ANALYSIS
https://doi.org/10.1038/s41893-018-0079-z

nature sustainability

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

Helen A. Hamilton[1], Diana Ivanova[1], Konstantin Stadler[1], Stefano Merciai[2], Jannick Schmidt[2], Rosalie van Zelm[3], Daniel Moran[1] and Richard Wood[1*]

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. Here, 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 period 2000–2011. We find that 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. As economies develop, this points to the need for trade agreements and policies to consider the displacement of ecosystem impacts.

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

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[6–10]. 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[7,10]. Increasing food trade has exacerbated this trend by (1) reducing the prices of food[9], (2) increasing global access to animal feed and, thus, animal products[7], and (3) allowing countries to displace emissions by consuming food without bearing the environmental consequences of producing it[7,8,11–13].

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, such as textiles, clothing and furniture[14]. Therefore, with wealth, the environmental impacts associated with the consumption of non-food commodities becomes increasingly important[15]. 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[16]. This displaces emissions, which makes it particularly difficult to address pollution when the relevant actors are spread across several countries[12].

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 accounts for all impacts (both domestic and foreign) due to a given country's consumption, and can therefore initiate and motivate both demand- and production-side domestic and international policy development to target environmental issues. For these reasons, here we 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 ReCiPe[17], 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 analyse changes over time and aim to understand the role of food and non-food consumption, trade and wealth in driving eutrophication impacts.

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 1,520 kt P eq.; Fig. 1b). This is a

[1]Norwegian University of Science and Technology, Trondheim, Norway. [2]Aalborg University, Aalborg, Denmark. [3]Radboud University, Nijmegen, the Netherlands. *e-mail: richard.wood@ntnu.no

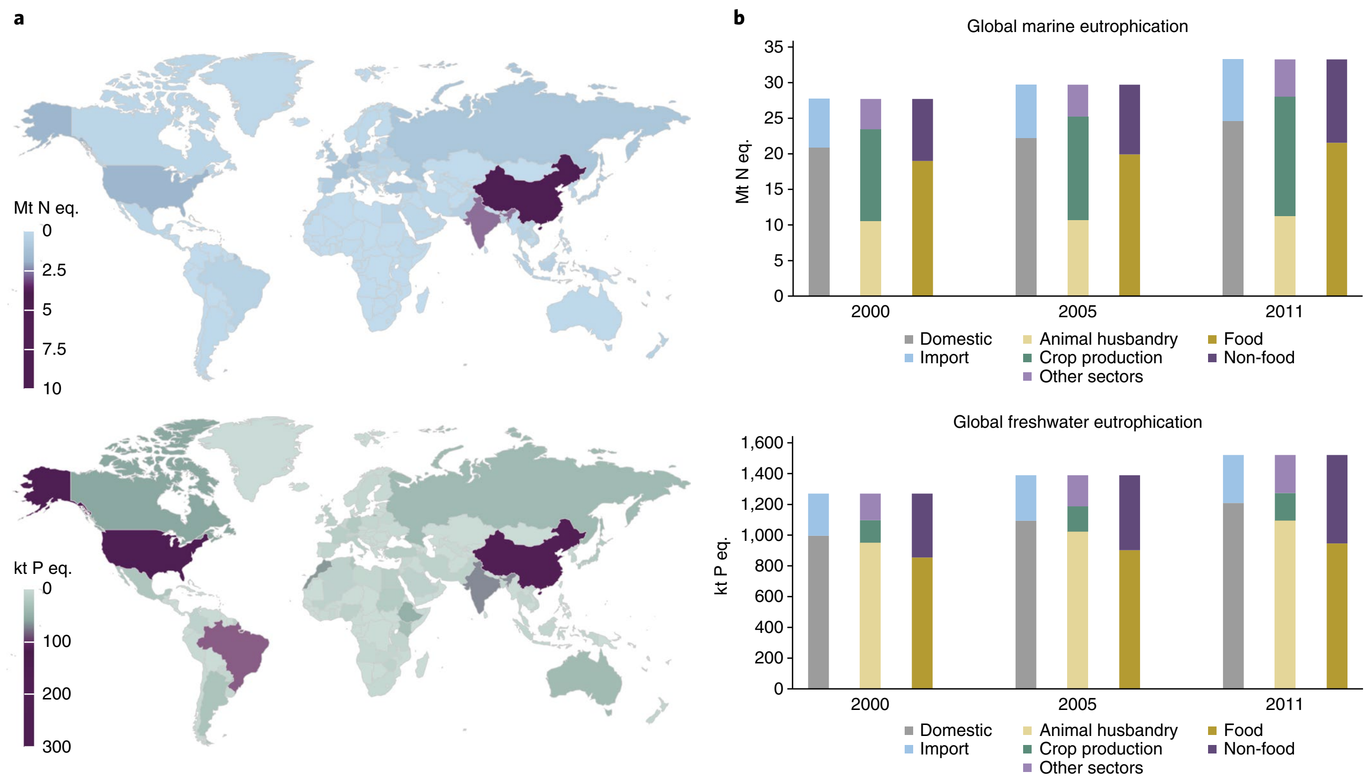

Fig. 1 | Global ME and FE footprints. a,**b**, Global ME and FE footprints for 2011 by country (**a**) and for the years 2000, 2005 and 2011 (**b**). Footprints are broken down based on whether they occurred domestically or from the consumption of imported products, by producing sector (crop production,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

