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Managing nitrogen to restore water quality in China

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The nitrogen cycle has been radically changed by human activities[1]. China consumes nearly one third of the world's nitrogen fertilizers. The excessive application of fertilizers[2,3] and increased nitrogen discharge from livestock, domestic and industrial sources have resulted in pervasive water pollution. Quantifying a nitrogen 'boundary'[4] in heterogeneous environments is important for the effective management of local water quality. Here we use a combination of water-quality observations and simulated nitrogen discharge from agricultural and other sources to estimate spatial patterns of nitrogen discharge into water bodies across China from 1955 to 2014. We find that the critical surface-water quality standard (1.0 milligrams of nitrogen per litre) was being exceeded in most provinces by the mid-1980s, and that current rates of anthropogenic nitrogen discharge (14.5 ± 3.1 megatonnes of nitrogen per year) to fresh water are about 2.7 times the estimated 'safe' nitrogen discharge threshold (5.2 ± 0.7 megatonnes of nitrogen per year). Current efforts to reduce pollution through wastewater treatment and by improving cropland nitrogen management can partially remedy this situation. Domestic wastewater treatment has helped to reduce net discharge by 0.7 ± 0.1 megatonnes in 2014, but at high monetary and energy costs. Improved cropland nitrogen management could remove another 2.3 ± 0.3 megatonnes of nitrogen per year—about 25 per cent of the excess discharge to fresh water. Successfully restoring a clean water environment in China will further require transformational changes to boost the national nutrient recycling rate from its current average of 36 per cent to about 87 per cent, which is a level typical of traditional Chinese agriculture. Although ambitious, such a high level of nitrogen recycling is technologically achievable at an estimated capital cost of approximately 100 billion US dollars and operating costs of 18–29 billion US dollars per year, and could provide co-benefits such as recycled wastewater for crop irrigation and improved environmental quality and ecosystem services.

Earth's biogeochemical cycles have been strongly affected by human activities. For example, the amount of reactive nitrogen entering the global environment increased from around 15 megatonnes (Mt) in 1860 to 185 Mt in 2010[1,5], while agricultural use of nitrogen fertilizers increased from 12 Mt in 1961 to 110 Mt in 2014[6]. Although it is a critical nutrient for crop yields and food production, human inputs of reactive nitrogen into terrestrial and freshwater ecosystems cause water pollution (for example nitrate, ammonium) and air pollution (ammonia, nitrogen oxides), as well as global warming and stratospheric ozone depletion (nitrous oxide)[7].

China has markedly increased its food production over the past four decades. Domestic grain production has increased from 132 Mt in 1950 to 607 Mt in 2014, without expansion of the total planting area (Supplementary Fig. 1). This has been achieved through substantial increases in the use of synthetic fertilizers (Fig. 1), along with increasingly efficient agricultural technologies and practices, and expanded irrigation infrastructure. China now accounts for around 32% of the world's consumption of nitrogen fertilizers[6]. To illustrate the critical role of nitrogen input on crop production, we use the nitrogen-based denitrification and decomposition (DNDC) biogeochemical model[8] and agricultural data from 2,403 counties to simulate major grain crops (rice, wheat and maize) during the period 1955–2014 (Methods; Supplementary Figs. 2, 3). Figure 1b shows that 45 ± 3% of current grain yields can be attributed to the use of synthetic nitrogen fertilizers. Similar conclusions have been reached after controlled long-term experiments in China[9] and other locations[10,11] (Supplementary Table 1).

Meanwhile, the traditional practices of recycling organic waste as fertilizer have been largely abandoned. The availability of subsidized synthetic nitrogen fertilizers and new sewage infrastructure have resulted in a sharp decrease in nutrient-recycling rates, from over 90% in the late 1970s to 36% in 2014 (Fig. 1c, d). Changing nitrogen management practices led to a considerable decrease in the average nitrogen use efficiency of China's croplands between the 1960s and the 2010s[2,3]. Increased nitrogen discharges from croplands and non-recycled organic wastes have resulted in pervasive water pollution.

