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LETTER

Valuing damages and benefits of the altered global nitrogen cycle; lessons for national to global policy support

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Abstract

Cost-benefit analysis (CBA) is increasingly used to inform environmental policy decisions by identifying interventions with the highest net societal benefits. Here we focus on CBAs for nitrogen (NCBA), explaining its history, presenting results of a recent first global NCBA and discussing opportunities and limitations. NCBAs have been conducted since the late 1990s for various geographic regions in Europe, the US, and China, primarily to support air quality and eutrophication policies. A first valuation of damages and benefits of the full nitrogen (N) cycle was conducted for the European Nitrogen Assessment in 2011, followed by NCBAs for the USA, the Netherlands and Germany. Here we present a first comprehensive global NCBA. Total global damage cost of N pollution in 2010 was estimated at US$1.1 trillion, primarily from increases in premature mortality by N derived $PM_{2.5}$ (35%), terrestrial biodiversity loss by N deposition (33%), and marine eutrophication by N river loads (21%). Global benefits of N in 2010 were estimated at US$ 2.2 trillion with >95% from increased crop yields. By 2050, global N-related costs will rise faster than N benefits because underlying models project that economic growth (GDP) increases willingness-to-pay to prevent N pollution more than crop prices. The geographical distribution of N-related costs will also shift, with China and India surpassing Europe and North America as regions contributing most to global N-related costs. The estimated N cost range for 2010 was US$ 0.6–2.2 trillion with uncertainty largely in dose-impact and damage cost relations. Given the large uncertainties, when using valuation and NCBA to select a N mitigation option, the net benefits should be substantially higher than the costs and markedly better than for a rejected alternative option. Use of NCBA is discouraged to compare international policy options that involve regions with very different levels of GDP, cultures and political systems.

1. Introduction

1.1. Altered global N cycle

The global environmental impacts of reactive nitrogen (N_r) losses are high and increasing in many parts of the world (Galloway *et al* 2003, Sutton *et al* 2011, 2013, 2019, Einarsson 2024, Sutton 2025). N_r refers to all N compounds other than unreactive dinitrogen (N_2), which constitutes 78% of the mass of our atmosphere. The most important reactive N compounds are nitrogen dioxide (NO_2), ammonia (NH_3), nitrate (NO_3^-) and nitrous oxide (N_2O), which are characterized by high mobility in air, water and soil, causing multiple impacts on biodiversity, human health and the greenhouse gas balance. Since the industrial revolution, the global anthropogenic generation of N_r increased to over 200 Tg/yr of N (figure 1) in 2020, and was close to three times higher than the intensity of the pre-1850 natural terrestrial N cycle (Fowler *et al* 2013). The three major anthropogenic sources of reactive N are synthetic fertilizer, cultivation of legumes (especially soybean) and formation of nitrogen oxides (NO_x) at combustion of fossil fuels.

N use and enrichment cause both benefits and damages to society (Jones *et al* 2014, Sutton *et al* 2019). The underlying hypothesis for this paper and use of NBCA in general, is that these costs often come from excess or inefficient use of N, so it should be possible to substantially reduce the costs with relatively little impact on the benefits. These benefits are most evident for intentional N addition to agricultural land. Shibata *et al* (2025) identified almost fifty ways in which N impacts the environment. The most important ones are listed in table 1.

1.2. Economic valuation of N pollution for policy decisions

The rationale of economic valuation of damages and benefits for the N cycle derives from the concept of externalities or external costs by Pigou (1917), which prescribes a tax to be included in market transactions to maximize welfare in the case of pollution. The tax is equivalent to the marginal damage that is being done by one additional unit of pollution. In this approach, a marginal change of a (nitrogen) state or pressure is thus valued. As there is no market of supply and demand for setting a price on externalities, shadow prices are used that can for example be derived from surveys measuring preferences, for example, by willingness-to-pay (WTP) for prevention or resolution of environmental impacts. As surveys into preferences typically are local, large-scale application of valuation requires translation of prices to other contexts, e.g. by scaling using differences in GDP, population or area. This procedure is also referred to as benefit transfer and is an essential component for global valuation of disturbance of the nitrogen cycle (Daly and Farley 2011).

Monetization of environmental pollution and cost-benefit analysis (CBA) can be used to guide (1) policy decisions, (2) signal and raise public awareness, (3) support the process of environmental liability (thereby applying the 'polluter pays' principle), (4) sustainable financing and (5) implement true pricing. The 'polluter pays' principle was first introduced by the OECD in 1972 and enshrined in the EU Treaty since 1987. The principle is considered to be at the heart of EU environmental policy even though its practical application is lagging behind (European-Court-of-Auditors 2021). Ultimately, internalization of externalities should change behavior and practices of producers and consumers, in other words, CBA turns environmental protection and sustainability into a financial case. This ambition is often explained by that 'money speaks louder than words'. Daily *et al* (2000) distinguishes three steps in decision-making using CBA: (1) identification of possible mitigation or policy alternatives (2) identification of all relevant effects, both classical economic (labor, capital, natural resources) and externalities, and (3) valuation of the consequence of not taking actions versus the policy alternatives. The first case of effective use of CBA for a policy decision by the European Commission related to nitrogen pollution was for the revision of the NO_2 air pollution standards in the Air Quality directive. Olsthoorn *et al* (1999) found that the societal benefit for the EU of reduced disease and premature mortality by lowering the NO_2 standard amounted to 0.41–5.9 billion euro/year. This was so much higher than the implementation cost of 0.08 billion euro/year that it convinced the European Commission, in spite of strong protests from the car manufacturing industry. This shows that comparison of benefits and costs of mitigation can be effective to select the most beneficial policy options. However, there are alternative approaches to aggregate and weigh changes of nitrogen impacts (supplementary material SM1.)

