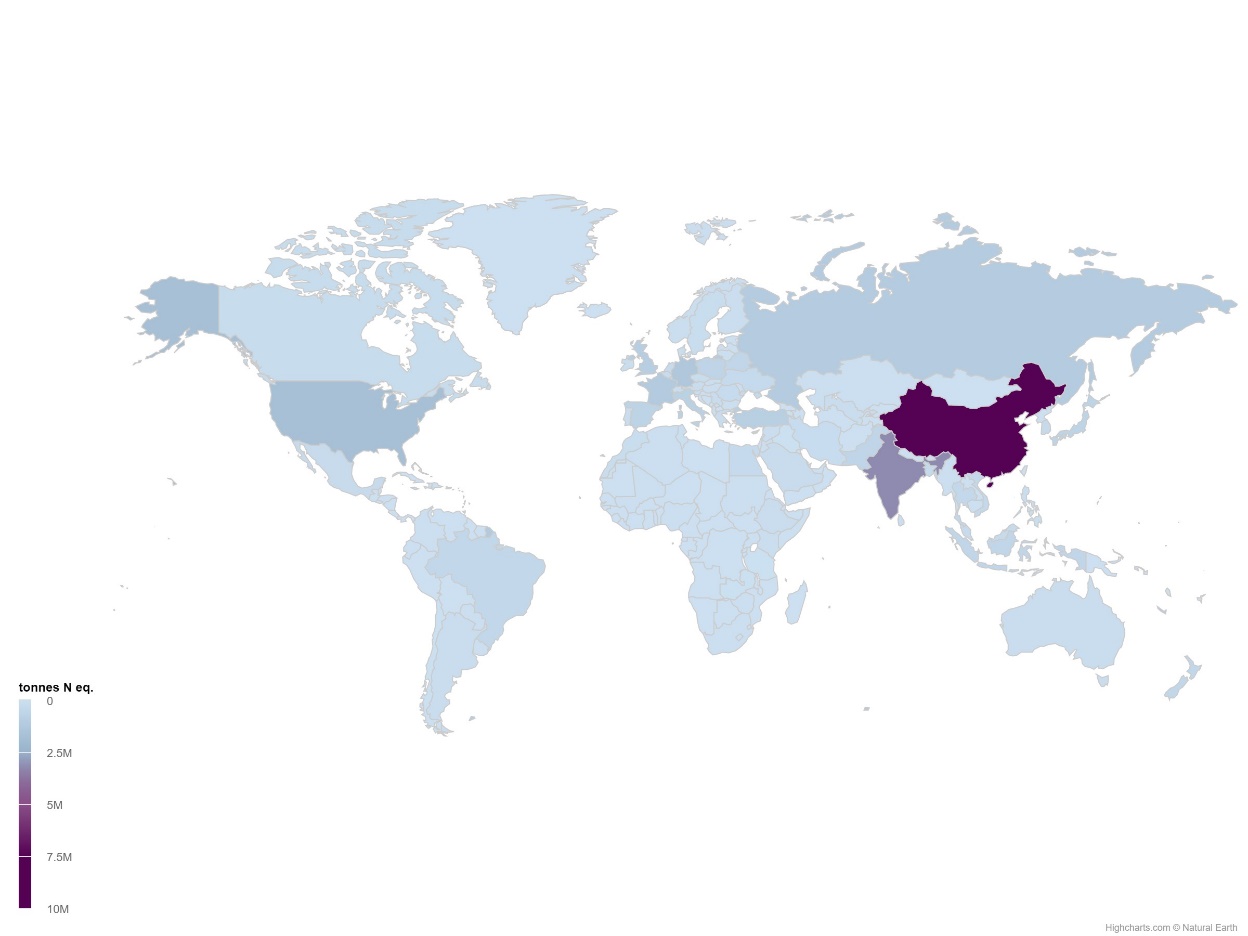
*Trade related food and non-food eutrophication impacts*