Quantifying safe levels[4] of nitrogen discharge to the environment is a prerequisite for effective nitrogen management. In 2009 it was proposed[12] that 25% (35 Mt N yr^{-1}) of the current level of human nitrogen fixation from the atmosphere could be considered as a global 'safe boundary', but this estimate was qualified as a 'first guess'. Recent studies have attempted to clarify the global safe boundary for nitrogen released by human activities using simplified nitrogen-budget models[5,13]; however, relationships between nitrogen discharge, nitrogen concentrations in freshwater bodies and biotic responses to nitrogen in water bodies are complex and often differ by region. Therefore, large uncertainties remain regarding regional 'safe' nitrogen-discharge thresholds, owing to insufficient understanding of spatiotemporal heterogeneity in biogeochemical and hydrological processes. The concept of a safe boundary is most useful when applied to regional and local scales, where practical management options are available. Here we use observations of water quality in representative rivers and lakes across China to characterize regional thresholds of total nitrogen discharge to the water environment, assess excess nitrogen, and evaluate the potential for reducing nitrogen pollution.

We collected observational data on the concentrations of total nitrogen in water bodies and used DNDC and other nitrogen balance models to estimate provincial nitrogen discharge into the inland aquatic environment for the years 1955–2014. Here, nitrogen discharge comprises anthropogenic sources from croplands, human and livestock excrement, organic garbage, and industrial waste (Figs. 2, 3; Methods; Supplementary Figs. 4–6; Supplementary Table 1). Nitrogen concentrations in water bodies were low before the 1980s (typically <1.0 mg N l^{-1}), but increased rapidly to amounts exceeding 15 mg N l^{-1} in many

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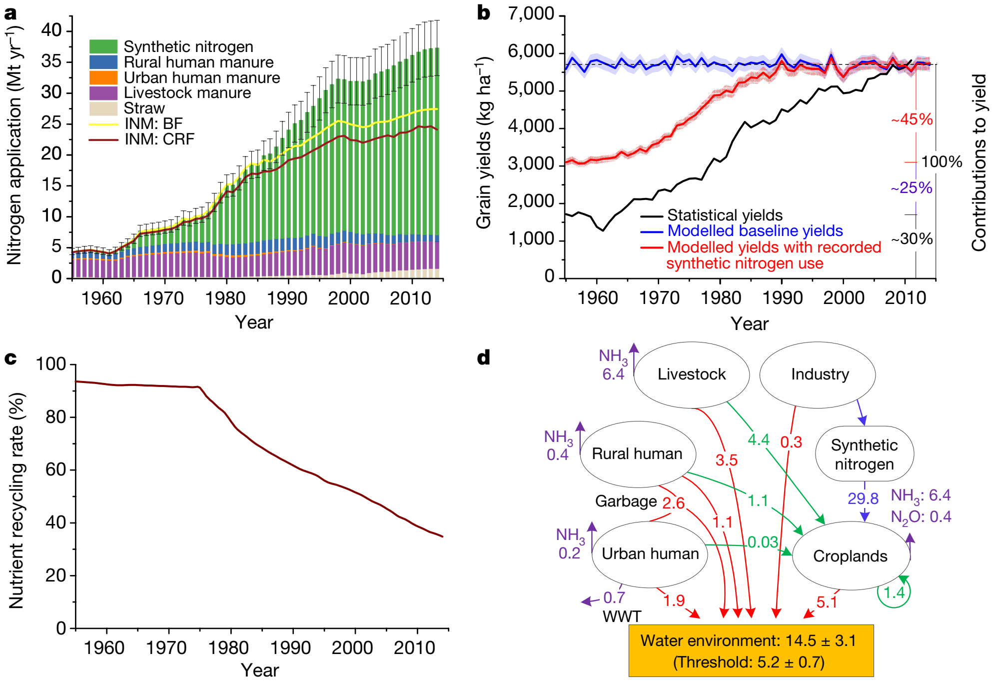

