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Environmental Research Letters

LETTER

Local and social facets of planetary boundaries: right to nutrients

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Abstract

Anthropogenic nutrient flows exceed the planetary boundaries. The boundaries and the current excesses vary spatially. Such variations have both an ecological and a social facet. We explored the spatial variation using a bottom-up approach. The local critical boundaries were determined through the current or accumulated flow of the preceding five years before the planetary boundary criteria were met. Finland and Ethiopia served as cases with contrasting ecology and wealth. The variation in excess depends on historical global inequities in the access to nutrients. Globally, the accumulated use per capita is 2300 kg reactive nitrogen (N_r) and 200 kg phosphorus (P). For Finland, the accumulated use per capita is 3400 kg N_r and 690 kg P, whereas for Ethiopia, it is 26 kg N_r and 12 kg P. The critical N boundary in Finland is currently exceeded by 40 kg $cap^{-1} a^{-1}$ and the accumulated excess is 65 kg $cap^{-1} a^{-1}$, while the global current excess is 24 kg $cap^{-1} a^{-1}$ and there is space in Ethiopia to increase even the accumulated flow. The critical P boundary is exceeded in Finland and (although less so) in Ethiopia, but for contrary reasons: (1) the excessive past inflow to the agrifood system in Finland and (2) the excessive outflow from the agrifood system triggered by deficits in inflow and waste management in Ethiopia. The critical boundaries set by Finnish marine systems are lower and those set by freshwaters are higher than the planetary boundaries downscaled per capita. The shift to dominance of internal loading in watercourses represents a tipping point. We conclude that food security within the safe boundaries requires global redistribution of nutrients in residues, soils and sediments and of rights to use nutrients. Bottom-up assessments reveal local dynamics that shed new light on the relevant boundary criteria and on estimates and remedies.

1. Introduction

Research suggests that anthropogenic nutrient flows have transgressed the upper tolerable limits of the Earth system (Rockström *et al* 2009, Carpenter and Bennett 2011, Steffen *et al* 2015a). Beyond such 'planetary boundaries' unpredictable changes that are difficult to reverse are probable. The safe boundaries vary according to the characteristics of local ecosystems. The excess in nutrient flows varies according to local critical boundaries and, in addition, according to local socio-economic features that affect nutrient use. Determining such spatial variations with their ecological and social facets can advance the understanding of the dynamics underpinning the planetary boundaries

and their transgression, as well as of equity issues (Steffen *et al* 2015b) when allocating the necessary reduction in the flows and tailoring the remedies to return to the safe space.

Planetary boundaries can be transgressed by bottom-up forcing through cumulative change or by emergent top-down forcing through a systemic change (Turner *et al* 1990, Barnosky *et al* 2012). Regarding nutrients, both of these two modes of forcing occur. Nitrogen-containing (N) greenhouse gas emissions and N contribution to acidification represent global forcing, while N and phosphorus (P) flows to coastal and freshwater systems constitute local forcing. In addition, the human dimension of transgressing the boundaries through the excess use of nutrients varies around the

globe, mainly according to affordability, illustrating the spatial inequity. Irrespective of the role of global forcing, planetary boundaries are transgressed by local actions and the gaps (i.e., the excess use of nutrients), must be bridged locally. In light of the interregional linkages of spatial equity, global actions may also be required.

Humanity's food supply depends on N and P, the two elements most limiting for agriculture. In addition, 74% of the conversion of atmospheric N_2 to reactive N (N_r) and 80% of P flows to water systems serve agrifood systems (Kahiluoto *et al* 2014a). The global spatial variation in crop yield is primarily nutrient and water-limited (Mueller *et al* 2012). The variation in nutrient use relative to crop uptake is marked: 32% of the N surplus and 40% of the P surplus occur on 10% of the global field area (Foley *et al* 2011). The historical wealth-related variation in nutrient use is reflected in the current local reserves in field soils, water systems and residues. These reserves essentially influence current crop yields and the requirement for fertilizer application to maintain agricultural productivity. The turbulence in global food prices in recent years has aggravated food insecurity most heavily in the countries relying on food imports and especially among the rural poor (Compton *et al* 2010, FAO 2011, Headey 2011), emphasizing the importance of resource distribution for food production.