28% increase from 2000 values, with 8.7 Mt N eq. and 420 ktP eq. for ME and FE, respectively. By comparison, the global food footprints only modestly increased from 2000 to 2011 values (ME: 19 Mt N eq. to 21 Mt N eq.; FE: 850 ktP eq. to 950 kt P eq.). In line with previous findings[12,18], we find agriculture to be the dominant production-side driver, accounting for 84% of the total footprints for both ME and FE; however, from a consumption standpoint in 2011, approximately one-quarter of these agricultural impacts were due to non-food consumption (see Supplementary Data for a list of non-food products).

To understand the regional drivers of food and non-food eutrophication impacts, we explore country-level food and non-food footprints. We find that the highest non-food ME footprint (related to the consumption of both imported and domestically produced non-food commodities) occurred in China, with 3 Mt N eq. out of a total footprint of 8.6 Mt N eq. (see Supplementary Data and Fig. 1a). This was double China's 2000 non-food ME footprint of 1.5 Mt N eq. A similar trend was also seen with China's food ME footprint, which increased by over 25% from 2000, peaking at 5.4 Mt N eq. for ME in 2011 and representing the highest country-level food footprint. For FE associated with non-food consumption, the US was the largest country-level contributor (263 kt P eq.) in 2011, which is nearly triple their 2000 value 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 value (33 kt P eq.). The highest country-level FE food footprints were observed in the US with 149 kt P eq. in 2000 and 140 kt P eq. in 2011. See Supplementary Data for a full breakdown of ME and FE footprints by sector and region.

In terms of the regional drivers for total eutrophication impacts (Fig. 1a), we find that Asia and the Pacific (in particular China and India), the US and Germany accounted for 54% of the global ME footprint (33 Mt N eq.). Africa, the US, China, Brazil and India accounted for 62% of the global FE footprint (1,500 kt P eq.). A Monte-Carlo-based sensitivity analysis confirms the robustness of the above and all following results (see Supplementary Information 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 s.d. = 6.8%) for FE footprints and 12.4% (1 s.d. = 4%) for the ME footprints. These decrease to 5.7% and 4.2% for FE and ME, respectively, in a less conservative scenario.

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 impact) and 313 kt P eq. (21% of global total) in 2011 for ME and FE, respectively (Supplementary Data and Fig. 1b). These global trade shares are comparable to previous studies. For example, one study[12] found, in 2010, 26% reactive nitrogen embodied in global trade, and another[11] found, in 2007, 41% of P and N emissions (termed grey water footprint) embodied in the 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 the Pacific account for 48% of all traded ME impacts.

Out of global traded impacts in 2011, 49% was due to non-food consumption for both ME and FE. This is a substantial increase from 2000 values, highlighting the increasing importance of non-food trade for both ME and FE (Fig. 2b). The ME impacts embodied in non-food trade even surpassed food trade in 2007, primarily driven

