Original Research

Minimizing nitrogen-related environmental harm while achieving food security in China

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Abstract

Elevated nitrogen (N) fertilizer inputs have been pivotal in sustaining a growing global population, but they have also caused significant environmental challenges, primarily driven by reactive nitrogen (Nr) emissions to air and water on a global scale—a situation particularly pronounced in China. To better understand the improvements to N management required to reduce impacts in China, a long-term N budget over the period 1961–2018 was compiled, historic changes in N inputs, crop uptakes and losses quantified, and compared with the N inputs required to satisfy food security and critical N inputs necessary to protect the environment, considering acceptable losses of ammonia emissions to air and nitrate leaching to water. The N inputs required to satisfy the food demand of China were 14, 21, 27, and 29 Tg N yr^{-1} in 1961, 1980, 2000, and 2018, respectively, while the actual N inputs were 4, 25, 39, and 48 Tg N yr^{-1}, respectively, exceeding the required input after 1980. Since 2000, the actual N inputs have also exceeded the critical N inputs in view of ammonia emissions to air (35 Tg N yr^{-1}), and nitrate leaching to groundwater (28 Tg N yr^{-1}) that are needed to protect biodiversity and human health. It is demonstrated that reducing N input from 48 to 31 Tg N yr^{-1}, thereby reconciling food production with environmental safety, can be achieved through enhanced manure recycling, balanced N fertilization, and integrated soil-crop system management. Such a 'win-win' strategy would improve air and water quality, protect the environment and human health, and enhance resource-use efficiency while reducing N fertilizer use and Nr losses.
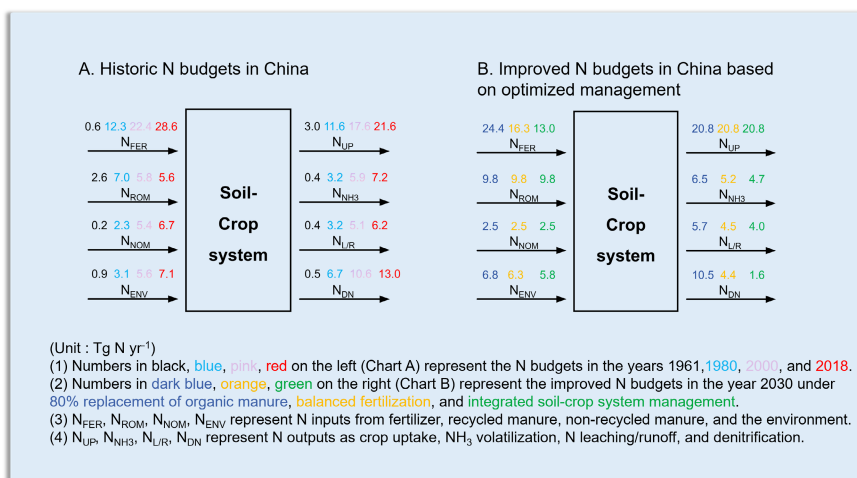
Highlights

- A long-term nitrogen (N) budget over the period 1961–2018 was systematically evaluated in China's agriculture.
- Actual N inputs tripled from 1961 to 2018, and surpassed the required N in 1980 (21 Tg N yr^{-1}) and critical N inputs in 2000 (28–35 Tg N yr^{-1}).
- Chemical fertilizer became the dominant source of N input compared with organic manure and environmental sources after 1980.
- N inputs could be reduced from 48 to 31 Tg N yr^{-1} to reconcile food demand with environmental safety through integrated soil-crop system management.

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Graphical abstract

A. Historic N budgets in China

B. Improved N budgets in China based on optimized management

(Unit : Tg N yr⁻¹)
(1) Numbers in black, blue, pink, red on the left (Chart A) represent the N budgets in the years 1961, 1980, 2000, and 2018.
(2) Numbers in dark blue, orange, green on the right (Chart B) represent the improved N budgets in the year 2030 under 80% replacement of organic manure, balanced fertilization, and integrated soil-crop system management.
(3) N_{FER}, N_{ROM}, N_{NOM}, N_{ENV} represent N inputs from fertilizer, recycled manure, non-recycled manure, and the environment.
(4) N_{UP}, N_{NH3}, $N_{L/R}$, N_{DN} represent N outputs as crop uptake, NH_3 volatilization, N leaching/runoff, and denitrification.

Introduction

The nitrogen (N) paradox—the stark contrast between N's role as a cornerstone of global food security and its emergence as a primary driver of environmental degradation—poses a critical challenge for sustainable development. Since the Haber-Bosch process revolutionized N fixation over a century ago, synthetic fertilizers have enabled a 300% increase in global crop yields, sustaining 4.5 billion people[1]. Yet, this agricultural triumph has come at a devastating cost: reactive nitrogen (Nr) emissions now account for 78% of global N pollution, fueling eutrophication, biodiversity loss, and greenhouse gas emissions equivalent to 1.5 gigatons of CO_2-equivalent annually[2–4]. This duality necessitates a paradigm shift in N governance. Feeding 10 billion people by 2050 within planetary boundaries requires simultaneously addressing two imperatives: (1) nutritional imperatives: Scaling plant-based diets and reducing food waste by 50% could cut agricultural N demand by 35%[5]; and (2) environmental imperatives: Implementing precision agriculture (e.g., variable-rate fertilization) and closing N loops through manure recycling could reduce losses by 40%–60% without compromising yields[6–8]. Thus, resolving the N paradox demands integrating food systems transformation with circular economy principles—a challenge as complex as balancing the Haber-Bosch process's dual legacy of life and death.

Nitrogen pollution risk is especially serious in China, where grain production has increased more than three times since 1960 in response to an approximately 100-fold increase in the application of N (NPK) fertilizers[9], but at the cost of very large emissions of Nr to the environment due to its overuse in intensive agricultural regions[10,11]. Excessive N application has caused significant acidification of agricultural soils[12], eutrophication of surface waters[13], and impacts on terrestrial ecosystems due to enhanced ammonia emissions and resulting N deposition to land[14–16]. In addition, excessive Nr poses significant risks to human health through two distinct pathways: (1) ammonia volatilization contributes to secondary particulate pollution; while (2) enhanced nitrate leaching contaminates groundwater and surface water resources that serve as drinking water supplies[17]. The potential health effects of high nitrate levels include reproductive problems, methaemoglobinaemia ('blue-baby' syndrome), and cancer of the digestive tract[17,18]. Increased levels of particulate matter (PM) can damage lung function. A recent study by Pozzer et al.[19] showed that a 50% reduction

in agricultural ammonia emissions could reduce the global mortality attributable to air pollution by ca 250,000 people per year, amounting to reductions in mortality of 30%, 19%, 8%, and 3% over North America, Europe, East Asia, and South Asia, respectively. The various impacts also impose large societal and economic costs, and the benefits of measures to reduce Nr losses are likely much higher than the abatement costs, as has been shown recently for ammonia[20,21]. China represents not merely a nation grappling with severe N challenges but also a globally significant 'hyper-accelerated case study' that provides critical predictive insights for developing countries navigating similar trajectories[22,23].

The intensifying demand for food security in China, driven by population growth and dietary shifts, necessitates sustainable intensification of agricultural systems. While the 'Zero Increase Action Plan' (2015) has curbed chemical N fertilizer use[24], its reliance on partial NUE improvements overlooks the critical knowledge gap in harmonizing fertilizer and manure systems. Zhang et al.[7] projected a 2050 N demand of 27 Tg N yr⁻¹ for China under 60% NUE, but their framework ignored manure's role in closing yield gaps while mitigating N losses. The present analysis quantifies, for the first time, the dual thresholds of N inputs for food production and critical N loss limits (e.g., NH_3 volatilization and nitrate N leaching), revealing that current practices are fall shorts of these benchmarks. By establishing an integrated framework, policy blueprints are derived that reduce fertilizer dependency and increase organic manure recycling while sustaining yields—a prerequisite for aligning China's agricultural targets with the Sustainable Development Goals (SDGs). However, no distinction was made between fertilizer and manure inputs, and incomplete manure recycling was not considered. Insight into combined fertilizer and manure management is crucial for assessing required changes in agricultural practices at the national scale. To this end, the necessary N inputs required to meet food production demands in China were first evaluated, comparing these requirements against critical N inputs and critical N loss thresholds to minimize environmental pollution. Variations in N source types were further distinguished, and historical trends in N inputs and losses spanning 1961−2018 were analyzed. Building on this analysis, actionable strategies for fertilizer and manure management were identified that could simultaneously fulfil food production targets and environmental protection goals, and the associated costs and benefits quantified.