1.3. Examples of previous regional and national nitrogen pollution cost valuations and NCBAs

The European N Assessment (Sutton *et al* 2011, Van Grinsven *et al* 2013) concluded that the total N pollution cost in 2008 for the EU27 amounted to €75–485 billion per year, equivalent to 1%–4% of the GDP of the EU27. Half of this cost was due to N pollution from agricultural sources, with major and comparable contributions by NH_3 emission and N runoff, whereas the cost of mortality and disease by nitrate pollution in drinking water was small. Using a similar approach as for the European N Assessment, Sobota *et al* (2015) quantified the total N pollution cost for the US in the early 2000s, Oehlmann *et al* (2021) for

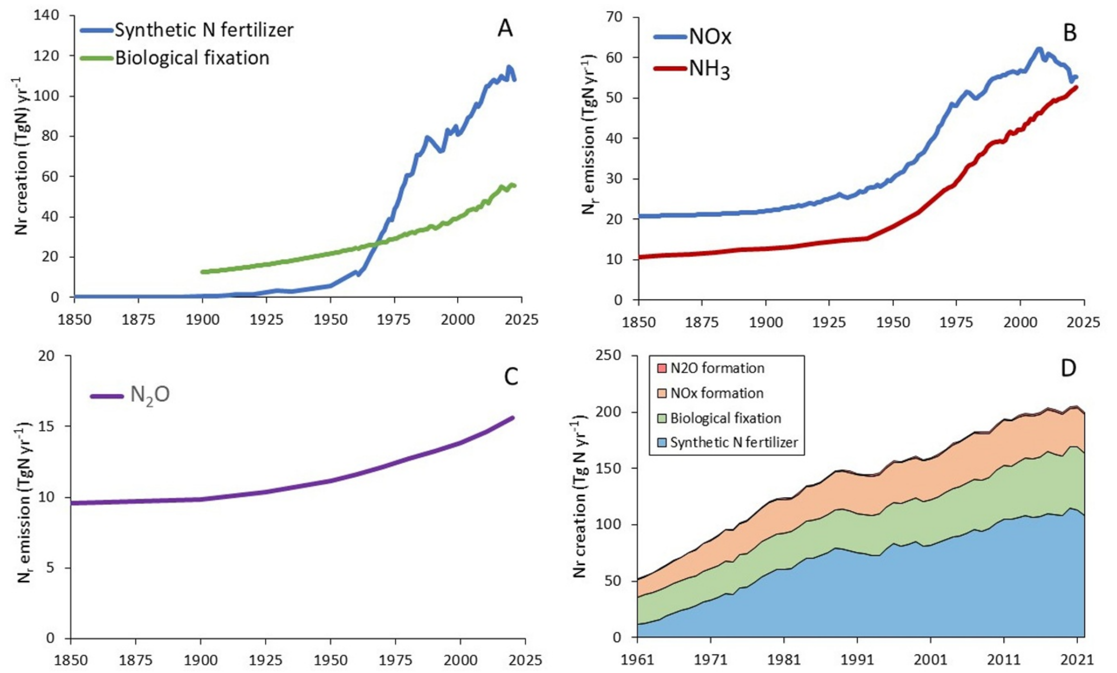

Figure 1. Evolution of global anthropogenic terrestrial formation of reactive nitrogen (N_r). Data sources: (A) Use of synthetic fertilizer after 1960 from FAOstat and before 1960 from Smil (2004); biological nitrogen fixation in agriculture from the IMAGE-GNM model from Beusen *et al* (2022). (B) Nitrogen oxides (NO_x) and ammonia (NH_3) emissions between 1970 and 2022 from EDGAR (https://edgar.jrc.ec.europa.eu/dataset_ap81) and the CEDS database (Hoesly *et al* 2018), Emissions between 1850 and 1970 are from Aardenne *et al* (2001). Emissions include natural emissions from biomass fires. (C) Nitrous oxide (N_2O) is total emission based on Aziz *et al* (2024) where the 1850 value is mostly natural. (D) Emission of N_r in N_2O only includes the minor fraction of newly formed Nr as derived from Tian *et al* (2020).

Table 1. Overview of negative and positive effects of increased nitrogen (N) input or environmental load.

Costs (Damages)
1. Human disease and premature death by N-induced $PM_{2.5}$ in ambient air pollution
2. Human disease and premature death by NO_2 in ambient air pollution
3. Human disease and premature death by N-induced ambient ozone (O_3)
4. Global warming by long-lived GHGs (N_2O, CO_2, CH_4) enhanced by N enrichment
5. Loss of terrestrial biodiversity by N deposition
6. Loss of ecosystem services due to marine eutrophication by N enrichment
7. Human disease by N_2O-driven depletion of stratospheric O_3
8. Human disease by nitrate and nitrite in drinking water
9. Loss of ecosystem services due to fresh surface water eutrophication by N enrichment
Benefits
1. Increased crop production through intentional addition of N to agricultural soils (in the form of synthetic N fertilizer and residues of N fixing crops or livestock manure)
2. Increased crop production through unintentional atmospheric N deposition
3. Increased C-sequestration by N enrichment and other processes influenced by N and contributing to global cooling
4. Increased production of woody biomass by unintentional atmospheric N deposition for use in construction, industry and as biofuel
5. Use of industrially produced ammonia as an energy carrier for renewable energy and as low emission shipping fuel

Germany in 2015 and Gu *et al* (2012) for China in 2008 (for more details see supplementary material SM 2).