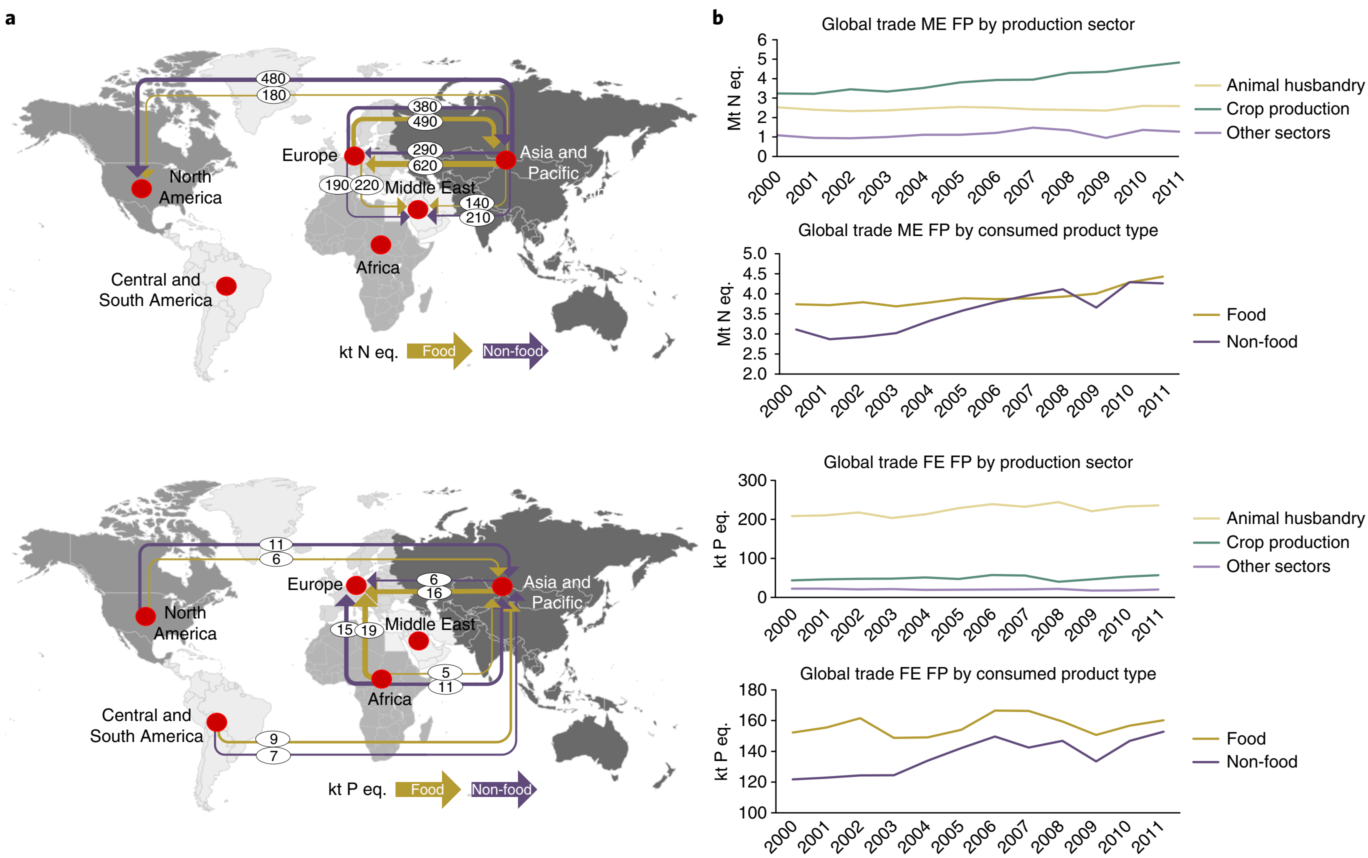

Fig. 2 | Eutrophication impacts embodied in trade. a, Top five continent-level displacements of marine (top, kt N eq.) and freshwater (bottom, kt P eq.) eutrophication associated with trade satisfying food and non-food demand. **b**, Global trade marine (top, Mt N eq.) and freshwater (bottom, kt P eq.) footprints (FPs) over time based on production sector and consumed product type. Arrows in **a** represent the gross flow of embodied impacts that occur in the country of origin (start of arrow) for the consuming country (point of arrow). Grey shading differentiates regions.

by the US consumption of Chinese-produced clothing, leather, fur and furniture, and Asian (excluding China) consumption of Chinese-produced textiles and clothing (see Supplementary 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 (Fig. 2b). The downturn was largely because of the respective 21 and 32% decreases in impacts embodied in non-food exports (primarily clothing, leather and furniture) from China and other Asian countries to the US. Despite the decline in US consumption over this period, growth was seen in a few regions—for example, the increased consumption of eastern European non-food products by the Middle East and Africa (mostly chemicals and construction materials). Nonetheless, after 2009, the US and other economies recovered and the importance of non-food trade appears to have been increasing ever since.

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 (Fig. 2a, top). 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; Fig. 2a, bottom). 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 the EU's high exposure to trade and their strong role in driving trade-related eutrophication impacts.

Product-level drivers

Underlying the growth in the overall footprints were the substantial increases in specific product groups (see Supplementary Data on product footprints for detailed results). For food impacts, the growth was primarily isolated to processed foods, which drove the modest increase in total food impacts from 2000 to 2011. In 2011, processed food (see Supplementary Data for aggregation key) accounted for 19% and 10.3% of total ME and FE impacts. However, this was a 35% and 20% growth for ME and FE, respectively, from 2000 values. Specifically, substantial growth was seen in the category of mixed processed food (product footprint code i15.i in the Supplementary Data), which accounted for 12.8% and 5.5% of total ME and FE impacts in 2011, a growth of 39% and 17% from 2000 values.