To address the need for strategic depth, the N budget was redefined as a diagnostic tool for evaluating historical phases of China's agricultural development. By quantifying national-scale N inputs in 1961, 1980, 2000, and 2018, systemic imbalances in NUE across four developmental epochs were diagnosed. They represent: (1) the early stage of the New China (1961, manure dominant); (2) the post-reform era (1980, wide use of fertilizer N); (3) the moderate prosperity phase (2000, overuse of fertilizer N); and (4) the primary developed stage (2018, peak N surplus). This diagnostic approach revealed critical N thresholds through a multi-objective assessment framework, including the environmental, economic, and social pillars. By aligning these metrics with China's SDG targets (SDG 2 and 13), a novel sustainability framework that balances food security with ecological resilience was established. As a first step, the fate of N fertilizer in agroecosystems was summarized by data-mining and national-scale N budgets calculated for major crop systems in China for the years 1961, 1980, 2000, and 2018, which serves as an important tool to quantify the N surplus and the need for sustainable N management[25–26]. The results in terms of N inputs, N uptake, and N losses to air and water at the four time-points are then discussed in the context of the required N inputs for food demand and the critical N inputs and losses for minimizing environmental pollution. Critical N inputs were estimated based on acceptable thresholds for ammonia emissions to air and nitrate leaching to groundwater, using critical N loads for biodiversity protection and critical limits for groundwater quality (drinking water) to avoid impact on human health[4,27]. The potential for management practices to achieve sustainable development by minimizing the gap between actual and required/critical N inputs were quantified. Finally, the direct and indirect economic benefits of saving fertilizer N and reducing Nr emissions to the environment were analyzed.

Material and methods

Soil N budgets for croplands

Cropland N inputs comprise fertilizer, manure, and environmental sources (including N deposition, irrigation, and biological N fixation), while outputs comprise crop uptake, NH_3 emissions, leaching and runoff, and denitrification-nitrification processes (including N_2O, NO_x and N_2). N budgets were calculated using the mass balance approach[28]:

$$\sum_{h=1}^{m} N_{IN_h} = \sum_{g=1}^{n} N_{OUT_g} + \sum_{k=1}^{p} ACC_k \qquad (1)$$

$$\sum_{h=1}^{m} N_{IN_h} = N_{FER} + N_{OM} + N_{ENV} \qquad (2)$$

$$\sum_{g=1}^{m} N_{OUT_g} = N_{UP} + N_{SUR} \qquad (3)$$

$$N_{OM} = N_{ROM} + N_{NOM} \qquad (4)$$

$$N_{ENV} = N_{DEP} + N_{FIX} + N_{IRR} \qquad (5)$$

where, IN_h and N_{OUT_g} represent N input and N output; ACC_k represents N accumulation; N_{IN} denotes N input into the system; N_{UP} means N uptake by crops; N_{FER} and N_{OM} denote N input from chemical fertilizer and manure, respectively, where N_{OM} comprises two components: recycled manure (N_{ROM}) and non-recycled manure (N_{NOM}) (For N_{NOM}, it is assumed to be applied to fallow land with half is lost directly to water and the remaining half lost through NH_3 volatilization and denitrification—each process accounting for approximately half of this portion)[29,30]; and N_{ENV} denotes N from the environment, including N deposition (N_{DEP}), biological fixation (N_{FIX}) and irrigation (N_{IRR}).

Neglecting small changes in soil N pools over the long term, N input and output budgets in China between 1961 and 2018 were systematically evaluated. The N surplus (N_{SUR}) was calculated as the difference between N input and N uptake (N_{UP}) by harvested crops:

$$N_{sur} = N_{IN} - N_{UP} \qquad (6)$$

where, N_{SUR} (N surplus, defined as the difference between N inputs and N uptake) comprises NH_3 volatilization (N_{NH3}), and N_{DN} (N_2O, NO and N_2 emissions from denitrification-nitrification) and NO_3^- to water (leaching and runoff).

Nitrogen fertilizer use (N_{FER}) and manure N (N_{OM}) in China were estimated using data from the *China National Statistics Yearbook* and EARTHSTAT[31,32]. N_{DEP} was obtained from published data on N deposition in China as described in Liu et al.[14], Xu et al.[33], and Wen et al.[34]. N_{IRR} was calculated according to relationships between N_{IRR} and years, derived from published data[35]. The N uptake (N_{UP}) was calculated in three steps: (1) N_{UP} for 16 main crops (i.e. wheat, rice, maize, soybean, barley, sorghum, millet, rapeseed, groundnut, sunflower, sugarcane, potato, cassava, oilpalm, rye, and sugarbeet) (Supplementary Fig. S1) was calculated by the sum of N_{UP} of each crop's N_{UP}, which is multiplied by national crop production and N content; (2) N_{UP} for other crops (i.e. vegetables) was estimated using the same step as for the 16 main crops; and (3) the total N_{UP} for all crops in China was obtained by summing the values from the previous steps. To assess the effects of changing N inputs on N losses to air and water from China's cropping systems, a literature survey was conducted to assess the fate of N fertilizer and its contribution of NH_3 emission, N_2O emission, N leaching, N runoff, and N uptake (Supplementary Figs S2 & S3c as well as the references in the Supporting Information). Crop production data were derived from the Chinese National Statistics Yearbook[32], while the crop N_{up} (including N content), and N loss fractions were obtained from a literature survey (Supplementary Fig. S2).

Required N input

Required N input (N_{req}) denotes the necessary N input to supply the normal human protein-N requirement under a targeted N use efficiency[27]. N_{req} was calculated according to:

$$N_{req} = N_{int,req} \times T_{pop} / NUE_f \qquad (7)$$

where, N_{req} denotes the required N input; $N_{int,req}$ denotes the required annual human body N intake per year; T_{pop} denotes the national population (billion), and NUE_f denotes the national average N use efficiency (N consumption/N input) in the whole food chain (%).

$N_{int,req}$ was defined as 3.5 kg N capita^{-1} yr^{-1}, based on Chinese national dietary guidelines, slightly higher than the value of 3.0 kg N capita^{-1} yr^{-1} proposed by Westhoek et al.[36] and De Vries et al.[27]. T_{pop} in China was 0.66, 0.99, 1.27, and 1.40 billion in 1961, 1980, 2000, and 2018, respectively. The value for NUE_f in the food system was taken from Ma et al.[37] as the average value of 0.16 (0.09–0.20). The fixed NUE_f value of 0.16 (achieved in 1980) was used as a theoretically achievable and nutritionally balanced benchmark, reflecting a state where plant and animal protein intake are in equilibrium. This value provides a reference for calculating the minimum critical N inputs needed to sustain both food production and environmental sustainability in China. While actual NUE_f has declined over time, this benchmark remains relevant for evaluating progress toward improved N management practices.

Critical N losses to water

Critical N losses to water were calculated from the N surplus leached to groundwater or surface water (runoff)[25]. Critical N losses to
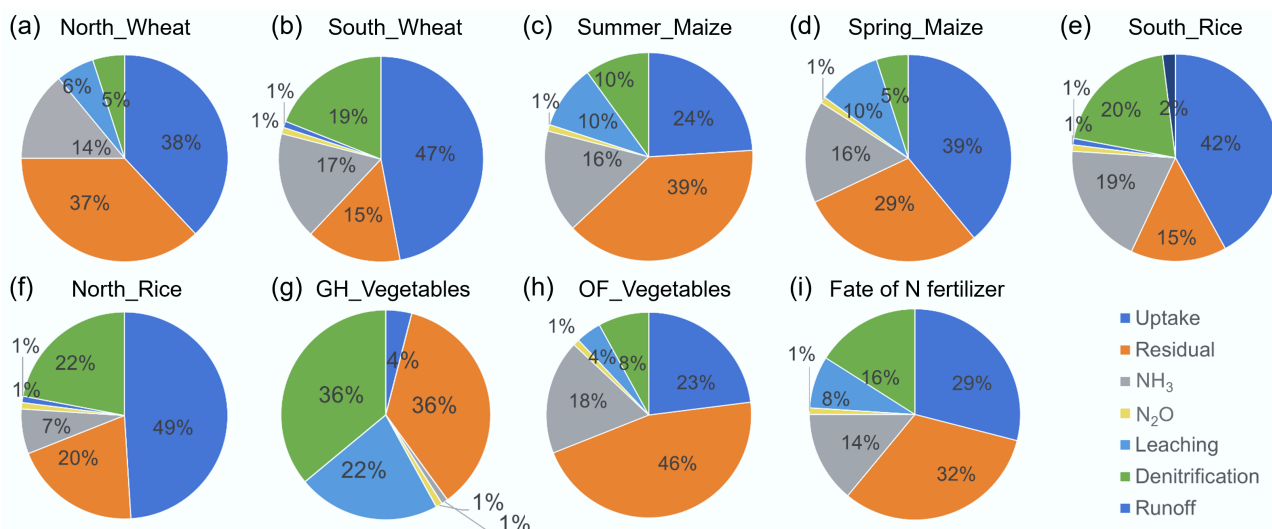
Fig. 1 Fate of fertilizer N in different crops. (**a**) North_Wheat. (**b**) South_Wheat. (**c**) Summer_Maize. (**d**) Spring_Maize. (**e**) South_Rice. (**f**) North_Rice. (**g**) Greenhouse(GH)_Vegetables. (**h**) Openfield(OF)_Vegetables and the overall fate of N fertilizer in China's crop production (**i**). Data were summarized from literature and introduced for details in the Material and methods section as well as the supplementary information.