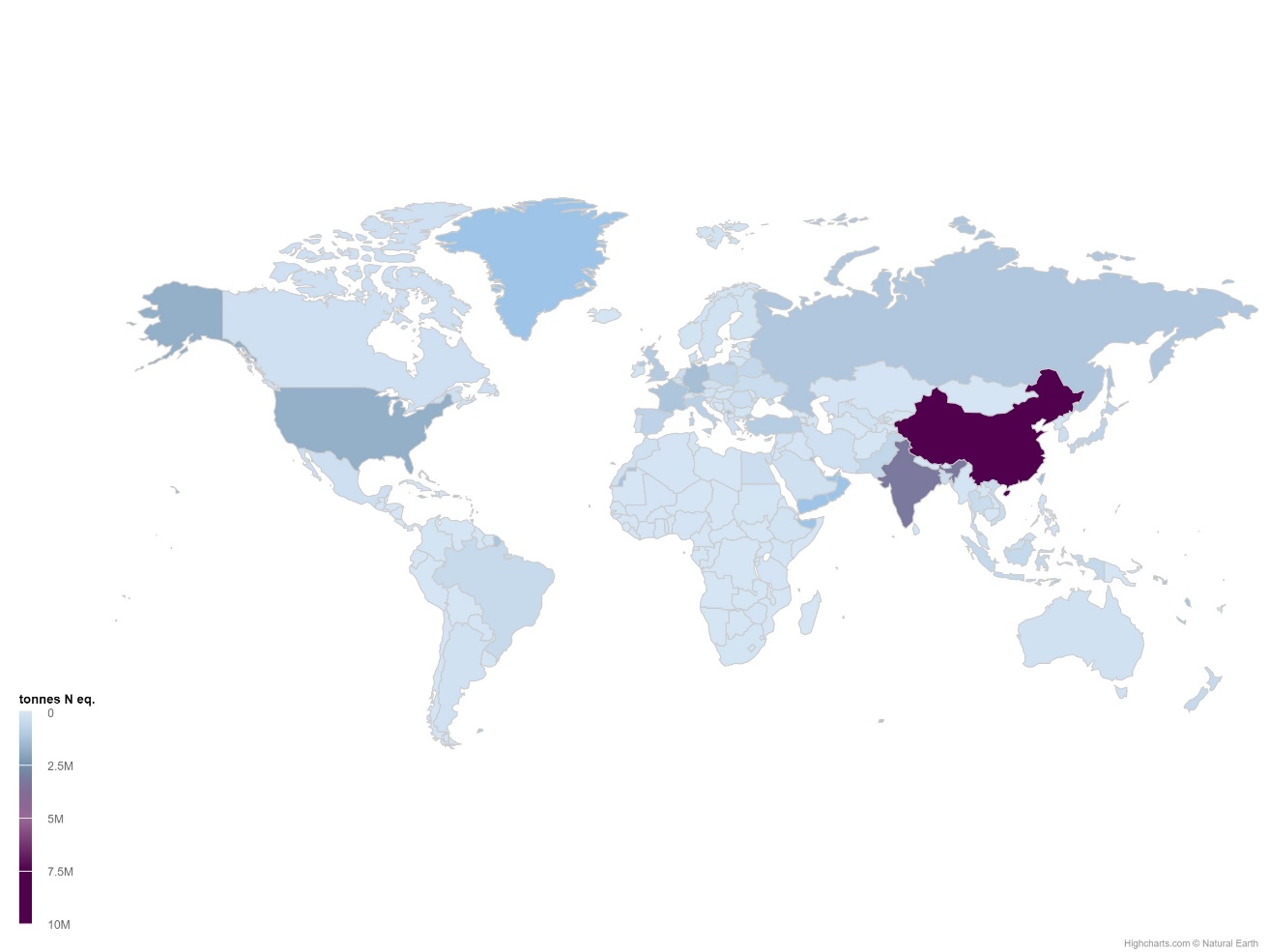
Overall, the impacts from the production of traded goods and services have increased from 6.9 Mt N eq. and 274 kt P eq. in 2000 to 8.7 Mt N eq. (26% of global impacts) and 313 kt P eq. (21% of global total) in 2011 for ME and FE, respectively (see SI, figure 1, right). These global trade shares are comparable to previous studies. For example, Oita et al. (2016)21 found, in 2010, 26% reactive nitrogen embodied in global trade and Mekonnen et al. (2016)20 found, in 2007, 41% of P and N emissions (termed grey water footprint) embodied in the European Union’s (EU) consumption of imported goods, where we find 46% for ME and 70% for FE for the same year. Furthermore, we find that only a few countries and regions bear the majority of impacts for the production of traded commodities. For example, China, Eastern Europe and Asia and Pacific account for 48% of all traded ME impacts.

Out of the global traded impacts, 49% was due to non-food consumption for both ME and FE in 2011. This is a substantial increase from 2000 values, highlighting the increasing importance of non-food trade for both ME and FE (figure 2, right). The ME impacts embodied in non-food trade even surpassed food trade in 2007, primarily driven by the U.S. consumption of Chinese produced clothing, leather, fur and furniture and Asian (excluding China) consumption of Chinese produced textiles and clothing (refer to the SI data file for data). However, this upsurge was abruptly stalled due to the economic recession, as seen from the sharp decline in traded embodied non-food impacts from 2008 to 2009 (figure 2, right). The downturn was largely because of the respective 21% and 32% decrease in impacts embodied in non-food exports (primarily clothing, leather and furniture) from China and other Asian countries to the U.S. Despite the decline in U.S. consumption over this period, growth was seen in a few regions, e.g. the increased consumption of Eastern European non-food products by the Middle East and Africa (mostly chemicals and construction materials). Nonetheless, after 2009, the U.S. and other economies recovered and, since, the importance of non-food trade appears to be ever increasing.