In terms of identifying specific non-food product drivers, one challenge is that many of the impacts arrive to consumers through services. In 2011, one-third of the non-food impacts occurred via public services, which accounted for 9 and 12% of global ME and FE footprints, respectively. These services, which include defence and education, often have low environmental intensities but represent such a large amount of monetary and economic activity that purchases of implicated goods in these sectors drives substantial impacts[14]. Public administration

Table 1 | Income elasticities of footprints based on 49 countries and RoW regions by consumption category (2000–2011)

		Marine eutrophication (ME)					Freshwater eutrophication (FE)				
		Income elasticity (95% CI)		Sig.	R^2	% share $_{2011}$ ($\Delta\%_{2000\text{-}11}$)	Income elasticity (95% CI)		Sig.	R^2	% share $_{2011}$ ($\Delta\%_{2000\text{-}11}$)
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	19.6	0.59	(0.16, 1.01)	***	0.14	5.0
	Animal-based food	0.88	(0.59, 1.17)	***	0.36	26.0	0.69	(0.24, 1.14)	***	0.31	46.9
	Processed food	1.21	(0.94, 1.48)	***	0.58	19.1	1.07	(0.75, 1.40)	***	0.62	10.3
	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	2.2	0.97	(0.58, 1.35)	***	0.51	2.6
	Shelter	0.90	(0.61, 1.20)	***	0.40	7.2	0.96	(0.49, 1.43)	***	0.48	5.2
	Manufactured products	1.01	(0.81, 1.22)	***	0.71	7.6	0.87	(0.54, 1.20)	***	0.63	4.7
	Waste services	0.72	(0.13, 1.32)	**	0.11	4.7	0.20	(−0.36, 0.75)		0.01	9.0
	Other services	1.16	(0.82, 1.50)	***	0.63	13.6	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	(−3.2)	0.03	(−0.51, 0.57)		0.90	(−1.0)
	Animal-based food	−0.02	(−0.28, 0.24)		0.98	(−2.7)	0.03	(−0.20, 0.27)		0.97	(−4.0)
	Processed food	0.43	(0.10, 0.76)	**	0.98	(+2.1)	0.54	(0.25, 0.84)	***	0.97	(0)
	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	(+0.1)	0.94	(0.54, 1.35)	***	0.94	(+0.1)
	Shelter	0.83	(0.50, 1.16)	***	0.97	(+1.0)	1.03	(0.62, 1.44)	***	0.96	(+0.8)
	Manufactured products	1.25	(0.99, 1.51)	***	0.96	(+1.4)	0.94	(0.66, 1.22)	***	0.95	(+0.4)
	Waste services	1.18	(−1.46, 3.82)		0.84	(+1.7)	1.98	(−0.90, 4.86)		0.82	(+2.4)
	Other services	0.41	(0.03, 0.79)	**	0.97	(−0.4)	0.75	(0.28, 1.21)	***	0.97	(+1.3)

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^{-1} for ME and g P eq. cap^{-1} for FE). The independent variable is the log-transformed GDP per capita in PPP (constant 2011 USD) . Significance level (Sig.): $*P<0.1$; $**P<0.05$; $***P<0.01$. CI, confidence interval. We further explore the practical significance of consumption categories, with $\%_{2011}$ showing the relative importance of the category for ME and FE in 2011 (summing to 100%), and $\Delta\%_{2000\text{-}11}$ the percentage change in relative importance between 2000 and 2011 (summing to 0).

and defence were responsible for roughly half of public service impacts (i75; growing 17% and 24% from 2000 to 2011 for ME and FE, respectively), with education (i80; growing 5% and 37% from 2000 to 2011 for ME and FE) and health responsible for the remainder (i85; growing 39% and 46% from 2000 to 2011 for ME and FE). Waste services (all i90 codes) embodied 4.6% and 9% of total ME and FE respectively in 2011, mainly due to landfilling (ME) and wastewater (FE). The impacts from these sectors have undergone substantial increases, with 92% and 65% growth from 2000 values for ME and FE, respectively. For construction (i45), the contribution was 3.1% for both ME and FE in 2011, respectively. This was a 62% and 61% growth for ME and FE from 2000 values, as a result of the large and increasing material flows through this sector[19]. Roughly 1% of total ME impacts in 2011 were due to the electricity and heat sector (i40), mostly caused by high NO_x emissions. Substantial contributions are also seen in the consumption of (1) manufactured goods (i29–i36), with a 5.2% and 3.1% share of total ME and FE impacts, respectively, in 2011; (2) chemicals (i24.d), with a 1.5% and 0.9% share for ME and FE; and (3) clothing/textiles (i17–i19), with 2.1% and 2.6% for ME and FE. While other individual non-food products/industrial sectors represented insignificant contributions to global ME and FE impacts on their own, aggregating these minor contributions resulted in significant impacts at the consumption category level (see Table 1 for a test of relationships between income and impact on the consumption category level).