The European N Assessment (Van Grinsven *et al* 2013) estimated the direct farm benefits of increased N input for crop production for the EU27 in 2008 at €20–80 billion per year, which value was 2–3 times lower than N pollution cost of €35–230 per year. This suggests that the economic optimum N rate, where the marginal farm cost of fertilizer application equals the marginal benefit is not optimal for society as a whole. This was also concluded by Rodríguez *et al* (2024) for global cereal cultivation.

An NCBA for the Netherlands in the formal evaluation of the Dutch implementation of Nitrates Directive in the Fertilizer and Manure Act found that the benefits of the N policies for agriculture were up 7 times higher than the implementation costs (Van Grinsven and Bleeker 2017). In an NCBA for Denmark, Jacobsen et al (2024) found that the national health benefits of reduced colorectal cancers by complying to a stricter nitrate standard overwhelmed the additional mitigation cost.

Gu et al (2021) estimated that the ratio of global benefits over costs (BCR) of halving ammonia emissions would range between 2.6 and 4.5, depending on including fertilizer savings, as compared to a BCR of 0.4 for a 50% reduction of NO_x emissions alone. They concluded that national and international air pollution policies therefore should prioritize controlling NH_3 emissions. Liu et al (2019), Zhang et al (2020) and Guo et al (2020) came to similar conclusions for China (for more details, see supplementary material SM 3).

Here we present a first comprehensive global NCBA which is part of the International Nitrogen Management System project (www.inms.international/) and published in 2025 (Sutton 2025). Results are compared to a global NCBA which was part of a full CBA for the global agri-food system (FAO 2023). Uncertainties of damage cost are quantified and implications for policy use are discussed.

2. Method

The method to quantify environmental damages and convert these to monetary units is referred to as the 'Impact-Pathway-Approach (IPA)' (Bickel and Friedrich 2005). IPA has been most widely applied to evaluate air pollution impacts of energy generation. Models are used to translate emissions to changes in concentration levels (the air quality state) and next to impacts on humans and ecosystems, In the last step these impacts are converted to a monetary value (figure 2; Brink et al 2011, Rabl et al 2014, Silveira et al 2016), which can either be a cost (e.g. loss of recreational value) or a benefit (e.g. increased forest growth by nitrogen deposition or increased crop production by N fertilizer input).

Application of IPA to the disturbance of the global N cycle (Sutton et al 2013, Van Grinsven et al 2013) is demanding as the N cascade involves multiple environmental media, transport and exposure pathways and impacts. An overview of the used emission-dispersion-impact models is given in De Vries et al (2020) and a summary of application and valuation procedures for five major impacts in supplementary table 1. Global costs for N damage items as listed in table 1 were not included when these were small or very hard to quantify. Cost of human disease and premature death by NO_2 in ambient air pollution were excluded following (Van Dingenen et al 2018) and in view of complex interactions with effect of NO_2-induced formation of $PM_{2.5}$ and O_3 (Wang et al 2025) and risk of double counting (Castro et al 2023). Cost of human disease caused by N_2O driven depletion of stratospheric O_3, nitrate and nitrite in drinking water and damages by freshwater eutrophication by N enrichment were also excluded. One reason for these exclusions was the unavailability of required data and models to quantify impacts on global scale, partly because impacts of N on drinking water and on freshwater ecosystems depend on spatially highly variable pressures and practices. Further, practices determining impacts of nitrate in drinking water depend on highly variable quality of local resources and consumer choices for using untreated, public or bottled water for cooking and drinking. Previous work (Van Grinsven et al 2013) indicated that global costs for excluded items were small.

Damage costs can be estimated in multiple ways, e.g. prevention costs, restoration costs, revealed or stated preferences (de Vries et al 2024). WTP is a concept that is most closely related to welfare economics: it captures the preferences of individuals directly through e.g. surveys using choice experiments (Perman et al 2011). Regarding cost of eutrophication, the Baltic sea is one of the longest running studies (Markowska and Żylicz 1999, Ahtiainen and Öhman 2014). Preferences to mitigate eutrophication tend to, implicitly or explicitly, also address upstream impacts on freshwater and groundwater feeding into the Baltic Sea. People living in the Baltic region do not only recreate on the coast but also swim and fish in the many lakes and rivers (Vesterinen et al 2010) and are informed that sources and solutions for eutrophication of fresh and marine waters are interconnected. Therefore, surveys reveal that their WTP for improvements remains high regardless of how far they lived from the Baltic sea (Hyytiäinen et al 2013). Damage (or Dose) response functions often are nonlinear due to changing responses with increasing exposure (Shibata et al 2025, see supplementary material SM 4). Valuation (or Monetization) functions depend on GDP, using an income elasticity (ε), population and other scalers to allow benefit transfers to other regions or into the future (Daly and Farley (2011), see supplementary material SM 4). Table 2 summarizes and characterizes the effects of N use and losses as used for the global NCBA (for details see supplementary table 2).

Costs and benefits of nitrogen pollution in 2050 were explored for three contrasting scenarios from the shared socioeconomic pathways (SSPs) scenario group (Van Vuuren et al 2021). These scenarios consider different futures regarding climate forcing using the representative concentration pathways (RCPs) under different assumptions on future socio-economic developments and climate policies. On these SSPs different future N storylines (Kanter et al 2020) were superimposed. Due to lack of data, mitigation costs were not included. We used three combinations of SSPs, RCPs and N-policy ambitions: SSP1-$RCP_{2.6}$H representing high -policy ambitions for mitigating climate warming and N pollution (S7), SSP2-$RCP_{4.5}$M representing medium

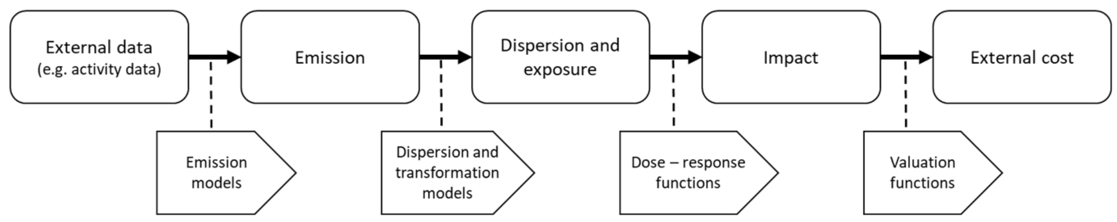

Figure 2. Schematic representation of Impact-Pathway-Approach (IPA) for the nitrogen cascade.