groundwater per hectare of land was calculated by multiplying a critical NO_3-N concentration for groundwater by an annual averaged water leaching rate, which is a fraction of precipitation plus irrigation, as shown in the following equation:

$$NO_3\text{-}N_{lea,ha}(crit) = NO_3\text{-}N_{dw}(crit) \times W_{AL} \times 10^{-2} \quad (8)$$

where, NO_3-$N_{lea,ha}$ (crit) denotes the critical N loss per hectare of land (kg N ha^{-1} yr^{-1}); NO_3-N_{dw} (crit) denotes the critical N concentration for drinking water (mg N L^{-1}), as defined by the WHO guideline; W_{AL} denotes the annual average leaching rate in China (mm yr^{-1}); 10^{-2} is a conversion coefficient from mg N L^{-1} × mm to kg N ha^{-1} yr^{-1}.

The critical N loss for the whole country was then calculated as:

$$NO_3\text{-}N_{lea,to}(crit) = NO_3\text{-}N_{lea,ha}(crit) \times Area_{crop} \times 0.001 \quad (9)$$

where, $Area_{crop}$ denotes the cropland area (Mha) in China; 0.001 is a coefficient to convert Million kg N yr^{-1} to Million tons N yr^{-1}, i.e. Tg N yr^{-1}.

11.3 mg N L^{-1} was used as NO_3-N_{dw} (crit) based on the WHO standard for the NO_3^- concentration in drinking water. Despite natural variations, this threshold is still appropriate as the present study specifically investigates the mitigation of human health risks associated with nitrate pollution from agricultural N fertilization, where groundwater contamination remains the primary exposure pathway. The national annual average leaching rate from croplands, W_{AL}, was calculated as the sum of precipitation and irrigation minus evapotranspiration for the different crops and integrated across the main agricultural regions in the country (Supplementary Table S1). Based on Huang[38] and Li & Huang[39], the average annual leaching rate was calculated to be 160 mm yr^{-1}. Multiplying the annual average leaching rate by the critical concentration of 11.3 mg N L^{-1} gives an annual average acceptable N loss (NO_3-N_{le} (crit)) to groundwater of 18.1 kg N ha^{-1} yr^{-1}. Multiplying this value by a cropland area of 125 Mha gives an acceptable annual loss of 2.3 Tg N yr^{-1}.

Critical NH$_3$ losses to air

As an N-indicator for eutrophication and related biodiversity (especially plant species diversity) decline in terrestrial ecosystems, the approach of De Vries & Schulte-Uebbing[40] was used, which took the area-weighted mean critical N load as a basis to assess critical NH$_3$ emission

rates. In this way, the critical NH$_3$ emission per hectare of land and the total emission for the country were derived according to De Vries et al.[25]:

$$NH_3\text{-}N_{em,ha}(crit) = N_{totdep}(crit) \times f_{NH3}/f(ag) \quad (10)$$

$$NH_3\text{-}N_{em,tot}(crit) = N_{em,ha}(crit) \times Area_{crop} \times 0.001 \quad (11)$$

where, NH$_3$-$N_{em,ha}$ (crit) denotes the critical NH$_3$ emission rate (kg N ha^{-1} yr^{-1}); N_{totdep} (crit) denotes the critical total N deposition (kg N ha^{-1} yr^{-1}); f_{NH3} denotes the fraction of NH$_3$ in total (NO_x + NH$_3$) deposition; f (ag) denotes the fraction of agricultural land in a region. According to Bobbink et al.[41], the N_{totdep} (crit) was set at a country average value of 10 kg N ha^{-1} yr^{-1}, a widely accepted conservative benchmark. For f_{NH3} and f (ag), national average values were used, i.e f_{NH3} as 0.54 according to Xu et al.[42] and f(ag) as 0.13 (125 Mha/960 Mha). The value of NH$_3$-N_{em} (crit) was thus calculated at 10 × 0.54/0.13 = 42 kg NH$_3$-N ha^{-1} yr^{-1}, which is equivalent to 5.2 Tg N yr^{-1}.

Critical N inputs

Critical N inputs related to either critical NO_3-N losses to groundwater or critical NH$_3$-N emissions to air were calculated from the above critical N losses divided by either the average N leaching fraction or NH$_3$-N emission fraction from croplands (relative to total N input):

$$N_{input}(crit) = N_{loss}(crit)/frN_{ave\_loss} \quad (12)$$

where, N_{input} (crit) denotes the critical N input (Tg N yr^{-1}); N_{loss} (crit) denotes the critical N loss (either N leaching or NH$_3$-N emission) rate (Tg N yr^{-1}); frN_{ave_loss} denotes the average N loss fraction (unitless), which is either the country average NO_3-N leaching fraction or NH$_3$-N emission fraction. The uncertainties in critical N inputs at the national scale were estimated at 25% assuming that uncertainties at the local scale average out. The 25% uncertainty estimate is based on previous studies: (1) Ma et al.[37] reported a 20%–30% uncertainty in national-scale N use efficiency (NUE) due to variability in agricultural practices and emission factors; (2) Oenema et al.[43] estimated > 30% uncertainty in the N budgets of the Netherlands, emphasizing spatial heterogeneity in soil properties and management. The limitations of current simple uncertainty estimate regarding spatial resolution and data granularity are also realized, and it is proposed that future work should refine uncertainties using high-resolution datasets.

Cost savings

The proposed changes in N management will lead to direct and indirect beneficial economic effects. Direct beneficial effects include the reduced costs of N fertilizer. Based on the costs of urea of about EUR€300–500 per ton, and an N content of 46%, N fertilizer costs were calculated to be EUR€0.6–1.0 per kg N, with an uncertainty of 50% to account for price differences for different types of fertilizers. The indirect benefits from reduced health costs and environmental improvement were based on the willingness to pay (WTP) in the EU27[44], corrected for the ratio of gross domestic product (GDP) *per capita* in China to that in EU27. The WTP for China was derived from the corrected values from the EU based on various N losses to air and water given in Supplementary Table S2.

Further methodological details and a complete list of references supporting the estimation of each N output are provided in Supplementary File 1 (methods) and Supplementary File 2 (references), respectively.

Results and discussion

Fate of N fertilizer in major crop systems

Based on data mining derived from literature, the fate of fertilizer N in China's cropland during the 2000s is summarized in Fig. 1. For every crop (Fig. 1a–h), N use efficiency (NUE; indicated by uptake in the figure) exhibited a significant variability (4%–49%), with cereals (wheat, maize, and rice) showing higher NUE than vegetables—specifically, openfield vegetables (23%) displayed markedly greater NUE than greenhouse vegetables (4%). Soil residual N was categorized into two groups: higher residual N (29%–46%) was observed in northern wheat, summer/spring maize, and greenhouse/openfield vegetables; lower residual N (15%–20%) occurred in southern wheat and rice in both southern and northern regions. Compared to N losses via N_2O and N runoff (0%–1%), NH_3 volatilization, N leaching, and total denitrification emerged as three primary N loss pathways across all crops. Exceptions included southern wheat (with only 1% leaching loss) and greenhouse vegetables (with 1% NH_3 loss). Nitrogen fate reflects integrated results of crop, soil, climate, and management[23]. Compared with cereals (i.e., rice, wheat, and maize) and openfield vegetables, the very low NUE (N_{UP}), and high N loss (esp. N leaching and denitrification) in greenhouse vegetables are mainly due to extremely high N and organic carbon inputs with frequent irrigation under shallow root systems, all of which restrict crop N uptake but favor N leaching and denitrification losses. In contrast, NH_3 volatilization is low in the greenhouse vegetables because of high soil moisture and a closed environment (greenhouse conditions).

At the national scale (Fig. 1i), the fate of fertilizer N was distributed as follows: 29% recovered by crops, 32% residual in soil, and 39% lost to the environment. Among these losses, total denitrification dominated (16%), followed by NH_3 volatilization (14%), N leaching (8%), and N_2O emissions/N runoff (both ≤ 1%). Collectively, these results underscore low NUE and elevated N loss rates, particularly in greenhouse vegetable production systems[45]. Based on these data-mining results, the historical N budgets in China are estimated below.