Income eutrophication relationships

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 performed cross-sectional (2011) and panel data regression analysis (2000–2011) at 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 find 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. Previous studies have noted dietary shifts to processed food associated with increased affluence[20,21]. See Supplementary Information 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 studies[6,7,10]; 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[22]. Domestic policies include improving agricultural practices through reduced fertilizer use and improved animal husbandry feeding practices. These can be enforced or incentivized through, for example, country/region-level regulatory standards (for example, the EU nitrate directive mandates the designation of sensitive farming areas[23]) and fiscal and economic incentives (for example, subsidies[24] and polluters pay tax)[22].

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 (1) drives the largest global non-food eutrophication displacements to Asia-Pacific and Africa for ME and FE, respectively, and (2) displaces a high percentage of non-food impacts. While the EU has developed frameworks and strategies for tackling eutrophication within Europe[23,25], policies that integrate international supply chains for addressing eutrophication abroad are lacking[12,26]. 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 yuan[27] and, in addition, frequently disrupt the natural drinking supply of coastal Chinese cities[27]. This analysis has shown that over 13% of these impacts are the result of producing products for export.

Consumption-based approaches should increase stakeholder engagement as well as the pressure for implementing policy through both demand- and production-side measures. Demand-side measures can include trade agreements, pricing mechanisms or green procurement. However, demand-side measures on specific product groups, such as clothing, manufactured products and even construction are unlikely to be acceptable or effective, but their aggregate effects should be considered in the quantification of our overall impact on the environment. Setting consumption-based targets (such as a 40% reduction in the EU's global eutrophication footprint) can motivate transfer of technology/skills to countries, for example, to improve fertilizer efficiencies or manage waste (a production-side mitigation). 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. Modelling 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 (for example, Eora[28]); however, these models lack the estimation of both N and P emissions as well as the product specificity necessary to characterize non-food commodities separately 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 demand[29]. 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[30]. 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 around ±10–20% at the national level[31]. 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, for example in 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[32]. 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 the Supplementary Methods and Supplementary Discussion 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[33,34]. In freshwater systems, studies suggest that nitrogen as well as iron could be co-limiting over shorter timescales[35]. Therefore, we recommend further research to focus on deriving eutrophication potentials for all nutrients in both marine and freshwater ecosystems.

Furthermore, previous work[36] 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, while recognizing that the relationships with ecological responses are more complex. The variability in FE used is mainly caused by variability in hydrological residence time between river basins[22,37]. While FE potentials were country specific, ME potentials were only available per continent. Results of previous work[38] show that eutrophication damage indicators in marine ecosystems can vary by up to four 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 the Supplementary Discussion for further details).

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', which spatially differentiates the level of impact in (1) N equivalents for marine eutrophication footprints and (2) P equivalents for freshwater eutrophication footprints[17]. 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. A detailed description of the methods and key assumptions made in this analysis are provided below.

Environmentally extended MRIO analysis. MRIO analysis is a widely used tool for calculating environmental footprints for various environmental pressures[39], such as carbon footprints[40], land footprints, water footprints[41] and labour footprints[42]. MRIO analyses inter-industry flows between economic sectors both domestically and abroad. This allows the distinction 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 in previous work[43]. We use a standard Leontief demand pull technique to allocate environmental pressure to final consumption category, using a full MRIO approach[44,45].

Here, we use the EXIOBASE (v3.4) MRIO model[46] 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 fibre 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 database[46], 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 non-food). See Supplementary Information for the sector aggregation key. Other MRIO models exist such as GTAP[47] and Eora[28]. 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%[31,48–51]. A further discussion of these issues is provided in the Supplementary Information.

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'[44,52]. 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 Information, 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 in the Supplementary Information. Here, we provide a summary of the 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 (crops, fodder and pasture). Production levels of individual crops, fodder crops and pasture are obtained at the national level from the FAOSTAT database[53]. The total domestic demand of nutrients is distributed to crops, fodder crops and pasture using distribution factors primarily obtained from FAO[54]. 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 United Nations Framework Convention on Climate Change (UNFCCC) inventories and emission factors applied to fuel combustion (for data specification see Supplementary Methods). 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 gives the amount of absorbed N. The IPCC's procedure[55] is then applied to determine the direct and indirect (leaching) water emissions of N_2O and NO_3 and the air emissions of NO_x, NH_3 and NO_3. We assumed that the remainder are N_2 emissions. If the calculated N_2 turned out negative, the protein content of the crop was adjusted to ensure a consistent N-balance.

With regard to agricultural P emissions, the quantity absorbed by crops is estimated using concentration data. The residual P (applied P minus the P absorbed by crops quantity) is assumed to accumulate in soil stocks, where P emissions to freshwater are calculated as 2.9% of this accumulated soil P. 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 emissions[56], which is an overestimate compared to previous work[57].

The modelling of emissions from land application of manure management (in the stable and storage) is based on the IPCC, chapter 11[55]. The input parameters for this, that is, the amount of manure, is calculated based on metabolic mass balances of animals for all animal categories and countries included in EXIOBASE. 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 Supplementary Methods.

