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Exploring the potential for nitrogen fertilizer use mitigation with bundles of management interventions

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Abstract

Mineral nitrogen (N) fertilizer use is essential to maintain high-yielding cropping systems that presently provide food for nearly half of humanity. Simultaneously, it causes a range of detrimental impacts such as greenhouse gas emissions, eutrophication, and contamination of drinking water. There is growing recognition of the need to balance crop production with the impacts of fertilizer use. Here we provide a global assessment of the potential to reduce mineral fertilizer use through four interventions: capping surpluses, enhancing manure cycling to cropland, cultivation of off-season green manures, and cycling of human excreted N to cropland. We find that the combined potential of these interventions is a reduction in global N fertilizer use by 21%–52%. The availability of interventions is spatially heterogeneous with most cropland having three to four interventions available with alternative N sources tending to be more abundant on cropland already receiving fertilizer. Our assessment highlights that these locally in part already practiced interventions bear great opportunities to mitigate synthetic N use and dependency globally. Yet, their limited adoption underpins the need for cross-sectoral policies to overcome barriers to their implementation and agronomic research on their robust scaling.

1. Introduction

Global agricultural production and food security have become highly reliant on mineral fertilizers over the past century. Synthetic mineral nitrogen (N) is estimated to support the nourishment of nearly half of humanity [1]. Due to its wide-spread use, N losses from agriculture have become the major cause for aquatic, marine, and terrestrial eutrophication, and a drinking water pollutant [2–4]. Moreover, the dependence on fossil fuel energy for producing synthetic fertilizers and its concentration in few countries globally has been identified as a risk for food security [5–7]. Contrasting to the contributions of N fertilizer to food security and its environmental externalities, large parts of global croplands are subject to plant nutrient deficits, contributing to food shortages, lock-ins in marginalized livelihoods, and amplified soil degradation [8, 9].

Having long been addressed in environmental [10] and agronomic research [11], sustainable crop nutrient management has more recently started taking an increasingly prominent role in policymaking, including the EU's Farm to Fork strategy [12], the USA's global fertilizer challenge [13], China's action plan for targeting zero growth of synthetic fertilizer use [14, 15], and various initiatives of the United Nations [16, 17].

Various options to alleviate synthetic N fertilizer use have been evaluated or proposed in earlier studies, most often focusing on a single approach. These include improvements in crop N use efficiency (NUE) through improved fertilizer management and technology [18], enhanced cycling of manure N to cropland [19], in-field biological N fixation [20], crop and fallow rotations [21, 22], crop residue management [23, 24], and recovery of N from waste streams such as sewage [25] among others. An earlier synthesis has been provided by the Global Partnership on Nutrient Management and the International Nitrogen Initiative [26]. Several of these interventions have earlier been combined in studies focusing on N pollution mitigation, e.g. across sectors in China [15] and from croplands globally [27]. Global and continental integrated modeling studies have evaluated strategies to mitigate global or regional fertilizer inputs or related outcomes of food system transformations, addressing implications of a global roll out of organic farming for N cycles [28], outcomes for food security of adhering to regional N surplus boundaries [29], feasibility of N cycle closure at regional scales [30], or future N requirements for contrasting demand and management scenarios [31]. These earlier studies provide insights into the multi-faceted outcomes of transformative change in the agri-food system for both human nutrition and N cycles considering demand and supply-side measures. Yet, they do not quantify the impact of individual interventions and their combinations within a consistent data framework and spatially explicit globally, which is vital to evaluate their potentials for sustainable N management at regional scales and to understand the role of bundles compared to individual interventions across spatial scales. It also avoids the loss of information and potential bias from using data aggregated at administrative levels such as the balancing out of spatially divergent nutrient surpluses and deficits or the potential overestimation of combined intervention potentials.

To overcome this gap, we herein employ comprehensive state-of-the-art spatial data and literature-derived parameters to quantify the potential for mitigating synthetic N inputs to cropland through four interventions (I) capping present fertilizer surpluses at attainable levels of NUE, (II) increasing the rate of N cycling from livestock manure to cropland, (III) substitution of mineral N fertilizer with N provided by side-season green manures, and (IV) substitution of mineral N fertilizer with N captured from human excretion (table 1). Through a chain of analyses (figure 1), the study provides novel insights on the spatial distribution of the interventions' availabilities and potentials as well as their global totals and individual shares, including the effectiveness of individual and combined interventions.

2. Methods

2.1. Study design

We assess both the individual and combined technical potential of the interventions in sequence based on the logic that inefficient resource use (*in-situ* management) should be addressed before investing in new N sources as the latter requires deeper economic transformations. While the availability of N from organic sources may be limited in initial years after their implementation, we assume here a steady state when N cycling is equilibrated [32]. A mid-point scenario serves as a medium variant for use in all evaluations beyond the intervention cascade (see section 2.7).

All analyses were performed for a reference year 2010 and at a spatial resolution of 5 arcmin (approx. 8.3 km × 8.3 km near the equator), assuming local implementation and no transport of N beyond this distance. Including transport would require further information and assumptions on specific implemented technologies, infrastructure, and socio-economic context, which are beyond the scope of this study. Still, further potentials from transport can be bracketed based on the global technical potentials (figure S8) of in principle mobilizable N sources manure and human waste. Brief rationales of the interventions are provided in supplementary text S1, a comparison of our estimates presented herein with earlier research is provided in supplementary text S2, and a discussion of key limitations of the methodology in supplementary text S3.

2.2. Soil surface nutrient balance

A cropland soil surface N balance serves for categorizing cropland with N surplus and deficit (e.g. figure S11) and provides the basis for the estimation of improvements in NUE (section 2.3). The pixel-specific soil surface N balance [33] (N_{bal}) was calculated as (all units [kg])

$$N_{bal} = (N_{fert} + N_{man} + N_{fix}) - \left(N_{crop} + N_{residue,\,removed} + N_{residue,\,burned}\right) \quad (1)$$

where N_{fert} is the amount of N applied in mineral fertilizer, N_{man} is the amount of N applied in manure, N_{fix} is the amount of biologically fixed N, N_{crop} ist the amount of N exported in harvested crops, $N_{residue,removed}$ is the amount of N removed in crop residues, and $N_{residue,burned}$ is the amount of N lost from burning residues.

We estimate spatially explicit crop-specific N fertilizer application rates around the year 2010 based on the respective rates around the year 2000 [34] and scaling them by national changes in N fertilizer application between 2000 and 2010 as reported in FAOSTAT [35]. Scaled application rates were capped at the maximum global rate for each crop in the original dataset to avoid unreasonable application rates. Subsequently, data for the 132 crops reported in [34] were aggregated to the 42 crop types (table S3) in the land use database SPAM 2010 v2.0 [36, 37], which provided spatially explicit crop yields, harvested, and physical cultivated areas. Manure N applied to cropland was adopted from [38]. Nitrogen embedded in harvested crop biomass (supplementary table S3) was