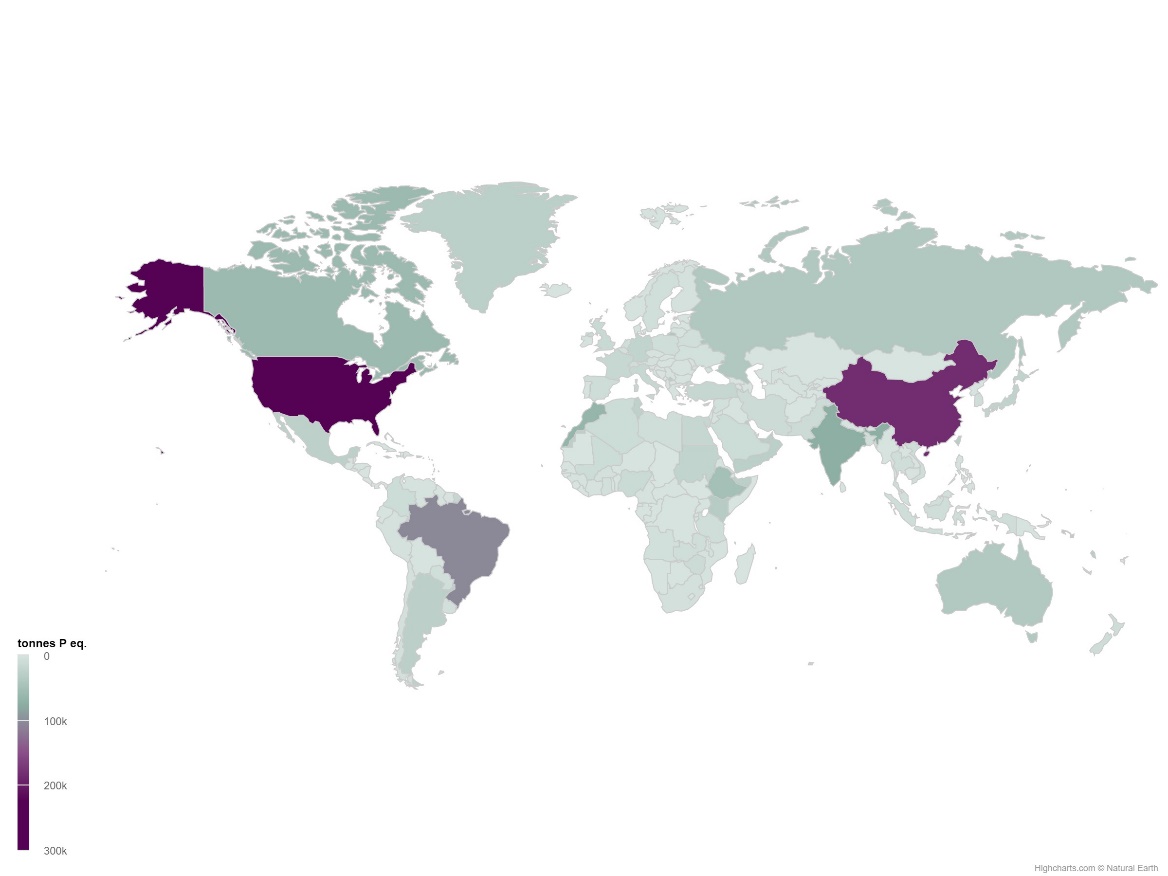
With regards to impacts from traded food, we find that for both ME and FE the impacts stay relatively stable over time. One exception is the slight upturn for ME traded food impacts from 2009 to 2011. This was primarily due to increased food imports by Spain and Russia from Eastern Europe and Asia, respectively.

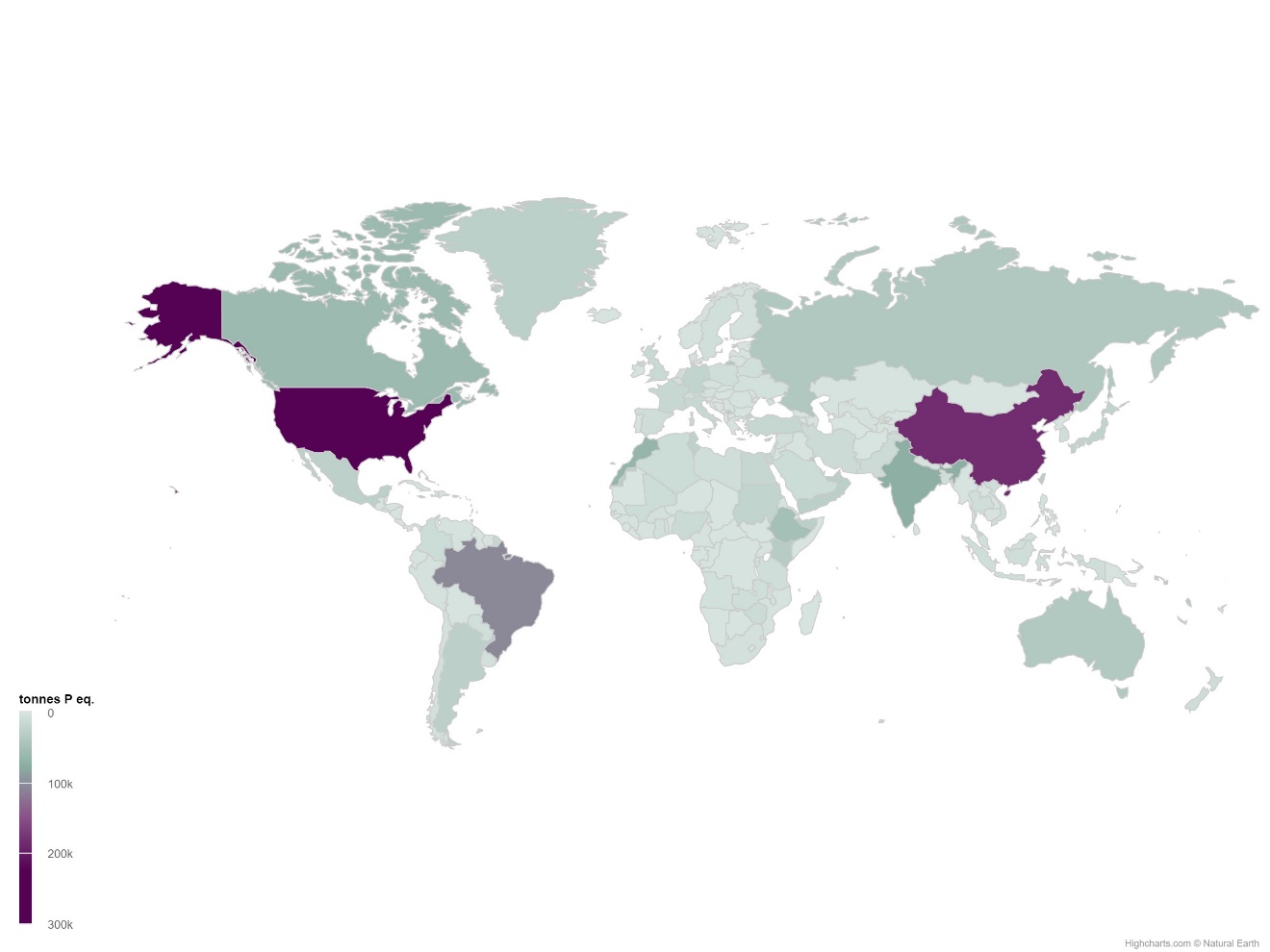
At a continent level we find that the top displacement of ME impacts is from Asia and Pacific to Europe, with 620 kt N eq. embodied in food and 290 kt N eq. embodied in non-food (figure 2, top left). For FE, we find that the top displacement of impacts is the European import of eutrophication embodied in commodities from Africa (19 kt P eq. for food and 15 kt P eq. for non-food; figure 2, bottom left). These European-level eutrophication displacements are primarily driven by EU consumption (96% of the total displacement for both ME and FE). In fact, EU consumption represented 28% and 33% of total ME and FE traded impacts (both food and non-food). This highlights i) the EU’s high exposure to trade and ii) their strong role in driving traded related eutrophication impacts.