Fig. 1 | The changing nitrogen cycle and its contributions to food production in China from 1955 to 2014. a, Nitrogen application between 1955 and 2014. The stacked bars represent the application of nitrogen to croplands; the yellow and red lines mark our model-based estimates of the minimum nitrogen application required to maintain annual food production under the two INM scenarios without changes in nutrient recycling rates (BF, broadcasting of fertilizer; CRF, controlled-release fertilizers). Error bars indicate upper and lower bounds of total nitrogen applied to croplands (see Methods). **b**, Grain yields (1955–2014) obtained from statistics and modelling. The contribution of synthetic nitrogen to grain productivity (around 45%) is shown by the red bar on the right. The blue line represents climate-driven variations in modelled grain yields (rice, maize and wheat) under the baseline scenario (current cropping with average agricultural inputs for 2007–2011; see Methods). The red line shows variations in modelled grain yields due to variations in climate and historical synthetic nitrogen inputs. The black line shows actual yields as recorded in the statistical yearbooks. The unexplained 25% growth in grain yields between 1955 and 2010 is attributed to additional contributions from the expansion of irrigation and technological advances (for example, pesticide, breeding and genetics) that affected the yield increase with nitrogen inputs simultaneously. **c**, The broken nitrogen cycle: increases in synthetic nitrogen application are associated with declining rates of nutrient recycling since the late 1970s. The recycling rate is defined as the ratio of nitrogen from organic waste returned to croplands to the amount of nitrogen in organic waste, excluding emissions to the atmosphere. **d**, The major anthropogenic nitrogen flows to croplands (green arrows, organic nitrogen; blue arrows, synthetic nitrogen), water (red arrows) and the atmosphere (purple arrows). Data are 2010–2014 averages. Nitrogen removal by WWT is based on data from 2014. Imported nitrogen is counted in our assessment of excretion (Supplementary Fig. 8).

catchments after the 1990s. Groundwater concentrations of nitrate also increased sharply after the 1980s, with stable isotope analysis implicating agricultural and domestic sources[14]. The Ministry of Water Resources classified 80.2% of groundwater samples (from 2,103 wells; Supplementary Fig. 7) as breaching the class IV standard in 2015[15].

We then empirically identified the critical nitrogen-discharge threshold for each province as the total anthropogenic nitrogen discharge during the year in which pollution levels first breached the water-quality standard (1.0 mg N l^{-1})[16,17] in representative catchments (Figs. 2, 3; Methods; Supplementary Table 3). Aggregating nitrogen-discharge thresholds from catchments to a national scale leads to a new estimate of the national nitrogen-discharge threshold of 5.2 ± 0.7 Mt N yr^{-1} (Fig. 3b). This input can be conceptualised as China's share of the total planetary safe nitrogen boundary. The current national anthropogenic nitrogen discharge rate is 14.5 ± 3.1 Mt N yr^{-1} (2010–2014 average), which is well above the threshold. Agricultural systems are responsible for 59% of current nitrogen discharge (35% croplands, 24% livestock). Another 39% is attributed to domestic waste (13% urban sewage, 8% rural sewage, 18% organic garbage) and the remaining 2% to industrial waste.

Although China's declared target[18] of zero growth in chemical fertilizer use by 2020 has already been achieved, synthetic nitrogen use was still 30.5 Mt in 2016. Proposed solutions for maintaining agricultural productivity and reducing negative environmental effects have centred on improved nitrogen management (INM) in croplands: applying the correct fertilizer products, at the correct rate, at the correct time, in the correct place[2]. To assess the potential benefits of INM on the croplands of China, we used the denitrification and decomposition model to estimate the minimum synthetic nitrogen input necessary to maintain historical yields (Methods) through the broadcasting of fertilizers or the use of controlled-release fertilizers (CRF). Figure 3 shows that INM could reduce cropland nitrogen discharge from the current level of 5.1 ± 0.3 Mt N yr^{-1} (2010–2014 average) to 2.8–3.0 Mt N yr^{-1} (range encompasses both the controlled-release and the broadcasting of fertilizer scenarios); in addition, fluxes of ammonia could be reduced by 39–72% and nitrous oxide emissions by 47–55%. INM could thus reduce current total cropland nitrogen inputs from 36.9 ± 0.4 Mt N yr^{-1} to 24.1–26.9 Mt N yr^{-1}, or the use of synthetic nitrogen from 29.8 ± 0.3 Mt N yr^{-1} to 17.0–19.8 Mt N yr^{-1}. If all smallholder farmers adopted INM practices[19], this alone could reduce the excess anthropogenic nitrogen discharge (9.3 ± 2.0 Mt N yr^{-1}; Methods) by 23–25%.