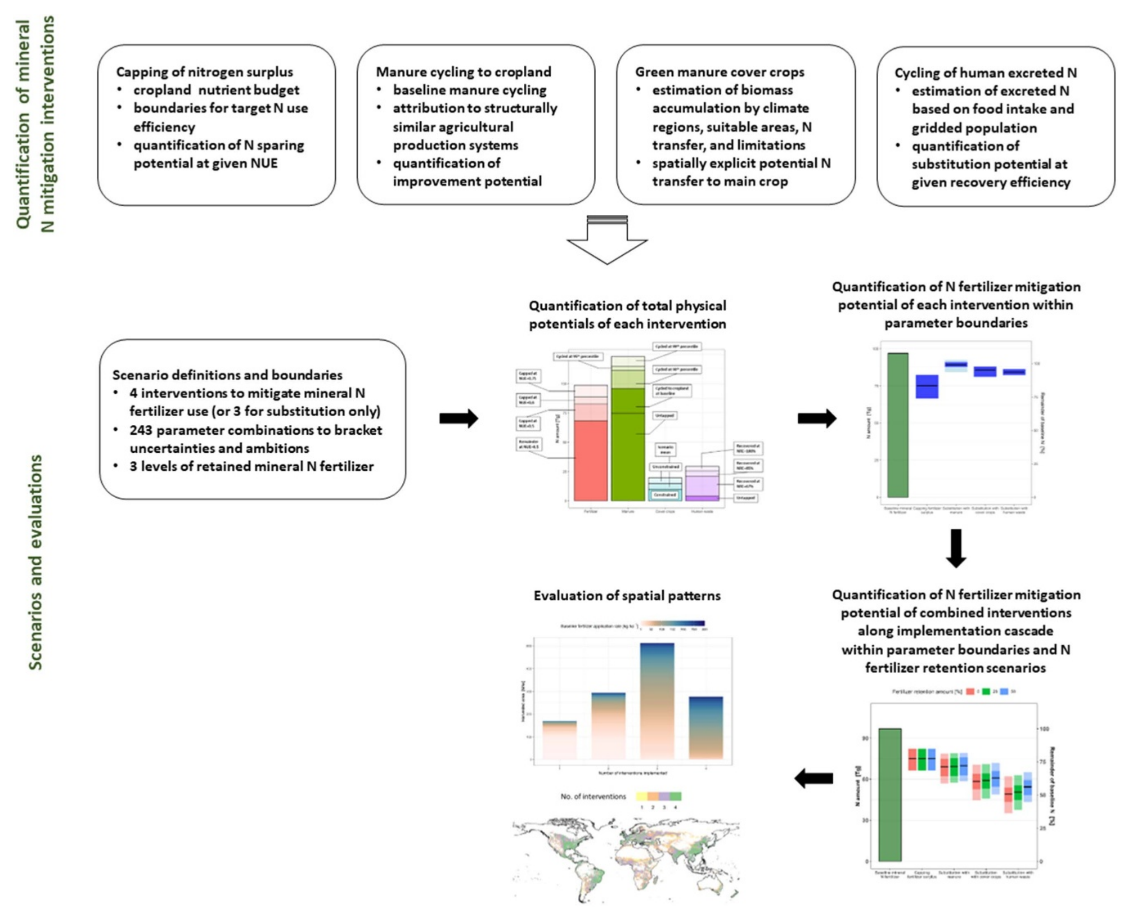

Figure 1. Study design schematic outlining the flow of conceptualization, quantification, and analyses. The top row outlines the interventions, which are in the bottom part combined to scenarios bracketed by parameter ranges, followed by a quantification of global total potentials, individual and combined implementation to mitigate N fertilizer use, and complementary evaluations on the requirement for bundles of interventions. See Methods for details and table 1 for an overview of intervention characteristics.

sourced from databases [39] and literature. Nitrogen embedded in crop residues was estimated based on the same sources and crop-specific harvest indices as implemented in the EPIC cropping systems model [40, 41]. Removal and burning rates of crop residues for major crops at the national scale were adopted from [42]. Residue burning rates for rice were adopted from a review for major producing countries [43] assuming an N removal fraction of 80% [44]. For leguminous crops, we assumed that N not applied in fertilizer is fixed from the atmosphere or long-term soil N cycling from earlier fixation.

2.3. Reduction potential for nitrogen fertilizer application surplus

The potential for mitigating mineral N fertilizer surplus was estimated based on the cropland soil surface N balance and achievable nitrogen use efficiencies (NUEs). There are various conceptualizations and accordingly approaches to estimate NUE as reviewed and exemplary quantified in [45]. Herein, we refer to fertilizer NUE in cropping systems in terms of the N removal in harvest relative to agronomic N inputs. We set an NUE range of 0.5–0.75 of which the lower end corresponds to lower targets commonly used in literature and policies and the upper end is a conservative but still highly ambitious estimate of achievable NUE as elaborated in supplementary text S1. To quantify the potential for N fertilizer sparing through NUE improvement, we first calculate the present pixel-specific N requirement for all crops ($N_{\text{req,base}}$) as (all units [t])

$$N_{\text{req, base}} = \left(N_{\text{crop}} + N_{\text{residue, removed}} + N_{\text{residue, burned}}\right) - \left(N_{\text{fix, crop}} + N_{\text{fix, rr}} + N_{\text{fix, rb}}\right) \qquad (2)$$

where $N_{\text{residue,removed}}$ is N removed in residue, $N_{\text{residue,burned}}$ is N lost in burned residue and $N_{\text{fix,rr}}$ and $N_{\text{fix,rb}}$ are the proportions of fixed N in the earlier. See equation (1) for other elements. Subsequently, this value was scaled to the hypothetic requirement at a given NUE ($N_{\text{req,nue}}$) according to

$$N_{\text{req,nue}} = N_{\text{req,base}} \div NUE \qquad (3)$$

where NUE is the target NUE parameter (table 1). The pixel-specific fertilizer sparing potential $N_{\text{spare,nue}}$ was

Table 1. Nitrogen fertilizer mitigation interventions, their rationales, and parameter boundaries reflecting uncertainties and levels of ambition. Quantifications were carried out for the 243 combinations of the three values for each of the five parameters. The mid-point scenario is based on the values in the respective row combined with the fertilizer retention scenario keeping 20% of the baseline application volume. NFRV is the nitrogen fertilizer replacement value, i.e., the fraction of organic N that can readily be taken up by a crop the same way as mineral fertilizer. See Methods section for details and supplementary text S1 for extended rationale and parameter ranges.

Intervention	Mineral N fertilizer surplus sparing	Increasing manure cycling to cropland		Cultivation of green manures	Cycling of human excreted N
Rationale and exemplary measures	Reduce mineral N surplus through improved management practices that result in increased N use efficiency (NUE), e.g. through better timing, tailored fertilizer formulations, and technology adoption	Additional manure N cycling to cropland by adopting practices from farms with higher manure N cycling rates and avoiding losses through inefficient handling or direct discharge		Cultivation of N fixing green manures outside the main crop growing season on climatically suitable cropland	Capture, processing, and application of human excreted N from wastewater and sanitation
Lower boundary	Achievable NUE = 0.50	Cycling ratio at 90th percentile	NFRV = 0.5	As upper boundary, modified by soil limitations (see Methods)	Recovery efficiency = 20%
Mid-point	Achievable NUE = 0.60	Cycling ratio at 95th percentile	NFRV = 0.75	Mean of outcome for lower and upper boundaries	Recovery efficiency = 55%
Upper boundary	Achievable NUE = 0.75	Cycling ratio at 99th percentile	NFRV = 1.0	Spatially explicit coefficients for N transfer constrained by aridity and cropping intensity	Recovery efficiency = 90%

then calculated as

$$N_{\mathrm{spare,nue}} = N_{\mathrm{fert}} - N_{\mathrm{man}} - N_{\mathrm{req,nue}} \text{ with } N_{\mathrm{spare,nue}} \geqslant 0. \quad (4)$$