Impact assessment methods. We characterize the N and P emissions following ReCiPe 2016v1.1[17], 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, that is, residence time in lakes and rivers determined by inflow, advection, retention and water use processes[37,58,59]. 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 a previous study[60] with the soil fate from other previous work[59]. Continent aggregated potentials for marine eutrophication were determined based on emission data[17]. 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[3]. 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 supplementary 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^{-1}), FE (in g P eq. cap^{-1}) and income (GDP cap^{-1} in PPP (constant 2011 USD)). This is done in order to simplify the analysis and isolate the income effect from population changes. Income coefficients are reported separately for ME and FE, and across consumption categories (total, food versus non-food, and disaggregated further by eight consumption categories). Previous studies have calculated and used income/expenditure elasticities in the context of other environmental indicators, such as carbon footprints[40] and land and water use[20].

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 approach[15]. 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 test[15] for the choice of panel data method (see Supplementary Methods). Further information on the estimated models, descriptive statistics and robustness checks (pooled OLS and random-effects models) is provided in the Supplementary Information.

Code availability. 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.

Data availability. This work uses the EXIOBASE dataset, which is a secondary data source released as a freely available dataset through 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.

Received: 27 November 2017; Accepted: 15 May 2018;
Published online: 14 June 2018

References

1. Tilman, D., Cassman, K. G., Matson, P. A., Naylor, R. L. & Polasky, S. Agricultural sustainability and intensive production practices. *Nature* **418**, 671–677 (2002).
2. Smith, V. H., Tilman, G. D. & Nekola, J. C. Eutrophication: impacts of excess nutrient inputs on freshwater, marine, and terrestrial ecosystems. *Environ. Pollut.* **100**, 179–196 (1999).
3. Potter, P., Ramankutty, N., Bennett, E. M. & Donner, S. D. Characterizing the spatial patterns of global fertilizer application and manure production. *Earth Interact*. https://doi.org/10.1175/2009EI288.1 (2010).
4. Diaz, R. J. & Rosenberg, R. Spreading dead zones and consequences for marine ecosystems. *Science* **321**, 926–929 (2008).
5. Steffen, W. et al. Planetary boundaries: guiding human development on a changing planet. *Science* **347**, 1259855 (2015).
6. Leip, A. et al. Impacts of European livestock production: nitrogen, sulphur, phosphorus and greenhouse gas emissions, land-use, water eutrophication and biodiversity. *Environ. Res. Lett.* **10**, 115004 (2015).
7. Lassaletta, L. et al. Food and feed trade as a driver in the global nitrogen cycle: 50-year trends. *Biogeochemistry* **118**, 225–241 (2014).
8. Schipanski, M. E. & Bennett, E. M. The influence of agricultural trade and livestock production on the global phosphorus cycle. *Ecosystems* **15**, 256–268 (2012).
9. Schmitz, C. et al. Trading more food: implications for land use, greenhouse gas emissions, and the food system. *Global Environ. Change* **22**, 189–209 (2012).
10. Xue, X. & Landis, A. E. Eutrophication potential of food consumption patterns. *Environ. Sci. Technol.* **44**, 6450–6456 (2010).
11. Mekonnen, M. M., Lutter, S. & Martinez, A. Anthropogenic nitrogen and phosphorus emissions and related grey water footprints caused by EU-27's crop production and consumption. *Water* **8**, 1–14 (2016).
12. Oita, A. et al. Substantial nitrogen pollution embedded in international trade. *Nat. Geosci.* **9**, 111–115 (2016).
13. MacDonald, G. K. et al. Rethinking agricultural trade relationships in an era of globalization. *Bioscience* **65**, 275–289 (2015).
14. Hertwich, E. G. The life cycle environmental impacts of consumption. *Econ. Syst. Res.* **23**, 27–47 (2011).
15. He, Q. et al. Economic development and coastal ecosystem change in China. *Sci. Rep.* **4**, 1–9 (2014).
16. Wood, R. et al. Growth in environmental footprints and environmental impacts embodies in trade: resource efficiency indicators from EXIOBASE3. *J. Indust. Ecol.* https://doi.org/10.1111/jiec.12735 (2018).
17. Huijbregts, M. A. J. et al. ReCiPe2016: a harmonized life cycle impact assessment method at midpoint and endpoint level. *Int. J. Life Cycle Assess.* **22**, 138–147 (2017).
18. Mekonnen, M. M. & Hoekstra, A. Y. Global gray water footprint and water pollution levels related to anthropogenic nitrogen loads to fresh water. *Environ. Sci. Technol.* **49**, 12860–12868 (2015).
19. Giljum, S. et al. Identifying priority areas for European resource policies: a MRIO-based material footprint assessment. *J. Econ. Struct.* **5**, 17 (2016).
20. Ivanova, D. et al. Environmental impact assessment of household consumption. *J. Ind. Ecol.* **20**, 526–536 (2016).
21. Steen-Olsen, K., Wood, R. & Hertwich, E. G. The carbon footprint of Norwegian household consumption 1999-2012. *J. Ind. Ecol.* **20**, 582–592 (2016).