Quantifying safe boundaries at small geographical scales is helpful to clarify more detailed environmental issues and local management options. Figure 4a–c summarizes the critical challenges for nitrogen management at the provincial level. Under current nitrogen management, cropland nitrogen discharge alone already exceeds critical pollution thresholds in 14 out of 31 provinces. INM-CRF could lower cropland pollution to below the critical thresholds in 29 provinces, but two provinces cannot reach safe thresholds without lowering food production (Inner Mongolia by 36%, Shaanxi by 12%).

Wastewater treatment (WWT) is another primary solution for the reduction of point-source water pollution, not only by nitrogen, but also by phosphorus and enteric bacteria in inland water bodies. Domestic

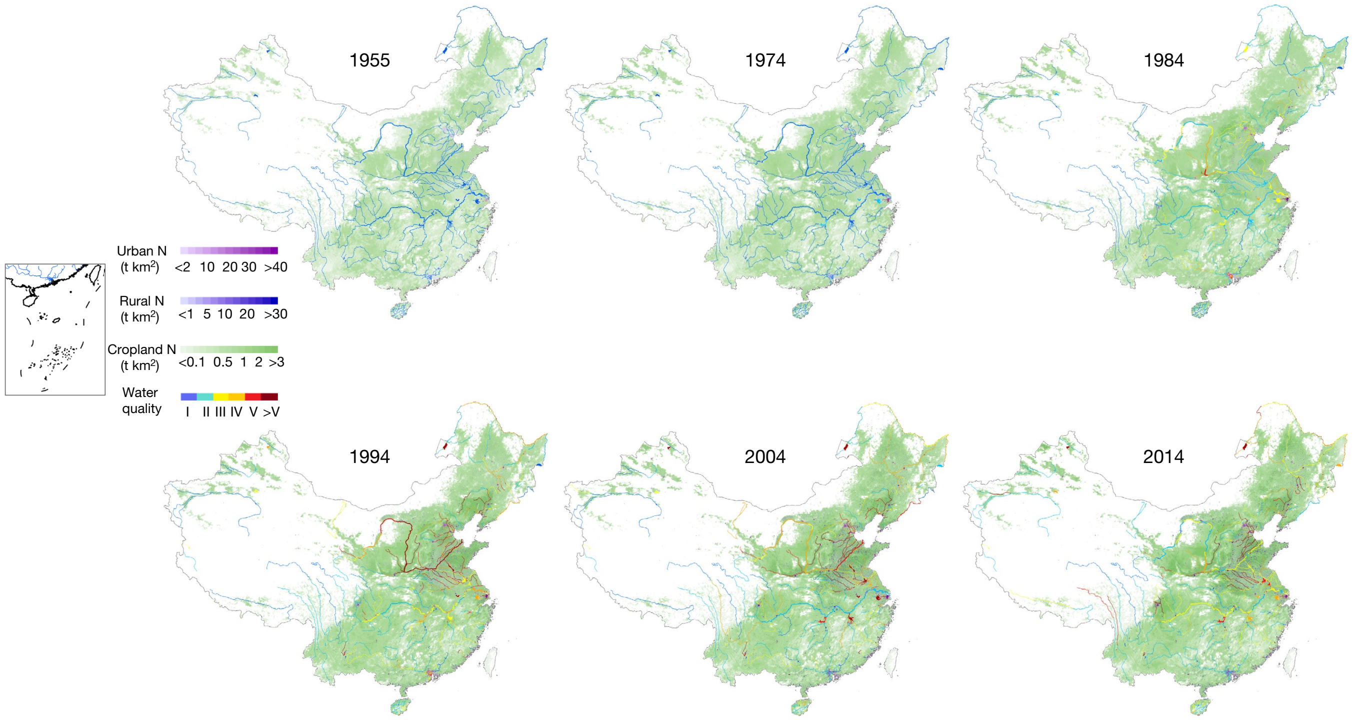

Fig. 2 | Reconstructing nitrogen discharge from 1955 to 2014 and the associated evolution of surface-water quality in terms of total nitrogen. Raster maps of cropland nitrogen are based on leaching and runoff simulated by the denitrification and decomposition model. Urban nitrogen includes discharges of human sewage, industrial waste and garbage at the provincial level. Rural nitrogen includes discharges of rural human sewage, livestock manure and garbage at the provincial level. The data sources can be found in Methods. For an animation of this figure showing nitrogen discharge over time, see Supplementary Video 1.