Table 2. Overview of impacts of N use or N loss, including the driving global N flow and the mean global unit damage cost. Codes 2nd column denote D = Damage, B = Benefit, X = Damage eXcluded in global NCBA. Characterization as Damage or Benefit is for increase of N flow. Only N in NO_x emission, synthetic fertilizer production and biological N fixation are new formation of Nr, all other impacting N flows are recycled or secondary. (*derived for the EU).

		Impact	N drivers	Global driving N flow 2010 (TgN)	Average global marginal cost (unit)
1	D	Increased mortality by ambient PM pollution	NO_x emission to air	35	3.6 (U$/kg NO_x-N)
2	D	Increased mortality by ambient PM pollution	NH_3 emission to air (recycled N)	49	3.6 (U$/kg NH_3-N)
3	D	Increased mortality by ambient O_3 pollution	NO_x emission to air	35	0.5 (U$/kg NO_x-N
4	D	Crop loss by ambient O_3 pollution	NO_x emission to air	35	0.4 U$/kg NO_x-N)
5	D	Terrestrial biodiversity loss (MSA) by N deposition	Atmospheric N deposition	32	12.8 (U$/kg Ndep)
6	B	Increased wood production	Atmospheric N deposition	32	0.8 (U$/kg Ndep)
7	B	Increased C sequestration forests for climate cooling	Atmospheric N deposition	32	8.2 (U$/kg Ndep)
8	D	Marine impacts (recreation, eutrophication	*N surplus agricultural soil*	*102*	
			N surplus natural soil	*91*	
			Net river N load to marine from:	40	8.4 (U$/kg N load)
			Groundwater	*18*	
			Surface runoff	*8*	
			N-point sources + other	*13*	
9	X	Freshwater eutrophication	N leaching and point sources	pm	pm (U$/kg N load)
10	X	Increased mortality and disease by nitrate drinking water	N leaching to drinking water sources	18	0.9* (U$/kg NO_3-N)
11	D	Climate damage	N_2O emission to air	4	29.7 (U$/kg$N_2O$-N)
12	X	Skin cancers and eye cataract	N_2O emission to air	4	0.8 (U$/kg N_2O-N)
13	B	Crop production (cereals)	N synthetic fertilizer input	101	15.3 (U$/kg N input)
			N biological fixation	44	
			N manure input (recycled N)	*97*	

policy ambitions (S3), and SSP5 $RCP_{8.5}$L (S1) representing overall low environmental policy ambitions (Rodríguez *et al* 2024).

3. Results

3.1. Nitrogen impacts and costs in 2010

The total global N pollution damage cost around 2010 was estimated at almost US$ 1.1 trillion (table 3, figure 2). China (30%) and Europe (22%, incl. FSU—Former Soviet Union) were the largest contributors to the global N pollution cost. Major costs resulted from premature mortality by N-induced formation of ambient particulate matter (35%), loss of terrestrial biodiversity by increased N deposition (33%) and marine eutrophication by increased river N loads (21%). The average global loss of life expectancy by N-driven air pollution in 2010 was 4 months, ranging between one month in Africa (incl. Middle East) and eight months in China. Average global loss of terrestrial biodiversity by increased N deposition was close to 7% (measured in MSA—Mean Species Abundance; Schipper *et al* 2020), with highest loss of 9% in China and a lowest value 5% in Oceania including South–East Asia (van Grinsven *et al* 2025). Net N-induced warming via long-lived greenhouse gases contributed 8% to the global N damage cost. Although the gross cost of N-induced warming of

US$ 414 billion was the largest cost (38% and equivalent to 3.5 Pg CO_2eq and mainly caused by emission of N_2O), this cost was largely compensated by N-induced climate cooling representing a global benefit of US$ 326 billion (30%, equivalent to 3 Pg CO_{2e} and mainly by N-induced C-sequestration; (Du and de Vries 2018, Schulte-Uebbing and de Vries 2018, de Vries *et al* 2025).

Given the large differences in GDP and population, the total societal cost of N pollution is not very informative for comparing the severity of N pollution damage in global regions. Expressing N pollution costs relative to GDP provides a better proxy for this (table 3). This relative global N cost in 2010 was 1.8% with highest values close to 3% for China and India and lowest values in the Americas. High relative N cost in China and India reflect both higher N pollution levels and lower GDP in these regions. N pollution cost per capita is another informative metric to compare regions and shows more contrast than N cost relative to GDP. Global average N pollution cost was 157 US$, ranging between 61 US$ in India and 650 US$ in Europe. N pollution cost per capita could be used as an indication of a maximum budget per capita for mitigation of N-related pollution.

3.2. Nitrogen costs and benefits in 2050

The projected anthropogenic formation of reactive N in 2050 under S7 and S3 is somewhat lower than in 2010 and close to the 1990 level, while under S1 it is 1.6 times higher than in 2010 (figure 3). The consolidation of Nr formation under S7 and S3 is the effect of policy, while the sharp increase under S1 is mainly caused by a massive increase of synthetic N fertilizer use, especially in Asia.