2.4. Improved manure cycling to cropland

This intervention assumes that all farms within similar production systems (e.g. low-input subsistence farming, mixed crop-livestock, high-input farming) can realize a similar rate of manure cycling to cropland through knowledge and technology transfer. To estimate attainable rates of manure cycling to cropland, we combined gridded estimates of baseline manure N cycling to cropland [38] (figure S1) with a global classification of agricultural systems, i.e. landscapes with structurally similar agricultural production systems [46]. The latter have been derived using spatially explicit data on agro-environmental characteristics (e.g. soil types, topography, native vegetation; level 1) and agricultural production systems (e.g. crop types, management intensities, livestock; level 2), which resulted in a total of 86 unique typologies with ten classes of varying agricultural management (level 2). We used this level 2 classification to estimate the manure cycling to cropland ratios for the 90th, 95th, and 99th percentile per class (figure S2). These attainable ratios were attributed to each pixel within a given management class if the baseline ratio was lower. Subsequently, we estimate how much of the applied manure N reaches the crop the same way as mineral N fertilizer assuming good management practice and a steady state of the system by specifying a nitrogen fertilizer replacement value (NFRV) [32, 47]. Accordingly, pixel-specific N available from improved manure cycling ($N_{\mathrm{avail,manure}}$) [kg] was estimated as

$$N_{\mathrm{avail,man}} = \left(N_{\mathrm{excreted,man}} \times cr_{\mathrm{pecentile}} - N_{\mathrm{applied,man}}\right) \times NFRV \quad (5)$$

where $N_{\mathrm{excreted,man}}$ is total manure N excreted in the pixel [kg], $cr_{\mathrm{percentile}}$ is the fraction of manure cycling for the given percentile (i.e. scenario) in the agricultural system typology the pixel belongs to [−], $N_{\mathrm{applied,man}}$ is the manure N applied in the baseline [kg], and NFRV is the N fertilizer replacement value as a fraction [−]. In the case of $N_{\mathrm{avail,man}} < N_{\mathrm{applied,man}}$, i.e. if the respective pixel is above the given percentile of cr, $N_{\mathrm{avail,man}}$ was set to 0.

2.5. Cultivation of green manure cover crops

We estimate potential N transfer from an off-season green manure to a main crop based on (a) average annual cover crop biomass production for legumes across cool to subtropical climates [48] ($n = 389$ publications), (b) an average N concentration of 2% in dry matter biomass [49], (c) an N transfer rate, and (d) suitable area. We digitized relevant figures from [48], extracted biomass data for legume species per temperature and humidity class, and calculated the mean for each climate (table S1). For actual N transfer, we assume an availability of green manure N for a main crop of 30%, which is at the lower end of experimental findings ranging from 27%–41% [22, 50, 51]. Resulting coefficients of green manure N transfer to the main crop for the temperature (cold, mild, warm) and humidity (semi-arid, humid) categories were assigned to Koeppen-Geiger climate regions over cropland (figure S3).

The area suitable for green manure cultivation was determined by (a) Koeppen-Geiger regions for which several green manure trials ($n > 2$) have been recorded in at least one of two extensive literature reviews (see below), (b) aridity index (AI) > 0.3, and (c) cultivation of annual crops, and (c) cropping intensity (i.e. the number of crops grown within a year consecutively on the same field). Rules (a) and (b) were determined by intersecting locations of off-season legume cover crop experiments ($n = 173$) from two databases on soil conservation practice trials [52, 53] with a dataset of Koeppen-Geiger regions [54], and determining the 90th percentile of the aridity index [55] for the same data. We made the exceptions to include Koeppen-Geiger region Am, for which no locations have been reported but which is climatically located between Af and Aw, and exclude hot steppes BSh, where trials are located but green manures may only be feasible to cultivate in niches. To account for cropping intensity, we scaled the potential for N transfer in each pixel by the ratio of harvested to physical cropland area (figure S5).

For a scenario with soil limitations (lower boundary; table 1), spatially distributed values of potential N transfer were scaled to 20% in pixels that have soils with high or very high P immobilization potential according to [56] or a negative cropland P budget according to [57], which may substantially limit the establishment of a green manure [58]. We hence consider P limitation a general limiting factor that frequently also reflects suboptimal pH, aluminum toxicity, and other soil related limitations typically co-occurring in strongly weathered or otherwise degraded soils [59]. Accordingly, pixel-specific N available from green manures ($N_{\mathrm{avail,green}}$) [kg] was estimated as

$$N_{\mathrm{avail,green}} = biomass_{\mathrm{green}} \times N_{\mathrm{conc,biomass}} \times area_{\mathrm{suit}} \times frac_{\mathrm{corr}} \times ci \times 0.3 \times 1000 \quad (6)$$

where $biomass_{\mathrm{green}}$ is the biomass per hectare for the respective climate region [t ha^{-1}], $N_{\mathrm{conc,biomass}}$ is the concentration of N in the green manure biomass, here 0.02, $area_{\mathrm{suit}}$ is the suitable area for green manures in the pixel [ha], $frac_{\mathrm{corr}}$ is the scenario- and pixel-specific scalar [−] with or without soil limitations (0.2 for soil limitations or 1 assuming no limitations), ci is the cropping intensity scalar, 0.3 is the N transfer coefficient [−], and 1000 is a conversion factor from [t] to [kg].

2.6. Cycling of human excreted N

National amounts of N consumed in food per capita were estimated based on per capita food supply for the >90 commodities reported in FAOSTAT [35] less food waste [60], and food-specific N contents (table S4). Resulting rates [kg N cap^{-1} yr^{-1}] were multiplied with gridded population data for the year 2010 [61] to obtain annual N excretion per pixel (figure S6). We adopted a span of N recovery efficiencies ranging from 20% as an ambitious but realistic trajectory under present technology regimes in sanitation [62] to 90% as the recovery potential considered feasible in earlier research [25], selecting a mid-point of 55%. Based on this, N potentially available from human excretion ($N_{\mathrm{avail,human}}$) [kg] was estimated as

$$N_{\mathrm{avail,human}} = N_{\mathrm{diet,natl,\,cap}} \times (1 - frac_{\mathrm{waste}}) \times cap \times NRE \quad (7)$$

where $N_{\mathrm{diet,natl,cap}}$ is the amount of N in the national average diet per capita and year [kg], $frac_{\mathrm{waste}}$ is the accumulated fraction of food waste from field to fork [−], cap is the number of people in a given pixel [−], and NRE is the nitrogen recovery efficiency [−].

2.7. Intervention cascade and boundaries

Interventions were implemented sequentially in the order of columns in table 1. For each intervention, we defined lower and upper parameter boundaries to bracket uncertainties and ambitions. For each intervention and parameter set ($n = 243$), the technical N supply or capping potential was estimated first (prior sections). Subsequently, these potentials were subtracted from mineral N fertilizer inputs per pixel in the defined sequence. Eventually, the median, 25th and 75th percentile, minimum, and maximum were calculated for each step of the sequence. We capped the maximum N fertilizer sparing rate per pixel at a value of 100 kg N ha^{-1} for the combined interventions, assuming that higher values would require more specific, locally tailored approaches to be sustainable while maintaining present crop yields. To quantify a solely substitution-based N fertilizer mitigation path, we performed complementary analyses excluding improvement in NUE.

Replacing all mineral fertilizer with alternative sources of N may not always be recommendable or feasible without yield losses. For example, the nutrient stoichiometry of manure may need to be adjusted to match crop requirements [63], or organic sources of N may fail occasionally, e.g. if the establishment of a green manure is impaired by adverse weather. Moreover, earlier research indicates that a mix or organic and inorganic N sources often provides the best outcomes in terms of nutrient use and crop yields [64, 65]. To account for this, we combined the interventions with three levels of minimum baseline N fertilizer retention: (I) allowing all N fertilizer to be replaced (0% retention), (II) retaining at least 20% of baseline mineral N fertilizer per pixel, and (III) retaining at least 50% of fertilizer per pixel. This resulted eventually in three realizations of the above 243 parameter combinations or a total of 729 parameter sets.