**Figure 1.** Global ME and FE footprints for 2011 by country (left) and global ME and FE footprints for the years 2000, 2005 and 2011 (right). Footprints are broken down i) based on whether they occurred domestically or from the consumption of imported products, ii) by producing sector (crop production, animal husbandry and other sectors) and iii) consumed product type (food and non-food). Global footprint totals are equal to global total direct impacts. See SI for numerical values.

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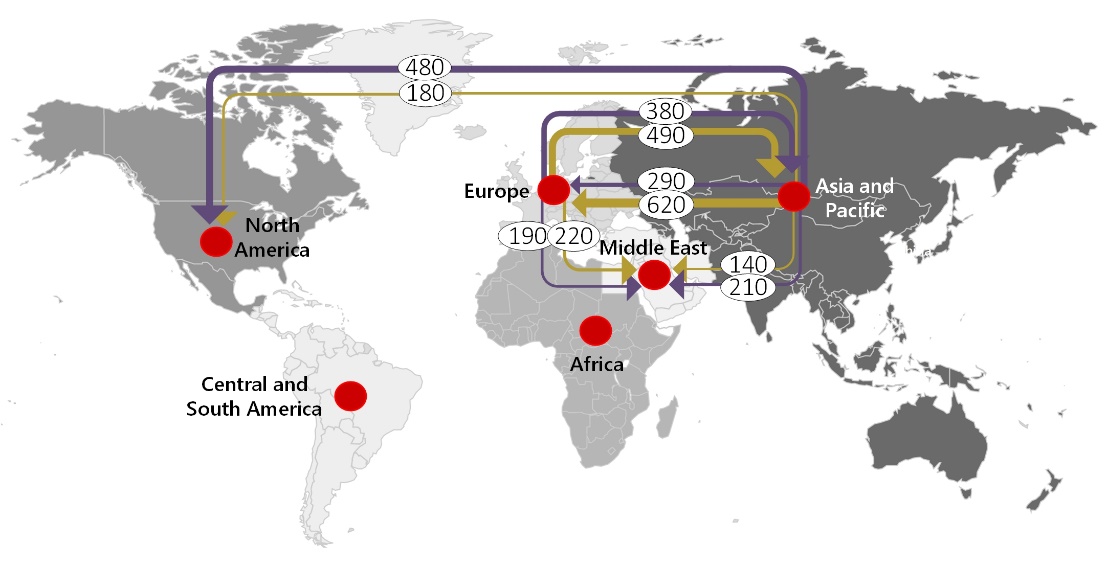
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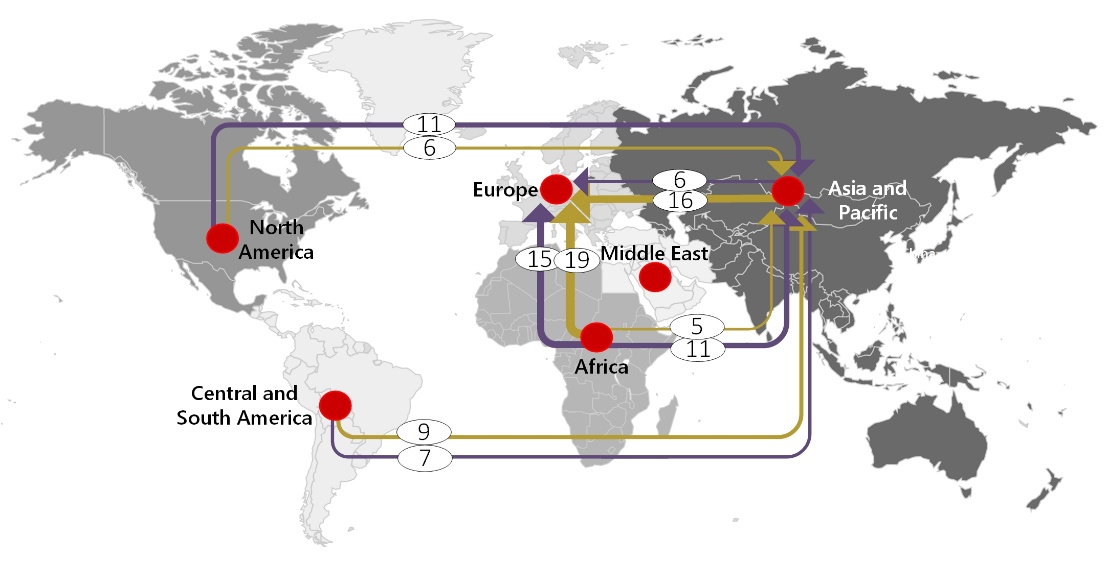
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kt P eq.

**Figure 2.** Top 5 continent-level displacements of marine (top left, kt N eq.) and freshwater (bottom left, kt P eq.) eutrophication associated with trade satisfying food and non-food demand. Arrows represent the flow of embodied impacts that occur in the country of origin (beginning of the arrow) for the consuming country (end of arrow). Gray shading differentiates regions; Global trade marine (top right, Mt N eq.) and freshwater (bottom right, kt P eq.) footprints over time based on production sector and consumed product type.