22. Selman, M. & Greenhalgh, S. *Eutrophication: Policies, Actions, and Strategies to Address Nutrient Pollution* (World Resources Institute, 2009)..
23. European Commission *Water Frameworks Directive* (The EU Nitrates Directive 1–4, 2010); http://ec.europa.eu/environment/water/water-nitrates/index_en.html
24. Shortle, J. S. & Abler, D. G. *Environmental Policies for Agricultural Pollution Control* (Centre for Agriculture and Bioscience International, 2001).
25. European Commission DG Environment *Joining Forces for Europe's Shared Waters: Coordination in International River Basin Districts* (The EU Water Framework Directive, 2008); http://ec.europa.eu/environment/water/water-framework/index_en.html
26. Sutton, M. A., Howard, C. M., Bleeker, A. & Datta, A. The global nutrient challenge: from science to public engagement. *Environ. Dev.* **6**, 80–85 (2013).
27. Le, C. et al. Eutrophication of lake waters in China: cost, causes, and control. *Environ. Manag.* **45**, 662–668 (2010).
28. Lenzen, M., Moran, D., Kanemoto, K. & Geschke, A. Building Eora: a global multi-region input–output database at high country and sector resolution. *Econ. Syst. Res.* **25**, 20–49 (2013).
29. Weinzettel, J., Steen-Olsen, K., Hertwich, E. G., Borucke, M. & Galli, A. Ecological footprint of nations: comparison of process analysis, and standard and hybrid multiregional input–output analysis. *Ecol. Econ.* **101**, 115–126 (2014).
30. Weinzettel, J. & Wood, R. Environmental footprints of agriculture embodied in international trade: sensitivity of harvested area footprint of Chinese exports. *Ecol. Econ.* **145**, 323–330 (2018).
31. Moran, D. & Wood, R. Convergence between the EORA, WIOD, EXIOBASE, and OPENEU'S consumption-based carbon accounts. *Econ. Syst. Res.* **26**, 1469–5758 (2014).
32. Tukker, A. Towards robust, authoritative assessments of environmental impacts embodied in trade: current state and recommendations. *J. Indust. Ecol.* https://doi.org/10.1111/jiec.12716 (2018).
33. Elser, J. J. et al. Global analysis of nitrogen and phosphorus limitation of primary producers in freshwater, marine and terrestrial ecosystems. *Ecol. Lett.* **10**, 1135–1142 (2007).
34. Howarth, R. W. & Marino, R. Nitrogen as the limiting nutrient for eutrophication in coastal marine ecosystems: evolving views over three decades. *Limnol. Oceanogr.* **51**, 364–376 (2006).
35. Sterner, R. W. On the phosphorus limitation paradigm for lakes. *Int. Rev. Hydrobiol.* **93**, 433–445 (2008).
36. Azevedo, L. B. et al. Assessing the importance of spatial variability versus model choices in life cycle impact assessment: the case of freshwater eutrophication in Europe. *Environ. Sci. Technol.* **47**, 13565–13570 (2013).
37. Helmes, R. J. K., Huijbregts, Ma. J., Henderson, A. D. & Jolliet, O. Spatially explicit fate factors of phosphorous emissions to freshwater at the global scale. *Int. J. Life Cycle Assess.* **17**, 646–654 (2012).
38. Cosme, N., Jones, M. C., Cheung, W. W. L. & Larsen, H. F. Spatial differentiation of marine eutrophication damage indicators based on species density. *Ecol. Indic.* **73**, 676–685 (2017).
39. Wood, R. et al. Global sustainability accounting-developing EXIOBASE for multi-regional footprint analysis. *Sustain* **7**, 138–163 (2015).
40. Hertwich, E. & Peters, G. Carbon footprint of nations: a global, trade-linked analysis. *Environ. Sci. Technol.* **43**, 6414–6420 (2009).
41. Zhang, C. & Anadon, L. D. A multi-regional input-output analysis of domestic virtual water trade and provincial water footprint in China. *Ecol. Econ.* **100**, 159–172 (2014).
42. Simas, M., Wood, R. & Hertwich, E. Labor embodied in trade. *J. Ind. Ecol.* **19**, 343–356 (2015).
43. Turner, K., Lenzen, M., Wiedmann, T. & Barrett, J. Examining the global environmental impact of regional consumption activities — part 1: a technical note on combining input-output and ecological footprint analysis. *Ecol. Econ.* **62**, 37–44 (2007).
44. Miller, R. A. & Blair, P. D. *Input-Output Analysis Foundations and Extensions* (Cambridge Univ. Press, Cambridge, 2009).
45. Murray, J. & Wood, R. (eds) *The Sustainability Practitioner's Guide to Input-Output Analysis* (Common Ground Research Networks, Champaign, IL, 2010).
46. Stadler, K. et al. EXIOBASE 3 Developing a time series of detailed Environmentally Extended Multi-Regional Input-Output tables. *J. Ind. Ecol.* https://doi.org/10.1111/jiec.12715 (2018).
47. Aguiar, A., Narayanan, B., & McDougall, R. An overview of the GTAP 9 data base. *J. Glob. Econ.* **1**, 181–208 (2016).
48. Owen, A., Steen-Olsen, K., Barrett, J., Wiedmann, T. & Lenzen, M. A structural decomposition approach to comparing MRIO databases. *Econ. Syst. Res.* **26**, 262–283 (2014).
49. Stadler, K., Steen-olsen, K. & Wood, R. The 'rest of the world'—estimating the economic structure of missing regions in global multi-regional input-output tables. *Econ. Syst. Res.* **26**, 303–326 (2014).
50. Steen-Olsen, K., Owen, A., Hertwich, E. G. & Lenzen, M. Effects of sector aggregation on CO_2 multipliers in multiregional input-output analysis. *Econ. Syst. Res.* 284–302 (2014).
51. Bouwmeester, M. & Oosterhaven, J. Specification and aggregation errors in environmentally extended input–output models. *Environ. Resour. Econ.* **56**, 307–335 (2013).