Prior assessments of planetary boundaries excluded the dimension of spatial and temporal social-ecological variation. In this study, we quantify the spatial variation of critical nutrient boundaries and the distance from the boundary, i.e., excess in nutrient flows or capacity to increase those flows, taking into account the historical nutrient accumulation. Then we assess the variation in the proportions of agrifood systems for the reactive nutrient flows. We downscale the planetary nutrient boundaries through even per capita distribution and compare them to the locally determined critical nutrient boundaries for Finland with a fragile ecology, and to the socio-ecologically contrasting Ethiopia. We posed the following questions:

1. How can planetary nutrient boundaries be made spatially explicit at the national scale?

2. How much do critical nutrient boundaries, excesses and proportions of agrifood systems vary in space?

3. Are the ecological or socio-economic and current or historical spatial variations more decisive to the excess?

2. Methods[4]

2.1. Steps in the assessment
The planetary nutrient boundaries proposed by Rockström *et al* (2009), the P boundary of freshwater

[4] References cited in this section available in supplementary material.

systems proposed by Carpenter and Bennett (2011), and the N boundary reassessed by de Vries *et al* (2013), revised by Steffen *et al* (2015a, b) and complemented by Kahiluoto *et al* (2014a) were applied in the socio-ecologically contrasting local cases of Finland and Ethiopia. First, the proposed planetary nutrient boundaries were *downscaled* through an equal distribution within the global population (table 1). Thereafter, the safe boundaries were *locally determined* for Finland for comparison, based on the characteristics of the local marine (N, P) and freshwater (P) ecosystems (see section 2.2). Regarding Ethiopia, the downscaled planetary boundaries were used due to lack of historical ecological data. To indicate the socio-economic spatial variation, the *current and accumulated* (1900–2010) *use* of N and P, and P flow to water systems, were estimated globally and for Finland and Ethiopia (see section 2.3). The excess was determined as the difference between the current or the accumulated flows and the critical boundary. Finally, the proportion of the agrifood systems for the local reactive nutrient flows was assessed. For the steps in the assessment, see also table 1.

2.2. Planetary nutrient boundaries: towards a bottom-up approach
Previously, the planetary N boundary including both the climatic and water system risks, was set at 20 Mt N_r a^{-1} (de Vries *et al* 2013, Steffen *et al* 2015a, b). For P in oceans, the proposed boundary criterion was to avoid an extensive anoxia in the near-bottom layers for the next 1000 years (Rockström *et al* 2009). For P in freshwaters, 24 mg P m^{-3} of water, which represents the meso-eutrophic interface, i.e., a typical target to limit the eutrophication of lakes and reservoirs, and for P flow from terrestrial ecosystems to freshwater systems, 1.2 Mt a^{-1} (Carpenter and Bennett 2011) was proposed. In this study, the proposed planetary boundaries were downscaled to the local case through an equal distribution within the global population of 7.0 billion in 2010 (UN 2011).

When attempting a bottom-up approach by locally determining a critical N boundary, the local conversion of the atmospheric N_2 to N_r in Finland was estimated. This attempt is in line with the statement of Steffen *et al* (2015a): 'The planetary boundary (PB) framework is not designed to be 'downscaled' or 'disaggregated' to smaller levels, such as nations or local communities.' In the assessment of the current N_r conversion, synthetic fertilizer use in agriculture and forestry (153 kt a^{-1}) (Grönroos and Nikander 2002, MMM 2008, Metla 2012, Tike 2013), other use of Haber-Bosch N (36 kt a^{-1}) (Galloway *et al* 2004, 2003, 2008), cultivation-induced biological N_2 fixation (C-BNF, 9 kt a^{-1}) (Antikainen *et al* 2005, Sipiläinen *et al* 2012) and fossil energy use (42 kt a^{-1}) (Statistics Finland, 2013, 2014a,b; Ministry of Employment and the

Table 1. Spatial variation in nutrient flows, critical nutrient boundaries and excess globally and locally (Finland and Ethiopia). Nutrients: nitrogen (N) (N_2 conversion to reactive N (N_r)) and phosphorus (P) flow to water systems.

	Flow		Boundary		Excess	
	kg cap^{-1}	kg cap^{-1} a^{-1}	kg cap^{-1}	kg cap^{-1} a^{-1}	kg cap^{-1}	kg cap^{-1} a^{-1}
Global						
N_r						
Downscaled[a]		27		2.9		24
P						
Downscaled$_{Oceans}$[a]		1.5		1.6		-0.1
Downscaled$_{Freshwaters}$[b]		1.3–4.6		0.2		1.1–4.4
Finland						
N_r						
Current$_{Conversion}$[c]		43		2.9		40
Current$_{Baltic Sea}$[d]		43		8.4		34
Current$_{Lakes}$[e]				Not limiting		
Accumulated$_{Conversion}$ 1900–2010[f]	3,400	68[g]	Not known	2.9[g]	> 3400	65[g]
P						
Current$_{Baltic Sea}$[h]		0.7		0.2		0.5
Current$_{Lakes}$[i]		0.7		0.7		0.01
Accumulated$_{Baltic Sea}$ 1900–2010[j]	44	1.3[g]	0.2	0.004[g]	44	1.3[g]
Accumulated$_{Lakes}$ 1900–2010[k]	39	1.5[g]	4.9	0.1[g]	34	1.4[g]
Ethiopia						
N_r						
Current$_{Conversion}$[c]		1.7		2.9		-1.2
Accumulated$_{Conversion}$ 1900–2010[l]	26	2.8[g]	51	2.9[g]	−25	-0.1[g]
P						
Current$_{Freshwaters}$[m]		0.2		0.2		0.1
Accumulated$_{Freshwaters}$ 1900-2010[l]	5.5	0.6[g]	3.1	0.2[g]	2.4	0.4[g]