*Income Eutrophication Relationships*

|  |  |  |  |  |  |  |  |  |  |  |
| --- | --- | --- | --- | --- | --- | --- | --- | --- | --- | --- |
|  |  | **Marine Eutrophication (ME)** | | | |  | **Freshwater Eutrophication (FE)** | | | |
|  |  | Income Elasticity (95% CI) | | Sig | R2 |  | Income Elasticity (95% CI) | | Sig | R2 |
| **Cross-sectional analysis (2011)** | **Total** | **0.93** | **(0.68, 1.18)** | **\*\*\*** | **0.51** |  | **0.79** | **(0.39, 1.20)** | **\*\*\*** | 0.44 |
| **Food** | **0.91** | **(0.61, 1.21)** | **\*\*\*** | **0.42** |  | **0.74** | **(0.34, 1.13)** | **\*\*\*** | 0.39 |
| Plant-based food | 0.72 | (0.27, 1.17) | \*\*\* | 0.21 |  | 0.59 | (0.16, 1.01) | \*\*\* | 0.14 |
| Animal-based food | 0.88 | (0.59, 1.17) | \*\*\* | 0.36 |  | 0.69 | (0.24, 1.14) | \*\*\* | 0.31 |
| Processed food | 1.21 | (0.94, 1.48) | \*\*\* | 0.58 |  | 1.07 | (0.75, 1.40) | \*\*\* | 0.62 |
| **Non-food** | **0.99** | **(0.74, 1.24)** | **\*\*\*** | **0.60** |  | **0.88** | **(0.43, 1.34)** | **\*\*\*** | 0.46 |
| Clothing | 1.04 | (0.79, 1.29) | \*\*\* | **0.62** |  | 0.97 | (0.58, 1.35) | \*\*\* | 0.51 |
| Shelter | 0.90 | (0.61, 1.20) | \*\*\* | 0.40 |  | 0.96 | (0.49, 1.43) | \*\*\* | 0.48 |
| Manufactured products | 1.01 | (0.81, 1.22) | \*\*\* | 0.71 |  | 0.87 | (0.54, 1.20) | \*\*\* | 0.63 |
| Waste services | 0.72 | (0.13, 1.32) | \*\* | 0.11 |  | 0.20 | (-0.36, 0.75) |  | 0.01 |
| Other services | 1.16 | (0.82, 1.50) | \*\*\* | 0.63 |  | 1.16 | (0.51, 1.81) | \*\*\* | 0.49 |
| **Panel analysis (2000-2011)** | **Total** | **0.27** | **(0.09, 0.45)** | **\*\*\*** | **0.99** |  | **0.35** | **(0.17, 0.52)** | **\*\*\*** | **0.98** |
| **Food** | **0.08** | **(-0.11, 0.27)** |  | **0.99** |  | **0.13** | **(-0.06, 0.31)** |  | **0.98** |
| Plant-based food | 0.06 | (-0.25, 0.38) |  | 0.97 |  | 0.03 | (-0.51, 0.57) |  | 0.90 |
| Animal-based food | -0.02 | (-0.28, 0.24) |  | 0.98 |  | 0.03 | (-0.20, 0.27) |  | 0.97 |
| Processed food | 0.43 | (0.10, 0.76) | \*\* | 0.98 |  | 0.54 | (0.25, 0.84) | \*\*\* | 0.97 |
| **Non-food** | **0.67** | **(0.38, 0.96)** | **\*\*\*** | **0.97** |  | **0.85** | **(0.54, 1.15)** | **\*\*\*** | **0.97** |
| Clothing | 0.75 | (0.46, 1.05) | \*\*\* | 0.96 |  | 0.94 | (0.54, 1.35) | \*\*\* | 0.94 |
| Shelter | 0.83 | (0.50, 1.16) | \*\*\* | 0.97 |  | 1.03 | (0.62, 1.44) | \*\*\* | 0.96 |
| Manufactured products | 1.25 | (0.99, 1.51) | \*\*\* | 0.96 |  | 0.94 | (0.66, 1.22) | \*\*\* | 0.95 |
| Waste services | 1.18 | (-1.46, 3.82) |  | 0.84 |  | 1.98 | (-0.90, 4.86) |  | 0.82 |
| Other services | 0.41 | (0.03, 0.79) | \*\* | 0.97 |  | 0.75 | (0.28, 1.21) | \*\*\* | 0.97 |

**Table 1**. Income elasticities of footprints based on 49 countries and RoW regions by consumption category (2000-2011). Cross-sectional ordinary least squares (OLS) model for 2011 data (using robust standard errors) and fixed-effects linear model for 2000-2011 panel data (using clustered standard errors). The dependent variables are log transformed values of footprints by consumption domain (in g N eq./cap for ME and g P eq./cap for FE). The independent variable is the log-transformed GDP per capita in 2011-constant USD PPP per capita. Significance level (Sig): \* p <0.1; \*\* p <0.05; \*\*\* p <0.01. CI = Confidence interval

We test wealth as a potential factor affecting the distribution of ME and FE across countries and time. To increase the robustness of our results, we perform cross-sectional (2011) and panel data regression analysis (2000-2011) on the per capita level. Positive and significant coefficients (and 95% confidence intervals) suggest that affluence, measured by per-capita GDP, increases ME and FE footprints (table 1).