WWT has increased considerably since the 1980s. In 2014, around 49.43 billion m^3 of municipal wastewater (75% of urban water use) was treated, with an energy consumption of 14.8 billion kWh. The total economic cost (infrastructure and operations) of WWT was about US\$20.8 billion (US\$0.42 m^{-3}) (ref. [20]), equivalent to 2.2% of national rural household income (population 619 million) or 0.2% of gross domestic product (GDP) in 2014. However, the amount of nitrogen removed by WWT was only 0.70 ± 0.1 Mt N (around 26% of total municipal sewage nitrogen), because only 56% of WWT plants had nitrogen-removal facilities[21] and their average nitrogen removal rate was 55% (ref. [22]). Average efflux concentrations in treated water released to the environment were 14.3–16.5 mg N l^{-1} (refs [21,23]). Further improvements in the efficiency of nitrogen removal (for example, tertiary WWT) are feasible and have been achieved in some wastewater treatment works, although at additional capital[24] and operational costs[25].

China needs a holistic strategy to mobilise and integrate all relevant socio-economic sectors to effectively cut nitrogen pollution, not

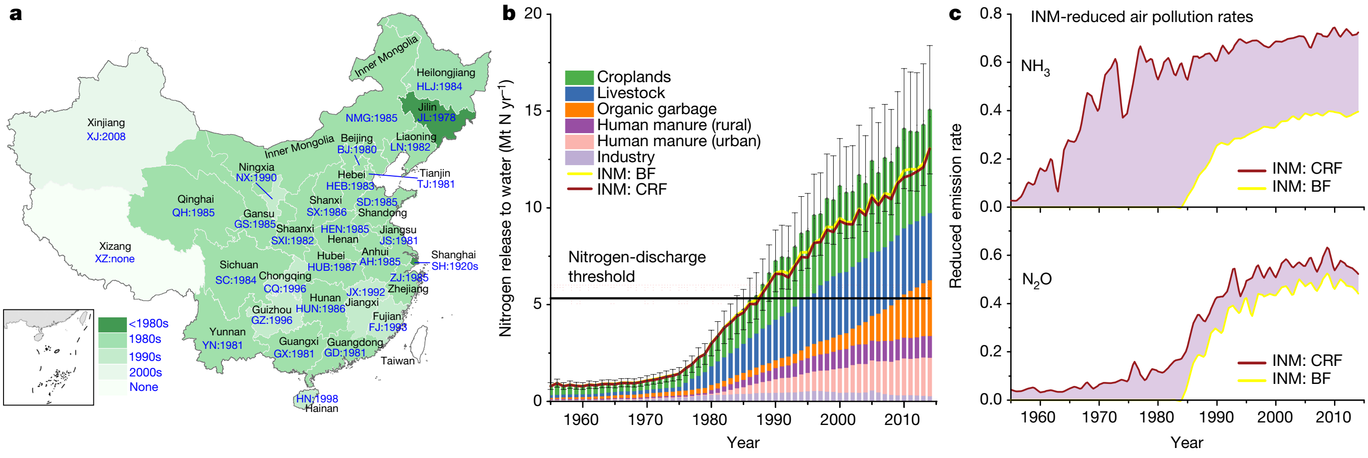

Fig. 3 | Thresholds of nitrogen discharge to the water environment and the potential contributions of INM in reducing nitrogen pollution. **a**, The years in which nitrogen pollution first exceeded the national standard in each province (see Supplementary Table 3 for representative catchments and references). **b**, Sixty-year records of nitrogen discharge (runoff and leaching) to the water environment in mainland China from anthropogenic sources. The yellow and red lines mark simulated minimum nitrogen discharge amounts through INM under the BF and CRF scenarios, respectively. The national critical threshold (5.2 ± 0.7 Mt N yr^{-1}; horizontal black line) is estimated by aggregating provincial-level nitrogen discharge amounts from the years marked in **a**. Error bars indicate estimated upper and lower bounds of total nitrogen discharge into the water environment (see Methods). **c**, Predicted reductions in emissions of air pollutants (ammonia and nitrous oxide) from croplands are estimated by substituting current nitrogen management with INM. Under INM, controlled-release fertilizers cannot substantially reduce nitrogen discharge to the aquatic environment relative to the broadcasting of fertilizer, but do reduce nitrogen loss in the form of ammonia.