[a] The planetary boundaries for N (de Vries *et al* 2013, Steffen *et al* 2015a, b) and for P Rockström *et al* (2009) equally divided for the global population in 2010; for the current flow estimate see Kahiluoto *et al* (2014a).

[b] The planetary boundary estimate of Carpenter and Bennett (2011) equally divided for the global population in 2010; for the current flow estimate see Kahiluoto *et al* (2014a).

[c] The downscaled planetary boundary see [a]. The local current average flow 2006–2010.

[d] The local boundary based on the carrying capacity of the Baltic Sea and on the local average flow 1926–1930. The local current flow see [c].

[e] P limiting factor for eutrophication.

[f] The boundary for accumulated use: the global downscaled boundary; the local flow exceeded that already before 1900. The flow accumulated 1900–2010.

[g] Assuming that the excess will be depleted during as many years as it needed to evolve.

[h] The local boundary based on the carrying capacity of the Baltic Sea, equally divided for the population of the Baltic Sea drainage in 2010, and on the local average flow to water systems 1926–1930. The local current average flow to water systems 2006–2010.

[i] The local boundary based on the carrying capacity of local lakes and on the local average flow to water systems 1956–1960. The local current flow see [h].

[j] The boundary for accumulated use: the flow to water systems accumulated (since 1900) when exceeding the local boundary (in the end of 1930s). The flow accumulated 1940–2010.

[k] The boundary for accumulated use: the flow to water systems accumulated (since 1900) when exceeding the local boundary (1960). The flow accumulated 1961–2010.

[l] The boundary for accumulated use: the global downscaled boundary. The flow accumulated 1993–2010.

[m] The downscaled planetary boundary see [b]. The local current average flow to water systems 2006–2010.

Economy 2012) were taken into account (see Kahiluoto *et al* 2014a). The mean values for 2006–2010 were used for all data unless otherwise stated. Regarding the accumulated local N_r conversion for Finland, the global boundary downscaled to the Finnish population (16 kt a^{-1}) was adopted as our proposed critical N boundary. The N_r use accumulated after the accumulated boundary was transgressed estimated (N_r conversion for the global boundary downscaled, 1900–2010, 16 000 kt). When assessing the necessary annual reduction of the accumulated excess, it was

assumed that the excess would be reduced during as many years as it needed to evolve.

The ecology of the Baltic Sea was hypothesized to set an additional boundary to that by the N_r conversion because 'it has been debated whether N or P is the most limiting nutrient in the Baltic Sea' (HELCOM 2009). Reductions in both P and N are required, and the primary production for the spring period is N limited in the open basins, except for in the Bothnian Bay and Bothnian Sea. The decade of the rapid expansion of hypoxia in the Baltic Sea was identified (Savchuk
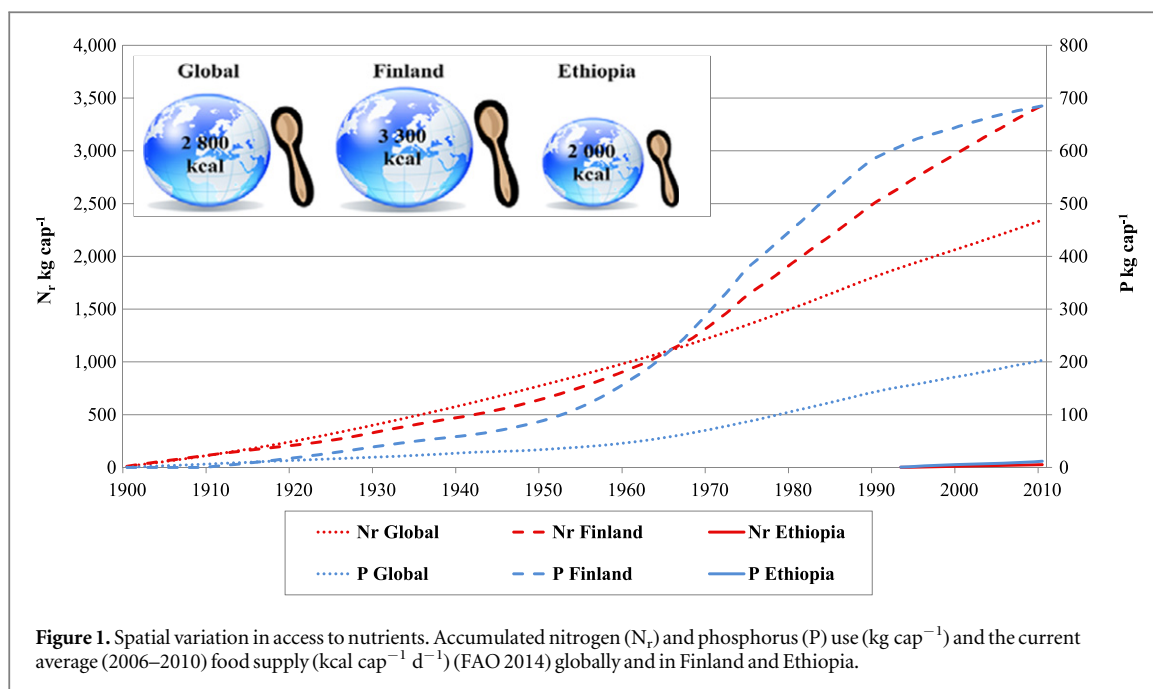
et al 2008; Carstensen *et al* 2014), indicating the transgression of the carrying capacity. The mean conversion of N_2 to N_r per capita in Finland for the preceding five-year period (1926–1930, 47 kt a^{-1}) was estimated as the critical N boundary set by the Baltic Sea. Additionally, the critical N boundary was assessed in analogy with the assessment of the critical P boundary, i.e., based on the mean N flow in the Baltic Sea catchment during that period (331 kt a^{-1}) (HELCOM 2011, Gustafsson *et al* 2012, Baltic Stern, 2013). The excess was estimated as the difference between our proposed critical boundary and the current flow in Finland (73 kt a^{-1}) (Biodiversity/SYKE, 2014).