The projected costs of N pollution in 2050 are 2.6 times higher than in the base year under S7, 2.7 times higher under S3, and 4.7 times higher under S1 (figure 3(B)). The increases under S7 and S3 are primarily caused by increasing unit damage costs (UDCs) (figure 6) with increasing GDP due to the effect of income elasticities. Global $GDP_{ppp2005}$ is projected to increase by a factor of three (under S3) to five (under S1) between 2010 and 2050 (figure 4). Under S1, N pressures and impacts also increase. The projected total costs of N pollution in 2050 expressed as a percentage of GDP are fairly stable (figure 4), with the lowest share in 2050 under the high ambition N policy scenario (S7).

China accounted for 30% of the global N pollution cost in 2010. This share is projected to increase to 33%–40% in 2050, reflecting that the decreasing effect of future N policies on N cost is smaller than that of economic growth (figure 5). The share of Europe and North America in total global N pollution decreases from 35% in 2010 to 16%–18% in 2050, mainly due to an increase in total global N cost. The largest relative increase of N pollution costs is projected for India, where the share in total global cost increases from 9% in 2010 to 15%–21% in 2050.

3.3. Direct nitrogen benefits

Global benefits of increased N use exceed global costs for the year 2010, but not in 2050 under scenario S7 and S1 (figure 6). More than 95% of global benefits of increased N use are provided by increased crop production by increased use of N fertilizer. Benefit contributions of increased crop production by N deposition (2%–3%) and of wood production by N deposition (0.2%–0.4%) are minor. Cereals contribute about half to global crop production (Rodríguez *et al* 2024). Projected total global cereal grain production increased from 2.8 Gton in 2010 to 3.4, 3.8 and 4.1 Gton in 2050 for S7, S3 and S1, respectively. N benefits for all crops increase from 2 trillion USS_{2010} in 2010 and 2050 under S7, to about 4 trillion USS_{2010} under S3 and S1. The change of monetized crop benefits in 2050 reflects the combined effect of increases in global crop yields and projected region dependent crop prices (Rodríguez *et al* 2024).

Applying IPA also allows to establish the contribution of N to the total impacts (N-share hereafter), and thereby to the total cost or benefit. The global mean N-share around the year 2010 for increase of harvested cereal yields by synthetic N fertilizer was estimated at 45%, for premature mortality by air pollution by $PM_{2.5}$ at 30% and for biodiversity loss in terrestrial ecosystems at 7% (for details see supplementary material SM 5).

3.4. N damage costs in the context of total externalities of the global agri-food system

FAO (2023) estimated the total external (or hidden) cost of the global agri-food system in 2020 at $US\$_{ppp2020}$ 12.7 (CI_{90} 10.8–16.0) trillion (table 4), equivalent to almost 10% of the global GDP and close to the market value of the global food system. By far, the largest contribution (73%) to the global external cost was for disease and mortality caused by unhealthy dietary patterns. The external cost caused by N pollution was estimated at $US\$_{ppp2020}$ 1.5 trillion (CI_{90} 0.5–4.3), inferring a N-share of 12% in the total external cost of the global food system. The FAO study identified N pollution as the major environmental cost, contributing almost half. The included impacts and approaches (Lord *et al* 2023) were similar to this study (table 1). A large part of the almost 50% higher estimate of total global N cost by (Lord 2023) for 2020 as compared to our study for 2010 (table 3) can be explained by the increase of the global GDP by more than 30% and its effect on UDC through the GDP elasticity.

4. Uncertainties

Given the large uncertainties in dose-response relations and UDCs based on WTP, it is no surprise

Table 3. Societal cost of nitrogen pollution for N flows in 2010 in global regions. Costs are expressed in $USD_{ppp2015}$ and expressed as % of GDP and $USD_{ppp2015}$ per capita (ME is Middle East, FSU is Former Soviet Union, ANZ is Australia and New Zealand, L is Latin and N is North)[16].

2010	Population (billion)	GDP_{ppp} (trillion USD_{2005})	Mortality ambient PM_Nr	Mortality ambient NO_x	Crop yield loss by O_3	Terrestrial biodiversity loss	Marine pollution	Net Nr land warming	Net Nr aquatic warming	Net Nr warming	Total N cost
					Billion USD_{2015}						
World	**6.98**	**69.1**	**421.4**	**23.9**	**14.6**	**406.5**	**258.5**	**88.6**	**1.5**	**90.1**	**1215**
Africa + ME	1.33	6.1	34.7	0.8	1.2	42.4	4.5	11.6	0.3	11.8	95
China	1.41	11.6	110.1	12.6	4.6	99.8	82.6	18.3	0.3	18.6	328
Europe + FSU	0.81	17.4	141.7	3.0	1.0	77.2	78.3	−5.4	0.2	−5.2	296
India	1.23	3.7	28.0	2.5	1.9	50.7	6.4	15.6	0.3	15.8	105
L America	0.59	6.2	20.3	0.6	0.5	30.1	11.1	21.5	0.1	21.6	84
N America	0.34	13.6	39.2	5.3	4.1	40.3	54.4	10.4	0.1	10.4	154
R Asia + ANZ	1.27	10.5	42.4	1.8	1.2	66.1	21.3	16.7	0.3	17.0	150
					% of GDP						
World			**0.62%**	**0.05%**	**0.03%**	**0.59%**	**0.37%**	**0.13%**	**0.00%**	**0.13%**	**1.78%**
Africa + ME			0.59%	0.02%	0.03%	0.70%	0.07%	0.19%	0.00%	0.20%	1.62%
China			0.92%	0.11%	0.10%	0.86%	0.71%	0.16%	0.00%	0.16%	2.86%
Europe + FSU			0.87%	0.03%	0.01%	0.44%	0.45%	−0.03%	0.00%	−0.03%	1.77%
India			0.79%	0.10%	0.16%	1.38%	0.17%	0.42%	0.01%	0.43%	3.03%
L America			0.31%	0.01%	0.02%	0.49%	0.18%	0.35%	0.00%	0.35%	1.35%
N America			0.29%	0.05%	0.03%	0.30%	0.40%	0.08%	0.00%	0.08%	1.15%
R Asia + ANZ			0.40%	0.02%	0.01%	0.63%	0.20%	0.16%	0.00%	0.16%	1.43%
		GDP_{ppp} (USD_{2005}/cap)					USD_{2015}/cap				
World		**9895**	**60.3**	**3.4**	**2.1**	**58.2**	**37.0**	**12.7**	**0.2**	**12.9**	**174**
Africa + ME		4557	74.7	15.1	0.7	77.5	116.5	52.8	19.4	72.1	357
China		8270	65.9	44.7	2.0	91.2	84.5	43.2	16.3	59.4	348
Europe + FSU		21 367	186.7	51.6	4.8	97.1	139.9	138.7	31.5	170.3	650
India		2976	26.2	0.6	0.9	31.9	3.4	8.7	0.2	8.9	72
L America		10 461	6.4	1.6	0.2	30.9	19.6	45.4	19.4	64.7	123
N America		39 761	6.9	2.9	0.5	38.3	13.1	37.5	16.3	53.8	115
R Asia + ANZ		8310	24.0	5.1	0.7	40.4	22.3	115.6	31.5	147.1	240