This analysis indicates that a 1% increase in GDP per capita leads to an increase of 1.0% and 0.9% in the ME and FE footprints (cross-section elasticity coefficients for non-food: 0.99 and 0.88 respectively, table 1 top). All cross-sectional models share positive and highly significant coefficients, suggesting that eutrophication impacts are responsive to increases in income with regards to both food and non-food consumption. We cannot conclude significant differences between food and non-food coefficients (table 1, top).

The panel data analyses indicate that non-food consumption is significantly more income-elastic than food consumption for the period 2000-2011. For non-food models, the ME- and FE- income coefficients are 0.67 (0.38, 0.96) and 0.85 (0.54, 1.15), respectively (table 1, bottom). Clothing, shelter, and manufactured products are particularly elastic and significant at the 1% level (table 1, bottom). While the share of global impacts across these consumption categories is relatively small, they have increased rapidly with economic growth: the global ME shares for clothing, shelter, and manufactured products have grown from 2.1 to 2.2%, 6.2 to 7.2%, and 6.2 to 7.6% from 2000 to 2011. For FE, the shares have increased from 2.5 to 2.6%, 4.4 to 5.2%, and 4.3 to 4.7% for the above categories, respectively. Services and, in particular, waste services are less responsive to changes in income (table 1, bottom). In terms of plant- and animal-based food, income elasticities are insignificant at the 5% level both for ME and FE. However for processed food, we find statistically significant coefficients of 0.43 (0.10, 0.76) and 0.54 (0.25, 0.84) for ME and FE, respectively. Prior studies have noted dietary shifts to processed food associated with increased affluence23,29–31. See SI for a further discussion of regression analyses.

**Discussion**

The role of food and diets as drivers of eutrophication has been the focal point of recent studies8,12–16,32; this is justified considering that agriculture accounts for the vast majority of production-side eutrophication impacts and has a clear link to food production. However, we find that focusing on food consumption and diets alone would lead to a systematic and significant underestimation of eutrophication impacts from a consumption viewpoint. With non-food consumption growing over time, being highly responsive to changes in wealth and subject to large global supply chain fragmentation, it is increasingly important to consider these environmental concerns in policy development.

To adequately address eutrophication, a variety of policy instruments and strategies are needed. This is because N and P emission pathways are complex: they originate from point and non-point sources, are emitted by a number of sectors and have a variety of drivers.33 Domestic policies include improving agricultural practices through reduced fertilizer use and improved animal husbandry feeding practices. These can be enforced or incentivized through e.g. country/region-level regulatory standards (e.g. the EU Nitrate directive mandates the designation of sensitive farming areas34) and fiscal and economic incentives (e.g. subsidies35 and polluters pay tax and).33

However, our analysis has shown that the strong and increasing trade component of agriculture prompts the need for accounting for all eutrophication due to a country’s consumption - both domestic and international. Otherwise, reducing eutrophication domestically could be achieved through outsourcing impacts to other countries. For example, our results show that the vast majority of all eutrophication related to the EU’s non-food consumption occurs in other regions. The EU both i) drives the largest global non-food eutrophication displacements to Asia-Pacific and Africa for ME and FE, respectively and ii) displaces a high percentage of non-food impacts. While the EU has developed frameworks and strategies for tackling eutrophication within Europe34,36, policies that integrate international supply chains for addressing eutrophication abroad are lacking.21,37 Such policies are especially important when developing countries are the primary recipients of displaced impacts and these impacts potentially impede the country’s ability to sustainably grow. For example, in China, the total economic losses due to FE are valued at billions of yuan38 and, in addition, frequently disrupt the natural drinking supply of coastal Chinese cities.38 This analysis has shown that over 13% of these impacts are the result of producing products for export.

We argue that a consumption-based approach provides key insights into consumer drivers, increases stakeholder engagement as well as increases the pressure for implementing policy through both demand- and production-side measures. Demand-side policies can include consumption-based targets (e.g. a 40% reduction in the EU’s global eutrophication footprint), and measures can include trade agreements and pricing mechanisms. Furthermore, the transfer of technology/skills to countries for e.g. improving fertilizer efficiencies or managing waste (a production-side mitigation) can be motivated through a consumption approach. This is especially so considering that consumer-driven environmental policy development often comes from wealthy regions, who can more easily afford the resources needed to support the implementation of policies in developing countries.

*Uncertainty and spatial variability*. Modeling eutrophication impacts at a global scale is inherently associated with uncertainty and is highly limited by data resolution. The MRIO model used provides data detailed at the product-level (15 agricultural goods, 12 food commodities, around 40 manufactured goods [non-food/agricultural/energy goods], and 25 services) the geographical resolution is restricted to 44 countries and 5 rest of world regions. Alternative MRIO models provide a higher country granularity (e.g. Eora39), however, these models lack the estimation of both N and P emissions as well as the product specificity necessary to characterize non-food commodities separate from food commodities. Other methods based on physical flow accounting have captured the physical dimension of agricultural trade, but have not been able to quantify non-food demand40. In the calculation of environmental footprints, MRIO approaches use relative monetary relationships, which reflect the economic demand for goods; this is in comparison to physical trade approaches, which would reflect relative demand for goods in mass terms. There is no general consensus on the best approach, but it is considered that consumption based impacts due to the economic MRIO data have an impact on uncertainty of ca +/-10-20% at the national level41. A difference in scope is found in that MRIO approaches distinguish intermediate from final food consumption. Using hypothetical extractions (see methods) our results showed that intermediate demand for food, in for example, government services, contributes to roughly 9 % of the global total of FE and ME footprints respectively and roughly 15-20% of the non-food footprints. Future MRIO development should focus on both disaggregating world regions and improving product level detail in order to fully assess the importance of individual products for driving environmental impacts.42 The N and P accounts derived in this work can be further refined to capture spatial variations in agricultural practices to the point of emission. See SI for further details.