Regarding a critical P boundary for the Baltic Sea, the same boundary criterion of extensive hypoxia as for N was used. P is the limiting factor for toxic summer blooms of cyanobacteria in the Baltic Sea both currently and, to an even greater extent, previously (Vahtera *et al* 2007, Savchuk *et al* 2008, Conley 2012). Cyanobacteria contribute 'tremendous quantities' of N to the ecosystem, making it especially important to reduce P loads. The P flows in the Baltic Sea catchment were estimated for the preceding five years' period of the rapid expansion of hypoxia (1926–1930, 18 kt a^{-1}) (HELCOM 2011, Gustafsson *et al* 2012, Baltic Stern, 2013) to define a critical P boundary for the Baltic Sea. The excess was estimated based on the current P flow in Finland (4,1 kt a^{-1}) (Biodiversity/SYKE, 2014). The expanding hypoxia has here been temporally related to P flows and not to soil P stocks or fertilizer inputs, as Steffen *et al* (2015a) suggest for local/regional scales because P flows are directly related to the risk of eutrophication and are comparable with N boundary and across spatial (e.g., globe v. Finland v. Ethiopia) and temporal (1960s v. current) scales; by contrast, the relative significance of the soil nutrient stocks and fertilizer input varies among those scales (see Carpenter and Bennett 2011, Steffen *et al* 2015a). The Finnish P flow for 1900–1954 was based on the agricultural flow interpolated relying on the fertilizer P use in 1900–1954 and on the ratio of P flow and fertilizer P use in 1956–1960. The fertilizer P use in 1900–1910 was assumed to be equal to that estimated for 1910 (Antikainen *et al* 2008). For the accumulated P flow, our proposed local critical boundary for the Baltic Sea was determined based on the accumulated annual P flow for 1900–1939 (0.02 kt a^{-1}), i.e., until the local P boundary was transgressed. The P flow accumulated after the accumulated boundary was transgressed was estimated (1940–2010, 250 kt)

Regarding the Finnish lakes, to determine a critical P boundary, the same boundary criterion of extensive hypoxia was used as for the Baltic Sea and N (Rockström *et al* 2009). Eutrophication of Finnish lakes has since the 1960s been considered largely P-limited. While the thresholds between the multistability stages of shallow lakes are multiple (Scheffer and van Nes 2007), the upper tolerable limit for P flows was determined based on the criterion of