N budgets and food security

To gain an overall insight into long-term changes of China's N budgets, a long-term dataset of China's N inputs (from N fertilizer, manure, and the environment) and outputs (crop uptake, NH_3 emissions, leaching/runoff, gaseous losses from denitrification and nitrification) between 1961 and 2018 were systematically compiled based on a mass balance

approach[28]. The years 1961, 1980, 2000, and 2018 were chosen as four exemplar time points to show the changes in N flows (Fig. 2a–d), while annual changes of total N input and output are given in Supplementary Fig. S2a and b.

In 1961, China was experiencing a three-year period of economic hardship, facing serious societal issues (e.g., food scarcity) (Supplementary Fig. S4). Under this background, the use of N fertilizer (N_{FER}, 0.6 Tg N yr^{-1}) was very low, and major N inputs were recycled manure (2.6 Tg N yr^{-1}) plus atmospheric N deposition and N in irrigation water, together defined as N from the environment (N_{ENV}, 0.9 Tg N yr^{-1}); crop uptake (N_{UP}) was 3.0 Tg N yr^{-1} and the N surplus (N_{SUR}, 1.3 Tg N yr^{-1}) was relatively small (Fig. 2a). N losses from manure (N_{OM}, both recycled to croplands, i.e. N_{ROM} and not recycled, i.e. N_{NOM}) were included in the overall N budget analysis (Supplementaty Table S2). The overall N use efficiency (NUE, defined as the percentage of the total N input that is retained in the harvested parts of the crop) was 70%, with 30% of total N input being lost to the environment as NH_3, N leaching, runoff, and denitrification (N_{DN} including N_2O, NO, and N_2). However, cereal production was only 140 Mt yr^{-1}, equal to 235 kg capita^{-1} yr^{-1}, and daily protein intake was only 40 g capita^{-1} (Supplementary Fig. S2), less than the WHO recommendation (50 g capita^{-1} d^{-1})[46]. At that time, most of the population of China was suffering from hunger and was chronically under-nourished due to food shortages and famine. Food production was the top priority for the Chinese government at this time.

By 1980, China was in the post-reform era, achieving initial industrialization and self-sufficiency in agricultural products. The demand for food and economic development had greatly stimulated the widespread use of N fertilizer. N fertilizer (12.3 Tg N yr^{-1}) had become the largest N input to cropping systems, followed by N inputs from organic manure (9.3 Tg N yr^{-1}), and the environment (3.1 Tg N yr^{-1}). Meanwhile, national N uptake (11.6 Tg N yr^{-1}) was lower than the N surplus (13.1 Tg N yr^{-1}) (Fig. 2b), with NUE decreasing to 47%. Total cereal production, however, had increased to 321 Mt yr^{-1}, equalling 325 kg capita^{-1} yr^{-1}, and consequently the average daily dietary protein consumption had reached 53 g capita^{-1}, but with only 12% coming from animal products (Supplementary Fig. S4). Overall food demand was largely met, yet the proportion of animal protein in the diet remained below the WHO recommendation (20%–30%).

After 1980, rapid economic development and an increasingly wealthy lifestyle in China strongly stimulated agricultural expansion. The 'Meeting vegetable/fruit/animal-products in market (Cai-Lan-Zi)' project, launched by the government in 1988, greatly increased the production of vegetables, fruits, and animal products through a combination of increases in per unit-area yield and in the harvested areas of vegetables and fruits. This was accompanied by a massive increase in N fertilizer production and consumption[22].

In 2000, China's economy entered a moderate prosperity phase and gradually became the 'world factory', resulting in a large amount of N surplus in agriculture. Total N inputs to cropping systems grew to 39.2 Tg N yr^{-1}, with a very large input from fertilizers (22.4 Tg N yr^{-1}), followed by N from organic manure (11.2 Tg N yr^{-1}) and from deposition and irrigation (N_{ENV} of 5.6 Tg N yr^{-1}) (Fig. 2c; Supplementary Table S2). Crop uptake (17.6 Tg N yr^{-1}) accounted for 45% of the total N input. The N surplus (21.6 Tg N yr^{-1}) via NH_3 volatilization (5.9 Tg N yr^{-1}), N leaching and runoff (5.1 Tg N yr^{-1}), and denitrification (10.6 Tg N yr^{-1}) almost doubled compared with that in 1980. Total grain production increased to 462 Mt yr^{-1}, equal to 364 kg capita^{-1} yr^{-1} with an average daily dietary protein consumption of 75 g capita^{-1}, including 25% from animal products (Supplementary Fig. S4). Large losses
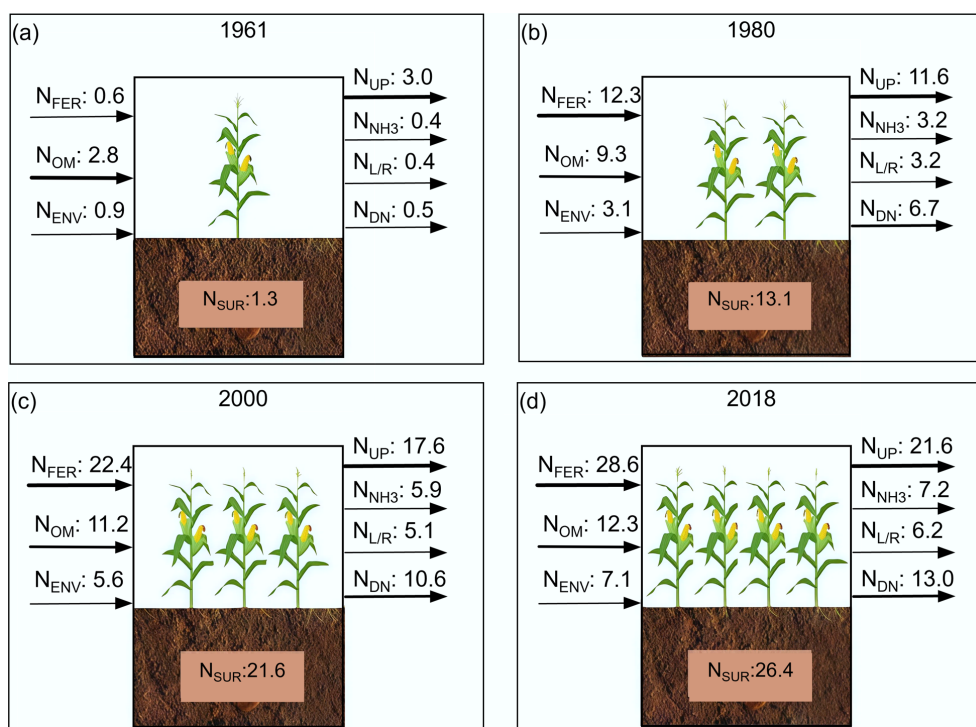
Fig. 2 Nitrogen inputs and losses to the environment in China's crop production systems: (**a**)–(**d**), N flows (Tg N yr^{-1}) in 1961, 1980, 2000, and 2018; (**e**) and (**f**) changes of total N inputs and outputs. Abbreviations: N_{FER}, N from fertilizer; N_{ENV}, N from environment (sum of N fixation, deposition and irrigation); N_{OM}, N from manure; N_{UP}, N uptake by crop; N_{NH3}, N loss from NH_3 emission; $N_{L/R}$, N loss from leaching and runoff; N_{DN}, N loss from denitrification and nitrification (N$_2$, NO and N$_2$O); N_{SUR}, N surplus equals the sum of all N losses to air and water (sum of N_{NH3}, $N_{L/R}$ and N_{DN}).

mainly attributed to direct discharge of animal excreta to surface water or to N leaching, NH$_3$ volatilization and denitrification from the soil and manure management (55%).

In 2018, China completed its primary modernization, and its agricultural input and production showed further increases. Between 2000 and 2015, there was a continuous increase in N inputs from 39.2 Tg N yr^{-1} to 52.3 Tg N yr^{-1}, and the N surplus increased by 22% from 21.6 to 28.8 Tg N yr^{-1} (Fig. 2c; Supplementary Fig. S2a, b). Total grain production (621 Mt yr^{-1}), annual grain consumption (452 kg capita^{-1}), and daily dietary protein intake (98 g capita^{-1}) in 2015 increased substantially by 24%–34% relative to 2000, including 35% protein intake from animal products (Supplementary Fig. S4). China had greatly improved its grain production, a very significant achievement, but this was accompanied by significant N losses to air and water. Between 2015 and 2018, a decline of 8% in total N inputs was observed (from 52.3 Tg N yr^{-1} in 2015 to 48.0 Tg N yr^{-1} in 2018) (Fig. 2d; Supplementary Fig. S2a), which was mainly explained by the implementation of the 'Zero Increase Plan' for N fertilizer by the Chinese Ministry of Agriculture and Rural Affairs in 2015[24]. The N surplus also decreased by 8% from 28.8 to 26.4 Tg N yr^{-1}, suggesting successful policy interventions for reducing agricultural N pollution.