The impact assessment methodology is based on the generic assumption that P is the limiting nutrient for primary production of biomass in freshwater systems and N is the limiting nutrient for primary production in marine systems. Research has challenged this assumption and has shown that N and P can be both limiting for marine and fresh waters.4,43,44 In freshwater systems, studies suggest that nitrogen as well as iron could be co-limiting over shorter time-scales.45 Therefore, we recommend further research to focus on deriving eutrophication potentials for all nutrients in both marine and freshwater ecosystems.

Furthermore, Azevedo et al.43 showed that spatially-explicit methodologies for impact assessment are crucial for accurate assessments of P emission impacts in freshwater systems. Currently, we account for the spatial variability of P inputs to freshwater systems with country-specific eutrophication potentials. Thereby, we account for the difference in residence time in water bodies, whilst recognizing that the relationship to ecological responses are more complex. The variability in FE used is mainly caused by variability in hydrological residence time between river basins.33,46 While FE potentials were country-specific, ME potentials were only available per continent. Results of the work by Cosme et al.47 show that eutrophication damage indicators in marine ecosystems can vary up to 4 orders of magnitude for emissions to rivers and marine waters. This could significantly influence the results, particularly for determining which country contributes most to ME impacts (see SI for detail).

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**Author contributions**

H.H. and R.W. designed the study. R.W. prepared the IO model and basic results. S.M. and J.S. developed the phosphorus and nitrogen accounts. R.Z. prepared the impact assessment method. H.H. and D.I. conducted the analysis. D.M. conducted the sensitivity analysis. H.H. made the figures. H.H., R.W., K.S. and D.I. contributed to the data interpretation. H.H., D.I., and R.W. wrote the paper. H.H., R.W., D.I., R.Z., K.S., S.M., D.M. and J.S. contributed to manuscript editing.

**Competing interests**

The authors declare no competing interests.

**Methods**

We apply environmentally extended multi-regional input output analysis (MRIO) to quantify the summed supply chain emissions of nitrogen and phosphorus due to consuming a given good or service. Spatially explicit life cycle impact assessment methods were used to characterize the physical emissions into a standard unit termed ‘eutrophication potential’ that spatially differentiates the level of impact in i) N equivalents for marine eutrophication footprints and ii) P equivalents for freshwater eutrophication footprints.48 This was done for each year in the period between 2000 and 2011. Finally, we explore the relationship between the observed impacts and wealth through a set of univariate regression models. For the sensitivity analysis, the primary variables of the model – the trade flows, consumption patterns, P and N releases, and eutrophication impact of these releases – were perturbed and the model result sensitivity to those perturbations were measured. The below paragraphs provide a detailed description of the methods and key assumptions made in this analysis.

*Environmentally Extended Multi-Regional Input-Output analysis*

MRIO analysis is a widely used tool for calculating environmental footprints for various environmental pressures49, for example carbon footprints24, land footprints, water footprints50 and labor footprints51. MRIO analyzes inter-industry flows between economic sectors both domestically and abroad. This allows to distinguish between domestic impacts versus impacts that occur in other countries due to the consumption of traded goods and track the onward processing of goods across multiple borders (for example, the production of soy in South America, to the production of beef cattle in North America, to the import of leather to Indonesia for textile production to the final demand of clothing in Europe). The use of MRIO for calculating environmental footprints has been documented extensively and further information can be found, for example, in Turner et al.52 We use a standard Leontief demand pull technique to allocate environmental pressure to final consumption category, using a full MRIO approach.53,54

Here, we use the EXIOBASE (v3.4) MRIO model49,55,56 to calculate freshwater and marine eutrophication footprints. EXIOBASE provides globally consistent disaggregation of agricultural and food products with the principal goal to separate out animal, crop and fiber crop based supply chains. EXIOBASE provides full coverage of individual EU countries and 15 other major economies, whilst modelling five other rest of the world (RoW) regions grouped by continent. Fifteen agricultural industries are modelled and 12 food processing industries as well as separate industries for forestry and textile production. We use version 3 of the database56, for years 2000-2011 in the industry by industry classification. We conduct the analysis at the full level of database disaggregation before aggregating results into sector groups based on production method (animal husbandry versus crop production versus other industries) and product type (food versus nonfood). See supporting information for the sector aggregation key. Other MRIO models exist such as GTAP57 and Eora39. Eora does not have the product resolution for disaggregating food and non-food drivers, whilst GTAP lacks time-series data. A comparison of results across choice of MRIO model focusing on regional and product level aggregation effects for national footprints has been undertaken by a number of authors for other environmental indicators, finding country level differences commonly in the range of 10-20%41,58–61. A further discussion of these issues is provided in the supplementary material.

In MRIO modelling food is treated as both a good for intermediate and final consumption. In order to avoid double counting, all impacts of intermediate consumption are allocated to final consumption in the Leontief demand calculation. Hence, food provided by the workplace, by education or health facilities will be shown in the footprint of the consumption of the respective service. It is possible to estimate the magnitude of such effects using a technique known as “hypothetical extraction”.53,62 In such cases, the flows of interest are set to zero in the intermediate coefficient matrix, and the difference between the results from the full coefficient matrix and the adjusted coefficient matrix gives the contribution of the flows to the overall footprint. In this case, we set all food flows into the service sector to zero across the whole MRIO database and calculate the resultant FE and ME footprints. In the supplementary material, we present results as a fraction of the total FE and ME footprint globally.