expanding hypoxia (Rockström *et al* 2009) and/or abrupt eutrophication and then temporally related to the P flow into the freshwater system. The decade of rapid eutrophication and expansion of hypoxia was identified based on (1) biological changes in the sediments of observed lakes (Räsänen *et al* 1992, Meriläinen and Hamina 1993, Kauppila *et al* 2002, Kauppila and Valpola 2003, Kauppila 2005, Heikkinen and Väisänen 2007, Koivunen 2008, Kihlman and Kauppila 2010, Kauppila unpublished) and (2) the development of the concentration of total or bioavailable P (Mitikka and Ekholm 2003, Räike *et al* 2003, Ekholm and Mitikka 2006). The mean P flow for a preceding five year period (1956–1960, 4.0 kt a^{-1}), originating from scattered sources (agriculture 1.1 kt a^{-1}, rural dwellings 0.5 kt a^{-1}, forestry 0.03 kt a^{-1}) and point sources (industry 0.6 kt a^{-1}, municipalities 1.8 kt a^{-1}, and fish farming, fur farming and peat production 0.0 kt a^{-1}) was then estimated (Biodiversity/SYKE 2014). The P flow from agriculture in 1955–2010 was estimated relying on the agricultural soil P status in 1955–2000 (Mäntylahti 2002) and 2005–2009 (Uusitalo *et al* 2014) and on the estimates for the flow available for 1995–2010 (Biodiversity/SYKE 2014). The P load from field soils to the watercourses linearly depends on soil extractable P (Ekholm *et al* 1999), and more than 90% of the agricultural load originates in field soils (Rekolainen *et al* 1995). The estimates of P flow from population centres for 1971–2010 and from scattered dwellings for 1995–2010 (Biodiversity/SYKE 2014) were extrapolated for 1955–1970 and 1955–1994, respectively, based on the degree of urbanization (Statistics Finland 1997). The estimates for P flow from industry for 1972–2010 and from forestry for 1995–2010 (Biodiversity/SYKE 2014) were extrapolated for 1955–1971 and 1955–1994, respectively, based on the growth rate of the forest industry (Kahiluoto 1986). The P flow from other point sources was assumed zero before the first official estimation in 1976 (0.025 kt a^{-1}; Biodiversity/SYKE 2014). For the accumulated P flow, a local critical boundary was determined based on the accumulated annual P flow for 1900–1960 for the freshwaters (0.4 kt a^{-1}), i.e., until the local P boundary was transgressed. The P flow accumulated after the accumulated boundary was transgressed was estimated (1961–2010, 220 kt).

For Ethiopia, the planetary N boundary (de Vries *et al* 2013, Steffen *et al* 2015a, b) and the planetary P boundary for freshwaters (Carpenter and Bennett 2011) were translated to the per capita allocation by dividing the global boundary with the current world population (2.9 kg N_r, and 0.2 kg P cap^{-1} a^{-1}). The current N_r conversion in Ethiopia (133 kt a^{-1}) was estimated, including the synthetic fertilizer use (91 kt a^{-1}) (FAO 2013), other use of Haber-Bosch N (0.2 kt a^{-1}) (Galloway *et al* 2003, 2004, 2008), the C-BNF (37 kt a^{-1}) and the fossil energy use (4.7 kt a^{-1}). The C-BNF in Ethiopia was assessed

Figure 1. Spatial variation in access to nutrients. Accumulated nitrogen (N_r) and phosphorus (P) use (kg cap^{-1}) and the current average (2006–2010) food supply (kcal cap^{-1} d^{-1}) (FAO 2014) globally and in Finland and Ethiopia.

assuming the global ratio of C-BNF and the synthetic fertilizer use. The conversion of N_2 to N_r in fossil energy use in Ethiopia (World Bank 2014c) was assessed assuming the ratio of N_r creation per fossil energy use in Finland. The current P flow to water systems in Ethiopia (21 kt a^{-1}) was assessed based on the scarce available data on the P flow from agriculture (17 kt a^{-1}) (Haileslassie *et al* 2005, 2007) and from sewage of urban households (3.4 kt a^{-1}) (Nyenje *et al* 2010, World Bank 2014a). For Ethiopia, the minor accumulated N_r conversion (1800 kt) was estimated for 1993–2010. The accumulated P flow in Ethiopia (460 kt) was estimated considering the temporal (1993–2010) development of the arable land area (FAO 2015) and urban population (World Bank 2014a). Conclusions regarding the precise estimates warrant caution.

2.3. Spatial variation in nutrient use: the socio-economic facet

In terms of the Gross Domestic Product (GDP, purchasing power parity) in international dollars (Int $) per capita, Finland is ranked 24th (36 000 Int$) and Ethiopia 172nd (1 240 Int$) among the 182 countries reported (World Bank, 2014b). The accumulated N_r and P use for 1900–2010 globally and for Finland and for 1993–2010 for Ethiopia were estimated (including the current use, see section 2.2).