Required and critical N inputs

To quantify the environmental impact of N used in food production, the required N input for food demand and the critical N input for minimizing environmental harm for China was calculated, and the time estimated when the actual N input exceeded the critical N input (Fig. 3a–d) based on environmental thresholds[27,47].

Briefly, the required N input denotes the amount of necessary N to guarantee the required human protein intake for the whole food

chain. The required annual N intake from food is 3.5 kg N capita^{-1} according to the daily protein intake (60 g capita^{-1}) recommended by the Chinese Health and Nutrition Association (slightly higher than the WHO recommended value of 50 g protein intake per capita per day). The ratio of N consumption to N inputs in the complete food chain was 0.16 in China, close to the global average value of 0.167 given in De Vries et al.[27], and equal to the average value given by Ma et al.[37]. Between 1961 and 2018, the required N input ranged from 15 to 30 Tg N yr^{-1} (Fig. 3a). Considering the major N overuse regions located in eastern China (Fig. 3b), and the decreased N uptake proportion with time (Fig. 3b), we expect a quicker increase in required N input in eastern China. The required N input was negatively correlated with NUE along the food chain—i.e., the higher the NUE within the food chain, the lower the required N input (Fig. 3d).

The critical N input represents the maximum acceptable N input to achieve an N surplus that meets environmental protection targets, i.e. a NO$_3^-$ concentration in leachates below 11.3 mg NO$_3$-N L^{-1}, and a critical NH$_3$ emission based on a critical N load for terrestrial ecosystems. Using the WHO limit for drinking water (11.3 mg N L^{-1}) as the critical N concentration permitted to be leached from croplands leads to a critical NO$_3$-N loss to groundwater of 18.1 kg N ha^{-1} yr^{-1} or 2.3 Tg N yr^{-1}. Using a leaching fraction of total N input of 0.08 (0.06–0.10), the total critical N input for China was calculated as 28 (24–32) Tg N yr^{-1}. For the critical NH$_3$ emission from croplands using an area-weighted mean critical N load (10 kg N ha^{-1} yr^{-1}) to natural and semi-natural ecosystems[25,41], a critical NH$_3$ loss to air was estimated at 42 kg N ha^{-1} yr^{-1} or 5.2 Tg N yr^{-1}. Using an average NH$_3$ loss fraction for China of 0.15 (close to 0.14 or 14% in Fig. 1), the total critical N input was calculated to be 35 (30–40) Tg N yr^{-1}.
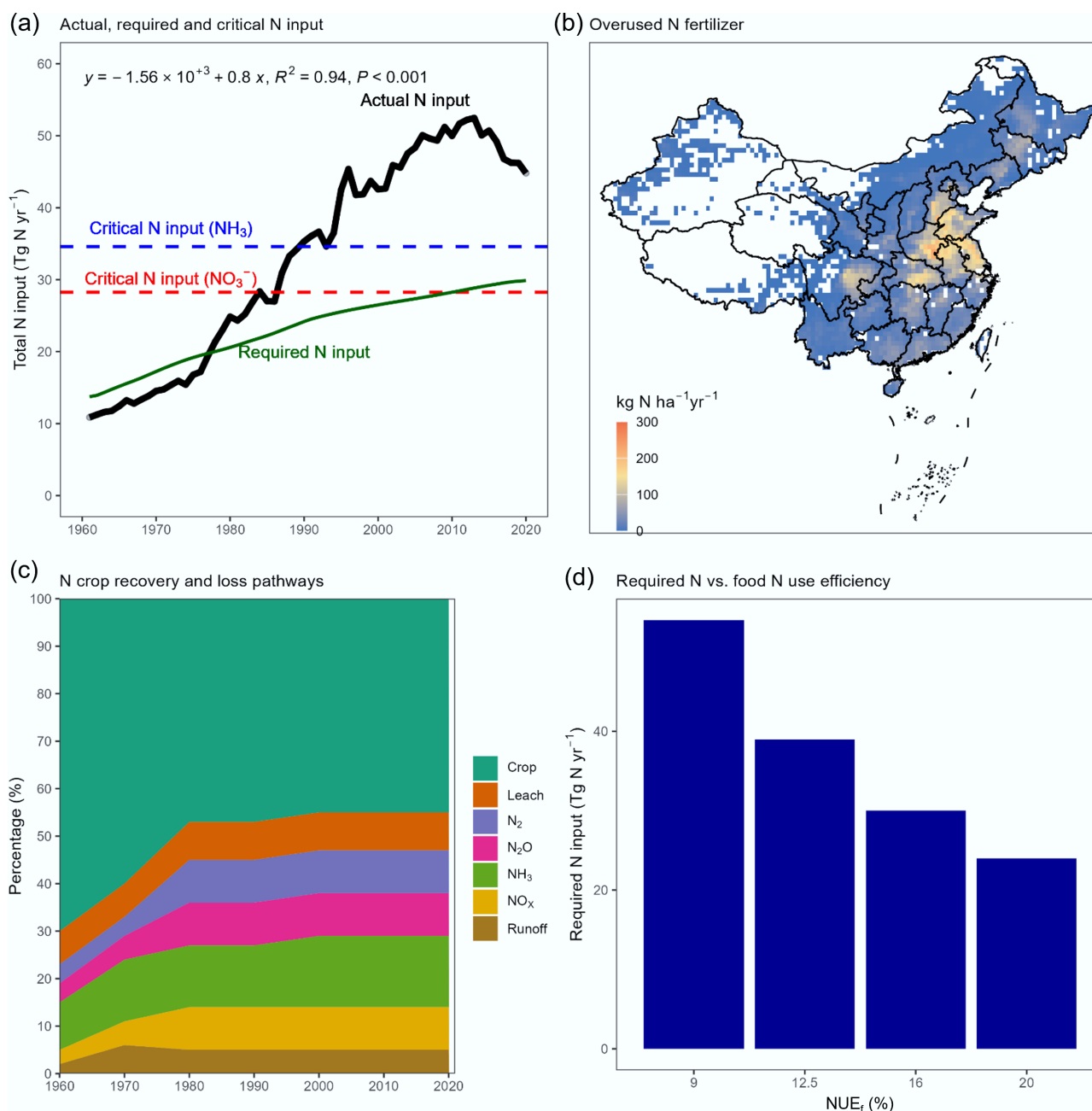
(a) Actual, required and critical N input

$$y = -1.56 \times 10^{+3} + 0.8\,x, \; R^2 = 0.94, \; P < 0.001$$

Actual N input

Critical N input (NH_3)

Critical N input (NO_3^-)

Required N input

(b) Overused N fertilizer

kg N $ha^{-1}yr^{-1}$

(c) N crop recovery and loss pathways

Crop
Leach
N_2
N_2O
NH_3
NO_X
Runoff

(d) Required N vs. food N use efficiency

NUE_f (%)

Fig. 3 Actual, required and critical N inputs. (**a**) Annual changes in actual, required and critical N inputs. The required N input to cropland is calculated based on the human requirement for daily protein intake and N use efficiency in the food chain. (**b**) The spatial distribution of fertilizer N overuse for 16 major crops in China. (**c**) The contribution of N output by each pathway in China between 1961 and 2018. (**d**) Required N inputs under different food N use efficiency levels in China.

Based on the critical N loss with respect to surface water limits (1.0 mg N L^{-1}) used by Yu et al.[13], and the years 1987–1988, when China's actual N runoff was equal to the critical N loss, the critical N input was found to be 27–28 Tg N yr^{-1}; actual N input surpassed the critical N input for NH_3 around 1990–1995 (Fig. 3a). The actual N input was much lower than the required and critical N inputs in 1961, and close to those inputs in 1980, but substantially higher after 2000 (Fig. 3a). Meanwhile, the required N input was less than the critical N input, but became similar, after 2010, due to demands from a continuously increasing population. The actual N input was 18–20 Tg N higher than the required or critical N input (assuming the lowest value to be the 'real' critical input) in 2018, equal to 37%–39% of total N input. Approximately 38% of N inputs were in

excess in China based on a yield-N input response model[15]. The current N input to crop production in China is therefore excessive, especially in the North China Plain and Yangtze River Basin (Fig. 3b; Supplementary Figs S1, S3, S5), and is far from safe and must be optimized to protect both the environment (e.g., biodiversity), and human health (e.g., drinking water quality).

Strategies to improve N management

Large differences between actual and required/critical N inputs in China have been identified. Closing these gaps is a huge challenge for the country. However, progress has been made in parts of Europe, achieved through the introduction of the European Union Directive on

NO_3^- in water, national directives on N fertilizer and manure inputs, and the National Emissions Ceilings Directive (NECD) with limits for NH_3 emission[41,48]. The EU's Nitrates Directive and NECD offer the most actionable templates for China's agricultural pollution control. By integrating EU-style compliance frameworks with China's digital governance strengths (e.g., IoT-based fertilizer tracking), a hybrid model could achieve predictable NUE improvement in high-risk regions. This alignment ensures policy coherence while addressing China's unique challenges in food security and ecological conservation.