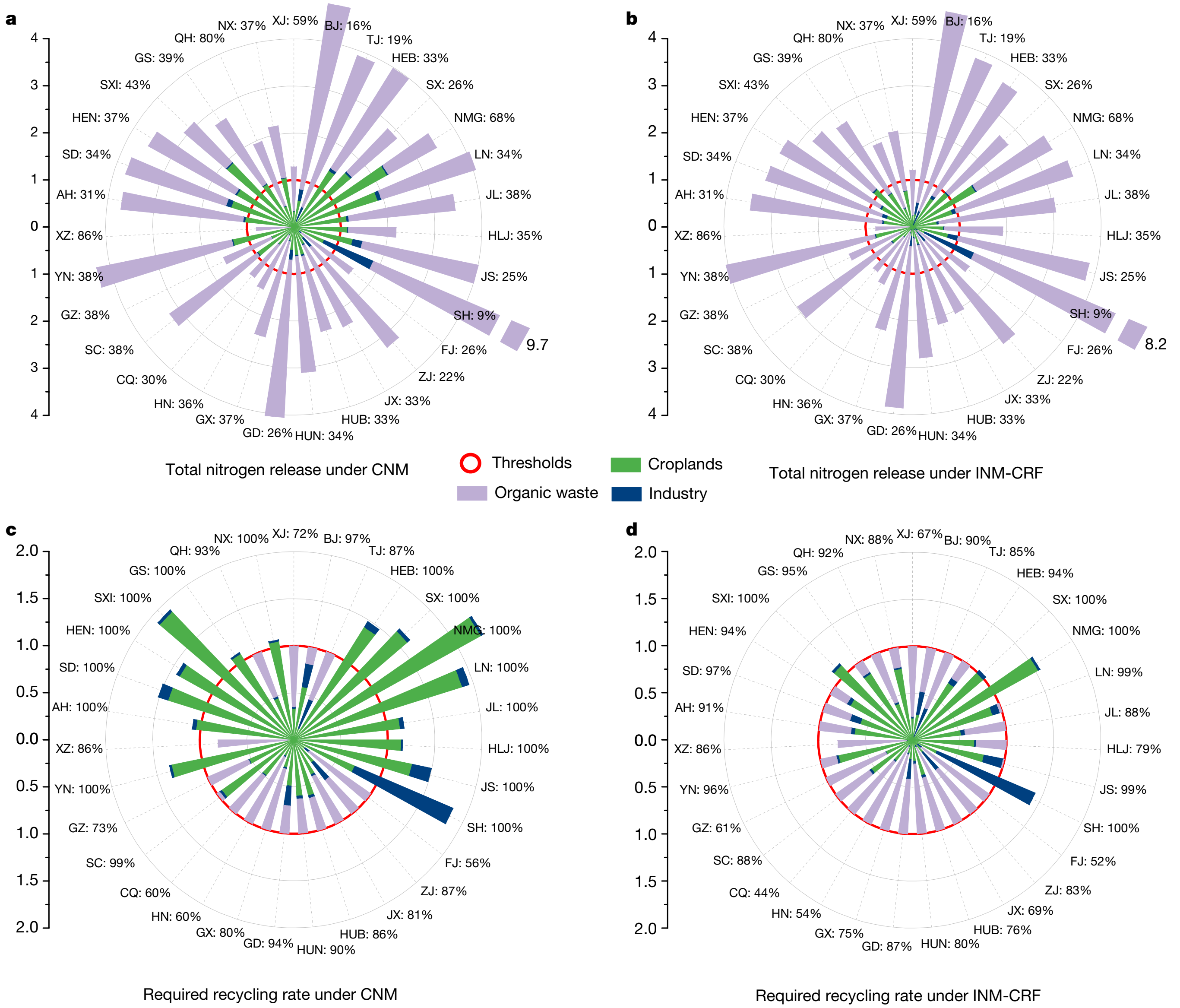

Fig. 4 | Anthropogenic nitrogen discharge and requirements to meet the critical threshold in each province of mainland China. **a**, **b**, Provincial-level nitrogen discharge (see Fig. 3a for locations) into the water environment under current nitrogen management (CNM; **a**) and improved nitrogen management (INM-CRF; **b**) in croplands based on 2010–2014 mean values. Percentages listed in **a** and **b** are current nutrient recycling rates. **c**, **d**, The minimum nitrogen-recycling rates (in percentage of recyclable solid–liquid organic wastes) required for each province to meet its critical pollution threshold under CNM (**c**) and INM-CRF (**d**). All discharge and recycling rates are normalized against the critical pollution threshold (marked as a red circle).