The mineral N fertilizer remaining after intervention implementation ($N_{\text{remain,tot}}$) [t] was estimated in each pixel as

$$N_{\text{remain,tot}} = N_{\text{fert}} - N_{\text{spare,nue}} - N_{\text{avail,man}} - N_{\text{avail,green}} - N_{\text{avail,human}} \text{ with } N_{\text{remain,tot}} \geqslant 0 \tag{8}$$

where N_{fert} is baseline mineral N fertilizer input [t], $N_{\text{spare,nue}}$ [t] is N sparing potential through improved fertilizer NUE [t], $N_{\text{avail,man}}$ is N available from improved manure cycling [t], $N_{\text{avail,green}}$ is N available from green manures[t], and $N_{\text{avail,human}}$ is N available from human excretion [t]. With the above constraints of (a) a maximum sparing rate of 100 kg N ha^{-1} per pixel and (b) the respective N fertilizer retention fraction, interventions occurring later in the above chain were only considered if the N remainder stayed above these thresholds and remained non-negative in each pixel.

3. Results

3.1. N fertilizer mitigation potentials and patterns

The four interventions have a combined potential to spare 21%–52% of mineral N fertilizer inputs to cropland globally across the three levels of N fertilizer retention and intervention boundaries (figure 2(a); see also table 1). If the total N sparing potential is not limited to 100 kg N ha^{-1}, which we implemented here as a safeguard to avoid overly optimistic sparing potentials and to acknowledge that very high N fertilizer substitution may rather require tailored solutions, the sparing potential would be slightly higher with 21%–59% (figure S12).

Capping the N fertilizer surplus by increasing NUE to a range of 0.50–0.75 would lead to a reduction in global synthetic N use by 11%–28% (11–27 Tg). This agrees well with the range of earlier studies suggesting sparing potentials of 14%–30% with improved N fertilizer management including the abatement of gaseous emissions, better timing and placement of applications, and more efficient chemical formulations [34, 66].

Livestock manure N provides an additional savings potential of 3%–9% of the baseline, corresponding to 2.6–8.3 Tg N in addition to the approx. 25 Tg manure N presently applied to cropland [38]. While this is a minor share of the estimated 120 Tg manure N excreted by livestock globally, it needs to be considered that approx. 50 Tg N in animal fodder are sourced from grassland [29] and need to be replenished, including losses unavoidable in free-range grazing via runoff and volatilization [67]. This leaves a rather moderate fraction of manure N disposable as an input to cropland in addition to the baseline (see also figure S8).

Nitrogen transfer from the cultivation of green manures outside the main season allows for a further decrease in N fertilizer requirement by 4%–8%. This amount is primarily the result of the cropland area estimated to be suitable for green manures (herein 880 Mha), whereas N transfer rates themselves range from 1.4–32 kg N ha^{-1} with an area-weighted average of 8–16 kg N ha^{-1} globally for soil-constrained and unconstrained boundary scenarios.

Finally, recovering N from human food consumption and reusing it locally as a fertilizer can substitute 3%–7% of the baseline N fertilizer consumption, harnessing a volume of 2.4–6.8 Tg N out of a total potential stock of nearly 30 Tg N (figure S8). This volume results from reuse in rural and moderately urbanized areas coinciding with cropland; highly urbanized areas outside food-producing regions (>80% built-up land cover) would contribute another up to 7.5 Tg N (figure S8) from spatially highly concentrated sources. Yet, this is not accounted for in the present study, which does not consider spatial transfer of N, the feasibility and destination of which depends on the specific technology and underlying economics.

If improvement in NUE is not included but only N fertilizer substitution interventions (figure 2(b)), the total mitigation potential decreases to 12%–32%, highlighting the importance of improved fertilizer management but also the still sizable potential of alternative N sources combined. The contribution of livestock manure increases to 3.7–12.2 Tg (4%–13%), that of green manures to 5.1–9.6 Tg (5%–10%), and that of human excreted N to 2.9–10.3 Tg (3%–11%) as in regions that have potential for fertilizer surplus capping, a larger volume of fertilizer can be substituted by the alternative N sources in this scenario. Evidently, the alternative N sources can in part compensate a lack in NUE improvement but not to the full extent, rendering the latter the main lever at global scales among these interventions *in-situ*.

Spatially, rates of N fertilizer reduction range from close to zero up to the defined maximum of 100 kg N ha^{-1} for the mid-point scenario (figure 3). The highest rates occur where N can be sourced from concentrated livestock or human settlements, coinciding with high mineral N fertilizer use. Such

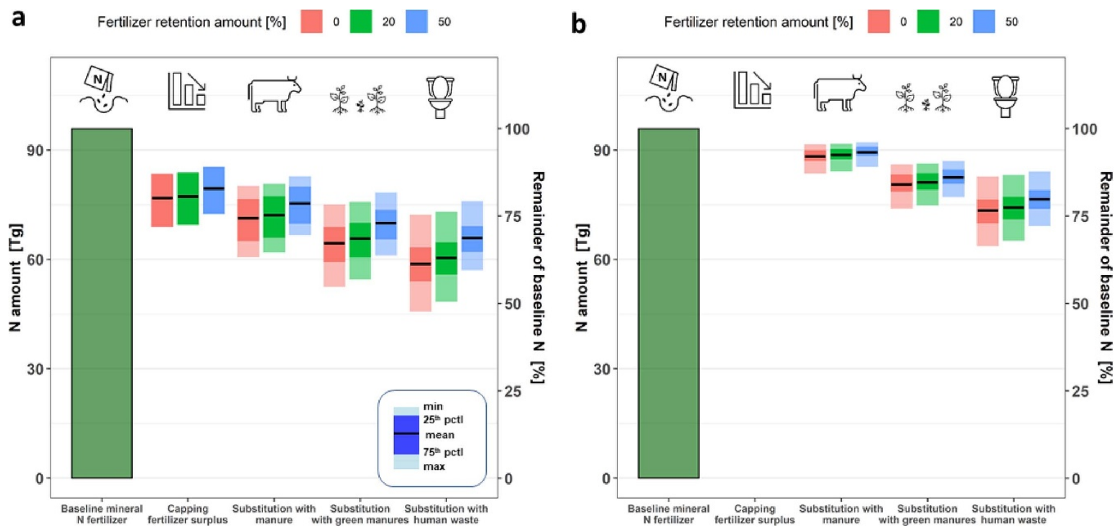

Figure 2. Combined mineral N fertilizer mitigation potential along a chain of interventions (a) including capping of N fertilizer surplus or (b) considering mineral fertilizer substitution with alternative N sources only. Later occurring interventions may bear a larger potential that is not fully exploited (see also figures S5 and S6). Interventions may hence be considered in part interchangeable leading to the same combined potential (i.e. result for last intervention). The green bar indicates the mineral N fertilizer consumption around 2010. Boxes show the total range (light color), 25th and 75th percentile (dark color) and mean (black line) of remaining N fertilizer requirement after the implementation of each intervention across its parameter ranges (table 1).