A three-step strategy for China is proposed (Fig. 4a): (i) replacing some fertilizer N with effectively recycled manure (as shown by 'Manure replacement' or '80% manure recycled'); (ii) balancing N fertilizer inputs and other non-fertilizer N sources by reducing excessive N fertilizer use based on the crop N demand (as shown by 'Balanced fertilization'); (iii) integrated soil-crop system management (as shwon by 'Integrated approaches'), including optimized N fertilization techniques (i.e., 4R strategy) and optimized crop and soil management (best cultivars; optimal rotations; improved soil productivity)[23,49,50]. The potential of a range of Nr-saving technologies for crop production in China to close the gaps were assessed, and 2018 was taken as a baseline for assessing the N reduction potential in 2030.

First, one major issue in China is the inefficient use of manure, which is only partly recycled in crop production. The total amount of manure N in 2018 was 15.4 Tg N yr^{-1}, of which 80% (12.3 Tg N yr^{-1}) was accessible to cropping systems and 20% was accessible to grazed grasslands. The amount of recycled manure N available for croplands was estimated at 9.8 Tg N yr^{-1}, assuming a recycling
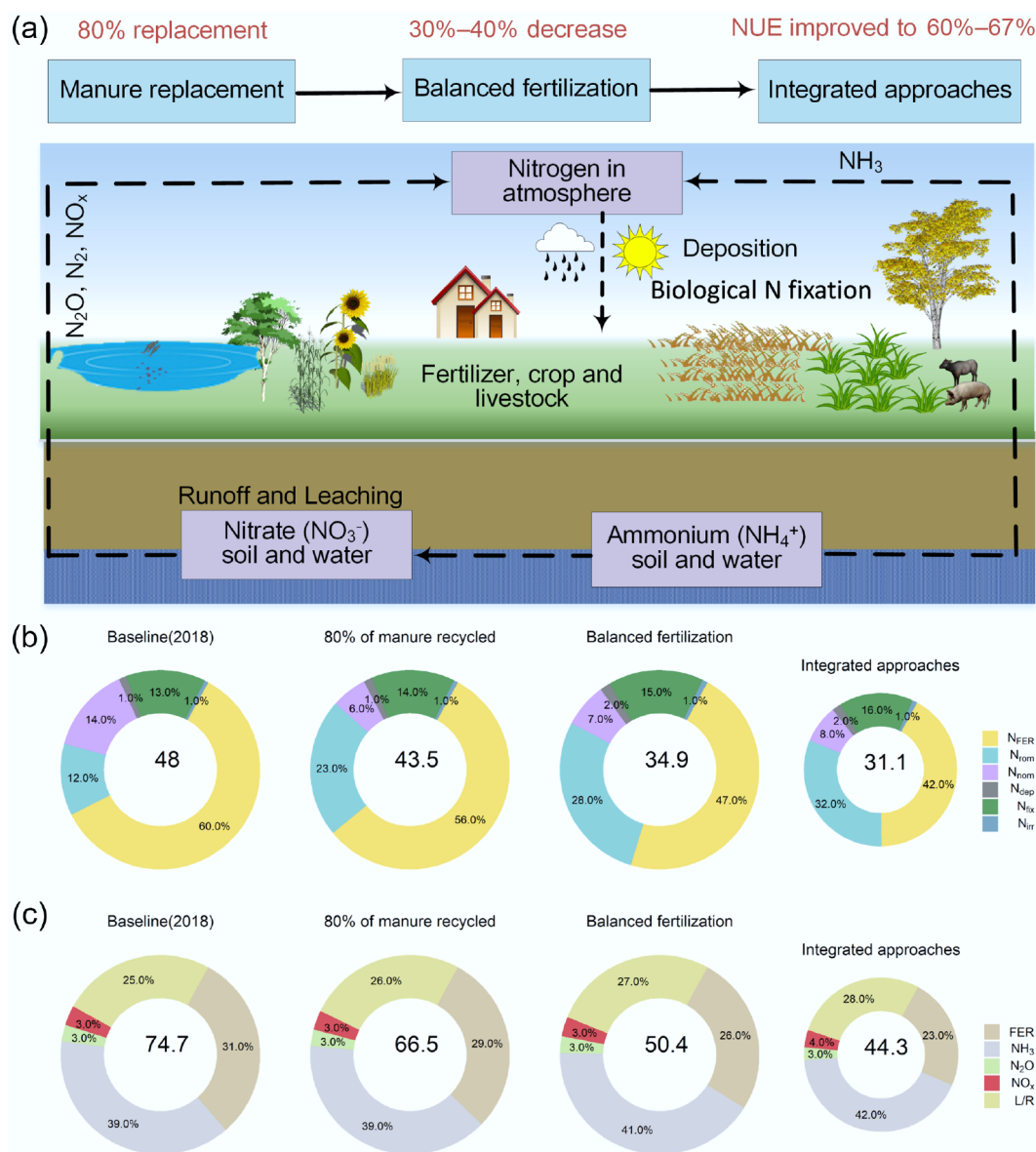

Fig. 4 Reduction of N inputs and associated costs through improved nutrient management practices: (i) 80% of manure recycled; (ii) option (i) + balanced fertilization; (iii) options (i) + (ii) + integrated approaches through soil-crop system management. (a) A scheme of N flows and steps to reduce N fertilizer use and N loss. (b) Reduction in N inputs by improved recycling of manure, plus balanced fertilization (Right amount), plus integrated management (Right time, Right place and Right product with soil-crop management). (c) Cost-benefit analysis induced by N fertilizer saving and by lower N_r losses to water and air. Notes: (1) N_{FER}, N_{ROM}, N_{NOM}, N_{DEP}, N_{FIX}, and N_{IRR} in sub-chart B denote fertilizer N, recycled manure, non-recycled manure, N deposition, biological N fixation, and N from irrigation, respectively; (2) FER, NH_3, N_2O, NO_x and L/R in sub-chart C denote direct and indirect costs induced by fertilizer N use, and by NH_3, N_2O, NO_x, and N leaching/runoff losses, respectively.

efficiency of 80%, while current manure N input was estimated at 5.6 Tg N yr^{-1} in 2018 (Supplementary Table S3). Thus, 80% recycling of N in manure (the policy target by 2025) would lead to an additional saving of 4.2 Tg N yr^{-1} (plus the already recycled 5.6 Tg N yr^{-1}), approximately equal to 34% of fertilizer N (28.6 Tg N yr^{-1}). We note that the 80% manure recycling target by 2025 is ambitious. The main barriers to the recycling of manures are the decoupling between crop and livestock production and the high economic costs associated with manure returning to cropland. To realize the recycling of N in manure, we need to reform the currently industrialized livestock-dominated systems towards more coupled crop-animal systems. Meanwhile, governments should allocate targeted subsidies to stimulate the development and adoption of advanced machinery for livestock manure management, thereby promoting environmental compliance and operational efficiency in farming systems.

The second step is to balance N fertilization and non-fertilizer N sources when calculating total N inputs. China's average NUE is 46% (Supplementary Table S2). This could result in a further 30%−35% reduction in N fertilizer without impacting current yields[51], which would present an enormous improvement, implying a saving of 8.1 Tg N yr^{-1} and an additional reduction in NH$_x$ deposition of 0.5 Tg N yr^{-1}, thus reducing the total N input by 8.6 Tg N yr^{-1} (Fig. 4b; Supplementary Tables S3 & S4). This implies that an improved NUE of 62% is achievable in China, a level comparable to that in Europe, near 60%−65%[52].

The third step is to improve management of the whole soil-crop system by increasing the NUE and thereby decreasing N loss. This can be achieved through practices such as by applying 4R methods, best cultivars, and optimal rotations[23,49]. This would further reduce N fertilizer use by 20% (Supplementary Table S3) and increase the NUE to 67%, which is slightly higher than the NUE in Europe[52], which would deliver a more efficient use of all the N available and higher soil productivity[51,53].