*Phosphorus and Nitrogen Accounts*

The P and N emissions due to various sector activities are included in the EXIOBASE dataset and the calculation procedure is described in full by Merciai and Schmidt.63 Here, we provide a summary of their approach. P and N emissions that result from crop production are calculated using a mass balance approach, where the emissions are the difference between the nutrient inputs (chemical fertilizer and manure) and outputs (i.e. crops, fodder and pasture). Production levels of individual crops, fodder crops and pasture are obtained at the national level from the FAOSTAT database64. The total domestic demand of nutrients, as defined by the International Fertilizer Industry Association65, are distributed to crops, fodder crops and pasture using distribution factors obtained from FAO66 and Heffer67. Emissions from fodder crops, pasture and manure not spread on land are allocated to the respective livestock activities. N and P emissions from sewage treatment plants and landfills are modelled based on estimated food consumption and other relevant industrial inputs, and allocated directly to the relevant waste sector in EXIOBASE. Non-agricultural N emissions to air are based on UNFCCC inventories and emission factors applied to fuel combustion68. It is important to note that our methods were specifically developed to only consider the anthropogenic system (excluding the natural background losses). This is so that we can isolate the effect of consumption and income to improve our understanding of eutrophication drivers.

With regard to agricultural emissions of N, the protein content of crops, converted to N using the factor 6.25 kg protein/kg N69, gives the amount of absorbed N. The IPCC’s procedure70 is then applied to determine the direct and indirect (leaching) water emissions of N2O and NO3 and the air emissions of NOx, NH3 and NO3. We assumed that the remainder are N2 emissions. If the calculated N2 turned out negative, the protein content of the crop was adjusted to ensure a consistent N-balance.

With regard to P, the quantity absorbed by crops is determined from a database on food compositions71 supplemented by Ketterings and Czymmek72. We assume the residual part is emitted, with 2.9% of total emissions leaching into water based on the rest accumulating in soil stocks. Whenever the quantity of absorbed P is higher than the applied quantity, no emissions occur. Because these factors vary spatially, there is a strong need for further research in this domain. The factors we use represent common practice in life-cycle assessment and can be compared against the global average estimate of 10% leaching of P inputs to soil to water emissions73, which is an overestimate compared to Bennet (2001)74.

The modelling of emissions from land application of manure management (in the stable and storage) is based on the IPCC, chapter 11.70 The input parameters for this, i.e. the amount of manure, is calculated based on metabolic mass balances of animals for all animal categories and countries included in EXIOBASE.63 These emissions are allocated directly to the corresponding input-output sector of each animal type. Further information regarding this approach can be found in the SI.

*Impact assessment methods*

We characterize the N and P emissions following ReCiPe 2016v1.148, converting them to respective marine and freshwater eutrophication potentials. In contrast to previous versions, ReCiPe 2016v1.1 provides country-specific eutrophication potentials that capture the differing impacts P emissions have in different freshwater ecosystems, while it includes continent-specific factors for N emissions. River-basin specific eutrophication potentials include the fate of P and N in aquatic systems, i.e. residence time in lakes and rivers determined by inflow, advection, retention and water use processes.46,75,76 As emissions to air are not characterized for marine eutrophication in ReCiPe, we determined the potentials by including the atmospheric fate and chemistry of the compounds from Roy et al.77 with the soil fate from Cosme et al.76 Continent aggregated potentials for marine eutrophication were determined based on emission data.27 Country-aggregation for freshwater eutrophication in ReCiPe was based on gridded population estimates, representing wastewater treatment plant emissions in urban areas. For the purpose of this paper, we recalculated the country aggregates for emissions to agricultural soil based on gridded P fertilizer and manure application.6 ReCiPe methods are based on the underlying assumption that freshwater eutrophication is P limited and marine eutrophication is N limited. Therefore, it only considers P emissions as relevant for freshwater eutrophication and N emissions as relevant for marine eutrophication. See SI data file for data.

*Regression analysis*

We present income elasticities on the freshwater and marine eutrophication footprints based on data from 44 countries and 5 rest-of-the-world regions. Our analysis is conducted on per capita values for ME (measured in g N eq./cap), FE (in g P eq./cap) and income (GDP/cap in 2011-constant USD PPP). This is done in order to simplify the analysis and isolate the income effect from population changes78. Income coefficients are reported separately for ME and FE, and across consumption categories (total, food vs non-food, and disaggregated further by eight consumption categories). Prior studies have calculated and used income/expenditure elasticities in the context of other environmental indicators, e.g. carbon footprints24,79,80, land and water use23,81,82, and energy requirements83.

We study the relationship between ME/FE and income using two approaches: regression analysis on the cross-sectional data for 2011 across individual countries, and a panel analysis conducted on individual countries over time (2000-2011). The cross-sectional analysis explores inter-country variation in a single year (2011) using ordinary least squares (OLS) regressions. In addition, we examine the temporal dimension of the data (2000-2011) using the fixed-effects approach25. Our models broadly agree about the importance of income for ME and FE impacts driven by non-food consumption. Using the fixed-effect approach, we control for the time-invariant differences across countries that have an effect on FE and ME. Such factors include soil type, soil heavy metal content and precipitation amounts, amongst others. These country effects rise the explanatory power of our model significantly (table 1). It is worth noting, however, that our panel is relatively short for the economic growth effect to unfold and cause a significant footprint change within a country. We consult results from the Hausman test25 for the choice of panel data method (see SI). Further information on the estimated models, descriptive statistics, and robustness checks (pooled OLS and random-effects models) is provided in the SI.

**Data availability statement**

This work uses the EXIOBASE dataset which is a secondary data source released as a freely available dataset through [www.exiobase.eu](http://www.exiobase.eu). See references for data specification. All figures are based on model results from this dataset. The latest version of the dataset is available on request, or through [www.exiobase.eu](http://www.exiobase.eu).

**Code availability statement**

Two codes are used in the text. One to generate results (MATLAB), and one to perform the statistical analysis (STATA). Code is available directly from the authors on request.

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