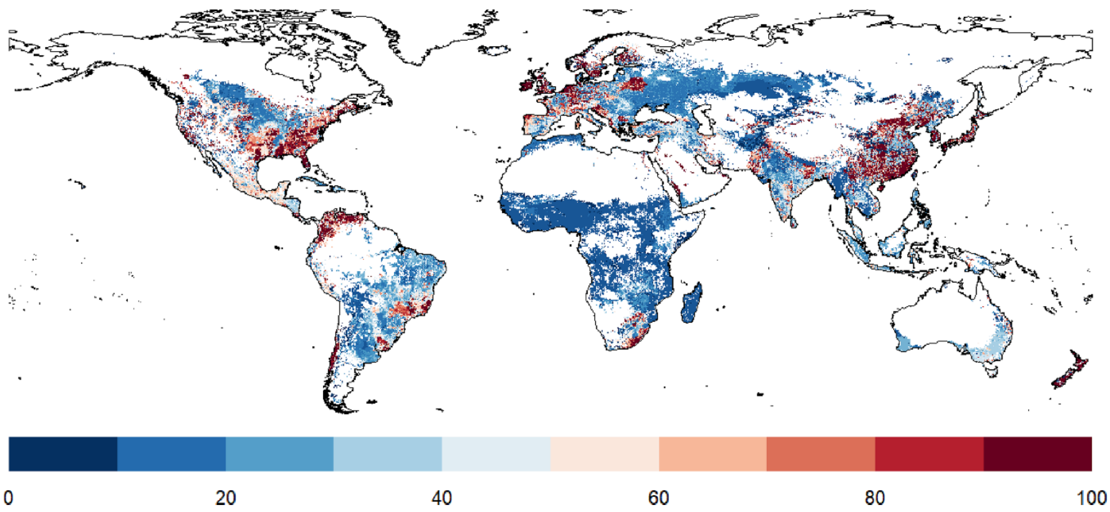

Figure 3. Rate of spared N [kg N ha^{-1}] for the bundles of interventions in the mid-point scenario (see table 1) with retention of at least 20% baseline mineral N fertilizer application.

hotspots are located in parts of China, the USA, Europe, New Zealand, South America, the Arab peninsula, and South Africa. Improvements in NUE or green manures in turn provide moderate to minor rates locally but are available more ubiquitously across global croplands.

Vis-à-vis the mitigation of mineral N application, we estimate complementary the availability of interventions on cropland with baseline N surplus or deficit separately (figure S11) to evaluate theoretical potentials for N input improvement also where additional sources may contribute to achieving higher productivity outcomes. We estimate that presently 34% of global croplands experience an N deficit and conversely 66% have a neutral or positive soil surface N balance. All four interventions are inherently only available on cropland with a positive N balance and improvements in NUE allow most often for the highest N mitigation rates. If only the three interventions that provide alternative N sources are considered, most areas in both N balance categories have three interventions potentially available (66% and 71% of cropland with negative or positive N balance, respectively), followed by cropland with two interventions (29% and 25%), and eventually one (5% and 4%).

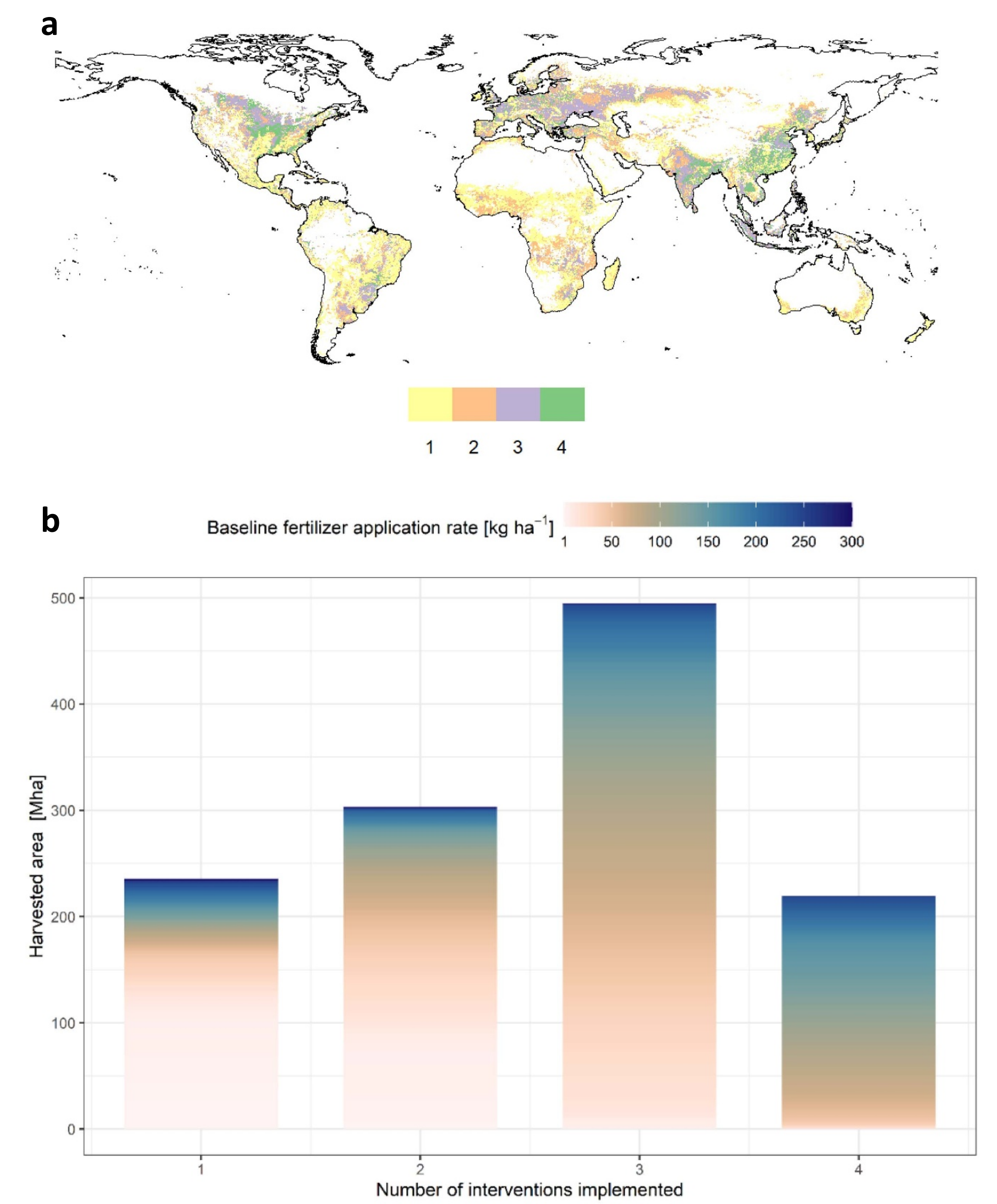

Figure 4. Number of interventions implemented in each pixel in the mid-point scenario (see table 1) (a) and extent of harvested cropland area with a given number of implemented interventions ordered by the baseline mineral N fertilizer application rate, which is also represented by the color scale (b). Larger numbers of interventions are available (figure S9) and required in high-input regions of North America, Europe, and Southern and East Asia. Three to four interventions are implemented on the majority of cropland (63%) and foremost in regions with moderate to high fertilizer application volumes.

In turn, the amount of N potentially available from the alternative sources (total of 58 Tg) is less balanced distributed among deficit and surplus cropland (not shown). Additional manure sources are foremost available on cropland with a positive N balance (83% of total) indicating limited potential in regions with present N fertilizer deficiency and the requirement for more transformative approaches in those. The distribution of N available from green manures in turn is more balanced with 63% to 37%, whereas the availability of human excreted N is again moderately skewed towards cropland with high N supply (77% vs 23%).

3.2. Requirement for intervention bundling

Each of the individual interventions has sizeable technical potential in terms of total N volumes globally (figure S8). However, they stay well below the combined N mitigation potential if implemented individually for fertilizer substitution (figure S7) as their spatial distributions vary greatly and their coincidence with fertilized cropland controls their effectiveness locally. Consequently, only bundles of interventions allow for mitigating the present levels of fertilizer use across most croplands globally (figure 4).

On the major share of cropland, three to four interventions are available (figure S9) and implemented simultaneously in the mid-point scenario (figure 4(a)). These are foremost located in intensively managed cropping regions of North America, Europe, and East and Southeast Asia. One or two interventions are primarily implemented in regions with low baseline fertilizer inputs, which typically cancels at least the option of surplus reduction, mostly in sub-Saharan Africa, Central Asia, and South America. Also, the spatial disconnectedness of croplands from alternative nutrient sources such as sufficiently large livestock herds and human populations leaves fewer options in many of these regions, rendering the cultivation of green manures the most ubiquitously realizable intervention there (figure S10).