Through this three-step strategy, total N inputs can be reduced by 17 Tg N yr^{-1} (around 35% reduction) and brought to a level close to the required/critical N input (31 vs 28−35 Tg N yr^{-1}). The improvement in N management will require considerable effort but would also lead to large direct and indirect benefits (Supplementary Table S5). Direct benefits include the reduced requirement for N fertilizer by farmers: an annual reduction of 15.6 Tg of N fertilizer results in direct savings for farmers of EUR€14.0 (13.4−16.0) billion, assuming an N fertilizer cost of approximately EUR€0.6−1.0 per kg N and the cost of urea of about EUR€300−500 per ton and an N content of 46% (Fig. 3c; Supplementary Table S5). The indirect benefits include reduced costs for treating water (removing nitrate from water), reduced health expenses, and environmental improvements, such as reduced emissions of greenhouse gases (contributing to carbon reduction targets). These were estimated at EUR€17.9 billion (Fig. 4c; Supplementary Table S5), based on an adjusted willingness to pay in the EU27 for the reduction of impacts from N losses to air and water[44]. In summary, the total direct plus indirect benefits due to N savings through integrated approaches could be up to EUR€30.4 (28.5−35.8) billion, or CNY¥212 (199−250) billion in China. In addition, the required and critical N inputs are not fixed; instead, they may decrease or increase alongside an aging population and reduced NH$_3$ emission and NO$_3^-$-N leaching fractions under improved N management in the future.

Conclusions and implications

To reach this ambitious target of minimising N-related environmental harm associated with food security, China needs a thorough nationwide reform of its agricultural production systems. This requires substantial advances in nutrient management technology and policy, particularly for the large number of smallholder farmers[54,55]. The decrease in N inputs will likely be associated with an increase in farm size and a reduction in the farmer population[56,57]. We believe, however, that farmer education may be just as important for improving N use efficiency, encouraging the adoption of such practices as better crop rotations and mixed cropping, using N-fixing plants as green manures, and applying N fertilizers, and manures at the right time and at the right place.

Recent national actions and changes in policies are bringing about a rapid transition of agriculture towards green agriculture development in China[58], including initiatives such as National Soil-testing and Fertilizer Recommendations, Zero-increase Action Plan for Fertilizer Use, National Action for Soil Quality Improvement, and amalgamating smallholder croplands[9,21]. To accelerate this transition, the following policy-related actions in line with the abovementioned proposed three-step strategy are recommended.

The most important issue, as the first-step strategy, is to promote the full utilization of various nutrient resources, including organic manures and environmental nutrients (e.g., N from deposition, biological fixation, and irrigation). We appeal to redirect subsidies from synthetic fertilizers to manure recycling infrastructure and enhance the integration of livestock and crop production systems for improved utilization of manure nutrients[59]. We also emphasize the importance of developing a field-scale manure nutrient recommendation system. Such a system should incorporate manure nutrient composition (encompassing total and available nutrients) and account for the impacts of application timing, method, rate, and soil type on manure N availability, as these factors directly influence post-application NH$_3$ volatilization, N$_2$O emissions, and N leaching losses. At the same time, it is essential to implement stricter environmental protection regulations to control N inputs and losses from croplands and livestock farms. Coordinated spatial planning to colocated intensive livestock farms with crop production areas, along with the establishment of nitrate vulnerable zones, will also be important[60].

It is also extremely important to develop systematic tools—from decision to application systems—for applying nutrients (either inorganic or organic resources) to the cropland. The development of intelligent machinery and site-specific soil-crop management should be encouraged as a priority, especially considering diversified cropping and different plot sizes in China. This can be achieved by promoting cooperative farming management schemes to accelerate the application of innovative/green technologies, such as 4R technologies.

Another significant reform for sustainable food systems will involve a structured shift in dietary patterns toward alignment with China's National Nutrition Guidelines and WHO's protein intake benchmarks (< 10% of total energy from animal proteins). Reducing and optimizing animal-based food consumption will significantly mitigate environmental pressures from livestock production. The proportion of legumes (e.g., soybean) with a high N fixing capacity in novel/new crop rotations should be increased to reduce N fertilizer inputs, with legume protein being an important component of a healthy diet[61]. This transition requires an integrated Food-Chain-N-Management approach for green transformation of agriculture in China[23].

Finally, we believe that developing international markets and trade policies is important to reconcile the conflict between agricultural productivity and environmental protection in China, ensuring that efforts to reduce N-related pollution benefit both national and global sustainability. We also suggest explicitly incorporating

uncertainty analysis (Supplementary Fig. S6) into future research and policy recommendations to enhance the robustness of conclusions and facilitate informed decision-making under varying environmental and economic scenarios.

Supplementary information

Author contributions

The authors confirm their contributions to the paper as follows: study design: Liu X, de Vries W, Zhang F; data analysis: Zhang Y, Liu X, Liu L; conducting the research: Liu X, Liu L, Zhang Y, Ma L; writing the manuscript: Liu X, Liu L, de Vries W, Zhang Y; all authors participated in the discussions and commented on the paper. All authors reviewed the results and approved the final version of the manuscript.

Data availability

The datasets used or analyzed in the current study are available from the corresponding author upon reasonable request.

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Declarations

Competing interests

The authors declare that they have no conflict of interest.

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References

[1] Erisman JW, Sutton MA, Galloway J, Klimont Z, Winiwarter W. 2008. How a century of ammonia synthesis changed the world. *Nature Geoscience* 1:636−639

[2] Galloway JN, Townsend AR, Erisman JW, Bekunda M, Cai Z, et al. 2008. Transformation of the nitrogen cycle: recent trends, questions, and potential solutions. *Science* 320:889−892

[3] Sutton MA, Oenema O, Erisman JW, Leip A, van Grinsven H, et al. 2011. Too much of a good thing. *Nature* 472:159−161

[4] Schulte-Uebbing LF, Beusen AHW, Bouwman AF, de Vries W. 2022. From planetary to regional boundaries for agricultural nitrogen pollution. *Nature* 610:507−512

[5] Springmann M, Clark M, Mason-D'Croz D, Wiebe K, Bodirsky BL, et al. 2018. Options for keeping the food system within environmental limits. *Nature* 562:519−525

[6] Chen X, Cui Z, Fan M, Vitousek P, Zhao M, et al. 2014. Producing more grain with lower environmental costs. *Nature* 514:486−489

[7] Zhang X, Davidson EA, Mauzerall DL, Searchinger TD, Dumas P, et al. 2015. Managing nitrogen for sustainable development. *Nature* 528:51−59

[8] Gerten D, Heck V, Jägermeyr J, Bodirsky BL, Fetzer I, et al. 2020. Feeding ten billion people is possible within four terrestrial planetary boundaries. *Nature Sustainability* 3:200−208

[9] Jiao X, Lyu Y, Wu X, Li H, Cheng L, et al. 2016. Grain production versus resource and environmental costs: towards increasing sustainability of nutrient use in China. *Journal of Experimental Botany* 67:4935−4949

[10] Ju X, Xing G, Chen X, Zhang S, Zhang L, et al. 2009. Reducing environmental risk by improving N management in intensive Chinese agricultural systems. *Proceedings of the National Academy of Sciences of the United States of America* 106:3041−3046

[11] Chen B, Zhang X, Gu B. 2025. Managing nitrogen to achieve sustainable food-energy-water nexus in China. *Nature Communications* 16:4804

[12] Guo JH, Liu XJ, Zhang Y, Shen JL, Han WX, et al. 2010. Significant acidification in major Chinese croplands. *Science* 327:1008−1010

[13] Yu C, Huang X, Chen H, Godfray HCJ, Wright JS, et al. 2019. Managing nitrogen to restore water quality in China. *Nature* 567:516−520

[14] Liu X, Zhang Y, Han W, Tang A, Shen J, et al. 2013. Enhanced nitrogen deposition over China. *Nature* 494:459−462

[15] Liu L, Xu W, Lu X, Zhong B, Guo Y, et al. 2022. Exploring global changes in agricultural ammonia emissions and their contribution to nitrogen deposition since 1980. *Proceedings of the National Academy of Sciences of the United States of America* 119:e2121998119

[16] Zhu J, Jia Y, Yu G, Wang Q, He N, et al. 2025. Changing patterns of global nitrogen deposition driven by socio-economic development. *Nature Communications* 16:46

[17] Van Grinsven HJ, Ward MH, Benjamin N, de Kok TM. 2006. Does the evidence about health risks associated with nitrate ingestion warrant an increase of the nitrate standard for drinking water? *Environmental Health* 5:26

[18] Espejo-Herrera N, Gracia-Lavedan E, Pollan M, Aragonés N, Boldo E, et al. 2016. Ingested nitrate and breast cancer in the Spanish multicase-control study on cancer (MCC-Spain). *Environmental Health Perspectives* 124:1042−1049

[19] Pozzer A, Tsimpidi AP, Karydis VA, de Meij A, Lelieveld J. 2017. Impact of agricultural emission reductions on fine-particulate matter and public health. *Atmospheric Chemistry and Physics* 17:12813−12826

[20] Zhang X, Gu B, van Grinsven H, Lam SK, Liang X, et al. 2020. Societal benefits of halving agricultural ammonia emissions in China far exceed the abatement costs. *Nature Communications* 11:4357

[21] Gu B, Zhang L, Van Dingenen R, Vieno M, Van Grinsven HJ, et al. 2021. Abating ammonia is more cost-effective than nitrogen oxides for mitigating $PM_{2.5}$ air pollution. *Science* 374:758−762