only from croplands but also from livestock, domestic and industrial wastes. Because 63% of current nitrogen discharge to freshwaters is from non-recycled livestock and domestic waste, future policies should pursue a transformational expansion of nutrient-recycling systems, together with water, sanitation and hygiene programmes. Results from long-term (>20 years) fertilization experiments in China indicate that combining synthetic fertilizers with manure can improve soil quality and generate yields of rice, maize and wheat that are 8.2–9.9% larger than those obtained using synthetic fertilizers alone[9]. Figure 4c, d demonstrates that reducing nitrogen discharge by sufficient amounts to return to provincial safe thresholds, although daunting, could nearly be achieved by implementing INM and increasing the national nutrient-recycling rate to 86–88%. Nutrient-recycling rates in nine provinces would need to exceed 95% under INM-CRF. Even with a recycling rate of 100%, Shanghai would still need to decrease industrial nitrogen discharge by 37%, and Shanxi must either reduce food production by 3% or decrease industrial nitrogen discharge by 80%. Increasing national nutrient-recycling rates to 86–88% could enable food productivity to be maintained at current levels while further reducing synthetic nitrogen requirements under INM from 17–19.8 Mt N yr^{-1} to 10.8–13.6 Mt N yr^{-1} (assuming equivalent nitrogen-use efficiency for organic and synthetic nitrogen). The side effects of increased recycling could include an increase in nitrous oxide emissions[26].

We evaluate the costs of three strategies to achieve our proposed increase in nutrient recycling: traditional wet manure recycling; dry compost recycling; and direct wastewater recycling, with consideration of the potential effects of new toilet technologies and infrastructure changes (Methods). We estimate that the operational costs of recycling organic waste would range from US$8–29 billion yr^{-1} based on 2010 prices (Supplementary Table 4). Traditional recycling is the most expensive option and carries sanitation and hygiene risks. A more practical approach would be to deliver domestic wastewater for irrigation, by separating industrial wastewater from domestic wastewater and connecting household sewage systems to existing irrigation systems. This approach would cost around US$104 billion for infrastructure, with operational costs of US$18–29 billion yr^{-1}.

Overall, the costs of building nutrient recycling systems are small relative to the annual cost of water pollution at current levels, which is estimated to be 1.5% of national GDP in 2010 (US$91 billion; comprising US$20 billion from treatment and US$71 billion from the effects of environmental degradation)[27]. Farmers could benefit more from nutrient recycling than from WWT because the former offers potential increases in job opportunities and income in rural areas. The risks to human health from increased nutrient recycling could be reduced by sealing domestic wastewater systems to minimize physical contact and by increasing wastewater disinfection (for example, by chlorination, aerobic treatment, ozonation and/or ultraviolet light). Livestock waste contributes around 14.3 Mt N yr^{-1} from excretion, and further improvements in its management—for example by the manipulation of animal diets, the trapping of particulate emissions and the installation of methane tanks—could reduce air and water pollution, increase manure-nitrogen recycling rates and improve sanitation. These initiatives could create new market opportunities for the fertiliser industry[28] and for farmers.

The substantial challenges that China must overcome in order to restore safe and sustainable levels of nitrogen in water bodies are also shared by other regions in which water pollution is increasing because of excessive nutrient discharge. This includes many parts of Asia, South and Central America, and sub-Saharan Africa[7]. Well-defined targets for nitrogen release into the local environment are essential for formulating effective regional policies to reduce pollution, which are in turn essential for progress at the global level.

Online content

Any methods, additional references, Nature Research reporting summaries, source data, statements of data availability and associated accession codes are available at https://doi.org/10.1038/s41586-019-1001-1.

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Additional information

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