[16] ppp: purchase power parity units, so corrected for regional difference in purchase power, here for year 2005.

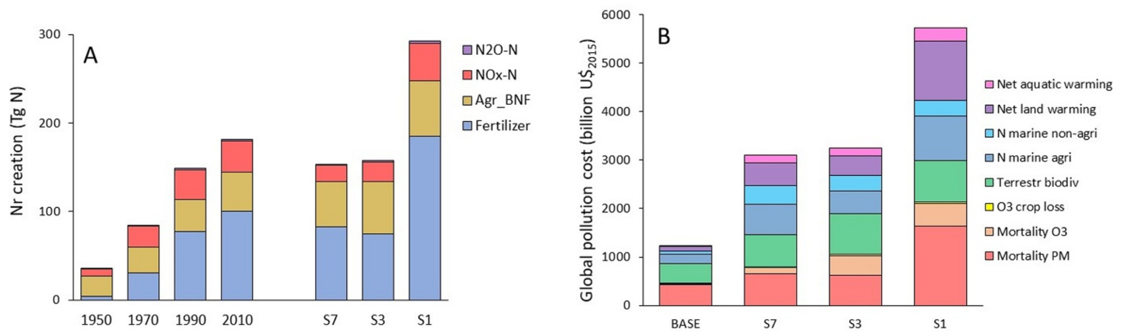

Figure 3. History of global anthropogenic terrestrial formation of reactive N (A) and monetized impacts of related N pollution in base year 2010 (B) and projected changes in 2050 under three INMS scenarios (S7, S3 and S1) which combine SSP1, SSP2 and SSP5 with RCP 2.6, 4.5 and 8.5 and with high, medium and low N policy ambitions.

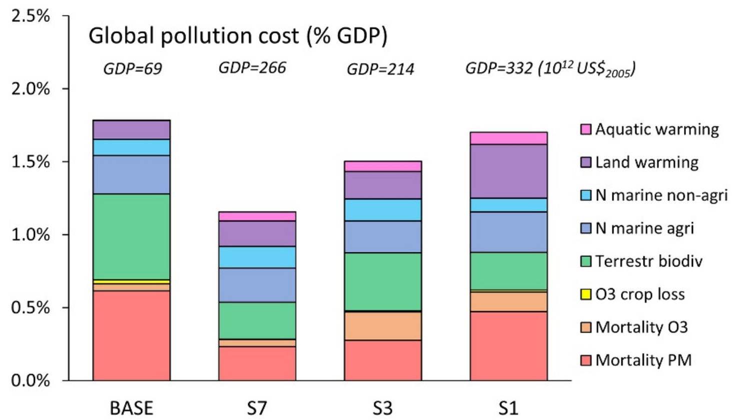

Figure 4. Monetized impacts of global N pollution expressed as share of GDP for the base year (2010) and for 2050 under three contrasting scenarios. Total GDPs are given in italic.

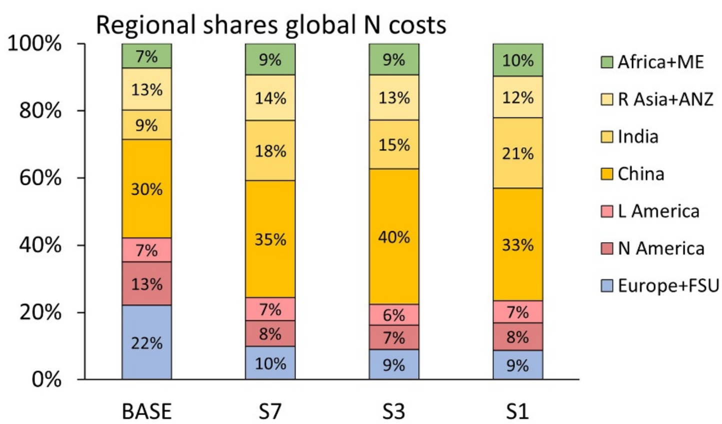

Figure 5. Projected regional shares in global N pollution costs in the base year (2010) and for 2050 under contrasting scenarios.