52. Dietzenbacher, E. & Lahr, M. L. Expanding extractions. *Econ. Syst. Res.* **25**, 341–360 (2013).
53. *FAOSTAT* (FAO, accessed 25 February 2016); http://faostat.fao.org/site/567/DesktopDefault.aspx#ancor
54. *Fertilizer Use by Crop* Fertilizer and Plant Nutrition Bulletin 17 (FAO, 2006).
55. De Klein, C. et al. in 2006 *IPCC Guidelines for National Greenhouse Gas Inventories* Vol. 4 (eds Eggleston, H. S., Buendia, L., Miwa, K., Ngara, T. & Tanabe, K.) Ch. 11 (IGES, 2006).
56. Bouwman, A. F., Beusen, A. H. W. & Billen, G. Human alteration of the global nitrogen and phosphorus soil balances for the period 1970-2050. *Global Biogeochem. Cycles* **23**, (2009).
57. Bennett, E. M., Carpenter, S. R. & Caraco, N. F. Human impact on erodable phosphorus and eutrophication: a global perspective. *Bioscience* **51**, 227 (2001).
58. Cosme, N., Koski, M. & Hauschild, M. Z. Exposure factors for marine eutrophication impacts assessment based on a mechanistic biological model. *Ecol. Modell.* **317**, 50–63 (2015).
59. Cosme, N., Mayorga, E. & Hauschild, M. Z. Spatially explicit fate factors of waterborne nitrogen emissions at the global scale. *Int. J. Life Cycle Assess.* https://doi.org/10.1007/s11367-017-1349-0 (2017).
60. Roy, P. O., Huijbregts, M., Deschênes, L. & Margni, M. Spatially-differentiated atmospheric source-receptor relationships for nitrogen oxides, sulfur oxides and ammonia emissions at the global scale for life cycle impact assessment. *Atmos. Environ.* **62**, 74–81 (2012).

Acknowledgements
We would like to thank K. Bjørset (Norkart), K. Steen-Olsen (Norwegian University of Science and Technology (NTNU)) and M. Simas (NTNU) for their technical support, M. Huijbrets (Radboud University) for his valuable comments and feedback, and G. Majeau-Bettez, C. Bulle (CIRAIG) and F. Verones (NTNU) for help with characterization factors. We would also like to thank R. Lonka (NTNU) for his assistance with the visualization tools.

Author contributions
H.A.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.v.Z. prepared the impact assessment method. H.A.H. and D.I. conducted the analysis. D.M. conducted the sensitivity analysis. H.A.H. made the figures. H.A.H., R.W., K.S. and D.I. contributed to the data interpretation. H.A.H., D.I. and R.W. wrote the paper. H.A.H., R.W., D.I., R.v.Z., K.S., S.M., D.M. and J.S. contributed to manuscript editing.

Competing interests
The authors declare no competing interests.

Additional information
Supplementary information is available for this paper at https://doi.org/10.1038/s41893-018-0079-z.

Reprints and permissions information is available at www.nature.com/reprints.

Correspondence and requests for materials should be addressed to R.W.

Publisher's note: Springer Nature remains neutral with regard to jurisdictional claims in published maps and institutional affiliations.