Data sources for these estimates included the conversion of N_2 to N_r in the Haber-Bosch process for synthetic N fertilizers, other N chemicals (Galloway *et al*, 2003, 2004, 2008), C-BNF (Vitousek *et al* 1997, Galloway *et al* 2003, 2008, Antikainen *et al* 2008, Sipiläinen *et al* 2012) and the fossil energy use (N emissions) (Statistics Finland 2013, 2014a, b, Galloway *et al*

2003, 2004, 2008, Antikainen 2007, Vitousek *et al* 1997, Ministry of Trade and Industry 1977, World Bank 2014c). Global synthetic fertilizer use was registered at 5–10 year intervals for the period 1900–1960 (Smil 2004, Smil 2000) and annually for the period 1961–2010 (FAO 2013). Synthetic fertilizer use in Finland in 1910–1950 (Antikainen *et al* 2008), 1961–1993 (FAO 2013) and 1994–2010 (FAO 2013, Tike 2013, Metla 2012, MMM 2008, Grönroos and Nikander 2002) was considered. The estimate of the fertilizer nutrient use in Ethiopia relied on the data that represent the period 1993–2010 (FAO 2013).

Regarding P use, the estimates of synthetic P fertilizer use were complemented by estimates on the use of synthetic P feed (Smil 2000; Poutiainen 2002, Linderholm *et al* 2012) and detergents (Ott and Rechberger 2012, Smil 2000).

The proportion of N_r conversion in the local agrifood system consisted of the synthetic fertilizer use, C-BNF and fossil energy use. The estimate of fossil energy use in the Finnish agrifood system was based on the GDP share of agriculture (2.8%) and food processing industry (7.1%) (Viitaharju *et al* 2014), in Ethiopia based on the GDP share of agriculture (47%) (World Bank 2015). The contribution of the Finnish agrifood system to the P flow to water systems was estimated relying on the flows from agriculture, fish farming, municipalities, scattered dwellings and industries apart from the pulp and paper industries (Biodiversity/SYKE, 2014). The portion of the agrifood system from P flows from municipalities, scattered dwellings and industries was estimated based on the determined (see above) proportion of synthetic P use in the agrifood system (89%).

3. Results

3.1. Planetary nutrient boundaries: towards a bottom-up approach

The ecology of the Baltic Sea did not set an N boundary based on the avoidance of a large-scale ocean hypoxic event additional to that based on the conversion of N_2 to N_r (table 1). If only the N flow to the Baltic Sea was considered (excluding conversion of atmospheric N_2), a flow of 13 kg, boundary of 3.7 kg and excess of 9.4 kg N $cap^{-1} a^{-1}$ was estimated. Consequently, the downscaled PB for N is also the proposed critical N boundary for Finland. Regarding P, applying the global boundary criterion of ten times the pre-industrial flow would allow a further increase in P flow to the sea (table 1). However, the P flow per capita tolerated by the local marine ecology is below the global average. Expanded hypoxia has been recorded since the 1930s in the Baltic Sea, and the current Finnish flow to the sea clearly exceeds our proposed critical boundary. For the local lake system, the upper tolerable P boundary was estimated to be transgressed in the mid-1960s. In contrast with the global state, the local P flow to lakes has currently returned to within the safe boundaries, but the accumulated flows exceed the corresponding safe boundary by 15 fold.

3.2. Spatial variation in nutrient use: the socio-economic facet

The higher excess in the local N_r flow in Finland compared with the global flow, relative to the safe boundary, is due to the difference in nutrient use, while the boundary per capita is equal (table 1). In 2010, the nutrient use in Finland was 41 kg N_r with 2.9 kg P, with the N_r use being one-third higher than but the P use slightly lower than the global average per capita. The nutrient use in Ethiopia in 2010 was 2.1 kg N_r and 1.0 kg P $capita^{-1}$. Fertilizer use represents 64% of the total N_r use and 74% of the total P use for Finland and 69% and 21% for Ethiopia, respectively. The excess cumulated across the last century reveals the entire global spatial disparity. The cumulated (1900–2010) N_r and P uses per capita are 2300 kg N_r and 200 kg P globally, 3400 kg N_r and 690 kg P for Finland, and 26 kg N_r and 12 kg P for Ethiopia (figure 1). While the per capita use of P in Finland has exceeded the global average since the beginning of the century, peaking in 1973, the local use of N_r has only exceeded the global average since the beginning of the 1950s, peaking in 1974. If the global historical nutrient equity per capita were induced, and the excess in the nutrient use were reduced at the same pace as it accumulated in Finland (1900–2010 for N_r and 1961–2010 for P), no further N_r conversion or virgin P flow to water systems would be possible globally or in Finland, contrary to Ethiopia (table 1). The Finnish agrifood system was responsible for 69% of the local N_r conversion and for 81% of the local P

flow to water systems, while the corresponding proportions for the Ethiopian agrifood system were 98% for N_r conversion and 83% for P flow to water systems.