More comprehensive bundles of interventions are evidently implemented on cropland with higher baseline N application rates whereas a single intervention is often sufficient on cropland with low baseline N application (figure 4(b); see also figure S13). A set of all four interventions is most frequently employed on cropland with N application rates of 50–300 kg N ha^{-1}. Still, there are few regions with moderately low fertilizer application that require all interventions, stressing that also their combined potentials can be limited locally. Conversely, the implementation of one or two interventions also occurs in part on cropland with N application of up to 300 kg N ha^{-1} highlighting that complementary options considered herein are not applicable across all intensively managed cropland globally.

4. Discussion

Our analysis highlights that optimizing fertilizer use and substitution bear great combined potential to reduce mineral N fertilizer consumption. Even if the potentials found herein cannot always be realized locally, several interventions exist in most places and harnessing them in a coordinated manner would nevertheless contribute to comprehensive global N mitigation potentials depending on the ambitions invested. This may be further enhanced by additional options not analyzed in this study such as the retention of presently burned or removed crop residue [29], N fixation and transfer within rotations [28, 68], or enhanced NUE and N fixation in intercropping [69] (see also supplementary text S1). Yet, while the first is considered to provide an overall low potential due to low N concentration and high C:N ratios in residues, the second and third would require a redesign of cropping patterns and likely the demand side. Beyond the supply-side perspective taken here, demand-side interventions such as food waste reduction and dietary change provide comprehensive levers for further reductions [29, 70] that may regionally eliminate the requirement for mineral N fertilizer according to earlier studies (see also supplementary text S2).

Although the interventions herein provide sizable potentials to mitigate N fertilizer use, their implementation can introduce new risks and externalities. In line with earlier studies, we assume that long-term average yields are not affected by switching from mineral to organic N sources [32]. However, a meta-analysis found that crop yields in organic farming typically have a higher inter-annual variability due to weather fluctuations and their impact on soil biological processes [71]. Also the biomass production and N fixation by green manures is subject to weather fluctuations and extremes, which affects the amount of N provided in adverse years. In part this can be addressed through plant mixtures that increase green manures' climate resilience [22, 72]. Similarly, livestock manure supply is subject to feed availability and animal health, which may be impaired by biotic and abiotic shocks [73] and improved manure cycling to cropland may in turn pose a serious risk of soil contamination with heavy metals [74]. While these risks are well known and management options exist, the adoption of the interventions at scales will have to be paired with appropriate foresight and risk management.

On the other hand, various co-benefits can be expected from the adoption of the interventions, starting from substantial energy and GHG emission savings for fertilizer production [22], mitigation of air pollution [66], carbon sequestration potential [9], prevention of eutrophication from capping fertilizer surplus [27], and public health improvements from upgrading wastewater infrastructure [75]. Furthermore, the diversification of N sources can provide supply chain resilience if the above-mentioned risks are managed properly.

While our study presents for the first time an investigation of N fertilizer mitigation potentials through bundles of interventions globally, the individual interventions have been known and studied regionally or conceptually [22, 32, 47, 66, 75]. The fact that such interventions are still not systematically adopted underpins the importance of economic, technological, social, and institutional barriers [76–80] that will require further scrutiny. Globally consistent and spatially disaggregated socioeconomic data may allow for a spatially explicit analysis of barriers and enablers but are thus far lacking sufficient detail. In this context, our analyses provide a first indication of potential hotspots for

N fertilizer use mitigation and the requirement for intervention bundling that may incentivize further research into regionally and locally specific barriers and enablers for adoption.

A range of local and regional studies provide exemplary insights into factors that can facilitate the adoption of such interventions or may hamper it. For example [81], found that the adoption of sustainable fertilizer management in China could be facilitated through the establishment of demonstration networks and targeted government subsidies for required machinery, while public-private partnerships enabled the provision of tailored fertilizer formulations. For cover crops in the US mid-west, besides market opportunities also cultural norms and the option space farmers are aware of pose barriers to adoption, indicating that capacity building and targeted extension services are key elements in the upscaling of such practices [82]. In regions that allow for the cultivation of several cash crops annually, green manures may compete with the production of another crop and be limited by economic constraints such as high labor requirement and the need for complementary agronomic inputs [83]. Also animal manure is in some regions subject to competitive uses, being sourced as a household fuel or for other non-agricultural purposes in India among others [84]. Nitrogen recovery from wastewater streams beyond the lower threshold herein finally requires substantial investments in upgrading existing infrastructures [62] or building new ones where they lack so far [85]. While economic support and incentives as well as knowledge transfer emerge as common themes, the local experiences highlight that efforts to promote the adoption of interventions for sustainable N management will need to be tailored towards the particular local context to ensure that the specific challenges faced are addressed [86]. Moreover, scaling and implementation of novel technologies such as nutrient cycling from waste streams commonly requires cross-sectoral integration of various actors (e.g. policymakers, private sector, and potential users) within the agri-food system and beyond, which is often posing the most substantial barrier [87, 88].

Data availability statement

All data that support the findings of this study are included within the article (and any supplementary files). Derived data are available upon reasonable request from the authors.

Conflict of interest

Authors declare that they have no competing interests. Views expressed in this work are those of the authors and not necessarily those of The Nature Conservancy.

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References

[1] Erisman J W, Sutton M A, Galloway J, Klimont Z and Winiwarter W 2008 How a century of ammonia synthesis changed the world *Nat. Geosci.* **1** 636–9
[2] Diaz R J and Rosenberg R 2008 Spreading dead zones and consequences for marine ecosystems *Science* **321** 926–9
[3] Hellweger F L, Martin R M, Eigemann F, Smith D J, Dick G J and Wilhelm S W 2022 Models predict planned phosphorus load reduction will make Lake Erie more toxic *Science* **376** 1001–5
[4] Spiertz J H J 2010 Nitrogen, sustainable agriculture and food security. A review *Agron. Sustain. Dev.* **30** 43–55
[5] Gross M 2022 Global food security hit by war *Curr. Biol.* **32** R341–3
[6] Pörtner L M, Lambrecht N, Springmann M, Bodirsky B L, Gaupp F, Freund F, Lotze-Campen H and Gabrysch S 2022 We need a food system transformation—in the face of the Russia-Ukraine war, now more than ever *One Earth* **5** 470–2
[7] Pimentel D, Hurd L E, Bellotti A C, Forster M J, Oka I N, Sholes O D and Whitman R J 1973 Food production and the energy crisis *Science* **182** 443–9
[8] Gomiero T 2016 Soil degradation, land scarcity and food security: reviewing a complex challenge *Sustainability* **8** 281
[9] Smith P, Calvin K, Nkem J, Campbell D, Cherubini F and Grassi G 2020 Which practices co-deliver food security, climate change mitigation and adaptation, and combat land degradation and desertification? *Glob. Change Biol.* **26** 1532–75
[10] Steffen W *et al* 2015 Planetary boundaries: guiding human development on a changing planet *Science* **347** 1259855
[11] Smil V 1999 Nitrogen in crop production: an account of global flows *Glob. Biogeochem. Cycles* **13** 647–62
[12] Schebesta H and Candel J J L 2020 Game-changing potential of the EU's Farm to Fork Strategy *Nat. Food* **1** 586–8
[13] White House 2022 Fact sheet: president Biden to galvanize global action to strengthen energy-security and tackle the climate crisis through the major economies forum on energy and climate *The White House* (available at: www.whitehouse.gov/briefing-room/statements-releases/2022/06/17/fact-sheet-president-biden-to-galvanize-global-action-to-strengthen-energy-security-and-tackle-the-climate-crisis-through-the-major-economies-forum-on-energy-and-climate/) (Accessed 22 August 2022)
[14] Wang X, Xu M, Lin B, Bodirsky B L, Xuan J and Dietrich J P 2022 Reforming China's fertilizer policies: implications for nitrogen pollution reduction and food security *Sustain. Sci.* **18** 407–20
[15] Yu C *et al* 2019 Managing nitrogen to restore water quality in China *Nature* **567** 516–20
[16] Raghuram N, Sutton M A, Jeffery R, Ramachandran R and Adhya T K 2021 From South Asia to the world: embracing the challenge of global sustainable nitrogen management *One Earth* **4** 22–27
[17] Sutton M A, Howard C M, Kanter D R, Lassaletta L, Móring A, Raghuram N and Read N 2021 The nitrogen decade: mobilizing global action on nitrogen to 2030 and beyond *One Earth* **4** 10–14