that uncertainties in the estimates of nitrogen damage costs for countries or regions are very substantial. FAO (2023), based on Lord (2023), reported a 90% CI of US$$_{ppp2020}$ 0.5–4.3 for global N pollution cost in 2020 (table 4); corresponding to a range of 33%–285% around the mean. Rodríguez *et al* (2025) quantified the uncertainty for the total N costs for global cereal systems using the same response and monetization functions as used for global N costs in table 3, applying a Monte Carlo approach. The 90% CI for global N damage costs in 2010 for cereal cultivation was US$$_{ppp}$ 0.090–0.172 trillion corresponding to a range of 71%–137% around the mean. This lower uncertainty compared to Lord (2023) could be caused

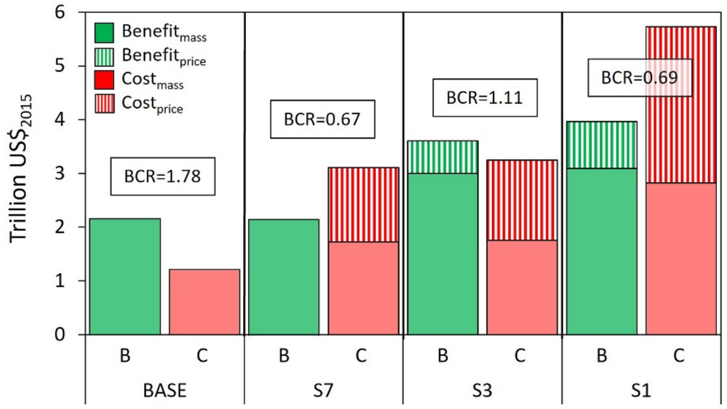

Figure 6. Total costs (C) and benefits (B) of global N use in the base year of 2010 and projected for 2050 under contrasting scenarios in real US$ for 2015 price levels, distinguishing effects of increase in flow and unit price. Also shown are the ratios of benefits over costs (BCR).

Table 4. Environmental, social and health hidden costs for the global agrifood system in 2020. Adapted from FAO (2023). CC BY 4.0.

		Environment				Social		Health	N-share	
	TOTAL	Climate	Water	Land	Nitrogen	Agrifood worker poverty	Under-nourishment	Dietary patterns	Total	Environment
	Billion US$$_{ppp2020}$									
World	12 749	855	105	392	1516	520	51	9310	12%	53%
Africa	953	154	4	43	57	285	19	392	6%	22%
America	2978	220	11	149	368	12	5	2212	12%	49%
S-America	*894*	*130*	*4*	*17*	*229*	*6*	*3*	*505*	*26%*	*60%*
N-America	*1711*	*68*	*6*	*128*	*73*	*0*	*0*	*1435*	*4%*	*27%*
Asia	5857	356	84	59	815	222	27	4294	14%	62%
China	*2555*	*104*	*9*	*6*	*382*	*3*	*0*	*2052*	*15%*	*76%*
India	*1123*	*77*	*36*	*24*	*144*	*157*	*15*	*669*	*13%*	*51%*
Europe	2862	113	5	139	261	1	0	2343	9%	50%
Oceania	99	13	0	2	14	0	0	70	14%	47%

by the assumption of no correlation between uncertainties of individual impacts. Relative uncertainties in global UDCs per individual N impacts could center around 50% of the mean UDC (see e.g. de Vries *et al* 2024 and Rodríguez *et al* 2025, see also supplementary tables 3 and 4). Assuming that uncertainties in these UDC values for individual impacts are fully correlated, this would imply a maximum uncertainty range for the total global N cost (table 3) of also 50%, implying a global cost range of 0.6–2.2 trillion US$$_{ppp}$ in 2010. Assuming no correlation and uniform distributions of UDCs this range would be 0.9–1.8 trillion US$$_{ppp}$. The uncertainties in UDCs partly derive from uncertainties in underlying surveys into preferences or WTP to prevent or resolve N damages. Voltaire *et al* (2013) and Braun *et al* (2016) included respondent's uncertainty in surveys into WTP to protect nature, respectively, into perceptions about solar radiation management, finding uncertainties again close to 50%. Another source of uncertainty in the UDCs used in our global NCBA is that WTPs for individual impacts are derived from separate surveys. Ideally, preferences to prevent or resolve the multiple impacts of N should be surveyed in one combined survey adding context allowing respondents to make an informed prioritization of the various impacts.

An uncertainty range for the mean total global N damage cost of ±50% could still be an underestimate because, we did not relate our calculated damages and costs to planetary boundaries, ecological tipping points or the possibility of irreversibility of some impacts, like biodiversity loss or soil degradation which carry the risk of disruption of the global food system.

5. Discussion

IPA and CBA for the global N cycle are potentially important tools to deal with the involved multiple N-sources, N-forms, dispersion and exposure paths and impacts. The first comprehensive global NCBA presented here provides new insights about the relative importance of impacts in global regions. Its relevance could be complemented with scenarios that include mitigation costs. Zhang *et al* (2020) and Gu *et al* (2021), for example, implemented

Marginal Abatement Cost Curves in their NCBA and convincingly showed that air pollution policies should prioritize reducing NH_3 over NO_x emissions. Nevertheless, policy makers could still be reluctant to use NCBA in important decisions about N policies. One reason is the presence of large uncertainties in monetized N damages amounting to a factor of two for the global and regional scale. Valuation and ultimately monetization, is a way of organizing information to help guide decisions and targeting main goals, but it does not provide a ready solution (Daily *et al* 2000). Another reason for policy reluctance could be the interdisciplinary nature of NCBA, demanding understanding and integration of social, natural and life sciences, which is challenging for reaching academic and political consensus. Putting monetary weights on the very diverse effects of N use may be conceived by politicians as a limitation of their political mandate to set priorities. While the aforementioned problems were not a barrier to use of NCBA to set new NO_x standards in the EU around 2000 (Olsthoorn *et al* 1999), they are increasingly posing barriers to take decisions today. An example is the current Dutch policiy to reduce ammonia emission to bring 74% of the N-sensitive nature areas in 2035 below the critical N load to comply with the EU Habitat Directive (Boezeman *et al* 2023). The acting Dutch cabinet values the economic risks for farmers and the farming sector higher than the risks of biodiversity loss, which challenges both the natural science and life science underlying the IPA linking ammonia emissions of farms to the critical status of Dutch nature.