4. Discussion

4.1. Planetary nutrient boundaries: towards a bottom-up approach

Nutrient flows act as drivers to other human-induced shifts in the Earth system, making it difficult to assess the critical thresholds. The 'first guess' on the quantitative planetary N boundary by Rockström *et al* (2009) has been revised by de Vries *et al* (2013), Kahiluoto *et al* (2014a) and Steffen *et al* (2015a). The global N_r use already had exceeded the boundary revised by Steffen *et al* 2015a (20 Mt a^{-1}) in 1909 and the Finnish use had already exceeded the downscaled planetary N_r boundary 16 kt a^{-1} before the beginning of the 20th century (31 kt a^{-1} in 1900). In Ethiopia, however, even the accumulated N_r use may be increased within the proposed boundary. N_r conversion rather than eutrophication as the boundary criterion was supported by the local situation in Finland where N_r is not a limiting factor for eutrophication either in the local freshwater or in the marine system (Savchuk *et al* 2008). Internal P loading and consequent N removal triggered by the largest anoxic seafloor in the world (Carstensen *et al* 2014) promotes N-fixing cyanobacteria (Vahtera *et al* 2007), maintaining the P limitation. The buffering capacity provided by the originally oligotrophic, shallow lakes with a regular ice cover (Scheffer and van Nes 2007) which represent 10% of the sparsely populated Finnish area, tolerates greater per capita P flows than the global average.

The past shift to dominance of internal loading both in the local marine system and in agricultural lakes has made eutrophication practically irreversible (Ekholm *et al* 1997, Räike *et al* 2003, Ekholm and Mitikka 2006, Vahtera *et al* 2007, Savchuk *et al* 2008, Aroviita *et al* 2014). This situation is reflected in the critical boundaries for P flow to the local lake system determined here: while the current flow already is within the boundaries, the accumulated flow by far exceeds them. A shift to dominance of internal loading represents a potential tipping point in the world's freshwater systems and estuaries, aggravated by climate change (Meier *et al* 2012). This observation serves one goal of bottom-up assessments of planetary boundaries also referred to by Steffen *et al* (2015b): 'We emphasize that our subglobal-level focus is based on the necessity to consider this level to understand the functioning of the Earth System as a whole.'

4.2. Spatial variation in nutrient use: the socio-economic facet

The historical access to nutrients—the critical resource for food production—illustrates the social

facet of planetary nutrient boundaries. Estimates show that 80% of the world's exports of the virgin, economically exploitable P reserves originated in 2011 in North Africa (de Ridder *et al* 2013) where little fertilizer P is used because of affordability issues. The excess in nutrient use is high in wealthy industrialized countries such as Finland; however, the differences between the wealthiest and poorest nations are declining, and therefore, the inequity is most reflected in the accumulated reserves. In Finland the originally low field soil P content increased by a factor of 2.2 from 1960 to 1995 (Mäntylahti 2002) with little change thereafter (Aakkula and Leppänen 2014). In many of the poorest regions of the world such as in sub-Saharan Africa, the use never really began and small P reserves and low organic matter content with N and P stocks occur in arable soils (Johnston 1986, Sanchez 2002) degraded by exploitation. Whereas the virgin P reserves were a reason for wars in the past (Burnett 2005), the recyclable reserves are a more important asset to reallocate as the world returns to the critical boundaries.

When considering the accumulated use, there is no longer any space within the critical boundaries globally and in the wealthy North to exploit virgin nutrients. Even in the South, such as in Ethiopia, where the current annual conversion rate of N_r can be nearly doubled, there is only a little space to increase the accumulated annual conversion rate of N_r. The 'zero tolerance' to N_r conversion poses a higher challenge due to the unavoidable gaseous losses than for the P flow. The N_r losses can be reduced, e.g., by replacing fossil energy with the anaerobic digestion of agrifood residues or with other renewable resources not competing for, or emitting, nutrients such as solar and wind power. The nutrient residues and reserves are most efficiently used where the need for a response to food security is highest and the local nutrient flow is least above the tolerable limit, as in Ethiopia.

4.3. Limits to the local food supply

The perspective of the planetary boundaries emphasizes the overall ecological role of agrifood systems relative to the dominant Finnish view to date, which was based only on the agricultural nutrient flows to waters. The local share of the nutrient flows that occur within the agrifood system (see section 3.2) is a third higher than the local share of the nutrient load to waters that originate in agriculture (for N 47%, P 57%) (SYKE 2014). This difference is due first to the entire 'agrifood system' and not only primary production ('agriculture') and, second, to gaseous emissions, not only leaching to water systems, being included in the planetary nutrient boundary perspective. P and N use, waste and emissions in various parts of the agrifood system such as in consumption, trading, food and feed processing and fish farming, in addition to agriculture, are included. The N_r conversion also covers gaseous

emissions from fertilizers and manure and from energy use in fertilizer manufacture, transportation, processing and trading, as well as loading to water systems. Only through such shifts in perspectives the major overall role of food for returning to the 'safe space for humanity' even in purely ecological terms (Garnett 2011) becomes evident, and the selection of means to return to the boundaries while simultaneously enhancing food security becomes broader. Innovations are necessary to fulfil this task, and the potential food security within the critical boundaries is not yet known.