[18] You L, Ros G H, Chen Y, Shao Q, Young M D, Zhang F and de Vries W 2023 Global mean nitrogen recovery efficiency in croplands can be enhanced by optimal nutrient, crop and soil management practices *Nat. Commun.* **14** 5747
[19] Spiegal S *et al* 2020 Manuresheds: advancing nutrient recycling in US agriculture *Agric. Syst.* **182** 102813
[20] Udvardi M *et al* 2021 A research road map for responsible use of agricultural nitrogen *Front. Sustain. Food Syst.* **5** 660155
[21] Barbieri P, Pellerin S, Seufert V and Nesme T 2019 Changes in crop rotations would impact food production in an organically farmed world *Nat. Sustain.* **2** 378–85
[22] Kaye J P and Quemada M 2017 Using cover crops to mitigate and adapt to climate change. A review *Agron. Sustain. Dev.* **37** 4
[23] Alghamdi R S and Cihacek L 2022 Do post-harvest crop residues in no-till systems provide for nitrogen needs of following crops? *Agron. J.* **114** 835–52
[24] Cassman K G, Dobermann A R and Walters D T 2002 Agroecosystems, nitrogen-use efficiency, and nitrogen management *AMBIO: A J. Hum. Environ.* **31** 10
[25] Trimmer J T and Guest J S 2018 Recirculation of human-derived nutrients from cities to agriculture across six continents *Nat. Sustain.* **1** 427–35
[26] Sutton M A (UNEP) ed 2013 *Our Nutrient World: The Challenge to Produce More Food and Energy with Less Pollution* Global Overview On Nutrient Management (Centre for Ecology & Hydrology)
[27] Gu B *et al* 2023 Cost-effective mitigation of nitrogen pollution from global croplands *Nature* **613** 77–84
[28] Barbieri P, Pellerin S, Seufert V, Smith L, Ramankutty N and Nesme T 2021 Global option space for organic agriculture is delimited by nitrogen availability *Nat. Food* **2** 363–72
[29] Chang J, Havlík P, Leclère D, de Vries W, Valin H, Deppermann A, Hasegawa T and Obersteiner M 2021 Reconciling regional nitrogen boundaries with global food security *Nat. Food* **2** 700–11
[30] Billen G, Aguilera E, Einarsson R, Garnier J, Gingrich S, Grizzetti B, Lassaletta L, Le Noë J and Sanz-Cobena A 2021 Reshaping the European agro-food system and closing its nitrogen cycle: the potential of combining dietary change, agroecology, and circularity *One Earth* **4** 839–50
[31] Bodirsky B L, Popp A, Lotze-Campen H, Dietrich J P, Rolinski S and Weindl I 2014 Reactive nitrogen requirements to feed the world in 2050 and potential to mitigate nitrogen pollution *Nat. Commun.* **5** 3858
[32] Li H, Feng W, He X, Zhu P, Gao H, Sun N and XU M-G 2017 Chemical fertilizers could be completely replaced by manure to maintain high maize yield and soil organic carbon (SOC) when SOC reaches a threshold in the Northeast China Plain *J. Integr. Agric.* **16** 937–46
[33] Foley J A, Ramankutty N, Brauman K A, Cassidy E S, Gerber J S and Johnston M 2011 Solutions for a cultivated planet *Nature* **478** 337–42
[34] Mueller N D, Gerber J S, Johnston M, Ray D K, Ramankutty N and Foley J A 2012 Closing yield gaps through nutrient and water management *Nature* **490** 254–7
[35] FAO 2022 FAO statistical database (available at: www.fao.org/faostat/en/#home) (Accessed 7 June 2022)
[36] International Food Policy Research Institute 2020 *Global Spatially-Disaggregated Crop Production Statistics Data for 2010 Version 2.0* (Harvard Dataverse) (https://doi.org/10.7910/DVN/PRFF8V)
[37] Yu Q, You L, Wood-Sichra U, Ru Y, Joglekar A K B, Fritz S, Xiong W, Lu M, Wu W and Yang P 2020 A cultivated planet in 2010—part 2: the global gridded agricultural-production maps *Earth Syst. Sci. Data* **12** 3545–72
[38] Zhang B, Tian H, Lu C, Dangal R S S, Yang J and Pan S 2017 Manure nitrogen production and application in cropland and rangeland during 1860–2014: a 5-minute gridded global data set for Earth system modeling. Supplement to: Zhang, B et al (2017): global manure nitrogen production and application in cropland during 1860–2014: a 5 arcmin gridded global dataset for Earth system modeling *Earth Syst. Sci. Data* **9** 667–78
[39] USDA 2019 Crop nutrient tool | USDA PLANTS (available at: https://web.archive.org/web/20221222155618/https://plantsorig.sc.egov.usda.gov/npk/databases/cropnutrientcontents-yieldunitchanges-oct-03.mdb) (Accessed 13 March 2019)
[40] Gassman P W, Williams J R, Benson V W, Izaurralde R C, Hauck L M and Jones C A 2004 Historical development and applications of the EPIC and APEX models *2004 ASAE Annual Meeting. American Society of Agricultural and Biological Engineers* p 1 (available at: www.card.iastate.edu/products/publications/synopsis/?p=763)
[41] Williams J R 1990 The erosion-productivity impact calculator (EPIC) model: a case history *Phil. Trans. R. Soc.* B **329** 421–8
[42] Köble R 2014 The global nitrous oxide calculator—GNOC—online tool manual v1.2.4 (available at: http://gnoc.jrc.ec.europa.eu/documentation/The_Global_Nitrous_Oxide_Calculator_User_Manual_version_1_2_4.pdf)
[43] Carlson K M *et al* 2017 Greenhouse gas emissions intensity of global croplands *Nat. Clim. Change* **7** 63–68
[44] Jamali M, Bakhshandeh E, Yaghoubi Khanghahi M and Crecchio C 2021 Metadata analysis to evaluate environmental impacts of wheat residues burning on soil quality in developing and developed countries *Sustainability* **13** 6356
[45] Quan Z, Zhang X, Fang Y and Davidson E A 2021 Different quantification approaches for nitrogen use efficiency lead to divergent estimates with varying advantages *Nat. Food* **2** 241–5
[46] Jung M, Boucher T M, Wood S A, Folberth C, Wironen M, Thornton P, Bossio D and Obersteiner M 2024 A global clustering of terrestrial food production systems *PLoS One* **19** e0296846
[47] Hijbeek R, ten Berge H F M, Whitmore A P, Barkusky D, Schröder J J and van Ittersum M K 2018 Nitrogen fertiliser replacement values for organic amendments appear to increase with N application rates *Nutr. Cycl. Agroecosyst.* **110** 105–15
[48] Ruis S J, Blanco-Canqui H, Creech C F, Koehler-Cole K, Elmore R W and Francis C A 2019 Cover crop biomass production in temperate agroecozones *Agron. J.* **111** 1535–51
[49] USDA N 2009 *Crop Nutrient Tool: Nutrient Content of Crops* (US Department of Agriculture, Natural Resource Conservation Service)
[50] Peoples M B *et al* 2017 Soil mineral nitrogen benefits derived from legumes and comparisons of the apparent recovery of legume or fertiliser nitrogen by wheat *Soil Res.* **55** 600–15
[51] Ladha J K, Peoples M B, Reddy P M, Biswas J C, Bennett A, Jat M L and Krupnik T J 2022 Biological nitrogen fixation and prospects for ecological intensification in cereal-based cropping systems *Field Crops Res.* **283** 108541
[52] Jian J, Du X and Stewart R D 2020 A database for global soil health assessment *Sci. Data* **7** 16
[53] Vendig I, Guzman A, De La Cerda G, Esquivel K, Mayer A C, Ponisio L and Bowles T M 2023 Quantifying direct yield benefits of soil carbon increases from cover cropping *Nat. Sustain.* **6** 1125–34
[54] Beck H E, Zimmermann N E, McVicar T R, Vergopolan N, Berg A and Wood E F 2018 Present and future Köppen-Geiger climate classification maps at 1-km resolution *Sci. Data* **5** 180214
[55] Trabucco A and Zomer R 2019 *Global Aridity Index and Potential Evapotranspiration (ET0) Climate Database V2* (https://doi.org/10.6084/m9.figshare.7504448.v3)
[56] Batjes N H 2011 Global distribution of soil phosphorus retention potential (ISRIC-World Soil Information)
[57] Zhang J, Beusen A H W, Van Apeldoorn D F, Mogollón J M, Yu C and Bouwman A F 2017 Spatiotemporal dynamics of