[22] Zhang F, Chen X, Vitousek P. 2013. Chinese agriculture: An experiment for the world. *Nature* 497:33−35

[23] Liu X, Cui Z, Hao T, Yuan L, Zhang Y, et al. 2022. A new approach to holistic nitrogen management in China. *Frontiers of Agricultural Science and Engineering* 9:490−510

[24] Liu X, Vitousek P, Chang Y, Zhang W, Matson P, et al. 2016. Evidence for a historic change occurring in China. *Environmental Science & Technology* 50:505−506

[25] De Vries W, Schulte-Uebbing L, Kros H, Voogd JC, Louwagie G. 2021. Spatially explicit boundaries for agricultural nitrogen inputs in the European Union to meet air and water quality targets. *Science of The Total Environment* 786:147283

[26] Yang Y, Liu L, Zhang F, Zhang X, Xu W, et al. 2021. Soil nitrous oxide emissions by atmospheric nitrogen deposition over global agricultural systems. *Environmental Science & Technology* 55:4420−4429

[27] De Vries W, Kros J, Kroeze C, Seitzinger SP. 2013. Assessing planetary and regional nitrogen boundaries related to food security and adverse environmental impacts. *Current Opinion in Environmental Sustainability* 5:392−402

[28] Gu B, Ju X, Chang J, Ge Y, Vitousek PM. 2015. Integrated reactive nitrogen budgets and future trends in China. *Proceedings of the National Academy of Sciences of the United States of America* 112:8792−8797

[29] Bai Z, Ma L, Jin S, Ma W, Velthof GL, et al. 2016. Nitrogen, phosphorus, and potassium flows through the manure management chain in China. *Environmental Science & Technology* 50:13409−13418

[30] Chadwick D, Jia W, Tong Y, Yu G, Shen Q, et al. 2015. Improving manure nutrient management towards sustainable agricultural intensification in China. *Agriculture, Ecosystems & Environment* 209:34−46

[31] Department of Rural Social and Economic Investigation of National Bureau of Statistics. 2005. *Compilation of Chinese Agricultural Statistics 1949−2004*. www.stats.gov.cn

[32] National Bureau of Statistics of China. 2019. *National Annual Statistical Bulletin (1978−2018)*. www.stats.gov.cn/sj/tjgb/ndtjgb

[33] Xu W, Zhang L, Liu X. 2019. A database of atmospheric nitrogen concentration and deposition from the nationwide monitoring network in China. *Scientific Data* 6:51

[34] Wen Z, Ma X, Xu W, Si R, Liu L, et al. 2024. Combined short-term and long-term emission controls improve air quality sustainably in China. *Nature Communications* 15:5169

[35] Liu X, Song L, Zhang Y, Zhang F. 2017. Diverse quantification techniques and environmental effects of root-layer nutrients. In *Innovation and Application of Nutrient Management Technology for Double High Agriculture*, eds Zhang F, Zhang C, etc. China Agricultural University Press, Beijing. pp. 51−64 www.cau.edu.cn/caup

[36] Westhoek H, Rood G, Van den Berg M, Janse J, Nijdam D, et al. 2011. The protein puzzle: the consumption and production of meat, dairy and fish in the European Union. *European Journal of Food Research & Technology* 1:123−144

[37] Ma L, Velthof GL, Wang FH, Qin W, Zhang WF, et al. 2012. Nitrogen and phosphorus use efficiencies and losses in the food chain in China at regional scales in 1980 and 2005. *Science of The Total Environment* 434:51−61

[38] Huang F. 2008. *Assessing basin-scale grain crop's water productivity at China's main breadbasket regions*. Thesis. China Agricultural University, Beijing www.lib.cau.edu.cn

[39] Li B, Huang F. 2010. Trends in China's agricultural water use during recent decade using the green and blue water approach. *Advances in Water Science* 21:575−583

[40] De Vries W, Schulte-Uebbing L. 2020. Required changes in nitrogen inputs and nitrogen use efficiencies to reconcile agricultural productivity with water and air quality objectives in the EU-27. *Proceedings - International Fertiliser Society* 842:39

[41] Bobbink R, Hicks K, Galloway J, Spranger T, Alkemade R, et al. 2010. Global assessment of nitrogen deposition effects on terrestrial plant diversity: a synthesis. *Ecological Applications* 20:30−59

[42] Xu W, Liu L, Cheng M, Zhao Y, Zhang L, et al. 2018. Spatial−temporal patterns of inorganic nitrogen air concentrations and deposition in Eastern China. *Atmospheric Chemistry and Physics* 18:10931−10954

[43] Oenema O, Kros H, De Vries W. 2003. Approaches and uncertainties in nutrient budgets: implications for nutrient management and environmental policies. *European Journal of Agronomy* 20:3−16

[44] Van Grinsven HJM, Holland M, Jacobsen BH, Klimont Z, Sutton MA, et al. 2013. Costs and benefits of nitrogen for Europe and implications for mitigation. *Environmental Science & Technology* 47:3571−3579

[45] Zhu X, Zhou P, Miao P, Wang H, Bai X, et al. 2022. Nitrogen use and management in orchards and vegetable fields in China: challenges

and solutions. *Frontiers of Agricultural Science and Engineering* 9:386−395

[46] Joint WHO/FAO/UNU Expert Consultation. 2007. Protein and amino acid requirements in human nutrition. In *Report of a Joint FAO/WHO/UNU Expert Consultation, WHO Technical Report Series 935*. pp. 1−265

[47] Steffen W, Richardson K, Rockström J, Cornell SE, Fetzer I, et al. 2015. Planetary boundaries: guiding human development on a changing planet. *Science* 347:1259855

[48] D'Haene K, Salomez J, De Neve S, De Waele J, Hofman G. 2014. Environmental performance of nitrogen fertiliser limits imposed by the EU Nitrates Directive. *Agriculture, Ecosystems & Environment* 192:67−79

[49] Chen X, Cui Z, Vitousek P, Cassman KG, Matson PA, et al. 2011. Integrated soil-crop system management for food security. *Proceedings of the National Academy of Sciences of the United States of America* 108:6399−6404

[50] Fan M, Lal R, Cao J, Qiao L, Su Y, et al. 2013. Plant-based assessment of inherent soil productivity and contributions to China's cereal crop yield increase since 1980. *PLoS One* 8:e74617

[51] Xia L, Lam SK, Chen D, Wang J, Tang Q, et al. 2017. Can knowledge-based N management produce more staple grain with lower greenhouse gas emission and reactive nitrogen pollution? A meta-analysis. *Global Change Biology* 23:1917−1925

[52] Bouwman L, Goldewijk KK, Van Der Hoek KW, Beusen AH, Van Vuuren DP, et al. 2013. Exploring global changes in nitrogen and phosphorus cycles in agriculture induced by livestock production over the 1900-2050 period. *Proceedings of the National Academy of Sciences of the United States of America* 110:20882−20887

[53] Wang G, van der Putten WH, Klironomos J, Zhang F, Zhang J. 2025. Steering plant-soil feedback for sustainable agriculture. *Science* 389:eads2506

[54] Cui Z, Zhang H, Chen X, Zhang C, Ma W, et al. 2018. Pursuing sustainable productivity with millions of smallholder farmers. *Nature* 555:363−366

[55] Kang J, Wang J, Heal MR, Goulding K, De Vries W, et al. 2023. Ammonia mitigation campaign with smallholder farmers improves air quality while ensuring high cereal production. *Nature Food* 4:751−761

[56] Wu Y, Xi X, Tang X, Luo D, Gu B, et al. 2018. Policy distortions, farm size, and the overuse of agricultural chemicals in China. *Proceedings of the National Academy of Sciences of the United States of America* 115:7010−7015

[57] Yu Y, Hu H, Gu B, Reis S, Yang L. 2022. Reforming smallholder farms to mitigate agricultural pollution. *Environmental Science and Pollution Research* 29:13869−13880

[58] Shen J, Zhu Q, Jiao X, Ying H, Wang H, et al. 2020. Agriculture green development: a model for China and the world. *Frontiers of Agricultural Science and Engineering* 7:5−13

[59] Chadwick DR, Williams JR, Lu Y, Ma L, Bai Z, et al. 2020. Strategies to reduce nutrient pollution from manure management in China. *Frontiers of Agricultural Science and Engineering* 7:45−55

[60] Bai Z, Lu J, Zhao H, Velthof GL, Oenema O, et al. 2018. Designing vulnerable zones of nitrogen and phosphorus transfers to control water pollution in China. *Environmental Science & Technology* 52:8987−8988

[61] Foyer CH, Lam HM, Nguyen HT, Siddique KH, Varshney RK, et al. 2016. Neglecting legumes has compromised human health and sustainable food production. *Nature Plants* 2:16112