The large share of the nutrients flowing through agrifood systems emphasizes the need for transformations in those systems to reduce the nutrient excess and secure the local access to food globally. In sub-Saharan Africa such as in Ethiopia, the vicious cycle of nutrient depletion and insufficient replenishment of N, P, potassium, sulphur and carbon drives the further decline of soil organic matter (Kirkby *et al* 2013, Kahiluoto *et al* 2014b) and thus nutrient losses through erosion (Haileslassie *et al* 2005) and carbon and N emissions, as well as leads to low productivity, limited nutrient flows in the agrifood systems and food insecurity. Consequently, the critical P boundary appears to be exceeded in Finland and Ethiopia for the following contrary reasons: (1) excessive past inflow to the agrifood system in Finland, and (2) excessive outflow from the agrifood system triggered by deficits in the inflow and in sewage management in Ethiopia, contributed to by improper management in both cases. While improper management in Finland implies inefficient recycling of nutrients due to manure concentration, low utilization of other agrifood residues and excessive fertilizer use (Kahiluoto *et al* 2011), it implies overexploitation of agricultural land in Ethiopia with insufficient nutrient inputs and low recycling of residues that serve as fodder and further as energy source (Rimhanen and Kahiluoto 2014).

Equity in the access to nutrients is necessary to increase the global efficiency in nutrient use for food. The added value of nutrient use for crop yield and combating hunger may be many times greater and the loads to waters smaller in areas with a low historical access to nutrient resources. Equity in the access to nutrients is also a step towards access to food despite global price volatility, conflicts, and poor local infrastructure and governance. The historical spatial variation in welfare limits the present agricultural production and consumption within the safe boundaries more than the local ecosystem or the current spatial inequity. The past inequity in access to resources manifests in the nutrient reserves available in soil and sediments and, more importantly, in the inequity in nutrition evident in the social and human capacities today. The under-nutrition of children leads to the irreversibly diminished physical development of the body and brains (stunting), reducing the cognitive

capacities; stunted women being more likely to give birth to stunted children (UNICEF 2014).

4.4. Bridging the gap

The current and projected future failure in reversing the local nutrient cycles through reducing P flow only (Ekholm and Mitikka 2006, Jarvis *et al* 2013, Aakkula and Leppänen 2014, Haygarth *et al* 2014) shows that for a transition to a low-nutrient society, major technological, social and institutional innovations are still to come. This is highlighted by the scenario that if nutrient inputs are reduced to the level that existed more than 100 years ago, the Baltic Sea would not be recovered even after 130 years (Savchuk and Wulff 2009). Recycling the reserves in soils (Sattari *et al* 2012), in water biomass (Kahiluoto *et al* 2011) and in agricultural lake and marine sediments may help to restore the ecology of water systems and the productivity of the degraded soils in the global South, if it is based on a firm ecological understanding. To meet the shown requirement to reduce nutrient use in the global North, many means in various parts of the agrifood systems, such as dietary shifts and waste prevention, are required in addition to recycling (Kahiluoto *et al* 2014a). For example, the exchange of nutrient quota (NEFCO 2008) or a combination of taxes (Goulder and Schein 2013) and subsidies, spatially differentiated in the global scale, may trigger, finance, and complement the trading of nutrient reserves and the rights to use nutrients.

The planet is a social-ecological system. The major complement to the PB framework required to serve as a sustainability target is the social dimension of global equity, tightly coupled with the boundaries of the key biophysical processes (Rockström *et al* 2009; Steffen and Stafford Smith 2013, Steffen *et al* 2015a,b). The tipping point for the pivotal social boundary, equity in the access to nutrients and thus to food, requires bottom-up considerations at lower hierarchical levels than the planetary scale. This critical boundary has been repeatedly transgressed in the past and is currently locally transgressed, highlighting the imperative of the local social-ecological context when assessing and combating these transgressions.

5. Conclusions

The unequal past distribution of nutrient flows hinders current equal access to nutrients and food. The reallocation of the recyclable reserves is a precondition for achieving food security. Spatial variation in critical nutrient boundaries and, particularly, in past nutrient use highlights the necessity of monitoring the cumulative transgression of the boundaries and for a world-wide financing framework to enable the redistribution of nutrients. Bottom-up assessments reveal local system dynamics that further elucidate the relevant boundary criteria, estimates and remedies.

Spatial equity in nutrient flows implies a critical factor of global equity and human rights, namely, the right to food.

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