soil phosphorus and crop uptake in global cropland during the 20th century *Biogeosciences* **14** 2055–68
[58] Giller K E and Cadisch G 1995 Future benefits from biological nitrogen fixation: an ecological approach to agriculture *Plant Soil* **174** 255–77
[59] Cardoso I and Kuyper T 2006 Mycorrhizas and tropical soil fertility *Agric. Ecosyst. Environ.* **116** 72–84
[60] Gustavsson J, Cederberg C and Sonesson U 2011 Global food losses and food waste FAO: 38
[61] Schiavina M, Freire S and MacManus K 2022 GHS-POP R2022A—GHS population grid multitemporal (1975–2030) (https://doi.org/10.2905/D6D86A90-4351-4508-99C1-CB074B022C4A) (Accessed 11 July 2022)
[62] Gottardo Morandi C, Wasielewski S, Mouarkech K, Minke R and Steinmetz H 2018 Impact of new sanitation technologies upon conventional wastewater infrastructures *Urban Water J.* **15** 526–33
[63] Goyette J-O, Bennett E M and Maranger R 2018 Low buffering capacity and slow recovery of anthropogenic phosphorus pollution in watersheds *Nat. Geosci.* **11** 921–5
[64] Miao Y, Stewart B A and Zhang F 2011 Long-term experiments for sustainable nutrient management in China. A review *Agron. Sustain. Dev.* **31** 397–414
[65] Zhang C, Liu S, Wu S, Jin S, Reis S, Liu H and Gu B 2019 Rebuilding the linkage between livestock and cropland to mitigate agricultural pollution in China *Resour. Conserv. Recycl.* **144** 65–73
[66] Ma R, Li K, Guo Y, Zhang B, Zhao X, Linder S, Guan C, Chen G, Gan Y and Meng J 2021 Mitigation potential of global ammonia emissions and related health impacts in the trade network *Nat. Commun.* **12** 6308
[67] Cai Y and Akiyama H 2016 Nitrogen loss factors of nitrogen trace gas emissions and leaching from excreta patches in grassland ecosystems: a summary of available data *Sci. Total Environ.* **572** 185–95
[68] Muller A *et al* 2017 Strategies for feeding the world more sustainably with organic agriculture *Nat. Commun.* **8** 1290
[69] Brooker R W *et al* 2015 Improving intercropping: a synthesis of research in agronomy, plant physiology and ecology *New Phytol.* **206** 107–17
[70] Kummu M, de Moel H, Porkka M, Siebert S, Varis O and Ward P J 2012 Lost food, wasted resources: global food supply chain losses and their impacts on freshwater, cropland, and fertiliser use *Sci. Total Environ.* **438** 477–89
[71] Knapp S and van der Heijden M G A 2018 A global meta-analysis of yield stability in organic and conservation agriculture *Nat. Commun.* **9** 3632
[72] Lawson A, Cogger C, Bary A and Fortuna A-M 2015 Influence of seeding ratio, planting date, and termination date on rye-hairy vetch cover crop mixture performance under organic management *PLoS One* **10** e0129597
[73] Godde C M, Mason-D'Croz D, Mayberry D E, Thornton P K and Herrero M 2021 Impacts of climate change on the livestock food supply chain; a review of the evidence *Glob. Food Secur.* **28** 100488
[74] Xu Y, Li J, Zhang X, Wang L, Xu X, Xu L, Gong H, Xie H and Li F 2019 Data integration analysis: heavy metal pollution in China's large-scale cattle rearing and reduction potential in manure utilization *J. Cleaner Prod.* **232** 308–17
[75] Trimmer J T, Cusick R D and Guest J S 2017 Amplifying progress toward multiple development goals through resource recovery from sanitation *Environ. Sci. Technol.* **51** 10765–76
[76] FAO 2022 *The Future of Food and Agriculture—Drivers and Triggers for Transformation* (FAO) (https://doi.org/10.4060/cc0959en)
[77] FAO, UNDP, and UNEP 2021 *A Multi-Billion-Dollar Opportunity—Repurposing Agricultural Support to Transform Food Systems* (FAO, UNDP, and UNEP) (https://doi.org/10.4060/cb6562en)
[78] Garrett R, Ryschawy J, Bell L, Cortner O, Ferreira J and Garik A V 2020 Drivers of decoupling and recoupling of crop and livestock systems at farm and territorial scales *Ecol. Soc.* **25** 24
[79] Liang X, Lam S K, Zhang X, Oenema O and Chen D 2021 Pursuing sustainable nitrogen management following the "5 Ps" principles: production, people, planet, policy and partnerships *Glob. Environ. Change* **70** 102346
[80] Reitzel K *et al* 2019 New training to meet the global phosphorus challenge *Environ. Sci. Technol.* **53** 8479–81
[81] Cui Z *et al* 2018 Pursuing sustainable productivity with millions of smallholder farmers *Nature* **555** 363–6
[82] Roesch-McNally G E, Basche A D, Arbuckle J G, Tyndall J C, Miguez F E, Bowman T and Clay R 2018 The trouble with cover crops: farmers' experiences with overcoming barriers to adoption *Renew. Agric. Food Syst.* **33** 322–33
[83] Falconnier G N, Cardinael R, Corbeels M, Baudron F, Chivenge P and Couëdel A 2023 The input reduction principle of agroecology is wrong when it comes to mineral fertilizer use in sub-Saharan Africa *Outlook Agric.* **52** 311–26
[84] Ravindra K, Kaur-Sidhu M, Mor S and John S 2019 Trend in household energy consumption pattern in India: a case study on the influence of socio-cultural factors for the choice of clean fuel use *J. Cleaner Prod.* **213** 1024–34
[85] Berendes D M, Sumner T A and Brown J M 2017 Safely managed sanitation for all means fecal sludge management for at least 1.8 billion people in low and middle income countries *Environ. Sci. Technol.* **51** 3074–83
[86] Piñeiro V *et al* 2020 A scoping review on incentives for adoption of sustainable agricultural practices and their outcomes *Nat. Sustain.* **3** 809–20
[87] Duquennoi C and Martinez J 2022 European Union's policymaking on sustainable waste management and circularity in agroecosystems: the potential for innovative interactions between science and decision-making *Front. Sustain. Food Syst.* **6** 937802
[88] Ross J Z and Omelon S 2018 Canada: playing catch-up on phosphorus policy *Facets* **3** 642–64