Further, the academic and ethical justifiability to add up and compare monetized values for impacts across very different impacts domains and global regions is contested. We give three examples. First, our NCBA finds similar monetary values for global premature mortality by N-derived ambient air pollution and for loss of terrestrial biodiversity by increased N deposition. The former relies on stated WTP for longer healthy life expectancy, while the latter reflects WTP for prevention or restoration of biodiversity. However, critics may argue that disease and premature death by ambient air pollution are less existential to society than loss of biodiversity. These deaths are derived from epidemiological studies that result in statistical values based on modeled attribution of total premature mortality to a suite of causes (see e.g. Stanaway *et al* 2018). Most of these premature deaths cannot be attributed directly to air pollution, as air pollution acts as a comorbidity. When someone dies prematurely, very rarely ambient air pollution can be identified as the primary cause of death. Loss of terrestrial biodiversity, however, is increasingly viewed as an existential threat for humanity (Richardson *et al* 2023). The consequences of degradation of ecosystem services like pollination and natural pest control for agriculture and food security can hardly be overstated. Second, another arbitrary element of global NCBAs is the choice to make the value of human life and biodiversity dependent on GDP. Therefore, in IPAs and NCBAs the value of human life in a low-income country is lower than in a high-income country. Particularly, this can become problematic when NCBA is used to justify moving N polluting activities to low GDP countries. While not based on NCBA per se, the latter has been a common practice in the past decades, based on classical economic arguments of lower wages and need for economic development in low GDP countries. Thirdly, pricing can be viewed as problematic as it conceives human well-being only in terms of utility or satisfaction of preferences (Wegner and Pascual 2011). In a typical NCBA, some people will be worse off, while others will be better off. Summing these individual effects into an aggregate implies the Kaldor–Hicks Compensation Principle. This principle assumes that the marginal utility of money is the same for all individuals in society, so that the people worse off can in theory be compensated for their loss of welfare. This certainly is not the case in an unequal world lacking democratic institutions. For example, Manero *et al* (2024) concluded that values of Indigenous people are very different and systematically under-represented. Munasinghe and Lutz (1992) concluded that estimates of values of externalities (non-use values) in the developing world were virtually non-existent. Therefore, the use of NCBA to set N policy priorities or to select the best N mitigation options in international arenas, including both high- and low- income countries with diverse cultures and political systems, is problematic as it ignores underlying differences in value structures. If access to sufficient food, water, housing, energy, medical care etcetera is not secured and the majority of people and the political representatives are not familiar with the western concepts of welfare, justification of policy decisions based on IPA and NCBA is not accepted.

However, NCBAs generally indicate when welfare gains for society can be expected by further reducing nitrogen loads. This would be a step forward, as political decisions are often based on lobbies or prevalent public opinions rather than on maximizing welfare for the society as a whole (Wegner and Pascual 2011). In this process, public participation, deliberative procedures and transparency of decision-making could enhance the results of the NCBAs. And we should not forget that there are alternatives for assigning societal N weights to different impacts or mitigation options, for example, by direct use of polls into public concerns or their consolidation in distance to policy targets. In a good government, these concerns should be reflected in the allocation of national budgets. Van Grinsven (2016) indeed found a correlation between WTP to prevent N pollution and proxies for budget allocation for Europe.

6. Conclusion

Valuation of N impacts and NCBA has the potential to become a useful tool for guiding policy on prioritizing strategies for joined-up mitigation of diverse N pollution impacts. Valuation helps to assess the relative importance (or 'societal weight') of these diverse N impacts, which is important to increase societal acceptance of N political action. For now, the potential of NCBA is highest in politically, socio-economically and culturally homogenous regions and therefore best established in western, educated, industrialized, fairly rich and democratic countries (so-called WEIRD countries, Henrich *et al* 2010). In such countries, money tends to 'speak louder than words', which demonstrates that nitrogen mitigation is welfare-enhancing and can be effective in driving ambitious environmental policies. Valuation and NCBA can justify shifting policy away from traditional N concerns, like local nitrate pollution of drinking water resources, toward more alarming global issues like biodiversity loss. Even when policymakers are reluctant to use the monetized N damages directly, NCBA also allows for the aggregation of a broad set of N impacts, offering insights into their variation over time and geography. However, there are also concerns. While the underlying IPA is well established, it relies on complex dispersion- and impact- models. Moreover, assigning monetary values to environmental goods remains contentious, particularly when comparing the 'hypothetical' non-market benefits of improved environmental quality with 'real' financial gains from increased agricultural production or the 'real' costs of N mitigation. A key takeaway from this first global NCBA is that three N-related impacts dominate: premature mortality from N derived $PM_{2.5}$, terrestrial biodiversity loss from N deposition and marine eutrophication from N river loads. Another robust message is that N damage costs in 2010 were around 2% of global GDP and that this percentage will not change very much in 2050 under three contrasting scenarios. Uncertainties of valuation results from application of IPA are large. Therefore, when using valuation and NCBA to select a N mitigation option, the net benefits for the preferred option should be substantially higher than for the rejected options.

Data availability statement

The data that support the findings of this study are available upon reasonable request from the authors.

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